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Response of stream macroinvertebrate communities to forest harvesting in the Piedmont region of North Carolina

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ABSTRACT

Forest disturbances have significant effects on water quality and quantity, river geomorphology, and the ecology of receiving waterbodies. Riparian forests provide numerous functions for aquatic communities including retaining fine sediments and nutrients, controlling water temperature, and providing food sources and habitat for aquatic organisms. Forestry Best Management Practices (BMPs) use riparian forests as buffers to mitigate potential sources of disturbance to aquatic ecosystems from forest management. The objective of this study was to quantify the impacts of timber harvest on stream macroinvertebrates in the Piedmont region. We assessed the changes in macroinvertebrate communities and identified their relationships with specific hydrologic and water quality parameters. We used a paired watershed approach to quantify the response of watershed hydrology and water quality to clearcut forest harvesting with the use of BMPs in the Hill Demonstration Forest and Umstead Research Farm in central North Carolina. We sampled macroinvertebrates and monitored water quality in the first-order streams, and surveyed vegetation within riparian zones one year preharvest (2010) and four years postharvest (2011–2014). We found more sensitive species (indicated by biotic index classifications), scrapers, and Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa from the harvested watershed than in the reference watershed at the Umstead Research Farm site. No differences were detected between the reference and treatment watersheds at the Hill Demonstration Forest site. More sensitive species were present in watersheds with the highest pine basal area and in-stream total organic carbon (TOC) loads. More scrapers were present in watersheds with low hardwood basal area, high stream flow, and more vines. EPT abundance was higher in watersheds with high stream flow, large pine trees, and thick leaf litter layer. EPT abundance was lower in watersheds with large hardwood trees and high TOC loads. Overall, sensitive species, scrapers, and EPT abundance were lower in the Umstead Research Farm reference watershed than in any other watershed. We conclude that, in contrast to similar studies in the region, clearcut harvesting with the use of BMPs did not cause detectable negative effects on macroinvertebrate communities. Moreover, water quality as determined by macroinvertebrates may even be improved in some cases following clearcutting. This study provides a better understanding of how macroinvertebrate communities in Piedmont streams change after harvesting and what watershed characteristics may be driving these changes. This information is useful in characterizing macroinvertebrates in headwaters in the Piedmont, and helps land managers protect aquatic resources across the region.

1. Introduction

Disturbances such as tree harvesting, road crossings, skid trail construction and use, site preparation, and prescribed burning have the potential to cause soil erosion and increase the amounts of sediment and nutrients delivered to streams during and following logging operations (Campbell and Doeg 1989, Webster et al. 1992, Martin et al. 2000). These changes may have significant effects on the water quantity and quality, stream geomorphology, and the ecology of receiving waterbodies. Consequently, forest upland disturbances may affect aquatic ecosystems including macroinvertebrate communities – important bioindicators of ecosystem health.

Headwaters streams are often closely linked to riparian forests (Newbold et al. 1980, Webster et al. 1992) and support biologically and hydrologically important processes (e.g., ecosystem services) (England and Rosemond 2004, Richardson and Danehy 2007, Yamashita et al.

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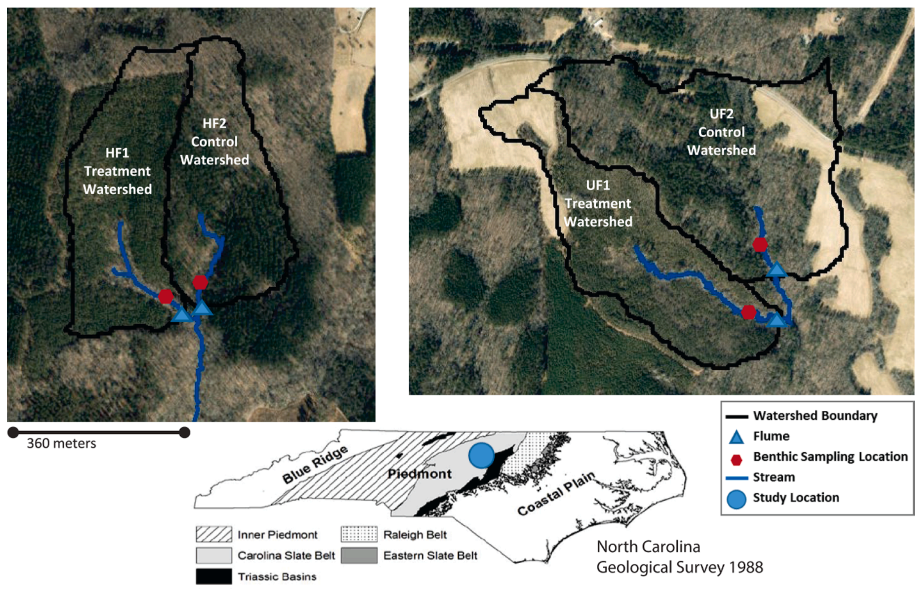

	Hill Demonstration Forest		Umstead Research Farm	
	HF1	HF2	UF1	UF2
Watershed size (ha)	12	12	19	29
Stream length (m)	300	260	550	200
Stand type	Mixed-pine hardwood		Mixed-pine hardwood	
Stand Age (years)	45		80	
Slope (%)	13		7	
Geologic Features	Carolina Slate Belt		Triassic Basin	
Dominate Soil Series	Tatum and Appling		Helena	
Soil characteristics	Non expansive clays, no perched water, deep soils, and drains water slowly throughout the year due to large amounts of stored water in bedrock and topographic control.		Expansive clays, perched water, thin soils, drains water slowly in growing season when soils were dry with an inactive confining clay layer, and fast in dormant season when soils were wet with an active confining layer.	

Fig. 1. Study watersheds and site characteristics, Piedmont North Carolina.

2011). Timber harvest can change several aspects of stream ecosystems including water temperature (Corbett et al. 1978, Gravelle et al. 2009a), light (Corbett et al. 1978, Campbell and Doeg 1989), sediment loads (Campbell and Doeg 1989, Karwan et al. 2007), nutrient loads (Ensign and Mallin 2001, Gravelle et al. 2009a), and flow regimes (Fuchs et al. 2003, Boggs et al. 2016). Changes in riparian forests can also greatly influence the timing, quantity, and quality of allochthonous coarse particulate organic matter inputs to streams (Santiago et al. 2011, Six et al. 2022). These changes can affect both abiotic and biotic components of stream ecosystems (Relyea et al. 2000, Kobayashi et al. 2010). To protect water quality from forest management and disturbances, best management practices (BMPs) have been widely used in the southern U. S. (Anderson and Lockaby, 2011; Jackson, 2014; Cristan et al., 2016). Among these practices, riparian buffers have proved to be useful for protecting water quality in headwater watersheds.

Changes in sediment, organic matter, temperature, and light regimes may cause subsequent changes in macroinvertebrate communities (Newbold et al. 1980, Brown et al. 1997, Haggerty et al. 2004). Disturbance by timber harvest can result in highly variable macroinvertebrate responses that are often linked to the forest type, physiography, hydroclimatic regime, and use of BMPs. For instance, in southwestern Oregon, there were higher densities but a lower richness of macroinvertebrates in harvested compared to reference streams (Gerth et al. 2022), whereas in coastal Washington there were higher densities of macroinvertebrates in harvested compared to reference streams but no difference in macroinvertebrate richness between stream types (Jackson et al. 2007). Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies) (EPT) abundance did not change after harvest in Louisiana or northern Idaho streams (Gravelle et al. 2009b, Klimesh et al. 2015), whereas EPT abundance increased after harvest in Arkansas streams (Brown et al. 1997). In the North Carolina Piedmont region harvested watersheds had lower macroinvertebrate richness and diversity, as well as fewer intolerant species than reference watersheds (Goodman et al. 2006). Studies also highlight the importance of riparian zone buffers, with fewer negative effects detected in buffered streams (Jackson et al. 2007; Gerth et al. 2022). Differences among studies suggest that conducting studies in specific forest types, physiographic regions, and hydroclimatic regimes while assessing differences between BMPs is critical to understanding disturbance effects on stream ecosystems.

The Piedmont region represents about 200,000 km^2 of the southeastern U.S. This region is known to have soil erosion and sediment problems due to land cultivation before the 1940s (Trimble et al., 1987; Riekerk et al., 1989). The region is now heavily forested from cropland abatement and reforestation for timber production. As the timber market continues to expand in the southeast, landowners may increase their investments in timber production in the Piedmont and other regions (Wear and Greis 2012). Local or site-specific conditions in the Piedmont should be accounted for when developing forest management actions during timber harvest operations to protect water quality (Jackson et al. 2004). Piedmont streams tend to have very different structural characteristics (e.g., moderate gradient streams) than coastal and mountain streams, leading to differences in habitat, food sources, and ultimately different macroinvertebrate community structures (Hax and Golladay

Table 1
Summary of field monitoring parameters. * indicates parameters that differentiated treatment and reference sites; ** indicates parameters that were used in macroinvertebrate assemblage correlations to watershed characteristics.

