EFFECTIVENESS OF FORESTRY STREAMSIDE MANAGEMENT ZONES IN THE SAND-CLAY HILLS OF MISSISSIPPI: EARLY INDICATIONS

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Abstract. During the past decade, compliance with initiatives to promote forestry best management practices (BMPs) has been monitored in most states of the southern U.S. and suggests an excellent level of acceptance throughout the region. However, effectiveness of these practices to protect water quality and aquatic habitat in streams that are potentially impacted by forest management activities has not been as thoroughly documented as the degree of compliance. The objective of this study was to determine effectiveness of streamside management zones (SMZs), a key element of BMPs designed for protection of water quality, aquatic habitat, and macroinvertebrate communities, in low-order streams within a region of north central Mississippi that is subjected to intensive forest management. Three SMZ treatments (undisturbed reference, clear-cut logging with an SMZ designated by forest managers, or clear-cut logging with no SMZ) were evaluated using a study with three replications of each treatment. Response metrics including water quality parameters, mineral soil exposure and net deposition/erosion within riparian zones, stream habitat indicators, and aquatic macroinvertebrate communities were comparable between streams receiving SMZs and undisturbed reference streams at all sampling intervals during the first year after treatment. Furthermore, significant elevation of streamwater temperature, decline in habitat stability rating, and increase in density of macroinvertebrates occurring in streams without an SMZ in comparison to reference streams provides additional evidence of SMZ effectiveness during the initial year after harvesting.

Keywords: aquatic macroinvertebrates, best management practices, forest management, logging, stream habitat, water quality

1. Introduction

Land disturbances adjacent to streams can be a major factor affecting surface water quality and aquatic habitat. It is well documented that temperature, flow, and habitat alteration can affect benthic macroinvertebrate communities (Lind, 1985; Hauer and Lamberti, 1996; Barbour *et al.*, 1999). Current questions related to macroinvertebrate studies and habitat assessment primarily revolve around their relationship to suspended sediment concentrations and substrate particle size distributions. In forestry, these issues are of concern because forest disturbances such

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as harvesting and road construction, have potential to impact these variables (e.g., increasing temperature and sedimentation) (Blackburn *et al.*, 1990).

Best Management Practices (BMPs) are aimed at preventing or reducing degradation of water quality from non-point source pollution, thereby decreasing erosion, protecting aquatic habitat, and maintaining aquatic communities. Streamside Management Zones (SMZs), a key component of BMPs, are buffers adjacent to streams designed to provide protection from disturbances on adjoining land. Thus far, studies of SMZs and the use of biotic indices for assessment of SMZ effectiveness have provided mixed results, showing both greater productivity as well as degradation of communities following perturbations (Wallace *et al.*, 1996; Wohl and Carline, 1996; Lammert and Allan, 1999; Shields *et al.*, 1995). Currently, Mississippi water quality standards, like those of many states, do not include narrative or numeric guidelines for protecting biotic communities (MS DEQ, 1995). Despite these facts, BMPs have been widely implemented throughout the southern United States, including Mississippi (Prudhomme, this issue).

Present concerns for water quality and aquatic communities, along with the possibility of future mandated regulations related to BMPs and total maximum daily loads (TMDLs) make it imperative that we gain a better understanding of relationships among land use, protective measures, aquatic communities, physical habitat, and stream hydrology. The focus of this paper is to address the effectiveness of SMZs to protect water quality, aquatic habitat and macroinvertebrate communities in low-order perennial streams within the Sand-Clay Hills Subsection of Mississippi during the first year after timber harvesting.

2. Methods

2.1. Site Description and Stream Characteristics

Nine first- or second-order streams within the Sand-Clay Hills subsection of Mississippi were selected for the study (Table I). Streams are located in Webster, Choctaw, and Calhoun Counties in North Central Mississippi (Figure 1). These counties are part of the Southeastern Mixed Forest Province, Middle Coastal Plain Section and Northern Loessial Loam Hills subregion. This region is dominated by the Clay Hills land type. Soils within the rolling to ruggedly hilly area are high in clay content, with A-horizons of either sandy loam or silt loam texture (Hodgkins *et al.*, 1979). Although the history of land management surrounding the streams is incomplete, it is evident the land was cleared for farming in the past and existing forests are the result of farm abandonment. This region of Mississippi is currently under intensive pressure from forest harvesting with approximately 5% of the land base subjected to harvesting annually (R. Daniels, Mississippi State University, personal communication).

The 30-year average precipitation within the study area is 1,119 mm/year, with mean winter (December, January, February) rainfall of 116 mm/month and summer

TABLE I

Characteristics of nine first- or second-order perennial streams in the Sand-Clay Hills subsection of Mississippi used to study SMZ effectiveness

Treatment	Stream type [a]	Watershed size (ha)	Watershed harvest unit area (ha)	SMZ area (ha)	Fraction of unit harvested (%) [b]	Mean hillslope gradient (%)	Mean channel cross-sectional area (m^2)	Mean water depth [c] (mm)
Reference1	A	98	0	N/A	N/A	51	14.0	63
Reference2	A	6,147	0	N/A	N/A	0	18.9	303
Reference3	A	76	0	N/A	N/A	4	2.0	88
SMZ1	A	43	14	8	65	5	16.3	44
SMZ2	G	101	25	13	66	15	1.6	186
SMZ3	G	45	26	9	74	6	1.4	105
No-SMZ1	A	200	14	0	100	2	6.6	53
No-SMZ2	A	42	14	0	100	3	6.1	169
No-SMZ3	A	27	3	0	100	8	6.2	68

[a] Stream type classification based on Rosgen (1994).
[b] Fraction of Unit Harvested = 1-((SMZ area/(Watershed harvest area + SMZ area)) × 100.
[c] Mean water depth within the thalweg.

