AN ABSTRACT OF THE THESIS OF

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Title: Aquatic Invertebrate-Habitat Relationships and Stream Channel Cross Section Area Change in Response to Streamside Management Zones in North Central Mississippi.

Abstract

Approved:____

Stephen H. Schoenholtz

The southern states lead the country in timber production and are subject to Best Management Practices (BMPs) designed to mediate the effects of forest harvesting on water quality. Small headwater streams on timberland in North Central Mississippi have received little attention with respect to effectiveness of the BMPs designed to protect them. With stand rotations of 25-30 years and with some counties held in nearly 30% private industrial forestry ownership, impacts of intensive forest management on water resources have the potential to be large. In North Central Mississippi, I evaluated the effectiveness of Streamside Management Zones (SMZs), corridors of the riparian zone along the stream left unharvested, as a component of BMPs. The streams sampled were low-order perennial headwaters either within a clearcut with no SMZ, a clearcut with an SMZ, or a site that had not been harvested. Aquatic invertebrate community composition, habitat and substrate composition, and stream channel cross-section areas were evaluated in 2002 and again in 2003. My objectives were 1) to describe the invertebrates and habitat conditions at each site, 2) to determine if there are relationships between invertebrates and water quality, cover metrics, or relative amounts of substrate size fractions, and 3) to determine whether or not there were detectable treatment effects on either substrate composition or stream channel cross-section areas. Water quality, cover, and substrate size classes were highly variable within each harvest treatment. No significant differences were found downstream of harvest treatments for relative distribution of substrate within each harvest treatment group. Even though the habitat and substrate were highly variable, the invertebrate composition at many sites was dominated by a single family, Chironomidae. Ordinations of the presence of invertebrate taxa using non-metric multidimensional scaling showed a separation of the Reference and No-SMZ treatment groups for 2003, indicating that treatment effects may only be expressed in the biota and not in physical stream habitat characteristics measured by this study three to five years after harvesting. Changes in stream channel cross-section area occurred for all treatments, with Reference treatments degrading and SMZ and No-SMZ treatments aggrading between 24 and 29 months after harvesting treatments were established. Overall, No-SMZ treatments did not have significant changes in stream channel cross-section between sampling intervals, while Reference and SMZ treated streams changed significantly throughout the study. High natural variability of the streams made it difficult to discern differences in stream habitat parameters for each harvest treatment. Differences between Reference and No-SMZ sites in the 2003 ordinations and between No-SMZ and the other treatments for stream channel cross-sections indicated that SMZs are functioning in these low-order streams.

Aquatic Invertebrate-Habitat Relationships and Stream Channel Cross Section Area Change in Response to Streamside Management Zones in North Central Mississippi.

By:

Terry A. Luecker

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APPROVED:

Major Professor, representing Forest Engineering

Head of the Department of Forest Engineering

Dean of Graduate School

I understand that my thesis will become part of the permanent collection of Oregon State University libraries. My signature below authorizes release of my thesis to any reader upon request.

Terry A. Luecker, Author

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1. <u>Introduction</u>

Forest practices have changed in response to public pressures and state regulations. Regulation compliance is mandatory in some states, but even where not mandatory, many forestry companies and private non-industrial landowners are following suggested best management practices (BMPs) (Carroll et al. 2004, Prud'homme and Greis 2002, Kilgore et al. 2004). Leaving trees unharvested in riparian areas to form streamside management zones (SMZs) is a component of BMPs used to fulfill several management objectives including protection of stream biota, maintenance of physical stream integrity, and reduced sediment movement into the stream. With nearly total compliance to BMPs (Prud'homme and Greis 2002), are SMZs effectively mitigating effects of harvesting on streams?

Sediment in the stream and suspended sediment in the water occur naturally in streams, but excess levels of sediment resulting from anthropogenic influences have the potential to negatively affect stream biotic communities including macrophytes, fishes, and aquatic invertebrates (Waters 1995). Mainly due to anthropogenic disturbance in watersheds, stream sedimentation is listed as one of the major pollutants of surface waters in the U.S. (USEPA 2000).

Use of biotic indicators, particularly macroinvertebrates, to assess water quality and biological condition shows promise, but can be time consuming, expensive, and is not as well documented for warmwater southern streams as for other areas of the U.S. There is a need to explore these communities in disturbed and undisturbed settings, determine their value as indicators of water quality and habitat quality, and explore their relationships to other components of stream ecosystems that may be sensitive to disturbance.