Data Category	Parameters	Frequency	Methods
Watershed hydrology			
Water Quantity	**Stream discharge (mm)	10-minute intervals	Calculated from stage and 2-H flumes with Sigma™ water level recorders
Riparian vegetation structure	Timber overstory *Pine stem count (stems/ha) **Pine basal area (m^2/ha) *Hardwoods stem count (stems/ha) **Hardwoods basal area (m^2/ha) *Total overstory basal area (m^2/ha) **Canopy cover (%) Timber midstory **Midstory stem count (stems/ha) Ground cover *Woody plants (%) *Herbaceous plants (%) **Vines (%) *Ferns (%) **Coarse woody debris (%) **Fine woody debris (%) *Leaf litter (%) *Rock (%) *Bare soil (%) **Moss (%) **Fungus (%) **Litter depth (cm)	Preharvest and postharvest	Modified Carolina Vegetation Survey Method, with 150 m^2 plots with 1 m^2 subplots
Water quality			
Water chemistry	Total Suspended Solids (TSS; mg/l) **Nitrate nitrogen (NO_3-N; mg/l) Ammonium nitrogen (NH_4-N; mg/l) Total phosphorus x(TP; mg/l) Total Kjeldahl nitrogen (TKN; mg/l) *Total nitrogen (TN; mg/l) Total organic nitrogen (TON) **Total organic carbon (TOC; mg/l)	Bi-weekly (baseflow) and storm-initiated (stormflow)	Grab samples (baseflow) and Sigma™ automated sampler (stormflow)
Water temperature	Temperature (°C)	10-minute intervals	Hobo™ Water Temp Pro V2 Logger
Aquatic community			
Benthic macroinvertebrates	Species richness (*N*) Species density (semiquantitative) Functional Feeding group (%) Species biotic index	Biannually	Semi-qualitative method described by NCDENR-DWR, Qual4 (2012)

1998, Kedzierski and Smock 2001, Goodman et al. 2006). Regional differences in precipitation, topography, and other landcover dynamics can limit our ability to extrapolate from hydrologic and water quality studies in other physiographic provinces (Riekerk et al., 1989; Sun et al., 2002; Sun et al., 2004).

The effects of forest management on water quality and quantity have been well studied in the Piedmont region (Hewlett et al. 1984, Grace 2004, Aust et al. 2015). Most recently, Boggs et al. (2016) assessed effects of timber harvest on water quantity and quality in six small watersheds in the Piedmont region of North Carolina. Discharge was more than doubled and stormflow peak nitrate reached its maximum concentrations postharvest in all treatment watersheds. Despite increases in nitrate, levels remained low and quickly returned to preharvest concentrations within two years. Total suspended solids (TSS), total phosphorus (TP), total organic nitrogen (TON), and total organic carbon (TOC) increased postharvest in some watersheds but not all treatment watersheds. Bioclassification of benthic macroinvertebrates indicated that stream water quality remained good/fair to excellent in treatment watersheds after harvest (Boggs et al. 2016).

Here, we evaluate the impacts of timber harvest on macroinvertebrate communities in Piedmont streams studied by Boggs et al. (2016) by assessing changes in macroinvertebrate community structure and identifying relationships of specific water quality properties to macroinvertebrate communities. Our goal is to assess the health of stream ecosystems in harvested (clearcut) and undisturbed Piedmont watersheds, using macroinvertebrate communities as indicators of ecosystem health. We addressed two questions: (1) are there differences between macroinvertebrate communities in watersheds that have been clearcut compared to those that have not, and if so, do these differences change over time? and (2) do water quality and quantity characteristics that are impacted by forest harvest also drive macroinvertebrate community structure?

2. Methods

2.1. Study watersheds

We sampled two pairs of watersheds in the North Carolina Piedmont region (Fig. 1; North Carolina Geological Survey, 1988) using a standard paired watershed approach (Boggs et al. 2016). The paired watersheds were in the Hill Demonstration Forest (treatment watershed—Hill Demonstration Forest-1 compared to reference watershed—Hill Demonstration Forest-2) and Umstead Research Farm (treatment watershed—Umstead Research Farm-1 compared to reference watershed—Umstead Research Farm-2) in the Neuse River Basin. Hill Demonstration Forest watersheds were in the Flat River Watershed at the North Carolina State University Hill Demonstration Forest in northern Durham County. Umstead Research Farm watersheds were in the Knap of Reeds Watershed at the North Carolina Department of Agriculture and Consumer Services Umstead Research Farm in western Granville County. Land use within both watersheds was primarily forest, with small amounts of agriculture in the Umstead Research Farm watersheds. Dominant overstory species in both watersheds included red maple (*Acer rubrum*), pignut hickory (*Carya glabra*), mockernut hickory (*Carya tomentosa*), white oak (*Quercus alba*), northern red oak (*Quercus rubra*), American beech (*Fagus grandifolia*), sweetgum (*Liquidambar styraciflua*), tulip poplar (*Liriodendron tulipifera*), sourwood (*Oxydendrum arboreum*), and loblolly pine (*Pinus taeda*).

Watersheds ranged from 12 to 29 ha each with a single first-order perennial stream 200 to 550 m long. Streams contained large woody debris and patches of aquatic vegetation with abundant detritus and small woody debris. Hill Demonstration Forest streams were about 1 m wide and 30 cm deep, connected to a narrow floodplain with rocky substrate. Hill Demonstration Forest stream channels had steep upland slopes ranging from 15 to 40% with watersheds underlain by Carolina Slate Belt (CSB) soil characteristics (Rogers 2006). Hill Demonstration Forest upland soils were well drained (>2 m depth to the water table) and functioned similarly in the growing and dormant seasons. Umstead Research Farm streams were about 2 m wide and 1.5 m deep, detached from a wide floodplain with a sandy substrate and gentle upland slopes averaging 7%. Umstead Research Farm watersheds were underlain by clayey Triassic Basin (TB) soil characteristics (US Department of Agriculture 1971). The clay layer in TB soils creates impermeable conditions that result in a perched water table during the dormant season and causes variability in how TB soils store, release, and generate water between the growing season and the dormant season. Additional details on stream channels can be found in Boggs et al. (2013), Boggs et al. (2016), and Dreps et al. (2014).

2.2. Timber harvest treatment

The upland of treatment watersheds was clearcut harvested using typical rubber tire-mounted logging equipment (Boggs et al. 2016), while no harvesting occurred in reference watersheds. Logging on Hill Demonstration Forest-1 and Umstead Research Farm-1 took place November 29, 2010–January 19, 2011, and July 7–September 8, 2010, respectively. A 15.2 m riparian buffer was retained on each side of the stream. High-value trees (pine trees ≥ 35.6 cm dbh and hardwood treed ≥ 40.6 cm dbh) were harvested from the riparian buffer as allowed by the Neuse River Basin Riparian Buffer Rule (NRR). Additional BMPs (i.e., trees skidded to the log deck without crossing stream channel and slash redistributed across the upland to limit soil disturbance) were deployed to prevent sedimentation and other forms of water quality pollution (details in Boggs et al. 2016). Hill Demonstration Forest-1 and Umstead Research Farm-1 were replanted with loblolly and shortleaf pine (*Pinus echinata*), respectively, between June 2011 and January 2012 (details in Boggs et al. 2016). Dramatic post-harvest hydrological changes were detected for both treatment watersheds (Boggs et al., 2016).

2.3. Macroinvertebrate sampling

We collected five years of benthic macroinvertebrate samples during the nongrowing (January –April) and growing (June–July) seasons at treatment and reference watersheds pre- and postharvest (Table 1). Watersheds were sampled twice preharvest (January and April 2010), and seven times postharvest, in the nongrowing and growing seasons of 2011–2013 and nongrowing season of 2014. Surveys were taken across seasons to capture differences in the life cycle of benthic macroinvertebrates. Benthic macroinvertebrate surveys were completed following the semiquantitative Biological Assessment Unit Qual-4 method outlined by the North Carolina Department of Environmental and Natural Resources (2012) Division of Water Resources (NCDENR). The Qual-4 method was designed for small streams (drainage area ≤ 8 km^2) and included four sampling types – kick netting through a riffle, sweep netting under rocks and gravel, collecting one leaf pack, and flipping over rocks to hand collect macroinvertebrate (visual sampling). About 5% of each stream was sampled. All samples were combined, sorted, and sent to Watershed Science, LLC, to be identified to the lowest possible taxonomic class. In general, macroinvertebrates were identified to genus, with most EPT, crustaceans, mollusks, and Odonata identified to species.

2.4. Water quantity and quality measurements

Beginning in 2008, we continuously measured stream discharge during both pre- and postharvest periods (Boggs et al., 2016) (Table 1). We used a 2-H flume as the flow reference structure at each watershed outlet. A Sigma 900 Max water sampler (Hach Company, Loveland, CO) with a depth sensor was used to measure and log discharge data every 10 min.

To determine the effect of harvesting on water quality, we measured stream temperature and water chemistry (Table 1). Stream temperature data were logged every 10 min using HOBO Pro v2 (Onset Computer Corporation, Bourne, MA) water temperature sensors. Water chemistry parameters included TSS, TOC, ammonium-nitrogen (NH_4-N), nitrate-nitrogen (NO_3 -N), TP, total Kjeldahl nitrogen (TKN), TON, and total nitrogen (TN). Water chemistry concentrations and loads were quantified from grab and storm-based water samples (Boggs et al. 2016). Grab samples were collected biweekly, under baseflow conditions. Storm-based samples were automated samples collected by Sigma 900 Max water samplers. Storm-based samples were collected on a stratified sampling program; intensive sampling (6 samples in 1 h) during the rising limb (initial increase in discharge in response to a storm event) and less intense sampling (6 samples over 6–10 h) during the recession limb (post-storm decrease towards baseflow conditions) of the hydrograph. Flow weighted concentrations and loads were determined to avoid overemphasizing one limb of the hydrograph (Boggs et al. 2016). Samples were preserved with sulfuric acid to a pH of < 2 and stored at 3.6° C before analysis. Constituents from each water sample were assayed at the North Carolina State University Soil Science Analytical Laboratory using standard methods (Greenburg 1992).

2.5. Riparian vegetation Assessment

We characterized the riparian buffer vegetation composition by establishing 152 m^2 survey plots across 10% of each buffer (Boggs et al. 2016). We assessed four plots in Hill Demonstration Forest-1, 6 in Hill Demonstration Forest-2, 10 in Umstead Research Farm-1, and 4 in Umstead Research Farm-2. Stem count of over- and midstory trees, diameter at breast height (dbh) of overstory trees, and percent canopy cover in each plot were measured annually (2009 [preharvest]; 2011–2013 [postharvest]) following protocols outlined in the Carolina Vegetation Survey (Peet et al. 1998). During the growing season, the percent canopy cover was measured at the plot center with hemispherical photography (Paletto and Tosi 2009). In each plot, we established six 1 m^2 subplots to estimate the percent groundcover and leaf litter depth.