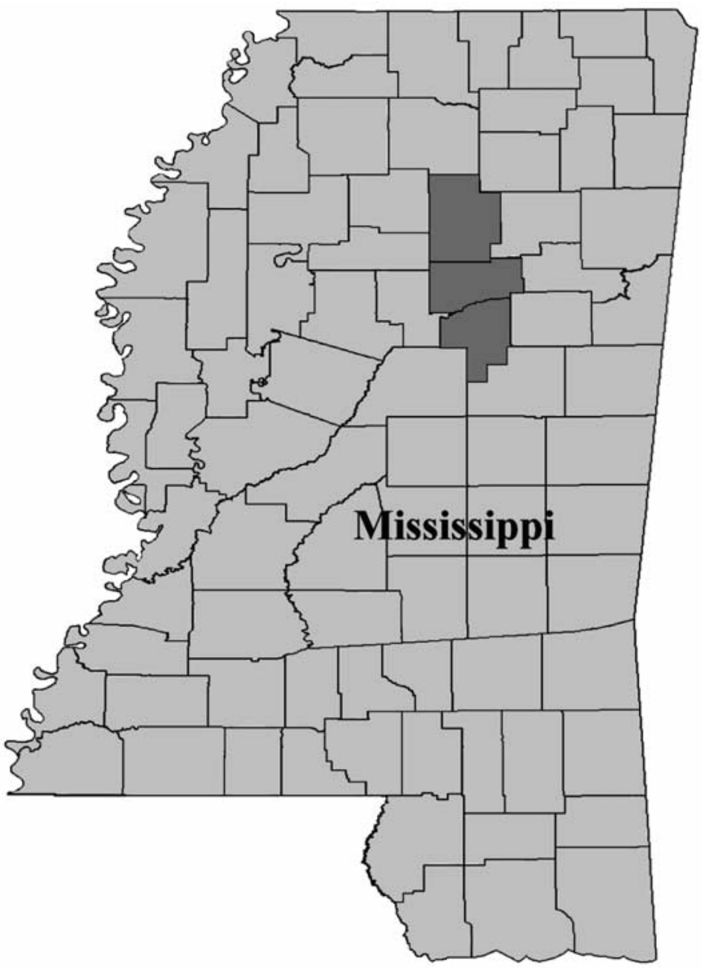

Figure 1. Three-county location of nine SMZ treatment streams in the Sand-Clay Hills subsection of Mississippi.

(June, July, August) rainfall of 87 mm/month (Table II). Mean winter temperature is 7 °C and mean summer temperature is 26 °C (Owenby and Ezell, 1992). Average monthly precipitation was determined using weather station data from each county containing a research stream. Long-term (30-year) average monthly precipitation ranged between 73 (August) and 164 mm (March) (Table II). For the duration of our study (1999–2001), lowest mean precipitation occurred in July 2000 (32 mm) after treatment and October 1999 (pre-treatment = 33 mm) and 2000 (post-treatment = 32 mm); and highest precipitation occurred in April 2000 (261 mm) after treatment. Overbank flooding was observed among all streams in April 2000 and in January 2001.

TABLE II

Monthly rainfall for the duration of SMZ study in the Sand-Clay subsection of Mississippi. Values are monthly means from weather stations within each county containing a research stream [a]

Year	Rainfall by month (mm)											
	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
1999									67	33	62	97
2000	97 [b]	55	137	261	64	116 [c]	32	38	57	32	223	97
2001	204	180	146									
Avg [d]	137	121	164	140	117	97	117	73	92	101	114	155

[a] Gauges 2896 Eupora, MS, 39 Ackerman, MS, and 1314 Calhoun City, MS are part of the cooperative network system of the U.S. National Weather Service.
[b] Installation of SMZ treatments initiated.
[c] Installation of SMZ treatments completed.
[d] Values are 30-year precipitation means for Eupora, MS.

A set of predetermined criteria was followed for stream selection to limit variation in response variables. Possible candidates were defined as small perennial streams, with at least 200 m in stream length within loblolly pine *(Pinus taeda* L.) plantations planned for future harvest on both sides of the stream. Upstream reaches were relatively free of adjacent land disturbances, such as roads, stream crossings, agricultural fields, or urbanization.

2.2. Treatments

Three treatments were used in this experiment to test the effectiveness of an SMZ for protecting water quality, aquatic habitat, and macroinvertebrate communities (Table I). Three replications of each of the three SMZ treatments were randomly assigned to nine streams. Three streams were randomly selected as reference sites, which remained unharvested and undisturbed. The remaining six streams are within plantations that were harvested. Three stands were randomly selected to have no SMZ. The No-SMZ treatment was a clear-cut logging operation with no accommodation for providing a designated riparian buffer. Basal area removal within the riparian zone approached 100% and the stand was often cut to the edge of the stream bank. Skidder traffic in the area was not regulated and no skid trails were designated before harvest. Skidders often ran adjacent to stream banks within the riparian zone but did not cross the stream channel. The remaining three streams had an SMZ designated by the forester who was managing the logging activity. Streamside management zones were set according to the contour of the land and the vegetation present, ranging from approximately 30 to 100 m in width on each side of the stream, depending primarily on site geomorphology and riparian vegetation. Boundaries of SMZs generally corresponded with borders between first terraces and the active floodplain. Harvesting of trees within SMZs was not allowed. Harvest of treatment watersheds began in late January, 2000 and was concluded by the end of June, 2000.

2.3. Sampling methodology

Streams were sampled for selected abiotic and biotic parameters prior to harvesting and again following treatment implementation. Sampling points immediately downstream of harvest boundaries within the adjacent loblolly stand were designated for water quality sampling in each stream and sampling points immediately upstream from the downstream harvest boundaries were used for measures of stream habitat indicators and macroinvertebrate communities. Sampling points for reference streams were chosen to resemble reach length and physical characteristics of treatment streams. Water quality parameters were measured bimonthly for five months prior to treatment and bimonthly for twelve months following treatment. Stream habitat indicators and aquatic macroinvertebrates were sampled monthly, four months prior to harvesting in seven streams and three months prior to harvesting in two streams. Post-harvest sampling of stream habitat indicators and macroinvertebrates was conducted in corresponding months of the year following treatment. Sites for the two streams sampled only three times were not selected when original sites were chosen and, due to the harvesting schedule, could not be sampled for four months prior to treatment. All sampling intervals were treated as subsamples to test for treatment effects in the statistical analyses. Air temperature was also measured adjacent to streams at the same time and location as biotic sampling.

2.3.1. *Water quality*

Electrical conductivity, dissolved oxygen (DO), pH, and stream water temperature were measured and recorded for each sampling location using a portable water analyzer (Autochek Model 51500) (Murphy and Wills, 1996; Plafkin *et al.*, 1989). Water samples were also collected at each sampling location to determine turbidity using a LaMott turbidity meter (Murphy and Wills, 1996).

Automatic water samplers (ISCO Model 2910 ISCO Company) were placed at each downstream location. Each sampler collected 250 ml of water every 12 hours. These samples were composited within the sampler and a representative subsample of 1 L was collected at biweekly intervals for analysis of total suspended sediment (TSS) (APHA, 1998; Keim and Schoenholtz, 1999). Grab samples were also taken during each biweekly visit, stored at 4 °C, and filtered through Acrodisc syringe filters. Grab samples were then analyzed within 48 hours of collection by ion chromatography (Ion Chromatograph Model DX-500, Dionex Corporation) to measure concentrations of dissolved nitrate, phosphate, and sulfate present in each stream (APHA, 1998).