Prud'homme and Greis (2002) surveyed 13 Southern states and found 12 of 13 had implementation of BMPs ranging from 63% to 96%. This documentation of BMP compliance is encouraging but does not address effectiveness of these practices in protecting water quality, stream habitat, and aquatic biota.

My research on stream insects and their associated habitat is part of an overall study of SMZ effectiveness in which water chemistry, sediment movement in the riparian zone, stream insect and fish community characteristics, and physical stream characteristics were quantified in response to three SMZ treatments (Young 2002, Carroll 2002, Clouse 2003). Land adjacent to fifteen streams was subject to either clearcut harvest with no SMZ (No-SMZ), clearcut harvest with at least a 30-m SMZ on both sides of the stream (SMZ), or no harvest serving as reference streams (REF). My specific areas of study within this project were to 1) describe streambed substrate and invertebrate taxa for three treatment groups, 2) discern SMZ treatment effects on streambed substrate and stream channel cross sectional areas, and 3) explore relationships between invertebrate taxa collected in streambed cores and measured habitat variables.

1.1. Effects of forest harvest

Many studies attempting to identify sediment sources from timber harvest activities have been conducted in regions with steeply sloping topography (e.g. Alsea (OR), Coweeta (NC), H.J. Andrews (OR), Hubbard Brook (NH) (Waters 1995)). Lewis (1988) identified harvesting activities in the Caspar Creek Watershed in Northern California which led to increased suspended sediment production that did not stabilize to near pre-treatment values until 12 years post-harvest. The proportion of disturbed watershed in this study had an approximately additive effect on the amount of suspended sediment, therefore sediment levels lower in the watershed might reach water quality thresholds when sites higher in the watershed do not. Beschta (1978) found that increased sediment movement into streams from logging in the Alsea Watershed was related to road building and subsequent mass failures related to roads.

The South currently produces more lumber than any other region in the US. The frequency of site disturbance that is associated with the intensive forest management characterized by short rotation times and site preparation involving soil disturbance has the potential to affect streams that originate in or traverse the stands in this region (Keim and Schoenholtz 1999). In efforts to monitor effectiveness of BMPs and to learn about the biotic and abiotic controls on the aquatic ecosystems, research has focused on identifying effects of harvest, such as increased turbidity, and mechanics of these effects, such as forest roads intersecting streams. Researchers and land managers have identified that timber harvest without SMZs can affect aquatic ecosystems through increased sediment movement into the stream and increased stream temperatures and have suggested the use of BMPs to reduce the impact on the streams (Carroll et al. 2004, Kedzierski and Smock 2001, Keim and Schoenholtz 1999). Evidence from many studies suggests that streambed instability and altered sediment dynamics have resulted from timber harvest through changes in flow regimes and dynamics of sediment delivery to streams (Angradi 1999, Carroll et al. 2004, Kedzierski and Smock 2001, Keim and Schoenholtz 1999, Zweig and Rabeni 2001). Streams in an unrestricted harvest unit (skidder traffic not controlled, trees felled up to stream bank) showed more instability through a greater amount of total change in channel cross sectional area than streams in unharvested reference areas in the loessial bluffs of western Mississippi (Keim and Schoenholtz 1999). Based on results of studies of sedimentation in the Northwest (Lewis 1998), also where timber harvesting is occurring in more than one part of a watershed, there is potential for cumulative negative effects on stream habitat from intensive forest management in southern industrial and private timberland.

Logging in the riparian zone can alter aquatic community structure by increasing amount of available light to the stream, which can stimulate macropyhte growth and abundance (Batzler et al. 2000, Kedzierski and

Smock 2001). The invertebrate community can also shift in response to a change in available substrate and food sources (Batzler et al. 2000, Kedzierski and Smock 2001). In one low-gradient system, the greatest effect from logging adjacent to the stream was a large increase in the amount of macrophytes which provided new substrate to colonize and higher macroinvertebrate production (Kedzierski and Smock 2001). Anthropogenic disturbance in a catchment can also have a destabilizing effect on stream habitat and lower the predictability of stream measures like organic material cycling (Stevens and Cummins 1999).

Storm flow sediment loads can vary by up to a three-fold increase between harvested and unharvested units (Keim and Schoenholtz 1999, Lewis 1998). Beschta (1978) also reported that the timing and magnitude of suspended sediment concentration was altered due to timber harvesting in the Alsea Watershed study in the Oregon Coast Range. Increasing the amount of fine sediment has been shown to affect the macroinvertebrate community even in streams that already have a proportion of substrate as sand or silt (Angradi 1999, Zweig and Rabeni 2001). Biomonitoring for fine sediment inputs into streams in the South is important because fish and macroinvertebrates are sensitive to changes in the substrate (Zweig and Rabeni 2001).