2.6. Data analyses

We examined the differences in macroinvertebrate communities between treatment and reference streams using both univariate and multivariate analyses. Because preharvest collections of macroinvertebrates were not taken during the same seasonal timeframe as postharvest collections, we assessed pre- and postharvest data separately. We hypothesized that if forest harvest did not affect macroinvertebrate communities, then the relationship of macroinvertebrate communities between treatment and reference streams should be similar pre- and postharvest. The lowest identifiable taxa levels were used in analyses (i.e., specimens identified to family or genus were counted the same as those identified to species). In each watershed, during each sampling round, macroinvertebrate taxa were assigned a semi-quantitative density (NCDENR, 2012): rare (1–2 specimen collected) = 1, common (3–9 specimen collected) = 3, and abundant (≥10 specimen collected) = 10. We used these values to summarize macroinvertebrate communities in seven ways: community structure (species × watershed matrices of semiquantitative densities), functional feeding group (FFG) structure (FFG × watershed matrices of FFG percentages), total richness, EPT richness, EPT abundance (semiquantitative density), intolerant taxa richness, and stream biotic index (BI; a weighted average of the tolerance values of samples with respect to their abundance, NCDENR, 2012). For univariate models, histograms of model residuals did not depart from normality, and for multivariate models, dispersion did not vary among groups.

2.6.1. Community and FFG structure comparisons

To test if macroinvertebrate community and FFG structures differed between treatment and reference streams, we used PERMANOVAs. We constructed separate models for pre- and postharvest collections and square root transformed species and FFG abundances to reduce the contribution of highly abundant groups in relation to less abundant groups (Anderson et al. 2008). We calculated Bray-Curtis similarity and Euclidean distance matrices for species and FFGs, respectively, to compare differences between each pair of streams. For preharvest analyses, we used PERMANOVAs to assess differences between community and FFG structures (response variable) of treatments (harvested versus reference streams; fixed effect) with treatments nested in site, and stream as a repeated measurements factor. For postharvest analyses, we used PERMANOVAs to test responses of assemblage structures to treatment, season, and year, with treatment nested by site, and stream as a repeated measurements factor. Models included 2-way interactions of treatment × season and treatment × year, as well as 3-way interactions with all variables. We used pairwise PERMANOVAs to obtain *p*-values for interactive effects between pairs of watersheds. We also conducted 2-dimensional non-metric multidimensional scaling (NMDS) (Clarke 1993) to visualize differences in assemblage structure identified by PERMANOVA. We used the PERMANOVA add-on (Anderson et al. 2008) in PRIMER 7.0 (Quest Research Limited) to conduct NMDS and analyze data, with 9999 permutations of residuals in both the main tests and post-hoc comparisons. When paired treatment and reference watersheds had distinctive communities as indicated by separation in ordination graphs, indicator species analyses (ISA) (Dufrêne and Legendre 1997) were used to determine which taxa were most responsible for assemblage differences. ISA were performed with the multipatt function of the *indicspecies* package (Cáceres et al. 2022) in R software (version 4.2.1; R Project Statistical Computing, Vienna, Austria).

2.6.2. Richness, abundance, and biotic index comparisons

To test if macroinvertebrate richness and abundance, as well as stream BI differed between treatment and reference streams, we used linear mixed-effect repeated-measures models (LMER) in separate analyses for pre- and postharvest collections. We constructed separate models for each scaled comparison and log transformed the data to meet normality assumptions for maximum likelihood tests. We used the same fixed and repeated measurements as described above in PERMANOVA models and used Tukey's honestly significant difference post-hoc tests for comparing means. Analyses were performed with the lmer function of the *lmerTest* package version 2.0 (Kuznetsova et al. 2015) in R software.

2.6.3. Watershed characteristic comparisons

To test the null hypothesis of no differences in watershed characteristics (water quality, water quantity, and riparian vegetation) between reference and treatment watersheds postharvest, we used canonical analysis of principal coordinates (CAP). This analysis identifies the axes that best discriminates a priori groups and tests these predictions using a permutation test (Heino 2013). Postharvest water quality measurements were averaged over a 6-month period for summer (April–September) and winter (October–March) estimates each year. We averaged the stem and basal area of riparian zone trees across sampling plots for each sampling round. We also averaged the percent ground cover and leaf litter depth across subplots. All watershed characteristics (Table 1) were $\log_e$ (variable + 1) transformed and normalized to zero mean and unit variance so that characteristics had comparable, dimensionless scales. We calculated Euclidean distance matrices between each pair of watershed's stream characteristics. To discriminate between treatment and reference watersheds, we ran CAP in PRIMER and identified watershed characteristics that were highly correlated ($r >$ 30%) with eigenvalues one or two. We visualized differences between treatment and reference watersheds using ordination plots and quantified separation watersheds using leave-one-out (LOO) allocation success (Bloom, 1991).

2.6.4. Macroinvertebrate assemblage correlation to watershed characteristics

To examine factors associated with postharvest macroinvertebrate community and FFG structure, we modeled the relationship of macroinvertebrate communities and FFGs to watershed characteristics. Because we were interested in differences between treatment and reference watersheds, only macroinvertebrate communities and watershed characteristics that differed between treatment and reference watersheds were analyzed. Highly correlated ($r \geq 0.8$) watershed characteristics were removed from analyses.

We measured the strength and significance of relationships between macroinvertebrate community and FFG structures and watershed parameters using multivariate, distance-based, linear models (McArdle and Anderson 2001). After square root transforming community and FFG structures, we assembled Bray-Curtis similarities and Euclidean distance matrices for community and FFG structures, respectively, between pairs of watersheds. All watershed characteristics were $\log_e$ (variable + 1) transformed and normalized. We used the 'Best' selection procedure where all possible combinations of predictors were tested and the best combination of parameters for each number of variables was selected. Because sample sizes were small relative to the number of estimated parameters, we based model selection on the corrected Akaike information criterion (AIC_c) (Burnham and Anderson 2004). Delta AIC_c values ≤ 2 represented the best-supported models (Hurvich and Tsai 1989). For each predictor variable, relative variable importance was calculated based on the variable's appearance in the AICc-best models (Burnham and Anderson 2004). Predictors with relative variable importance > 0.5 were considered important. Analyses were performed in PRIMER.

To examine factors associated with postharvest macroinvertebrate richness, abundance, and BI, we modeled the relationships between macroinvertebrate assemblage measures to watershed characteristics. We used LMER models fit with maximum likelihood estimations. In models, macroinvertebrate assemblage measures were dependent

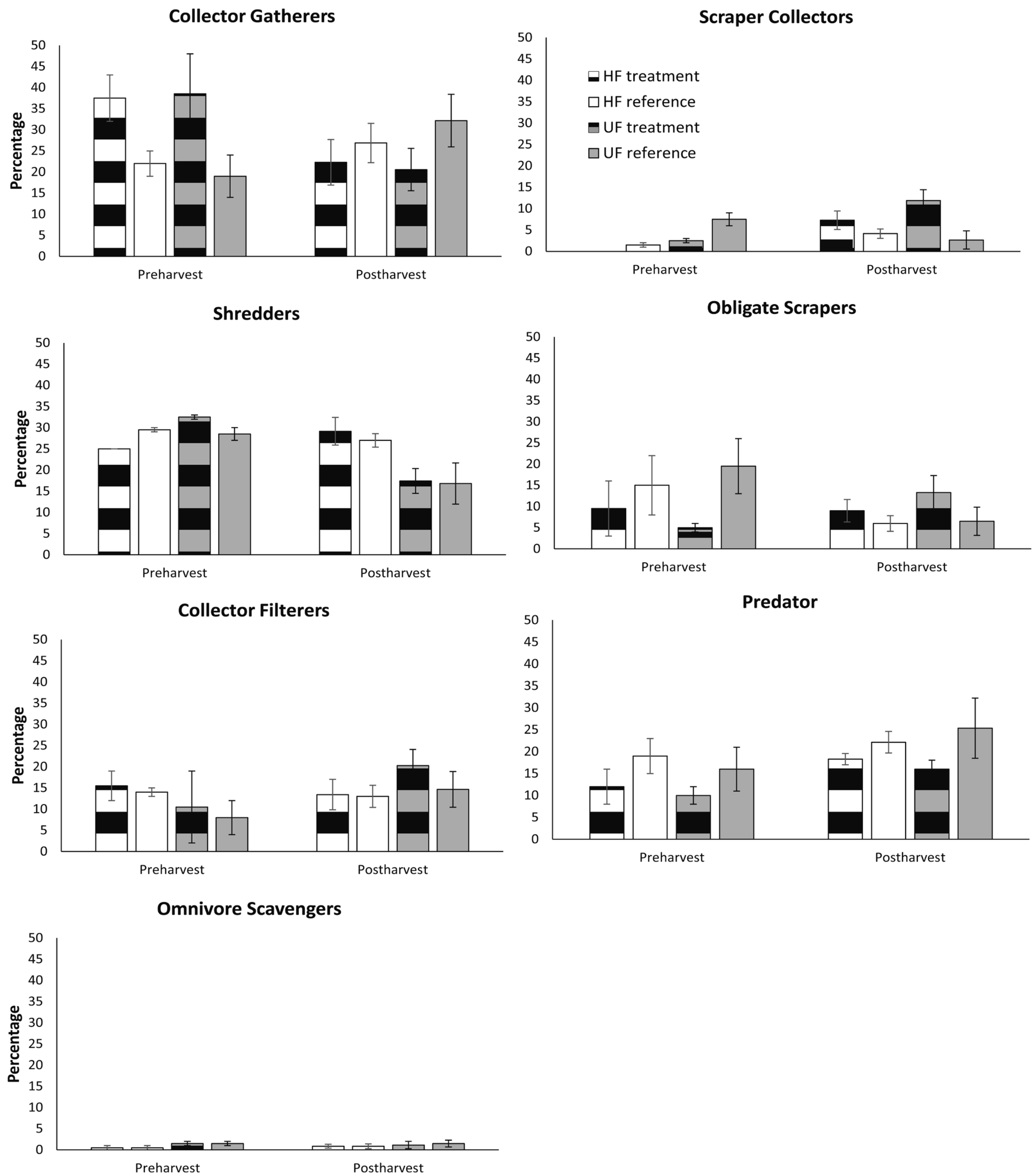

Fig. 2. Pre- and postharvest functional feeding group percentages for Hill Demonstration Forest (HF) and Umstead Research Farm (UF) treatment and reference watersheds.

variables and the stream was the repeated measurements factor. We used the MuMIn R package (Barton and Anderson 2002) to analyze all possible models. Model selection was based on AIC_c. We compared alternative models by weighting their level of data support (Hurvich and Tsai 1989) with all models with delta $\leq$ 2 of the lowest AIC_c value representing the best-supported models (Burnham and Anderson 2004).