2.3.2. *Mineral soil exposure and net erosion/deposition*

Net erosion or deposition within each riparian zone was measured using transects of erosion stakes (Brooks *et al.*, 1997) established immediately after treatments were completed. Stakes were marked at the mineral soil surface when placed into the soil. Net erosion or deposition after one year was measured by the length of exposed or buried stake, respectively, relative to the original groundline mark. Transects were spaced 40 m apart along the length of the study reach of each stream and ran perpendicular to the stream channel on both sides (Keim and Schoenholtz, 1999). Beginning at the stream bank, seven stakes were placed at 5-m intervals for a length of 30 m. Mineral soil exposure was estimated in 1-m^2 plots adjacent to each erosion stake at the time of erosion stake placement and was reassessed one year after treatment.

2.3.3. *Stream habitat indicators*

Stream channel cross sectional area and water depth within the thalweg were measured at monthly intervals for either three or four months prior to treatment and for the same months during the year following treatment. These measurements were taken at three equally spaced transects within each downstream sampling location (Peterson and Rabeni, 1995).

At each transect, a visual determination of percentage forest canopy cover, percentage bank vegetation cover, bank erosion rating, bank vegetative stability rating, and streamside cover quality rating was made for each side of the stream and averaged across both streambanks (Murphy and Wills, 1996; Plafkin *et al.*, 1989; Barbour *et al.*, 1999). Each of the three rating parameters was based on a scale of 0–10 as described by Plafkin *et al.* (1989). Low scale values indicate an unstable bank with high potential for erosion for the bank erosion rating, low percent vegetative cover of the stream bank for the bank vegetative stability rating, or poor quality cover that is unlikely to provide adequate protection from erosion for the streamside cover quality rating. A habitat stability index was calculated by summing the ratings for bank erosion, bank vegetative stability, and streamside cover quality.

Large woody debris (LWD) density was determined by dividing the number of large wood structures present by reach length (approximately 200 m) within the entire reach under study in each stream. Any structure >10 cm in diameter and >1 m in length which was in contact with the stream channel was considered LWD.

2.3.4. *Macroinvertebrates*

Macroinvertebrates were quantitatively collected using a Surber sampler from either two or three of the transects where aquatic habitat was assessed (Murphy and Wills, 1996). A 5-cm-deep sample was collected within the 30-cm square frame of the Surber sampler. Macroinvertebrates were preserved in a 10% formalin solution and returned to the laboratory for identification and enumeration.

Samples were rough-sorted by elutriation. Suspended invertebrates and detritus were poured out of a 1.0-L bucket following swirling (Payne and Miller, 1991). The rough-sorted sample was placed in a petri dish and divided into equivalent sections 1/16 in size. A 1/16 subsample was sorted until either 100 organisms were counted or a four-hour time limit was reached (Payne and Miller, 1991). Following rough-sorting, macroinvertebrates were identified at least to family using dichotomous keys (McCafferty, 1981; Merritt and Cummins, 1996) and stored in 70% ethanol. Shannon Weaver Diversity Index, Family Richness and number of individuals within pollution-sensitive Ephemeroptera, Plecoptera, and Trichoptera taxa (EPT Count) (Barbour *et al.*, 1999) were calculated for each sampling location to determine if community structure changed in response to treatment. Resolution of analysis was at the family level (Hilsenhoff, 1988).

2.4. Experimental design

A completely randomized experimental design was used with three replications of three SMZ treatments. Individual streams served as experimental units, either for each sampling interval or for entire pre-treatment or post-treatment periods. One-way analysis of variance was used to determine the presence of significant treatment effects at $\alpha = 0.10$ level. Observations of significance were made at $\alpha = 0.10$ because of high natural variability among the streams and in the interest of reducing the probability of a Type II error. Duncan's Multiple Range Test was used to separate means if treatment effects were significant.

3. Results

3.1. Water quality

No detectable differences were found for mean pH, DO, electrical conductivity, turbidity, TSS, air temperature, dissolved sulfate, dissolved nitrate, or dissolved phosphate either during pre-treatment sampling or post-treatment periods among the three treatments at downstream locations (Table III). Phosphate was not detectable in any of the samples taken during this study. Pre-harvest mean nitrate levels were not significantly different among treatment designations, although levels tended to be higher in streams scheduled for No-SMZ treatment. Mean nitrate levels increased slightly at downstream sampling sites for all treatments in the post-harvest period but showed no significant effects of treatment (Table III).

Mean streamwater temperature was the only physicochemical parameter to show significant differences following treatment (Table III). Pre-harvest measurements of mean streamwater temperature revealed no significant differences among streams designated for the three SMZ treatments. Once treatments were established, streams receiving No-SMZ treatment had significantly higher average downstream temperatures when compared to the two other treatments (Table III).

TABLE III

Effect of SMZ treatments on physicochemical parameters downstream locations in first- and second-order streams in the Sand-Clay Hills subsection of Mississippi[a]

SMZ treatment[b]	pH	DO (mg L^{-1})	Conductivity (μS cm^{-1})	Turbidity (NTU)	TSS (mg L^{-1})	Air temperature (°C)	Water temperature (°C)	Sulfate[g] (mg L^{-1})	Nitrate (mg L^{-1})
Pre-treatment[c]									
REF	7.58a (0.17)[f]	6.45a (0.49)	83.4a (33.8)	8.04a (2.32)	11.60a (2.01)	13.2a (3.0)	13.9a (1.5)	19.14a (16.20)	0.04a (0.01)
SMZ	7.88a (0.01)	6.03a (0.92)	115.9a (85.6)	7.14a (3.55)	11.77a (2.15)	13.2a (1.2)	14.1a (0.5)	12.19a (11.00)	0.06a (0.05)
No-SMZ	7.14a (0.50)	7.17a (0.53)	45.8a (22.9)	7.51a (2.49)	6.57a (4.06)	10.9a (2.0)	12.6a (1.6)	4.60a (3.24)	0.33a (0.28)
Post-treatment[d]									
REF	6.06a (0.21)	7.42a (0.28)	84.8a (35.1)	14.03a (4.78)	55.23a (32.79)	11.7a (3.4)	11.8b (1.5)	19.77a (14.31)	0.09a (0.06)
SMZ	6.42a (0.35)	6.91a (0.54)	126.5a (94.4)	9.95a (2.16)	29.21a (19.46)	9.3a (3.3)	12.1b (2.0)	13.30a (9.37)	0.07a (0.05)
No-SMZ	6.42a (0.39)	7.68a (0.68)	64.0a (38.5)	12.86a (3.43)	34.72a (18.59)	11.3a (4.7)	16.5a (0.7)	7.39a (4.63)	0.38a (0.19)
Change[e]									
REF	–1.52a (0.28)	0.97a (0.32)	1.5a (1.3)	6.00a (2.50)	43.63a (34.38)	–1.5a (6.0)	–2.1b (1.5)	0.62a (1.93)	0.05a (0.06)
SMZ	–1.46a (0.35)	0.88a (0.84)	10.5a (8.9)	2.82a (3.95)	17.45a (21.13)	–3.9a (4.4)	–2.0b (2.1)	1.11a (2.30)	0.01a (0.07)
No-SMZ	–0.72a (0.11)	0.54a (0.36)	18.2a (15.7)	5.35a (4.35)	28.16a (22.32)	0.4a (3.8)	3.9a (1.6)	2.79a (1.39)	0.05a (0.24)

[a] Values represent means of three replications. Values in parentheses are standard errors.
[b] REF = unharvested reference; SMZ = clear-cut harvesting with 30–100 m streamside management zone on each side of stream; No-SMZ = clear-cut harvesting without streamside management zone.
[c] Pre-treatment values are for September 1999 through January 2000.
[d] Post-treatment values are for September 2000 through January 2001.
[e] Change = Post-treatment value – Pre-treatment value.
[f] Within a column and subheading, means followed by the same letter are not significantly different at the $\alpha = 0.10$ level, according to Duncan's Multiple Range Test.
[g] Values reported as SO_4^{2-} and NO_3^-.