Reach type and flow, organic material dynamics, depth of scour, changed sediment-size distribution, and habitat stability have been shown to affect macroinvertebrate community structure, abundance, and production in streams with erosive sediments (Strommer and Smock 1989, Payne and Miller 1991, Kedzierski and Smock 2001, Carroll et al. 2004). In experimental sediment trays, the amount of fine sediment was correlated with the macroinvertebrate community that colonized the substrate, but organic material was not (Angradi 1999).

1.2. Macroinvertebrate Monitoring

Biological monitoring is the use of living organisms to determine the quality of the environment (Rosenberg and Resh 1993). Increased pressure on aquatic ecosystems from land use and management prompted the initial interest in biomonitoring late in the twentieth century (Cairns and Pratt 1993). Aquatic biological monitoring, or biomonitoring, procedures often use macroinvertebrates and fish in the assessment of habitat condition and water quality in freshwater lakes and streams. Macroinvertebrates are a diverse group of species that react strongly and often predictably to changes in the aquatic ecosystem (Cairns and Pratt 1993). These organisms are especially useful because they are ubiquitous, they exhibit a range of responses to environmental stress, and their community health and composition can reflect on both chemical water quality and stream habitat quality (Rosenberg and Resh 1993). Incorporating living organisms into a water quality monitoring plan may increase the sensitivity of the analysis and give greater temporal scope to the project (Rosenberg and Resh 1993).

Responses of macroinvertebrates may be at the individual, population, or community level, with most studies concentrating on the latter (Rosenberg and Resh 1993). Macroinvertebrate community composition is driven by a combination of habitat features. Studies have found significant correlations between invertebrate community structure and temperature (Hawkins et al. 1982), substrate composition (Reice 1980, Erman and Erman 1984, Bourassa and Morin 1995), current velocity and depth (Erman and Erman 1984, Quinn and Hickey 1994), organic inputs (Hawkins et al. 1982, Culp and Walde 1983), gradient (Danehy and Ringler 1999), topographic sub basin (Mauger 2001), and amount of fine sediment (Angradi 1999, Relyea et al. 2000, Zweig and Rabeni 2001). All of these factors contribute to the habitat for aquatic invertebrates. Consequently, invertebrate community composition changes may indicate a change in the habitat.

Substrate composition is often cited as the most important determinant of invertebrate composition, abundance, and distribution (Minshall 1984). Laboratory and in-stream experiments have attempted to test this hypothesis on the role of substrate and its influence on macroinvertebrates. Researchers have placed artificial substrates that vary in mean particle size, relative size distributions, and amount of organic detritus into streams and then compared the resultant taxa colonization to the substrate composition (Rabeni and Minshall 1977, Erman and Erman 1984). Insects were related to small substrates that served as the best food collecting device (Rabeni and Minshall 1977), and were related to median particle size, stream current, and detritus amounts (Erman and Erman 1984). Research using artificial substrates will minimize variance with respect to the independent variable, but may over-simplify actual conditions in the stream.

Some recent studies have used natural substrate to characterize communities, or have used invertebrate communities to characterize the substrate (Zweig and Rabeni 2001, Relyea et al. 2000). Relyea et al. (2000) examined the relationship between fine inorganic sediment and aquatic insects. They found species-specific responses to the amount of fine sediment in the streambed, identified a subset of taxa that showed definite preferences for or against fine sediment, and created the fine sediment bioassessment index (FSBI). Zweig and Rabeni (2001) compared substrate embeddedness and a visual survey of fine sediment coverage of the stream bottom to the invertebrate community at each site to develop the Deposited Sediment Biotic Index (DSBI). This index was created to characterize sediment impairment in the sampled streams and is a potentially useful tool for assessment of ecological effects of deposited sediment (Zweig and Rabeni 2001).

Few other studies have delved into quantitative natural sediment analysis where sediment is actually fractionated and size fractions are quantified. This may be because no methods are standardized and few studies can be directly compared. Many studies use qualitative measures to determine substrate composition. Studies that quantify substrate often use one of a suite of methods, not all of which are easily comparable. Quantification of substrate should accompany any macroinvertebrate research related to habitat. New research relating macroinvertebrates to substrate composition needs to be initiated to help facilitate more accurate biomonitoring protocols, especially on a regional basis as not all sampling paradigms relate to all systems.