3. Results

We collected 131 macroinvertebrate taxa, with 88 and 106 taxa collected in Hill Demonstration Forest and Umstead Research Farm streams, respectively (Table A1). EPT taxa comprised 30–63% of collections among streams pre- and postharvest. The dominant FFGs did not change in Hill Demonstration Forest streams pre- and postharvest, with collector gatherers and shredders most dominant (Fig. 2). Conversely,

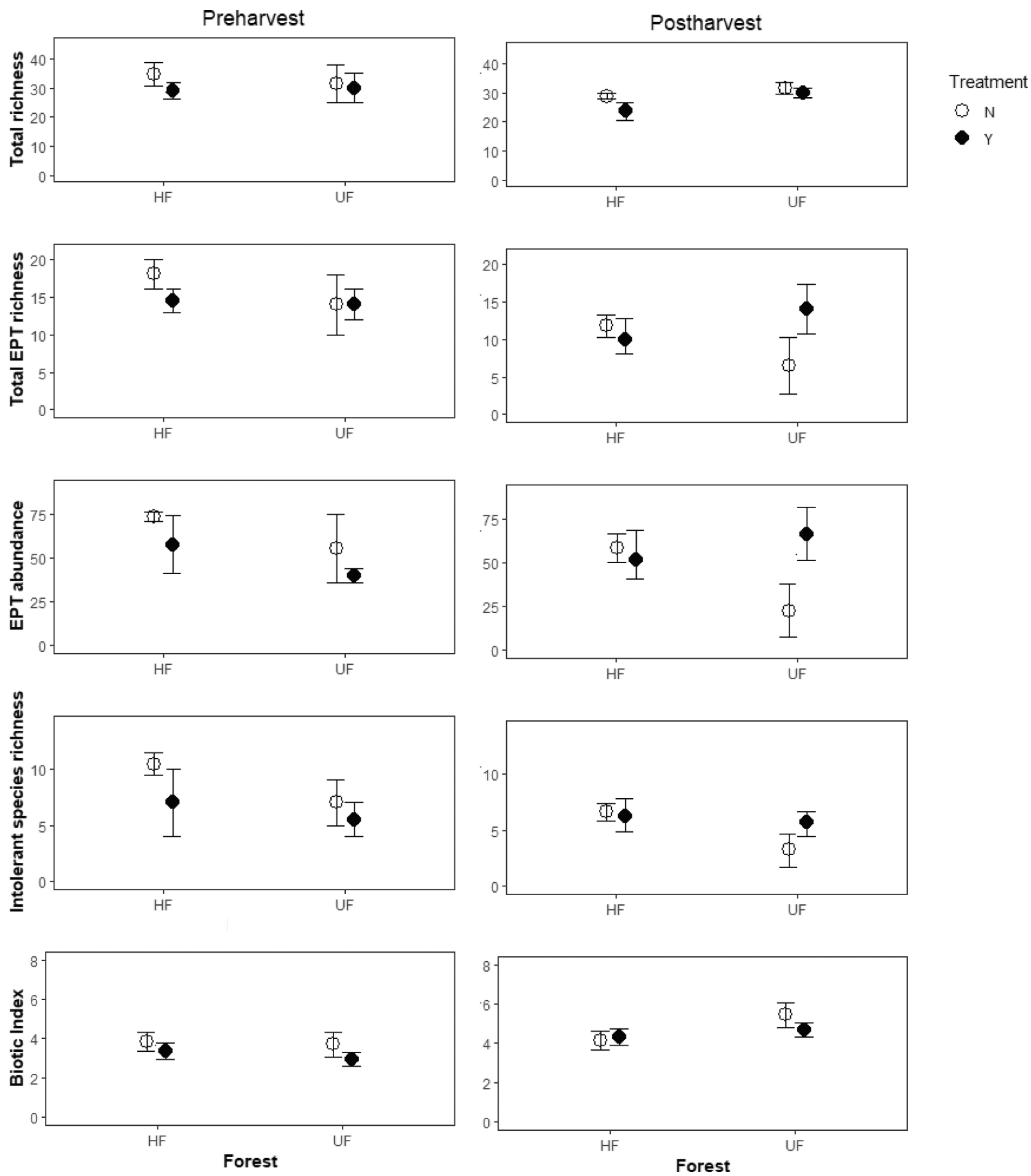

Fig. 3. Analysis of variance (ANOVA) results for total richness, total Ephemeroptera, Plecoptera, and Trichoptera taxa (EPT) richness, EPT abundance, intolerant species richness and biotic index pre- and postharvest for Hill Demonstration Forest (HF) and Umstead Research Farm (UF) treatment and reference watersheds. Y = treatment sites, N = reference sites.

the dominant FFGs did change between pre- and postharvest in Umstead Research Farm streams, with collector gatherers and shredders most dominant preharvest, and collector gatherers and predators the most dominant postharvest (Fig. 2). Omnivore scavengers were the least abundant FFG at all watersheds pre- and postharvest.

3.1. *Preharvest comparisons*

Reference and treatment watershed macroinvertebrate communities did not differ preharvest (all P values > 0.05). No differences were detected between macroinvertebrate community structure ($F_{2,7} = 1.05$, $P = 0.44$), FFG structure ($F_{2,7} = 1.55$, $P = 0.25$), richness (total, $F_{1,4} = 0.27$, $P = 0.63$; EPT, $F_{1,4} = 0.06$, $P = 0.82$; intolerant species, $F_{1,4} = 0.28$, $P = 0.63$), abundance (EPT, $F_{1,4} = 0.59$, $P = 0.48$), and biotic indices ($F_{1,4} = 0.28$, $P = 0.63$) preharvest (Fig. 3).

3.2. *Postharvest comparisons*

3.2.1. *Macroinvertebrate community*

Macroinvertebrate community structure differed between treatment and reference watersheds ($F_{2,27} = 4.15$, $P = 0.02$) in Umstead Research Farm ($t = 2.26$, $P = 0.05$) but not Hill Demonstration Forest ($t = 1.61$, $P = 0.19$) (Fig. 4). For Umstead Research Farm streams, ISA revealed six species that characterized communities in the harvested watershed and one species that characterized the reference watershed (Table 2). Eighty-three percent (5 of 6) of indicator species in the harvested watershed were either EPT species or species with a low BI (<3), indicating that these species are less tolerant of low water quality habitats than species that are not EPT or have a high BI. *Hydroporus* sp., a predatory beetle tolerant of low water quality, was the only indicator species for the reference watershed. In both forests, species communities differed between seasons in treatment and reference watersheds ($F_{1,27} = 7.77$, $P = 0.02$). Species assemblage structure did not differ between years ($F_{3,27} =$

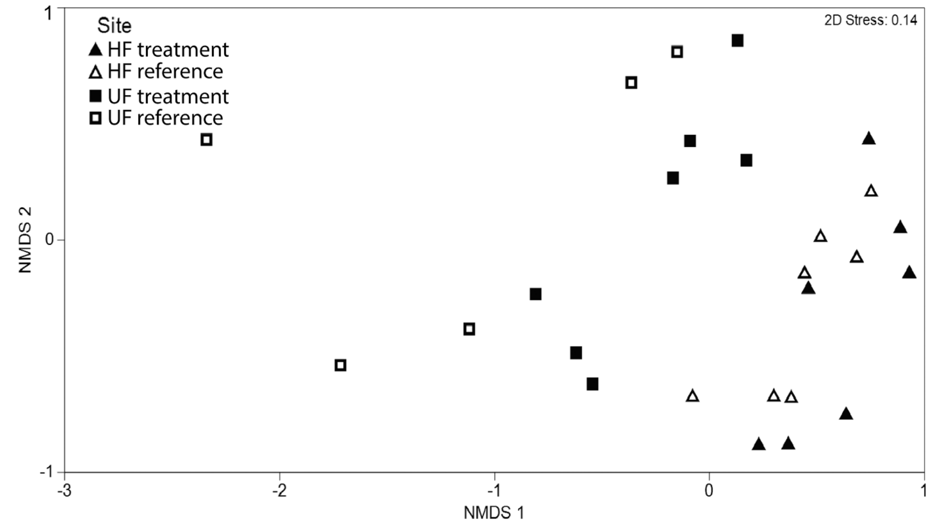

Fig. 4. Postharvest macroinvertebrate assemblage structure non-metric multidimensional scaling (NMDS) ordinations. Symbols represent collections from Hill Demonstration Forest (HF) and Umstead Research Farm (UF) treatment and reference watersheds. Distance between symbols reflects their Bray–Curtis dissimilarities based on species semiquantitative densities in 2-dimensional ordination space.