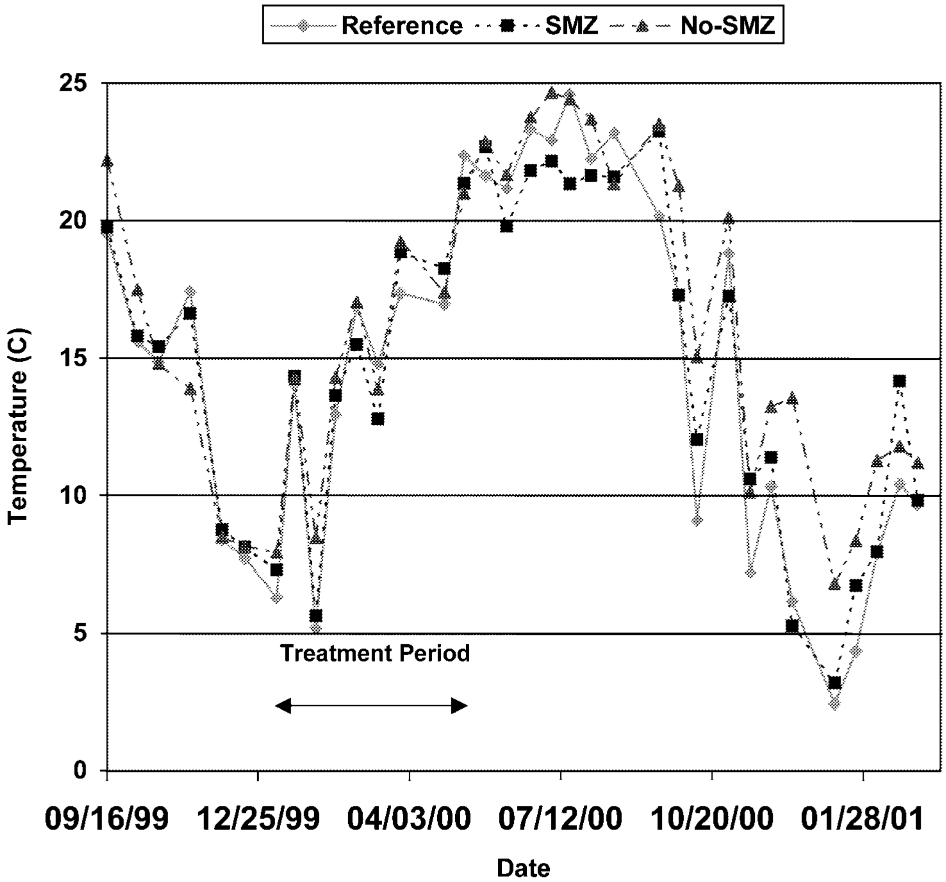

Figure 2. Mean streamwater temperature for each sampling date at locations downstream from SMZ treatments in first- or second-order streams of the Sand-Clay Hills subsection in Mississippi. Each value is mean of three streams.

Both the SMZ and Reference streams had average decreases of about 2 °C following harvest, whereas the No-SMZ treatment had an average increase of almost 4 °C. Following harvesting, the No-SMZ treatment had significantly higher temperature at 72% of the sampling dates (Figure 2).

3.2. Mineral soil exposure and net erosion/deposition

Immediately following treatment, mean mineral soil exposure was highest (47 and 45%) on the streambank at the 0-m position in the Reference and SMZ treatments, respectively, and lowest (10%) at this near-stream position in the No-SMZ treatment (Figure 3). At a distance of 5 m from the streambank, the SMZ treatment had significantly higher mean soil exposure than the Reference and No-SMZ treatments immediately following harvest (Figure 3).