Quantitative biomonitoring programs are needed to understand the impact of humans on the environment. Identifying and using indicator species that have particular requirements of physical or chemical variables is critical to aid in characterizing community responses and changes to anthropogenic effects on aquatic ecosystems (Johnson et al. 1993). This will aid in development of sustainable land management to preserve aquatic ecosystem structure and function. In addition to the use of indicator species, rapid bioassessment protocols are important. Rapid bioassessment procedures are often abbreviated sampling protocols that take less time, training, technical support, and therefore money to monitor stream systems (Resh and Jackson 1993).

1.3. Justification of Study

Interest in monitoring for impacts on aquatic habitat from agriculture, forestry, and urban development is increasing rapidly (Cairns and Pratt 1993). Much of this work relating to forestry has been concentrated in the Pacific Northwest and the East Coast of the U.S. in experimental watersheds such as the Alsea, H.J. Andrews, and Coweeta Watersheds (Waters 1995). Other areas of the country have been overlooked and the indices and other biomonitoring tools may not be suitable in these cases because of inherent differences in bioregions across the U.S. It is important to have support for local studies of BMP effectiveness that have an appropriate scope of reference. These local projects can provide a point of reference for local diversity and could be used in conjunction with projects of a broader scope.

Although not formally addressed in this study, much of the land in north central Mississippi has a long history of anthropogenic disturbance beginning with forest clearing for agriculture, then natural regeneration to pine after farm abandonment, logging, and replanting back to pine. Harding et al. (1998) found evidence of community diversity differences between logged and unlogged sites 40 years after disturbance. Thus, the response or lack thereof by the biotic community to contemporary forest management could be related to the legacy of disturbance in this region. In north central Mississippi saw-log timber requires stand rotations of about 27-30 years. Therefore the frequency of disturbance is high, which has the potential to contribute to the effect of legacy disturbance in the watersheds.

Nonetheless, impacts on aquatic systems associated with current land disturbance in this area need to be characterized and monitored. Many aquatic studies from the Pacific Northwest and Appalachians have centered on forest land use relations to stream ecology, but much of this information may not be applicable to small, low-order streams in north central Mississippi. Streams respond to change in different ways and these mechanisms have not been widely reported in forested headwater streams of the Upper Coastal Plain in the Southeastern U.S. Warmwater streams running through Mississippi forests may respond to disturbance differently due to the community composition (more sediment-tolerant taxa), habitat structure (more fine sediments, less wood in streams), and hydrology (flashier systems in deeply incised channels) and must be researched and understood so appropriate BMPs, monitoring procedures, and restoration efforts will be successful.

Basic information about aquatic invertebrate communities in relation to stream habitat features in north central Mississippi is needed if biomonitoring is going to be a useful tool to assess management effects. This study will incorporate north central Mississippi, an important timber-producing region, into a growing list of locations where new attention is paid to the effects of forest harvesting on aquatic biota in stream systems and the role of SMZs in maintaining or creating desirable stream conditions.

1.4. Null Hypotheses

1. Streamside management zones have no effect on substrate composition.
2. There is no relationship between invertebrates and water quality, cover metrics, or relative amounts of substrate.
3. Streamside management zones have no effect on stream channel cross-sectional area.

2. Methods

2.1. Study Area and SMZ Treatments

Study sites were located within the Sand-Clay Hills region of north central Mississippi and were selected based on three criteria: 1) sites were to contain low-order, perennial streams located within managed stands of loblolly pine (*Pinus taeda* L.), 2) stands were designated for harvest on both sides of the stream, and 3) upstream areas were free of roads, agriculture, and logging activity (Carroll 2002, Young 2002). Fifteen sites meeting these criteria were selected and were located across six counties: Attala, Calhoun, Choctaw, Webster, Winston, and Yalobusha County. Six streams were located in harvest units that incorporated a >10 m SMZ, six streams were located in unrestricted harvest units where harvest was allowed up to the stream bank, and three streams were in units not designated for harvest treatment until at least 2003 (Table 2.1). Two of the fifteen streams were not included in this study. One stream was dry during each sampling period, whereas a potential reference stream was third-order and not deemed comparable to the other lower-order streams. All streams included in this study were first- or second-order perennial streams. Not all streams were harvested on both sides of the stream. Many counties in this area list forestry as the primary or secondary agriculture activity. For further site description, see Carroll et al. (2004).

Two sampling sites, referred to as upstream and downstream, were designated for each stream. Each sampling site contained three permanent transects marked with steel rods (Figure 2.1). The upstream site was located above the harvest unit and the downstream site was located within the harvest unit adjacent to the downstream harvest boundary.