Table 2
Indicator species analyses at Umstead Research Farm. FFG = functional feeding group; EPT = Ephemeroptera, Plecoptera, and Trichoptera taxa; * = $P < 0.05$; ** = $P < 0.01$.

Taxa	Biotic Index	FFG	EPT	Specificity	Fidelity	Indicator Value
Harvested						
Chimarra spp.	2.8	Collector filterer	Y	*0.83*	*1.00*	*0.91***
Stenonema femoratum	7.2	Scraper collector	Y	0.82	1.00	0.90**
Paraleptophlebia spp.	0.9	Collector gatherer	Y	0.77	1.00	0.88*
Stenacron interpuctatum	6.9	Scraper collector	Y	1.00	0.71	0.85*
Pseudolimnophila	7.2	Collector gatherer	N	1.00	0.71	0.85*
Psephenus herricki	2.4	Scraper collector	N	0.83	0.86	0.85*
Reference						
Hydroporus spp..	8.6	Predator	N	0.92	0.86	0.89**

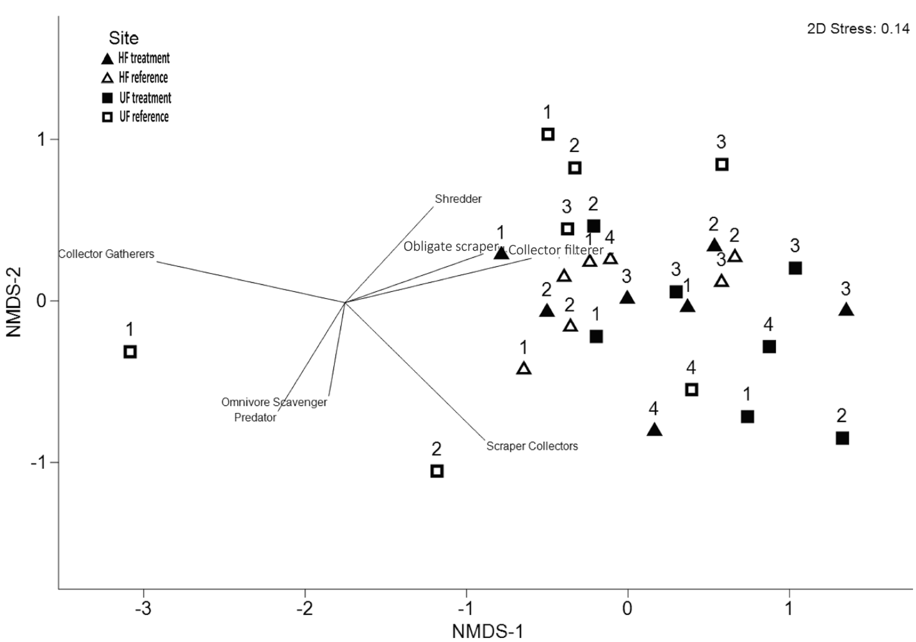

Fig. 5. Postharvest macroinvertebrate functional feeding group (FFG) structure non-metric multidimensional scaling (NMDS) ordinations. Symbols represent collections from Hill Demonstration Forest (HF) and Umstead Research Farm (UF) treatment and reference watersheds. Number labels above symbols represent spring and fall sampling each year postharvest, with 1 representing the first year postharvest, 2 representing the 2nd year postharvest, etc. Distance between symbols reflects their Bray–Curtis dissimilarities based on FFG percentages in 2-dimensional ordination space. FFG vectors show the relative association and magnitude of correlation for each group.

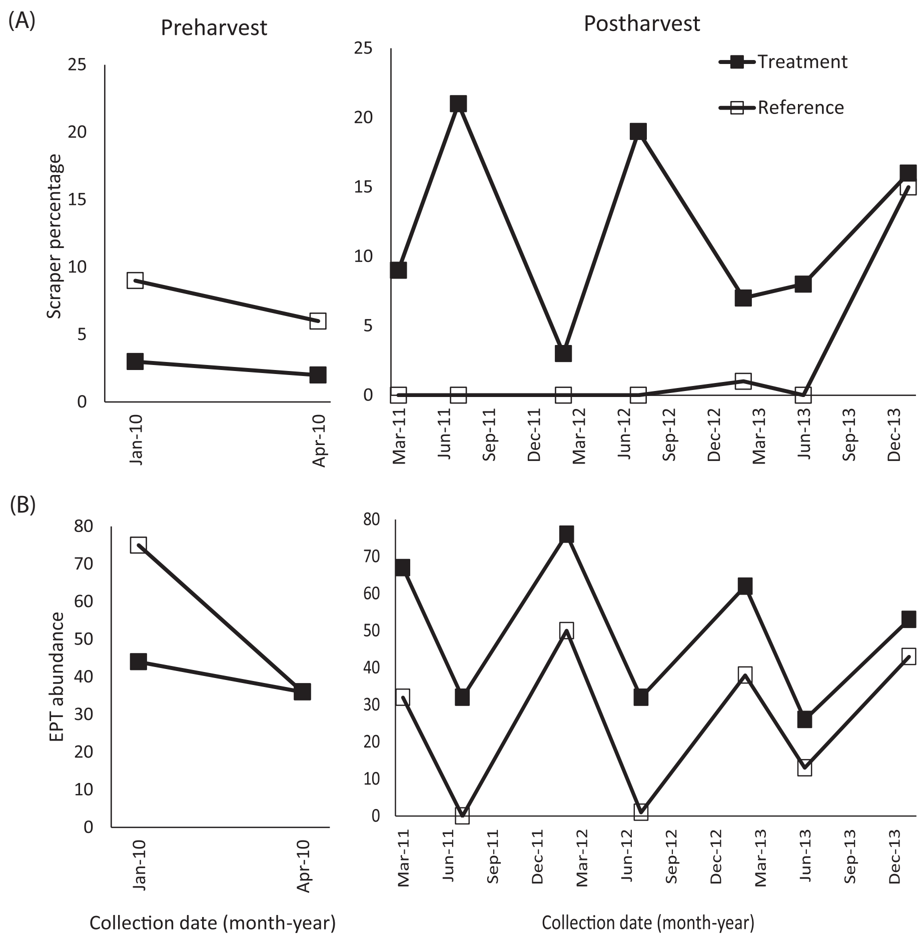

Fig. 6. Time series of scraper percentages (A) and Ephemeroptera, Plecoptera, and Trichoptera taxa (EPT) abundance (B) for pre- and postharvest collections in the Umstead Research Farm (UF) treatment and reference watershed. Only the UF watershed is displayed because no differences between treatment and reference watersheds were detected between the Hill Demonstration Forest watersheds.

1.70, $P = 0.06$).

Postharvest FFG structure differed between treatment and reference watersheds ($F_{2,27} = 4.36$, $P = 0.04$) in Umstead Research Farm ($t = 2.40$, $P = 0.05$) but not Hill Demonstration Forest ($t = 1.20$, $P = 0.33$) (Fig. 5). Scrapers were more dominant at Umstead Research Farm treatment watershed than reference watershed (Figs. 5, 6). In both forests, FFG structure also differed between years in treatment and reference watersheds ($F_{3,27} = 4.74$, $P < 0.01$).

Postharvest EPT abundance differed between treatment and reference watersheds ($F_{1,16} = 5.39$, $P = 0.03$) in Umstead Research Farm ($t = 2.70$, $P = 0.05$) but not Hill Demonstration Forest ($t = 0.50$, $P = 0.96$) (Figs. 3, 6). EPT abundance was 30% higher at Umstead Research Farm treatment watersheds than at reference watersheds. There were no differences in richness or BI measurements between treatment and reference watersheds (all $P > 0.05$) (Fig. 3). Richness (total [$F_{1,16} = 24.83$, $P < 0.01$] and EPT [$F_{1,16} = 15.11$, $P < 0.01$]), EPT abundance ($F_{1,16} = 13.68$, $P < 0.01$), and BI ($F_{1,16} = 5354$, $P = 0.03$) were higher in the winter than summer in both treatment and reference watersheds. Macroinvertebrate metrics did not differ between years (all $P > 0.05$).

3.2.2. *Watershed characteristics*

All watershed characteristics except TSS, NH_4-N, TP, TKN, TON, and water temperature differentiated treatment and reference watersheds postharvest (Table 1 and 3; Fig. 7). The treatment watersheds had more bare soil, understory plants, and coarse woody debris in riparian zones, and higher nitrogen (NO_3 and TN) and carbon (TOC) loads than reference watersheds. The reference watersheds had larger trees, as well as more canopy cover and leaf litter than treatment watersheds.

Canonical analysis of principal (CAP) coordinates supported the distinctiveness of treatment and reference watersheds (LOO = 100%). Postharvest comparisons within both forests revealed differences between stream characteristics in treatment and reference watersheds (Eigenvalue CAP 1 = 0.99; Eigenvalue CAP 2 = 0.97) (Fig. 7). Postharvest comparisons between forests showed similarities between treatment watersheds but differences between reference watersheds (Fig. 7).

3.2.3. *Macroinvertebrate and watershed characteristics*

Of the watershed characteristics that differentiated between treatment and reference watersheds, 13 were used in analyses due to high correlations ($r \geq 0.8$) (Table 1 and 3).

Three of the watershed characteristics that discriminated between treatment and reference watersheds were also correlated with macroinvertebrate community structure (Table 4). Pine basal area, TOC loads, and stream flow explained 34% of the variation in macroinvertebrate community structure. In Umstead Research Farm, larger pine basal area, higher TOC loads, and greater stream flows were associated with macroinvertebrate community structure in treatment but not in reference watersheds. In Hill Demonstration Forest, pine basal area was associated with macroinvertebrate assemblage structure, with larger pine basal area in reference relative to treatment watersheds.