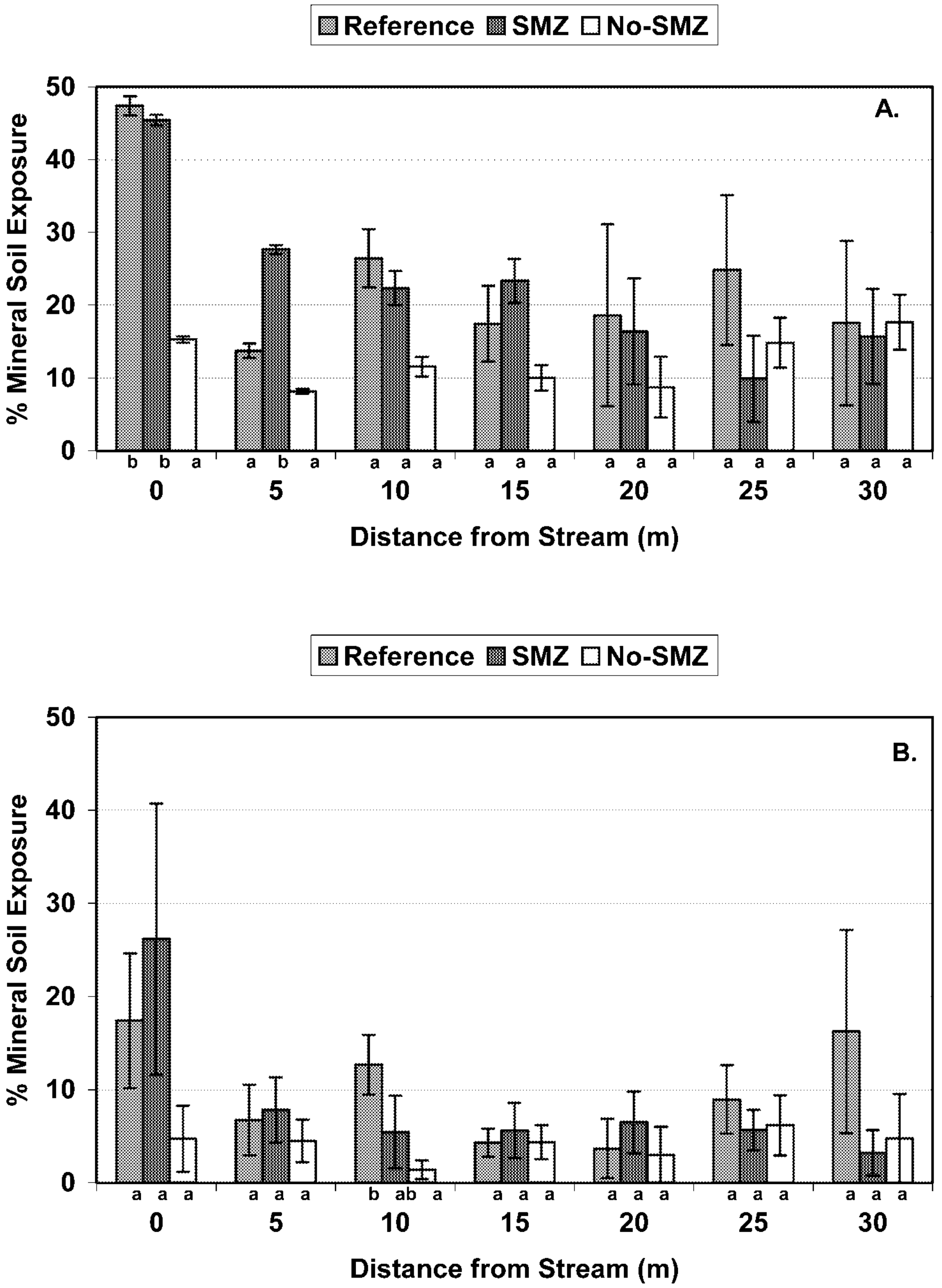

Figure 3. Comparison of percent mineral soil exposure among SMZ treatments for each riparian position (A) immediately following and (B) one year after logging in the Sand-Clay Hills subsection of Mississippi. Groupings are by distance from stream for three treatments. Values are means (+/-standard error) of three replications. Letters indicate statistically similar values within each group at the $\alpha = 0.10$ level, according to Duncan's Multiple Range Test.

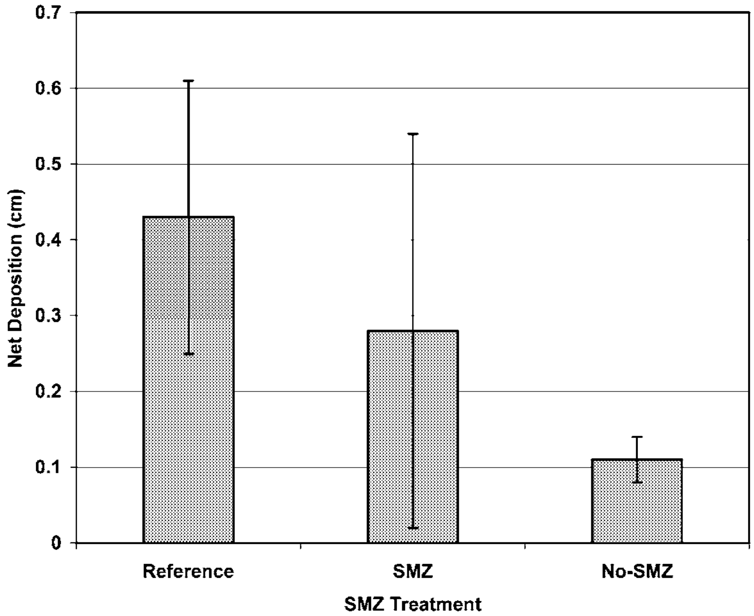

Figure 4. Effect of SMZ treatments on net deposition of sediment within the riparian zone one year after treatment establishment in the Sand-Clay Hills subsection of Mississippi. Values are means (+/-standard error) of three replications pooled across distances from the stream. There is no significant difference among treatments at the $\alpha = 0.10$ level, according to Duncan's Multiple Range Test.

Net erosion/deposition within riparian zones was not significantly affected by SMZ treatment (Figure 4) or position within the riparian zone (Figure 5) one year after treatment. Net deposition occurred at all positions among all three treatments, ranging from a mean of 0.12 cm in the No-SMZ treatment to 0.40 cm in the Reference treatment (Figure 4) and from 0.10 cm at 30 m from the stream bank to 0.76 cm on the streambank (Figure 5).

3.3. Stream habitat indicators

Forest canopy cover above the stream channel was comparable (~50%) among streams prior to treatment (Table IV). However, streams subjected to the No-SMZ treatment had a 42% decline of forest canopy cover from pre-treatment levels, resulting in significantly less canopy cover in comparison with Reference and SMZ forest canopy cover after the treatment period (Table IV).

Prior to treatment, percent vegetative cover present on stream banks was not significantly different among treatment designations, ranging from 32 to 50% (Table IV). Following treatment, No-SMZ streams had significantly more vegetative cover present (59%) on stream banks than Reference or SMZ streams with 43 and 30%, respectively. Cover on stream banks with No-SMZ treatment was

TABLE IV

Effect of SMZ treatments on aquatic habitat metrics in first- or second-order streams of the Sand-Clay Hills subsection in Mississippi [a]

SMZ treatment	Forest canopy cover (%)	Bank vegetation cover (%)	Bank erosion rating	Bank vegetative stability rating (scale) [b]	Streamside cover quality rating	Habitat stability index (scale) [c]	LWD density (count m^{-1})	Channel cross-sectional area (m^2)
Pre-treatment [d]								
REF	49a [g] (1.6)	35a (7.5)	3.96b (0.62)	3.78b (0.52)	4.57b (0.88)	12.31c (0.31)	0.07a (0.43)	12.12a (5.2)
SMZ	44a (5.0)	32a (3.3)	5.38ab (0.66)	4.74b (0.39)	5.53ab (0.34)	15.64b (0.82)	0.06a (1.00)	6.6a (5.1)
No-SMZ	52a (0.5)	50a (8.6)	6.25a (0.17)	5.90a (0.27)	6.50a (0.59)	18.65a (0.84)	0.04a (0.23)	6.8a (0.3)
Post-treatment [e]								
REF	70a (6.6)	43b (3.1)	3.81b (0.48)	4.72a (0.64)	6.04b (0.11)	16.52a (0.45)	0.13a (1.51)	11.0a (4.8)
SMZ	57a (8.8)	30b (6.2)	5.04ab (0.97)	4.99a (0.47)	7.88a (0.11)	17.15a (0.85)	0.09a (1.00)	6.4a (4.8)
No-SMZ	10b (3.7)	59a (7.7)	5.86a (0.55)	5.42a (0.39)	4.45c (0.46)	15.10a (0.79)	0.07a (0.28)	5.8a (0.1)
Change [f]								
REF	21b (7.6)	8a (4.7)	–0.15a (0.31)	0.94a (0.25)	1.47b (0.78)	4.21a (0.76)	0.06a (1.09)	–1.2a (0.4)
SMZ	13b (13.7)	–2a (4.7)	–0.34a (0.35)	0.25a (0.60)	2.35b (0.35)	1.51b (1.09)	0.03a (0.19)	–0.2a (0.3)
No-SMZ	–42a (4.0)	9a (8.5)	–0.39a (0.39)	–0.48a (0.52)	–2.05a (0.14)	–3.55c (0.60)	0.03a (0.20)	–1.0a (0.4)