2.2. Field Data Collection

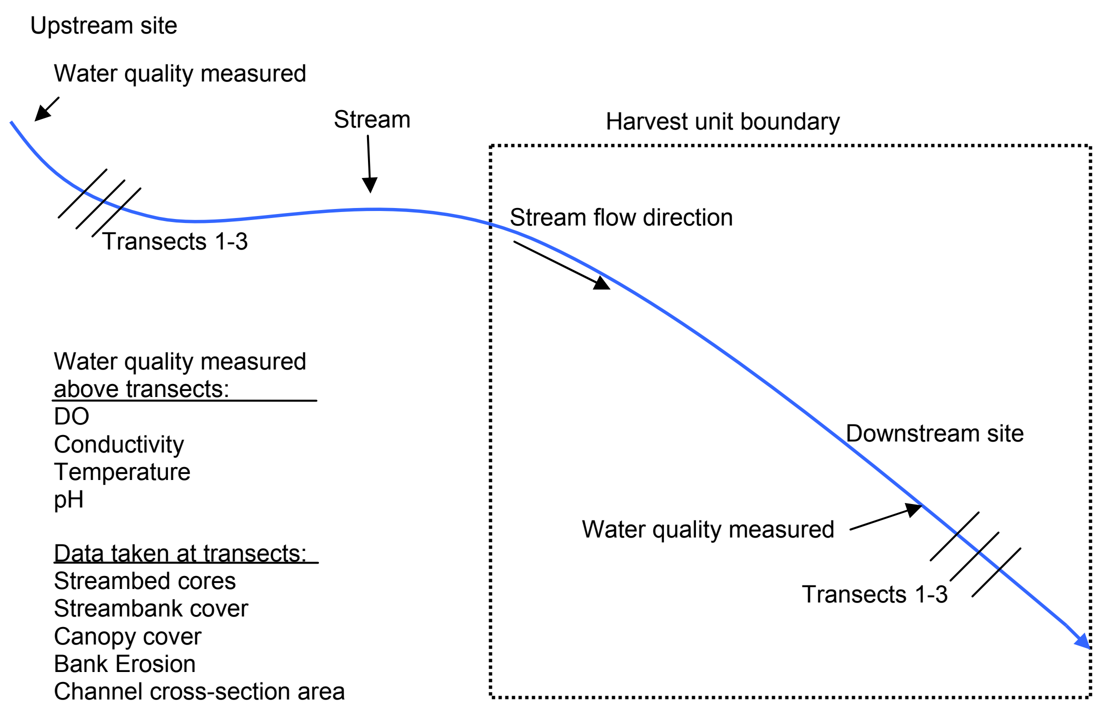
Field sampling was conducted during June and July of 2002 and 2003. Habitat metrics were visually assessed at each of the six transects in each stream on a scale of 1 to 10 for streambank vegetation stability, streamside

Table 2.1. Treatments and watershed characteristics for studied streams for low-order headwater streams of North Central Mississippi (after Carroll et al. 2004).

Treatment	Harvest date	Watershed area (ha)	Watershed harvest unit area (ha)
Reference 1	None	98	0
Reference 3	None	76	0
SMZ 1	Mar-00	43	14
SMZ 2	Jan-00	101	25
SMZ 3	Apr-00	45	26
SMZ 4	Nov-01	n.a.	n.a.
SMZ 5	Jun-01	n.a.	n.a.
No-SMZ 1	May-00	200	14
No-SMZ 2	Jan-00	42	14
No-SMZ 3	Mar-00	27	3
No-SMZ 4	Jul-01	n.a.	n.a.
No-SMZ 5	Sep-01	n.a.	n.a.
No-SMZ 6	Jan-01	n.a.	n.a.

n.a. - not available

Figure 2.1. Schematic of data taken for each studied stream streams for low-order headwater streams of North Central Mississippi .



cover, and bank erosion (Barbour et al. 1999). Scores were based on predetermined criteria where lower scores indicated poorer habitat (Table 2.2). The percent vegetative cover of the area surrounding the transect stake and canopy cover over the center of the stream were visually estimated. Initially, a second opinion from the field assistant was used to verify visual estimates and when both estimates were consistently similar estimates were conducted by one person. Wetted width of the stream was taken in the plane of the staked transects. Stream depth was taken at ¼, ½, and ¾ of the wetted width of the stream at baseflow conditions. Measurements of habitat variables were taken at the second and third transect at each site (Table 2.3). All habitat variables were measured in the plane of the transect.