Four of the watershed characteristics that discriminated between treatment and reference watersheds were also correlated with FFG structure (Table 4). Hardwood basal area, stream flow, vines, and coarse woody debris explained 38% of FFG structure variation. In Umstead Research Farm, smaller hardwood basal area and greater stream flow in

Table 3
Mean (standard error) of postharvest field monitoring parameters for Hill Demonstration Forest (HF) and Umstead Research Farm (UF) treatment and reference watersheds. * indicates parameters that differentiated treatment and reference sites; ** indicates parameters that were used in macroinvertebrate assemblage correlations to watershed characteristics.

	Parameters	HF treatment	HF reference	UF treatment	UF reference
Water quantity	**Stream discharge (mm)	1.01 (0.17)	0.53 (0.08)	1.55 (0.24)	1.28 (0.21)
Riparian vegetation structure	*Pine stem count (stems/ha)	56 (33.9)	246 (82.0)	31 (14.6)	0 (0.0)
	**Pine basal area (m^2/ha)	4.97 (2.99)	14.99 (5.06)	7.33 (3.92)	0.00 (0.00)
	*Hardwoods stem count (stems/ha)	394 (97.9)	438 (130.5)	333 (56.5)	453.9 (58.3)
	**Hardwoods basal area (m^2/ha)	17.9 (4.68)	18.1 (7.02)	15.4 (3.70)	40.0 (13.84)
	*Total overstory basal area (m^2/ha)	22.9 (3.84)	33.0 (6.04)	22.7 (3.81)	40.0 (6.91)
	**Canopy cover (%)	73.5 (5.18)	93.5 (1.15)	71.0 (5.38)	90.7 (1.16)
	**Midstory stem count (stems/ha)	1885 (391.4)	2510 (553.9)	2579 (542.4)	1613 (446.5)
	*Woody plants (%)	8.7 (2.16)	3.1 (0.57)	20.5 (3.41)	8.8 (3.56)
	*Herbaceous plants (%)	10.9 (2.39)	1.07 (0.60)	17.5 (2.76)	12.56 (4.02)
	**Vines (%)	3.8 (1.20)	2.9 (0.97)	12.6 (2.98)	10.68 (3.70)
	*Ferns (%)	3.0 (1.44)	0.3 (0.29)	7.0 (1.71)	5.3 (2.88)
	**CWD (%)	2.4 (1.23)	1.4 (0.70)	2.7 (0.79)	3.07 (1.66)
	**FWD (%)	10.5 (1.79)	8.2 (0.90)	8.2 (1.94)	5.6 (0.90)
	*Leaf litter (%)	44.3 (6.19)	78.8 (2.34)	25.9 (3.64)	53.8 (5.83)
	*Rock (%)	6.1 (3.24)	1.4 (0.97)	0.1 (0.07)	0.1 (0.06)
	*Bare soil (%)	8.0 (2.4)	1.2 (0.86)	5.4 (1.92)	0.1 (0.07)
	**Moss (%)	2.3 (1.09)	0.9 (0.35)	0.4 (0.15)	0.2 (0.05)
	**Fungus (%)	0.0 (0.05)	0.1 (0.08)	0.0 (0.01)	0.0 (0.01)
	**Litter depth (cm)	1.15 (0.13)	2.14 (0.29)	1.44 (0.16)	1.91 (0.21)
Water chemistry	TSS (mg/l)	31.07 (5.57)	29.53 (3.90)	32.67 (3.83)	33.44 (3.67)
	**NO_3-N (mg/l)	0.13 (0.07)	0.00 (0.00)	0.50 (0.20)	0.30 (0.07)
	NH_4-N (mg/l)	0.08 (0.04)	0.03 (0.01)	0.03 (0.01)	0.05 (0.07)
	TP (mg/l)	0.09 (0.03)	0.09 (0.02)	0.08 (0.01)	0.07 (0.02)
	TKN (mg/l)	0.70 (0.11)	0.66 (0.12)	0.78 (0.10)	0.75 (0.02)
	*TN (mg/l)	0.84 (0.10)	0.67 (0.12)	1.28 (0.20)	1.05 (0.10)
	TON (mg/l)	0.63 (0.09)	0.64 (0.11)	0.75 (0.10)	0.70 (0.07)
	**TOC (mg/l)	4.54 (0.60)	5.41 (0.50)	11.02 (1.09)	6.20 (0.41)
Water temperature	Temperature (°C)	14.41 (1.93)	14.11 (1.74)	14.04 (2.26)	13.86 (2.35)

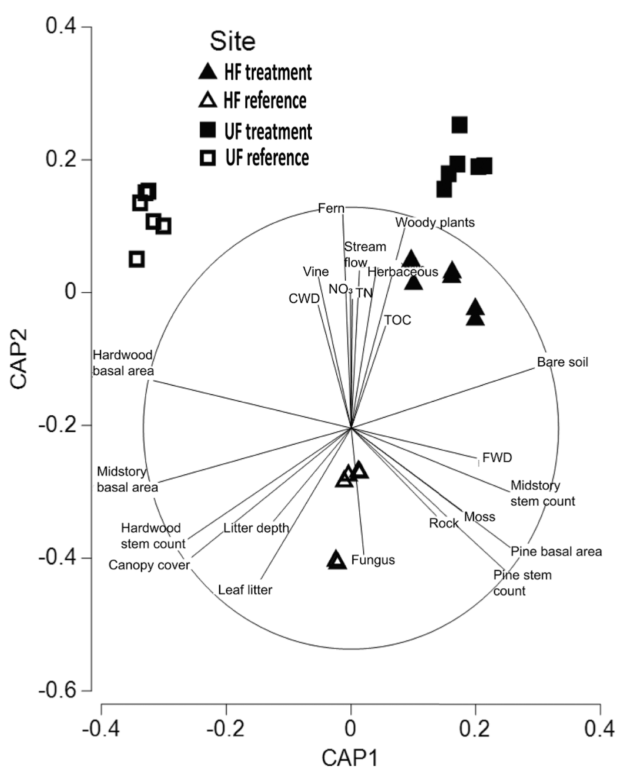

Fig. 7. Canonical analysis of principal coordinates (CAP) ordination plots based on watershed characteristics in Hill Demonstration Forest (HF) and Umstead Research Farm (UF) treatment and reference watersheds. Some symbols overlap. Black vectors represent raw Pearson correlations of stream properties that contributed > 30% to the dissimilarity between stream types. CWD = coarse woody debris, FWD = fine woody debris, TOC = total organic carbon, TN = total nitrogen.

treatment relative to reference watersheds were associated with FFG structure. No differences were detected between FFG structure in Hill Demonstration Forest watersheds.

Five of the watershed characteristics that discriminated between treatment and reference watersheds were also correlated with EPT abundance (Table 4). Stream flow, pine basal area, hardwood basal area, leaf litter depth, and TOC loads explained 46% of the variation in EPT abundance. EPT abundance was higher at watersheds with high stream flow, large pine trees, and thick leaf litter layer, and lowest in the watersheds with the largest hardwood trees and highest TOC loads.

4. Discussion

4.1. Differences between macroinvertebrate communities

Unlike previous studies of Piedmont streams where macroinvertebrate richness, diversity, and sensitive species decreased (Goodman et al. 2006, Helms et al. 2009), we found no detectable negative effects on macroinvertebrate community structure in this study. Few changes in macroinvertebrate community structure occurred after timber harvest, showing that clearcut harvesting with the use of BMPs did not greatly impact macroinvertebrate communities in study streams. Similarly, Boggs et al. (2016) did not find any differences between treatment and reference watershed bioclassifications, with bioclassification rankings of harvested watersheds remaining good/fair to excellent. The number of catchment pairs in the present study is not sufficient to fully analyze and explain differences between watersheds and studies, and further studies are needed to clarify the role of watershed characteristics in different responses to clearcutting. Differences in seasonal sampling of pre- and postharvest data also prevented direct analyses of pre- and postharvest data. Additionally, with only one treatment and reference pair in each watershed, we had low power to detect changes.

In this study, timber harvest increased the abundance of sensitive species (EPT species and species with low BI tolerance values) within the

Table 4
Model results of watershed characteristic correlations to macroinvertebrate communities. Results include variables from models that were within two AIC_c units of the best model. Watershed characteristics are listed by decreasing Pseudo-*F* values. *N* = number of models within 2 AICc units of the best model; SE = standard error; * = *P* ≤ 0.05; ** = *P* ≤ 0.01; RVI = relative variable importance (variables with RVI of 1.00 were included in all of the best models).

Model	Assemblage	Watershed characteristic	R^2	*N*	Estimate (SE)	*Pseudo-F*	RVI
Distance based linear model	Assemblage structure	Pine basal area	0.34	3		4.37**	1.00
		Total organic carbon				2.70**	0.67
		Stream flow				2.42*	0.33
	FFG structure	Hardwood basal area	0.38	5		3.55**	1.00
		Stream flow				2.13*	0.80
		Vine				1.98	0.60
		Course woody debris				0.94	0.20
Linear mixed-effects repeated measures model	EPT abundance	Stream flow	0.46	5	1.83 (1.14)	4.74*	0.60
		Total organic carbon			−1.26 (0.80)	2.94	0.40
		Pine basal area			0.61 (0.21)	1.74	0.40
		Hardwood basal area			−1.46 (0.48)	0.53	0.20
		Litter depth			−1.02 (0.66)	0.21	0.20

Umstead Research Farm harvested watershed. Streams in Umstead Research Farm differed from streams in Hill Demonstration Forest by having deeper channels detached from wide floodplains with gentle upland slopes (Boggs et al. 2016). Macroinvertebrate community structures in Hill Demonstration Forest treatment and reference watersheds were like assemblage structures in Umstead Research Farm treatment watersheds. Scrapers and other sensitive species were abundant in preharvest Hill Demonstration Forest streams. Furthermore, the shallow streams with narrower floodplains in Hill Demonstration Forest, as well as the less rapid lateral flow through CSB compared to TB soils, decreasing the discharge of stormflow and nutrients, may increase sensitive species in these streams. Streams in Umstead Research Farm were like the moderate gradient streams studied by Goodman et al. (2006) and Helms et al. (2009), but findings differed, with clearcut harvesting reducing diversity and sensitive species in their study streams. Changes to macroinvertebrate food sources with increased algae and grass inputs in clearcut streams may have driven the changes to macroinvertebrate communities seen by Goodman et al. (2006). We did not assess algal biomass or chlorophyll *a*, but we did assess the abundance of herbaceous plants in the riparian buffer. Like Goodman et al. (2006) herbaceous plants were more abundant at clearcut watersheds, however, the abundance of herbaceous plants was not correlated significantly with macroinvertebrate abundance, diversity, or richness.