[a] Values represent means of three streams. Values in parentheses are standard errors.
[b] Bank erosion scale: 0 (unstable) – 10 (stable). Bank vegetative stability scale: 0 (< 25% coverage) – 10 (> 80% coverage). Streamside cover quality scale: 0 (> 50% of the streamside bank has no vegetation) – 10 (dominant vegetation is shrub).
[c] Habitat stability index = Bank erosion rating + Bank vegetative stability rating + Streamside cover quality rating. Scale: 0 (poorest) – 30 (best).
[d] Survey conducted prior to harvest (Sample months: Sept–Dec 1999 and Jan 2000).
[e] Survey conducted after harvest (Sampling months: Sept–Dec 2000 and Jan 2001).
[f] Change = Post-treatment number – Pre-treatment number.
[g] Within a column and subheading, means followed by the same letter are not significantly different at the $\alpha = 0.10$ level, according to Duncan's Multiple Range Test.

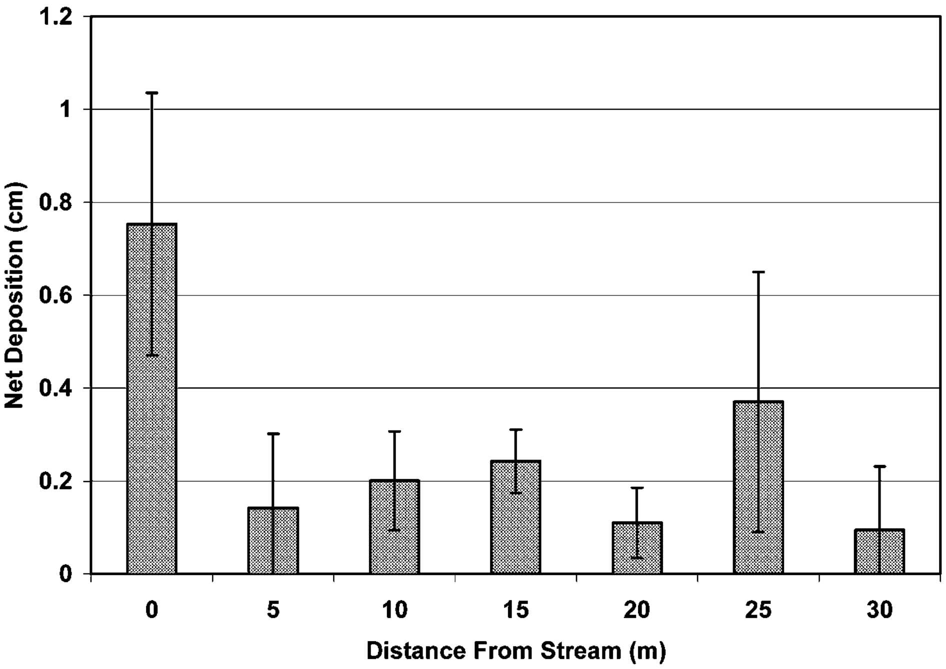

Figure 5. Effect of distance from stream on net deposition of sediment one year after establishment of SMZ treatments in the Sand-Clay Hills subsection of Mississippi. Values are means (+/-standard error) of nine replications pooled among SMZ treatments. There is no significant difference among distances at the $\alpha = 0.10$ level, according to Duncan's Multiple Range Test.

primarily provided by grasses and forbes, whereas streambank cover with SMZ and Reference treatments was primarily provided by shrubs and trees.

Bank erosion ratings showed significantly greater stability (higher scale values) in No-SMZ streams prior to treatment (6.25) and following treatment (5.86) than in Reference streams, which had erosion ratings of 3.96 and 3.81 for pre-treatment and post-treatment sampling, respectively (Table IV). Streams receiving SMZ treatment had intermediate bank erosion ratings of 5.38 and 5.04 before and after treatment, respectively (Table IV). All streams had increased likelihood of bank erosion (lower scale values) after treatment and did not demonstrate significant treatment effects (Table IV).

No-SMZ streams had significantly greater mean bank vegetative stability rating (5.90) than streams designated for SMZ or Reference treatments (4.74 and 3.78, respectively) prior to treatment (Table IV). However, post-treatment mean bank vegetative stability ratings were not different among treatments.

No-SMZ streams had the highest mean streamside cover quality rating (6.50) prior to treatment, which was significantly greater than the Reference mean (4.57) (Table IV). Post-treatment streamside cover quality was lowest in No-SMZ streams with a mean rating of 4.45 and highest in SMZ streams with a mean rating of

7.88. No-SMZ streams were the only treatment in which streamside cover rating decreased following harvesting (Table IV).

Summing bank erosion, bank vegetative stability, and streamside cover quality ratings into a habitat stability index showed that, prior to harvesting, No-SMZ streams had the highest mean habitat stability value of 18.65, compared to 15.64 for SMZ streams and 12.31 for Reference streams (Table IV). A significant decline in overall habitat stability between pre- and post-treatment sampling periods was observed in streams receiving the No-SMZ treatment (Table IV). In contrast, mean habitat stability ratings for Reference and SMZ streams increased after treatment.

There was no significant difference in mean density of LWD among stream groupings for pre- or post-treatment periods, with mean counts ranging from 0.04 pieces m^{-1} in No-SMZ streams before treatment to 0.13 pieces m^{-1} in Reference streams after treatment (Table IV).

Mean stream channel cross-sectional area was not significantly different prior to or following treatment implementation (Table IV). Mean net deposition occurred in all streams following harvesting, as evidenced by decreased cross-sectional area (Table IV).

3.4. Macroinvertebrates

Macroinvertebrate samples from pre- and post-treatment periods were dominated by the following taxa: Diptera (Chironomidae and Ceratopogonidae), Crustacea, Annelida, and Nematoda. In all, 19 orders and 53 families were collected prior to treatment, whereas 20 orders and 63 families were collected following treatment.

Mean number of macroinvertebrates collected at downstream locations during the pre-treatment period was not significantly different among treatment groupings, with counts ranging from 378,444 to 469,983 individuals m^{-3} (Table V). Significant differences occurred during the first year after treatment, with macroinvertebrate counts increasing by five-percent to 493,715 individuals m^{-3} in No-SMZ streams, whereas counts decreased by approximately 50% to 192,401 individuals m^{-3} in Reference and 167,185 individuals m^{-3} in SMZ streams (Table V). However, changes in macroinvertebrate counts between pre- and post-treatment showed no significant difference among treatments (Table V).