Summer 2002 water temperature, pH, and conductivity were measured with a YSI® Model 63 water analyzer and DO was measured with a YSI ® Model 55 DO meter. The DO meter was calibrated every morning before use and three-point pH and two-point conductivity were calibrated once at the start of data collection in 2002. Water temperature, pH, conductivity, and DO were measured with a YSI 556 multi-probe system in 2003. Three-point pH and two-point conductivity calibrations were made once at the start of data collection and DO was calibrated daily. For each day streams were sampled, all water quality data were collected once above the transect areas for the upstream and downstream sites at each stream (Figure 2.1).

Substrate cores were collected at approximately 1/3 of the stream wetted width either on the right or left side for 2002, and on both the right and left side for 2003. Substrate cores were collected by pounding a 5.4 cm diameter PVC pipe, open at both ends, approximately 5 cm into the streambed with a rubber mallet (Robertson and Piwowar 1985). A spatula was slid underneath the open bottom of the pipe, and the core was pulled from the stream. All contents of the core were placed in plastic bags for

Table 2.2 Habitat parameters and scores used in visual habitat assessment of stream banks streams for low-order headwater streams of North Central Mississippi . From Barbour et al. (1999) and Plafkin et al. (1989) as presented in Clouse (2003).

	Condition Category			
Habitat Parameter	Excellent	Good	Fair	Poor
Bank Erosion	Stable. No evidence of erosion or bank failure. Side slopes generally <30%. Little potential for future problems.	Infrequent, small areas of erosion mostly healed over. Side slopes up to 40% on one bank. Slight potential in extreme floods.	Moderately unstable. Moderate frequency and size of erosional areas. Side slopes up to 60% on some banks. High erosion potential during extreme flow	Unstable. Many eroded areas. Side slope >60% common. “Raw” areas frequent along straight sections and bends.
Bank Vegetative Stability	Over 80% of the streambank surfaces covered by vegetation or boulders and cobble.	50-79% of the streambank surfaces covered by vegetation, gravel, or larger material.	25-49% of the streambank surfaces covered by vegetation, gravel, or larger material	Less that 25% of the streambank surfaces covered by vegetation, gravel, or larger material
Streamside Cover	Dominant vegetation is shrub.	Dominant vegetation is of tree form.	Dominant vegetation is grass or forbes.	Over 50% of the streambank has no vegetation and dominant material is soil, rock, bridge materials, culverts, or mine tailings.
Score	10-9	8-6	5-3	2-0

Table 2.3 Habitat and water quality data collected for 2002-2003 at each stream and site streams for low-order headwater streams of North Central Mississippi.

Variable Measured	Description
Bank Erosion	Measured on a scale of 1-10[1]
Canopy Cover[2]	% of area over middle of transect shaded by the canopy
Conductivity[2]	Measured above sampling site
Depth	Cm deep of water at ¼, ½, and ¾ the wetted width
Dissolved Oxygen (DO)[2]	Measured above sampling site
pH[2]	Measured above sampling site
Stream Cross Section	Area in m^2 between permanent transects and streambed/floodplain
Streamside Cover[2]	Measured on a scale of 1-10[1]
Substrate[2]	Each size class represented as a % of the total
Vegetation Percent	% of ground covered by vegetation in 1 m^2 around transect marker
Vegetation Stability	Measured on a scale of 1-10[1]
Water Temperature[2]	Celsius measured above sampling site

[1] See Table 2.2 for further description.

[2] Variable used in statistical analysis and habitat cluster.

transport to the laboratory. Upon return to the field truck, formalin was added to make approximately 5% formalin solution in the bag prior to transport.

Stream cross sections were measured by stringing a measuring tape from bank to bank between the permanent transect markers. The distance was divided by seven and vertical measurements were taken from each of the seven distances. Cross-sections were measured in 2003 for comparison with previous measurements made between 1999-2002 (Carroll 2002, Clouse 2003, Carroll et al. 2004).

2.3. Laboratory methods

2.3.1. Invertebrates

Core samples containing aquatic invertebrates were prepared first by removing the formalin solution used to preserve, transport, and store samples. To do this, samples were placed in a Buchner funnel and filtered through Whatman 42 (2.5 µm) filter paper with suction. Samples were transferred to a bucket, water was added, the water was agitated, and the sample was then poured over both 250 µm and 63 µm mesh sieves to catch elutriated invertebrates and small sand grains that passed through the top sieve. This was repeated six times for each sample. The sample in the 250 µm sieve was transferred to a plastic bag and preserved with 70% ethanol and 5 ml rose Bengal biological dye solution for enumeration.