Scraper collectors were the only FFG that differed between treatment and reference watersheds. Normally macrophyte, periphyton, and algae growth are limited by light in forested headwater streams (Gregory et al. 1991). Growing season net radiation increased from 11.9 Wm^{-2} preharvest to an average of 24.3 Wm^{-2} postharvest in these watersheds (Boggs et al. 2016). This type of increase in light into a stream system can increase macrophyte periphyton, and algae growth, changing the available food sources in headwater streams. These food source changes can increase the biomass of scrapers which consume algae and other fine particulate organic matter in streams (Murphy et al. 1981, Silsbee and Larson 1983).

4.2. *Impacts of water quality and quantity characteristics on macroinvertebrate communities*

The major short-term effects of timber harvesting on macroinvertebrates occur due to increased sediment input and light in streams (Campbell and Doeg 1989). Treatment and reference streams did not differ significantly in temperature or TSS concentration, which is likely due to the 15 m buffer strip maintained along each stream and other BMPs used to reduce overland flow. The use of buffer strips has been shown to decrease the negative effects of clearcutting on temperature (Gregory et al. 1991, Osborne and Kovacic 1993) and sediment inputs (Grizzel and Wolff 1998) in streams. The amount of residual shade after the harvest in the riparian buffer likely kept stream temperatures from increasing significantly. Trees within buffer strips, as well as increased herbaceous and woody plant growth, could have improved soil stability in treatment streams (Wynn and Mostaghimi 2006, Boggs et al. 2016), reducing TSS inputs into streams. Furthermore, as shown in previous studies (Golladay et al. 1989, Gregory et al. 1991, Osborne and Kovacic 1993) the proper use and implementation of buffer strips in this study appears to have protected water quality in the streams from the negative effects that can be associated with clearcut harvesting.

Macroinvertebrate communities were linked to TOC loads. Umstead Research Farm treatment watersheds had the highest TOC load (Boggs et al. 2016), as well as more scrapers and communities with more sensitive species. Increased microbial activity in response to the decomposition of logging residues may increase TOC concentration after clearcutting, whereas decreases in litter and throughfall inputs could decrease TOC (Kalbitz et al. 2004, Palviainen et al., 2015). Increased TOC in streams may increase invertebrate food sources and potentially species diversity. Increased TOC in streams can also lead to decreased dissolved oxygen levels and abundance of macroinvertebrate taxa (Joyce et al. 1985).

Detrital inputs by riparian forests provide habitat (Collier and Halliday 2000) and an important energy source (Cummins et al. 1983, Whiles and Wallace 1997) to headwater stream communities. The mixed-pine hardwood stands that were sampled in this study contained litter from both hardwood and pine trees to the streams. Pine basal area was higher in watersheds with more sensitive species, suggesting that pine needle inputs did not have a negative impact on communities. Prior works show that the slow breakdown of pine needles increases the habitat available to macroinvertebrates (Collier and Halliday 2000), which can increase headwater stream richness and diversity (Goodman et al. 2006). Pine needles not only increase habitat but may also increase macroinvertebrate food sources. However, pine needles are generally considered to be a low-quality detritus food resource (Friberg and Jacobsen 1994) due to their low nutrient content (Klemmedson 1992). Although pine needles may be a low-quality food source, studies have shown that trichopteran shredders that exploit pine litter for food and case-making may dominate streams in conifers dominant watersheds (Grafius and Anderson 1980, Whiles and Wallace 1997). Trichopterans dominated the shredder community of our study streams and comprised 61% of Hill Demonstration Forest shredders and 52% of Umstead Research Farm shredders. Consumption of pine needles by trichopterans may increase FPOM availability to other macroinvertebrate FFG, such as collector-filters and gatherers (Short and Maslin 1977, Mulholland et al. 1985), and other sensitive species.

Macroinvertebrate community structure, richness, and abundance often vary seasonally and yearly (Linke et al. 1999, Haggerty et al. 2004, Bêche et al. 2006, Helms et al. 2009). In our study, macroinvertebrate richness and abundance were higher in winter than in summer, and assemblage structure varied yearly. Factors driving seasonal and yearly

Table A1

Semiquantitative abundance (pre- and postharvest data combined) of macroinvertebrate taxa collected from Hill Demonstration Forest (HF) and Umstead Research Farm (UF) treatment and reference watersheds. BI = biotic index, FFG = functional feeding group, CG = collector gatherer, SC = scraper collector, Sc = obligate scraper, Sh = shredder, Pr = predator, CF = collector filterer, OS – omnivore scavenger, – indicates unknown BI or FFG.

Taxa	BI	FFG	HF treatment	HF reference	UF treatment	UF reference
Ephemeroptera						
Family Baetidae						
Acentrella turbida	4	CG	0	0	1	1
Acerpenna pygmaea	3.9	CG	2	11	3	0
Baetis spp.	–	CG	0	1	0	0
Baetis tricaudatus	1.6	CG	0	0	0	10
Centroptilum spp.	6.6	CG	1	0	22	5
Diphetor hageni	1.6	CG	0	4	0	0
Plauditus dubius gp	5.8	CG	0	0	0	1
Pseudocloeon propinquum	5.8	CG	0	0	0	0
Family Caenidae						
Caenis spp.	7.4	CG	0	0	1	0
Family Ephemerellidae						
Ephemerella dorothea	6	CG	0	0	0	0
Eurylophella verisimilis	4.3	CG	21	17	6	4
Family Heptageniidae						
Maccaffertium (Stenonema) femoratum	7.2	SC	0	0	53	24
Maccaffertium (Stenonema) modestum	5.5	SC	30	28	1	0
Maccaffertium (Stenonema) pudicum	2	SC	0	0	0	0
Stenacrom pallidum	1.1	SC	1	1	0	0
Stenacron interpuctatum	6.9	SC	0	0	11	1
Family Leptophebiidae						
Leptophlebia spp.	6.2	CG	10	11	19	43
Paraleptophlebia spp.	0.9	CG	32	60	58	13
Family Siphlonuridae						
Ameletus lineatus	2.4	Sc	2	0	8	25
Plecoptera						
Family Capniidae						
Allocapnia spp.	2.5	Sh	2	1	0	3
Family Chloroperlidae						
Sweltsa spp.	0	Sh	0	1	0	0
Haploperla brevis	1	Sh	0	1	0	0
Family Leuctridae						
Leuctra spp.	2.5	Sh	17	23	19	11
Family Nemouridae						
Amphinemura spp.	3.3	Sh	12	3	17	27
Family Perlidae						
Acroneuria abnormis	2.1	Pr	0	0	0	0
Acroneuria arenosa	2.3	Pr	0	1	0	0
Eccoptura xanthenes	3.7	Pr	26	33	6	0
Perlesta spp.	4.7	Pr	0	0	0	0
Family Perlodidae						
Clioperla clio	4.7	Pr	0	1	2	4
Isoperla namata	2	Pr	0	0	0	0
Family Taeniopterygidae						
Strophopteryx spp.	2.7	Sh	0	12	5	0
Trichoptera						
Family Calamoceratidae						
Anisocentropus pyraloides	0.9	Sh	1	4	1	0
Family Dipseudopsidae						
Phylocentropus spp.	6.2	CG	3	2	0	0
Family Hydropsychidae						
Cheumatopsyche spp.	6.2	CF	0	0	18	2
Diplectrona modesta	2.2	CF	51	67	24	23
Symphitopsyche sparna	2.7	CF	0	0	0	0
Family Lepidostomatidae						
Lepidostoma spp.	0.9	Sh	5	20	0	0
Family Limnephilidae						
Hydatophylax argus	2.2	Sh	11	12	3	3
Ironoquia punctatissima	7.8	Sh	0	0	3	3
Pycnopsyche lepida	2.7	Sh	43	51	28	27
Pycnopsyche gentilis	0.6	Sc	25	11	4	0
Pycnopsyche guttifer	2.6	Sh	7	5	6	8
Family Molannidae						
Molanna blenda	6.1	Sc	3	7	2	0
Family Odontoceridae						
Psilotreta frontalis	0	Sc	12	28	23	12
Family Polycentropodidae						
Polycentropus spp.	3.5	Sh	0	1	0	0
Neureclipsis spp.	4.2	Sh	0	0	0	0
Family Philopotamidae						
Chimarra spp.	2.8	CF	1	1	41	11

(*continued on next page*)

Table A1 (*continued*)