Mean taxa richness of macroinvertebrates was not significantly different among treatment groupings prior to or after treatment, with number of taxa ranging from 9.39 to 11.83 before treatment and from 10.06 to 13.40 during corresponding months after treatment (Table V). Reference and No-SMZ streams gained an average of 4.01 and 0.31 macroinvertebrate taxa, respectively, after treatment, whereas SMZ streams lost an average of 0.99 macroinvertebrate taxa after treatment.

Comparison of mean Shannon Weaver Diversity Index (H') values prior to and after treatment revealed no significant differences (Table V). Index values increased slightly in all treatment groupings following treatment, with mean increases of

TABLE V

Effect of SMZ treatments on aquatic macroinvertebrates in first- and second-order streams within the Sand-Clay Hills subsection of Mississippi [a]

SMZ treatment	Macroinvertebrate count (count m^{-3})	Taxa richness (count)	Shannon Weaver Diversity Index [H′] [b]	EPT count (count m^{-3})
Pre-treatment [c]				
REF	427,816a [f] (104,676)	9.39a (0.81)	1.34a [f] (0.16)	156a (19.20)
SMZ	378,444a (84,970)	11.83a (0.45)	1.63a (0.10)	507a (481.68)
No-SMZ	469,983a (79,249)	9.75a (1.38)	1.54a (0.11)	436a (230.20)
Post-treatment [d]				
REF	192,401b (37,294)	13.40a (1.29)	1.81a (0.12)	108b (23.23)
SMZ	167,185b (43,862)	10.85a (1.04)	1.74a (0.27)	735a (191.19)
No-SMZ	493,715a (177,992)	10.06a (1.11)	1.63a (0.15)	258b (151.47)
Change [e]				
REF	–235,415a (141,546)	4.01a (0.64)	0.47a (0.02)	–48b (129.09)
SMZ	–211,259a (128,620)	–0.99a (1.36)	0.11a (0.16)	228a (470.96)
No-SMZ	23,732a (193,036)	0.31a (0.31)	0.09a (0.17)	–178b (197.93)

[a] Values represent means of three streams. Values in parentheses are standard errors.
[b] $H' = -\sum[(n_i/n)\ln(n_i/n)]$ where n_i = number of individuals within a taxon and n = total number of individuals among all taxa.
[c] Survey conducted prior to harvest (Sample months: Sept–Dec 1999 and Jan 2000).
[d] Survey conducted after harvest (Sampling months: Sept–Dec 2000 and Jan 2001).
[e] Change = Post-treatment number – Pre-treatment number.
[f] Within a column and subheading, means followed by the same letter are not significantly different at the $\alpha = 0.10$ level, according to Duncan's Multiple Range Test.

diversity ranging from $H' = 0.09$ in No-SMZ streams to $H' = 0.47$ in Reference streams.

Prior to harvesting, there were no significant differences in mean EPT count among streams designated for treatments, with counts ranging from 156 in Reference streams to 507 individuals m^{-3} in streams designated for SMZ treatment (Table V). After treatment, mean EPT count m^{-3} in SMZ streams increased by 228, whereas Reference and SMZ streams decreased by 48 m^{-3} and 178 m^{-3}, respectively (Table V). The most dominant pollution-sensitive order present among pre-treatment and post-treatment samples was Ephemeroptera, which consisted of five different families, but was predominantly made up of *Caenis* spp. in the family Caeniidae.

4. Discussion

4.1. Water quality

Few SMZ treatment effects on measured water quality parameters were observed in this study, with the exception of streamwater temperature. The significant increase in streamwater temperature observed without protection of SMZs is not surprising, and can be attributed to decrease in canopy cover and the small size of streams in this study. Because the streams had small channel cross-sectional areas and were relatively shallow during summer baseflow conditions (mean water depth in thalweg <303 mm in all cases), increases in solar radiation reaching No-SMZ streams as a result of canopy removal in the vicinity of stream banks had a detectable impact on streamwater temperature. Besctha *et al.* (1987) observed this trend in the Pacific Northwest. They noted streamwater temperature increases of 2 to 10 °C after removal of riparian vegetation. This observation is also supported by the findings of Brown and Krygier (1970), who showed that for periods when streamflow is normally low and air temperature is high, loss of riparian vegetation results in larger diurnal temperature variations and elevated monthly and annual temperatures.

Studies of timber harvesting effects on water quality have reported an increase in TSS and/or turbidity, attributable largely to sediment delivery from logging roads (Binkley and Brown, 1993) and channel aggradation (e.g., Keim and Schoenholtz, 1999). Thus, it was somewhat surprising that during the first year after harvesting, treatment effects on these parameters were not detected in our study. This lack of measurable treatment effect on TSS and turbidity may be, in part, a function of relatively high variability among experimental units (Table III). It may also be attributable to an absence of large-scale storm events with long return intervals during the post-treatment study period, as these relatively infrequent events often contribute a major portion of sediment export in low-order streams (e.g., Beschta, 1978; Grant and Wolff, 1991). Our observations suggest that the inherent instability and variability of hydrologic processes such as sediment export in these incised first- and second-order streams of the Sand-Clay Hills likely had more influence on TSS and turbidity levels than any treatment effects to date.

4.2. Mineral soil exposure and net deposition within riparian zones

Relatively high levels of mineral soil exposure along the stream banks of Reference and SMZ streams are likely a function of overbank scouring by floodwater and removal of forest floor litter accumulations, rather than a direct effect of treatments. Flooding was observed at all streams in April 2000 and January 2001. In contrast, relatively limited exposure of mineral soil along the stream banks in No-SMZ streams is likely the result of mineral soil surface coverage by logging slash that remained after removal of timber in the riparian zone. A similar pattern of mineral soil surface disturbance was also observed by Keim and Schoenholtz (1999) in a study of logging impacts to low-order streams in the steep loessial bluff region of Mississippi.

4.3. Stream Habitat Indicators

Responses of stream habitat indicators, including bank vegetation cover, as well as ratings for bank erosion, bank vegetative stability, streamside cover quality, and habitat stability index demonstrate their applicability as measures of SMZ effectiveness during the first year after treatment in these streams. In all cases, the similarity between Reference and SMZ streams for these metrics indicates that the SMZ treatment was effective in maintaining streambank conditions when compared with Reference streams. In contrast, streams without SMZs had significant alterations of streambank conditions in comparison to undisturbed Reference streams. Significant reductions in percent forest canopy cover, streamside cover quality rating, and habitat stability index occurred in No-SMZ streams when compared to Reference and SMZ streams.