Rough sorting began with rinsing the ethanol and rose Bengal dye off with water. The rinsed sample was placed in one or two divided Petri dishes. Approximately 1/16 of the sample was placed into either a Petri dish or into a zooplankton sorting wheel and invertebrates were picked out of the sand and organic material. Substrate remaining from the rough sorting was kept from 2002 samples and resieved. Amounts were minimal (< 2 gm total) and this was not repeated for 2003 samples. Invertebrates collected in 2002 were either sorted until all individuals were found, until 100 total individuals were found, or for four hours. In 2003, there were no time constraints for rough sorting and therefore rough sorting stopped when the sample was picked through or when 200 aquatic insects were found. Invertebrates from 2002 and 2003 samples were rough sorted using 10x magnification on a Meiji microscope and identified using dichotomous keys (Merritt and Cummins 1996).

Final identifications and recounting were made for 2002 invertebrates by Orlando Ferrar of Mississippi State University. 2003 invertebrates were recounted and identified by the author with assistance from Dr. David Lytle, professor of aquatic entomology, Oregon State University.

Family-level identification was used for both rough-sort and final identifications of insects where possible. Certain orders, especially Trichoptera and Plecoptera, were represented almost exclusively by individuals that were immature and/or damaged and were not identified to family. Conflicting views are apparent in the literature over the usefulness of taxa taken only to family or order levels, but family level is robust for many applications in stream biomonitoring (Bowman and Bailey 1997). For this study, because of time and monetary constraints, we chose to use mixed level taxonomy in our analyses. Individuals were tallied as an order where it was not possible to identify an individual to the family level. Therefore to express the number of invertebrate types or taxa richness within a sample core, an order and families within that order were counted as separate and equal entities. This was used to minimize the loss of information from coarse level identifications. The Shannon-Weiner Diversity index was calculated for invertebrate composition found in each streambed core using the formula:

$$H' = -\Sigma(pi)(\ln(pi)) \qquad \text{eq. (1)}$$

2.3.2. Substrate

After invertebrate elutriation, sieves were used to separate fine sand (63 μm), coarse sand (500 μm), and pebbles (2 mm) by washing water over a sieve stack to work particles into the appropriate size classes, which were based on a modified Wentworth scale (Cummins 1962). The rinse water was collected in a bucket under the sieves and recirculated with a small pump in order to determine the amount of particles smaller than fine sand. This fraction of the sediment was called silt/clay. Each sieve was rinsed into a pan and dried at 105 C for at least 24 hours. Silt/clay weight was determined by first weighing the bucket of rinse water, then taking a well mixed subsample of water, filtering it through pre-weighed Whatman 934-AH (1.5 µm) glass fiber filters, and then drying the sediment on the filter papers. The subsample weights were then converted to the weight of total sediment

2.4. Statistical Analyses

Descriptive statistics including means and ranges were calculated for each habitat variable at each site. Substrate size classes were relativized as percent total weight for each sample core. For statistical analyses, canopy cover and substrate size classes, represented as percentages were arcsin transformed to have equal variance throughout the range of values. Analysis of variance (ANOVA) was used to test the difference between means of relative percentages of each substrate size class for treatment effects. Analysis of variance was also used to test for treatment effects on the difference between upstream and downstream substrate by size class. Repeated measures ANOVA using the mixed model procedure was used to detect treatment effects on the change in cross section areas in nine of the streams over 12, 24, 29, and 43 months since the first measurements were taken (SAS Release 8.2 SAS Institute Inc, Cary NC 2000).

Linear regressions were run to test for relationships between relative abundance of invertebrate taxa and water quality and habitat metrics, between relative abundance of invertebrate taxa and percent substrate by size class, and between taxa richness and percent substrate by size class (S-PLUS Release 6.1 Insightful Corp., Seaattle, WA 2000). P-values less than 0.10 were used to determine presence of significant relationships, and correlation coefficients (r^2) were used to determine the fit of the data to the linear models. P-values less than 0.10 are suggestive but inconclusive of relationships and were used for these observational data. The high heterogeneity and relatively small sample size made relationships more significant than this (ex. $p<0.05$, $p<0.01$) rare in this data set.

Non-metric Multidimensional Scaling (NMS) was used to determine similarity of invertebrate composition among samples collected in 2002 and 2003 (Mather 1976, Kruskal 1964). Ordination techniques reduce data with many variables to the strongest patterns present. This is especially useful in community data where taxa occurrence or abundance may be non-linearly related across sites based on environmental gradients. The patterns NMS finds are the axes of the ordination graphs. By definition the axes are not related. Measured environmental variables can be related to the ordination graphs by coding related sample units with symbols, and also by examining how individual environmental variables correlate to each axis.