Taxa	BI	FFG	HF treatment	HF reference	UF treatment	UF reference
Dolophilodes spp.	0.8	CR	0	0	0	0
Wormaldia spp.	0.7	CF	1	8	5	2
Family Phryganeidae						
Ptilostomis spp..	5.9	Pr	0	0	0	1
Family Psychomyiidae						
Lype diversa	4.1	Sc	1	2	0	1
Family Rhyacophilidae						
Rhyacophila acutiloba	0	Pr	1	4	0	0
Rhyacophila carolina	0	Pr	6	0	0	0
Rhyacophila ledra	3.9	Pr	2	0	13	12
Rhyacophila nigrita	0	Pr	1	0	0	0
Family Uenoidae						
Neophylax atlanta	1.5	Sc	3	13	7	11
Neophylax consimilis	1.5	Sc	2	0	6	2
Diptera: Miscellaneous families						
Family Ceratopogonidae						
Palpomyia (complex)	6.9	Pr	2	4	1	3
Family Culicidae						
Aedes spp.	10	CG	0	0	10	0
Anopheles spp.	8.6	CG	0	0	1	3
Culex spp.	10	CG	0	0	0	3
Family Dixidae						
Dixa spp.	2.6	CF	29	16	7	1
Family Ptychopteridae						
Ptychoptera spp.	–	CG	5	6	0	0
Family Simuliidae						
Prosimulium spp.	6	CF	0	0	3	3
Simulium spp.	6	CF	2	0	18	18
Family Tabanidae						
Chrysops spp.	6.7	Pr	1	4	1	0
Family Tipulidae						
Dicranota spp.	0	Pr	4	4	0	0
Hexatoma spp.	4.3	Pr	5	7	6	2
Pseudolimnophila	7.2	CG	24	11	40	1
Tipula spp.	7.3	Sh	41	27	12	17
Diptera: Chironomidae						
Ablabesmyia parajanta/janta	7.4	Pr	0	1	0	0
Apsectrotanypus spp.	1	Pr	1	0	0	0
Brillia spp.	5.2	CG	1	0	0	0
Cardiocladius spp.	5.9	Pr	0	0	1	1
Chironomus spp..	9.6	CG	0	0	0	11
Conchapelopia Group	8.4	Pr	4	18	22	8
Corynoneura spp.	6	CG	0	4	2	5
Cladotanytarsus spp.	4.1	CF	0	1	0	0
Clinotanypus pinquis	8.7	CG	0	0	0	1
Cricotopus vieriensis gp (C/O sp 46)	4.4	CG	0	0	1	0
Cricotopus/Orthocladius sp gp 51	3.4	CG	0	0	1	0
Cryptotentipes spp.	6.2	CG	0	0	1	0
Cryptochironomus fulvus	6.4	Pr	1	0	0	0
Demicryptochironomus spp.	2.1	CG	0	0	0	0
Diplocladius cultriger	7.4	CG	0	1	1	9
Eukiefferiella brehmi gr (E sp 12)	2.7	CG	1	3	0	1
Eukiefferiella devonica gp (E sp 2)	2.6	CG	0	0	0	0
Eukiefferiella brevicalcar gp (E sp 6)	2.2	CG	0	1	0	0
Larsia spp.	9.3	Pr	2	10	0	1
Micropsectra spp.	1.5	CF	1	0	2	0
Microtendipes spp.	5.5	CF	4	10	9	8
Natarsia spp.	10	Pr	0	1	0	1
Orthocladius dorenus gp (C/O sp 7)	5.6		0	0	0	1
Orthocladius obumbratus sp (C/O sp 10)	8.5	CG	0	0	11	1
Orthocladius robacki: (C/O sp 12)	6.6	CG	0	0	1	1
Parakiefferiella spp.	5.4	CG	0	0	0	1
Paratendipes spp.	5.1	CG	1	0	1	1
Phaenopsectra spp.	6.5	Sc	0	0	1	1
Phaenopsectra flavipes	7.9	Sc	0	1	0	1
Polypedilum aviceps	3.7	Sh	0	4	8	2
Polypedilum illionense	9	Sh	2	1	1	0
Polypedilum fallax	6.4	Sh	0	0	0	0
Polypedilum halterale	7.3	Sh	0	0	1	0
Parakiefferiella spp.	5.4	CG	1	1	0	0
Parametriocnemus lundbecki	3.7	CG	34	42	7	3
Procladius spp.	9.1	Pr	0	0	0	1
Psectrocladius spp.	3.6	CG	0	0	0	3
Psectrotanytanpus spp.	10	CG	0	0	0	1
Rheocricotopus spp.	7.3	CG	0	0	0	0
Rheotanytarsus spp.	5.9	CF	0	3	2	0

(*continued on next page*)

Table A1 (*continued*)

Taxa	BI	FFG	HF treatment	HF reference	UF treatment	UF reference
Stempellinella spp.	4.6	CF	0	0	0	0
Symposiocladius lignicola	5.3	CG	1	0	0	0
Tanytarsus spp.	6.8	CF	1	4	1	3
Thienemaniella spp.	5.9	CG	1	2	2	0
Tribelos spp.	6.3	CG	1	0	1	1
Tvetenia bavarica gp (Esp1)	3.7	CG	4	1	0	0
Zavrelimyia spp.	9.1	Pr	2	6	3	2
Coleoptera						
Family Dryopidae						
Helichus spp.	4.6	Sh	11	9	11	13
Family Elmidae						
Optioservus spp.	2.4	Sc	0	0	1	1
Stenelmis spp.	5.1	Sc	4	5	1	0
Family Dytiscidae						
Acilius spp.	–	Pr	0	0	0	1
Agabus spp.	8.9	Pr	0	0	2	0
Hydroporus spp.	8.6	Pr	1	1	1	17
Family Hydrophilidae						
Enochrus spp..	8.8	Pr	1	0	0	2
Laccobius spp.	7.3	Pr	0	0	1	0
Family Psephenidae						
Ectopria nervosa	4.2	Sc	1	1	5	2
Psephenus herricki	2.4	Sc	0	2	14	4
Ptilodactylidae						
Anchytarsus bicolor	3.6	Sh	6	53	6	2
Odonata						
Family Aeshnidae						
Aeshna spp.	–	Pr	0	0	0	5
Boyeria grafiana	3.8	Pr	0	0	0	1
Boyeria vinosa	5.9	Pr	0	0	3	0
Family Calopterygidae						
Calopteryx spp.	7.8	Pr	4	1	8	2
Family Cordulegasteridae						
Cordulegaster spp.	5.7	Pr	23	55	2	4
Family Corduliidae						
Somatochlora spp.	9.2	Pr	0	0	1	1
Family Gomphidae						
Gomphus spp.	5.8	Pr	0	0	0	0
Lanthus spp..	1.8	Pr	0	1	0	0
Progomphus obscurus	8.2	Pr	0	0	0	1
Stylogomphus albistylus	4.7	Pr	1	0	7	1
Family Libellulidae						
Plathemis lydia	10	Pr	0	0	1	1
Oligochaeta						
Family Branchiobdellidae	6	CG	0	0	0	0
Family Enchytraeidae	9.8	CG	0	0	0	1
Family Lumbriculidae	7	CG	2	2	22	7
Family Naiidae						
Nais spp.	8.9	CG	4	2	0	0
Family Tubificidae						
immature Tubificidae		CG	1	0	0	1
Ilyodrilus templetoni	9.3	CG	1	0	1	0
Isochaetides curvisetosus	6.8	CG	0	0	0	0
Quistadrilus multisetosus	3.9	CG	1	0	0	0
Megaloptera						
Family Corydalidae						
Nigronia fasciatus	5.6	Pr	19	25	21	10
Family Sialidae						
Sialis spp.	7.2	Pr	2	2	5	10
Crustacea						
Family Asellidae						
Lirceus spp.	7.9	CG	2	0	5	5
Family Cambaridae						
Cambarus (immature crayfish)	7.5	OS	4	6	7	6
Cambarus bartoni	4.6	OS	0	1	1	0
Family Gammaridae						
Crangonyx spp.	7.9	CG	56	54	19	23
Mollusca						
Family Ancylidae						
Ferrissia spp.	6.6	Sc	0	0	1	1
Family Planorbidae						
Menetus dilatatus	8.2	Sc	0	0	0	11
Family Sphaeriidae						
Pisidium spp.	6.5	CF	0	3	0	0
OTHER TAXA						
Family Pyralidae	2	–	1	0	0	0
Family Planariidae						
Dugesia tigrina	7.2	Pr	0	0	1	0

variations include precipitation/discharge, temperature, and photoperiod (Cowx et al. 1984, Kerans et al. 1992, Bêche et al. 2006). Stream flow (i.e., discharge) increased in the treatment streams, and EPT abundance and the abundance of other sensitive species were higher at watersheds with higher stream flow. Although stream flow is connected to numerous stream physiochemical properties, as well as habitat and food availability, taxonomic richness of headwater stream communities generally increases with an increased stream flow due to the increase in habitat available (Dewson et al. 2007, Konrad et al. 2008, Elbrecht et al. 2016). Unlike findings in other clearcut harvest studies (Likens et al. 1970, McGurk and Fong 1995, Brosofske et al. 1997), temperature did not increase in our treatment streams. Increases in stream temperatures often occur after the removal of canopy cover, however, we did not detect any temperature differences between streams. Seasonal and yearly differences were similar between treatment and reference streams, indicating that clearcut harvest with the use of BMPs did not affect macroinvertebrate seasonal and yearly patterns and sensitive species stayed consistently higher in Umstead Research Farm treatment watersheds during at least 4 years postharvest.

5. Conclusions

This paired watershed study quantified the changes to macroinvertebrate communities due to forest harvest and linked these values to watershed characteristics affected by clearcut harvest. We conclude that forest harvesting with BMPs did not lead to detectable negative effects on macroinvertebrate communities. One harvested watershed had more sensitive EPT species than the unharvested watershed. Stream flow, organic carbon, and tree basal area explained most of the differences between macroinvertebrate communities in harvested and reference streams. We show that water quality is not always negatively impacted during clearcut harvesting with buffer strips and BMPs, and that water quality may even be improved in some cases. This study provides a better understanding of how macroinvertebrate communities in Piedmont streams change after harvesting and what watershed characteristics may be driving these changes. This information will provide land managers with a better understanding of the effects of clearcut harvesting on stream communities and water quality and give managers insight into the appropriate timber harvest management practices to use for protecting water resources across the region.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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Appendix

Table A1

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