Ocular estimates of forest canopy cover showed an expected decline in cover above No-SMZ streams and can be attributed to timber harvesting in the vicinity of stream banks. However, numerical increases in mean forest canopy cover from 49% to 70% in Reference streams and from 44% to 57% in SMZ streams between pre- and post-treatment, respectively, demonstrate the degree of inaccuracy when using a relatively subjective measure of forest canopy cover. These two treatments were not subjected to harvesting in the riparian zone and were sampled at the same location and time of year for pre- and post-treatment assessments. Therefore forest canopy cover was not expected to vary to this degree between pre- and post-treatment sampling. Use of densiometers or photographs from cameras with fish-eye lenses offers more accurate methods than ocular estimates for quantifying forest canopy cover in low-order forest streams.

The decline in streamside cover quality rating and habitat stability index of No-SMZ treatment streams can be attributed to elimination of streamside trees and woody shrubs, which resulted from clear-cut harvesting within the riparian zone. When harvesting occurs within the riparian zone, deep-rooted trees and shrubs that add stability to the streambank are often eliminated and the integrity of the streambank may be at risk as a result. A potential outcome of removing woody vegetation from streambanks was demonstrated in a study by Hartman *et al.* (1987), in which bank erosion did not occur until two years after cutting of trees along the streambank. The authors attributed the delay in bank erosion to the time necessary for tree roots responsible for bank stability to begin to decay and LWD to begin to decrease. Furthermore, riparian vegetation dominated by shrubs and trees (high streamside cover quality rating) provides critical cover for stream shading and aquatic communities as well as a source of allochthonous food chain support, particularly in low-order streams, where the role of riparian vegetation in stream functions is at a maximum (Plafkin *et al.*, 1989; Barbour *et al.*, 1999).

We expected an initial increase in LWD density within channels of No-SMZ streams as a result of treetops and debris left after logging. However, amount of LWD increased comparably (almost doubling) among all treatments, with Reference streams having slightly higher mean LWD occurrence during both pre- and post-treatment periods. Initial lack of significant treatment effect on LWD density is supported by research of timber harvesting of old-growth forests in western Washington, where LWD density was initially unaffected by timber harvesting (Ralph *et al.*, 1994). Equal density of LWD after treatment does not necessarily translate to equal quality for stream habitat. Ralph *et al.* (1994) observed a significant shift in location of LWD from instream to channel margins. Baillie *et al.* (1999) reported an average increase of 3-fold in LWD from pre-harvest counts in *Pinus radiata* (D. Don) and *Pinus nigra* (Arnold) stands in New Zealand, but found LWD had smaller diameter and shorter length than LWD prior to harvesting. Effects of the No-SMZ treatment on quantity and quality of LWD in our study are likely to become more evident over time because harvesting of trees within the riparian zone has eliminated the primary source of LWD for streams subjected to this treatment.

4.4. Macroinvertebrates

Channel characteristics, substrate composition, and instream cover are primary components shaping aquatic community structure. Two of these components may account for the increased numbers of macroinvertebrates observed after treatment in No-SMZ streams. First, less debris within the channel and decreased riparian vegetation may allow stream flow to flush fine particulates downstream and scour channel banks as observed by Newbold *et al.* (1980). This scenario is not likely given our initial post-treatment trends for maintaining LWD counts and aggradation of stream channel cross-sectional area in No-SMZ streams. Second, increased sunlight and temperature from canopy removal often increases detritus processing and algal production (Newbold *et al.*, 1980). Decreased instream cover in No-SMZ streams may explain the observed increase in macroinvertebrate density we observed. Small streams are greatly influenced by the surrounding forests (Maser *et al.*, 1988). In general, low-order forested streams receive much of their energy supply from allochthonous inputs of leaf litter and organic debris originating from adjacent riparian areas (Maser *et al.*, 1988; Gregory *et al.*, 1991). Minshall (1978) suggested that allochthonous and autochthonous food supplies probably function in the same way, with a detritus-based food economy, and can therefore support equal productivity. As the instream canopy opens, food base production may shift from an allochthonous to autochthonous supply (Minshall, 1978; Gregory *et al.*, 1991). Hence, increased light can increase the abundance of invertebrates (Gregory *et al.*, 1991). These results are similar to those of Wallace *et al.* (1996), who observed increased macroinvertebrate numbers following harvesting in small streams of North Carolina.

Dominant taxa in all treatments were Diptera (Chironomidae and Ceratopogonidae), Crustacea, Annelida, and Nematoda. Values for taxa richness did not reveal significant trends among treatments. However, changes in riparian forest structure that alter shade, stream temperature, and timing, quantity, and quality of allochthonous inputs resulting from our manipulations within the riparian zone suggest that treatment effects on taxa richness may be likely in the future.

Although Shannon Weaver Diversity [H'] values were comparable among treatment groupings prior to and after treatment, changes in EPT density between pre- and post-treatment showed significant increases in SMZ streams and a tendency to decline in No-SMZ and Reference streams. This suggests that the SMZ streams are maintaining or enhancing the density of sensitive macroinvertebrate taxa, but it is difficult to explain why this would occur at the same time that EPT counts declined in undisturbed Reference streams. It is noteworthy that EPT density in No-SMZ streams was dominated by the genus *Caenis*. This genus is one of the more tolerant EPT genera because their opercular gills provide an adaptation to survive in habitats with high percentages of fine sediment (Brigham *et al.*, 1982). Trends in our study are supported to varying degrees by observations of aquatic macroinvertebrate communities reported by Newbold *et al.* (1980) and Wohl and Carline (1996). In a study of 50 streams in northern California, Newbold *et al.* (1980) concluded that streams within logged areas had greater densities of macroinvertebrates, but decreased taxa richness and diversity after logging compared to unlogged streams and streams with wide buffer strips. Wohl and Carline (1996) investigated three streams within the Ridge and Valley physiographic province of central Pennsylvania and found streams with riparian grazing to have significantly fewer numbers of macroinvertebrates, but no differences in median numbers of taxa.

5. Conclusions

Early indications from this study support the use of SMZs as a component of forestry BMPs in order to maintain streamwater temperature, stream habitat, and macroinvertebrate density in low-order streams of north central Mississippi. Response metrics including water quality, mineral soil exposure and net deposition/erosion within the riparian zone, stream habitat indicators, and aquatic macroinvertebrate density, taxa richness, and diversity were comparable between streams receiving SMZs and undisturbed reference streams at all sampling intervals during the first year after treatment. In contrast, streamwater temperature, selected stream habitat indicators, and macroinvertebrate density showed significant alterations from reference conditions in the absence of an SMZ, providing further evidence of SMZ effectiveness.

Although these early indications of SMZ effectiveness are encouraging, it is important to consider that potential stream alterations linked to changes in water quality, LWD, sediment dynamics, stream habitat indicators, and aquatic communities have some degree of likelihood in the future. Responses to alterations in riparian structure and function may take longer than one year after treatment to be detectable considering the high degree of natural variation inherent in these low-order stream systems.

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