The 19 taxa in the 2002 data set represent 19 different axes, or dimensions, if the data were plotted. Distances between each sample unit plotted in the multi-dimensional space (in this case, 19 dimensions) represent the dissimilarity between each sample unit. Patterns in the data are extracted based on taxa occurrence, and the strongest patterns are the axes of the ordination graph. Samples plotted close together on the ordination graph are more similar than samples plotted further apart. The distance between sample units in the 19-dimensional space is compared to the distance between sample units plotted in the 1, 2, or 3-dimensional space to make sure that the reduced dimensionality of the data still represents the original relationships. This is termed the stress of the ordination. If the fit is good, the measure of stress calculated by NMS will be low. To examine how measured environmental variables relate to the sample units, symbols are assigned to each sample unit based on a relationship to a group, such as harvest treatment. If groups of symbols plot close together on the graph, then it can be inferred that groups of samples with similar invertebrate composition are related to the same harvest treatment. Individual variables, such as the percent canopy cover measured at each site, can also be overlaid on the graph to determine if sample order along the axis and the gradient in canopy cover are related. This ordination technique uses ranked distances and allows for any distance measure (Mather 1976, Kruskal 1964). It also alleviates the zero-truncation problem often associated with community composition data, caused by many shared zeros found in the data matrix (Beals 1984).

Raw counts of invertebrates were used to develop presence/absence data for each taxon. For 2002 and 2003 presence/absence invertebrate data, Sorenson's distance and the PC-ORD NMS autopilot mode, set on medium, tested the similarity among sample units (McCune and Mefford 1999). A Monte Carlo test used the difference in stress between real and randomized data to determine the number of axes to use.

The ordinations plotted sample units in family/order space (McCune and Mefford 1999). Graphical overlays, or joint plots, were used to show whether any habitat variables correlated to the axes. Vectors radiating from the centroid of the ordination show the strength (length of vector) and to which axis (vector direction) the variable most strongly correlated.

Because ordinations only allow the community data to be displayed in two or three dimensions, yet the data exists in many dimensions, hierarchical agglomerative (HA) cluster analysis was used for both 2002 and 2003 data to group sample units with similar habitat variables (McCune and Grace 2002).

Hierarchical agglomerative cluster analysis was run for canopy cover, conductivity, dissolved oxygen, streamside cover, water temperature, presence/absence of small and large wood, and substrate size classes (McCune and Mefford 1999). The groups of sample cores were symbol coded and overlaid on the ordination of the community data to determine whether groups of similar invertebrates related to groups of similar habitat. Harvest treatment was symbol coded and overlaid for the same purpose.

3. **Results**

3.1. Habitat and Substrate Composition

Upstream and downstream means of streambank cover, canopy cover, pH, temperature, dissolved oxygen (DO), and conductivity sampled at the time of invertebrate collections in 2002 and 2003 were calculated and are summarized in Tables 3.1 and 3.2. The 13 streams had a wide range of physical and chemical characteristics. Streambank cover ranged from 1 to 10 on a scale of 10, with 1 being poor cover and 10 being excellent cover (Table 3.1). Highest mean streambank cover in 2002 was 5.2 in the No-SMZ downstream location. Streambank cover for both upstream and downstream sites was highest in the No-SMZ treatment in 2003 with means of 5.1 and 5.3, respectively. Downstream canopy cover ranged from 0 to 90% for individual streams and was lowest for the No-SMZ treatment in 2002 and 2003 with 27% and 38% average canopy cover, respectively (Table 3.1).

Among water quality parameters at the time of invertebrate sampling, mean pH ranged from 5.0 to 8.3, mean streamwater temperature ranged from 17.3 to 24.4 °C, mean dissolved oxygen ranged from 1.3 to 9.1 mg/L, and mean conductivity ranged from 10 to 253 µs/cm for each treatment type (Table 3.2). Maximum streamwater temperature in individual streams at the time of invertebrate sampling was observed in the No-SMZ streams in 2002 and 2003 (Table 3.2). Mean conductivity values were highest in SMZ streams primarily because one SMZ stream had conductivity nearly one magnitude greater than the other streams.

Relative percentages of substrate size classes also ranged widely among treatments and between upstream and downstream sites (Figure 3.1, Table 3.3). Average substrate composition of Reference and SMZ treatment upstream sites in 2002 and 2003 were more similar than the No-SMZ sites because they contained more than 20% gravel each year, whereas the No-SMZ sites contained less than 14% gravel each year. Downstream Reference and SMZ sites were similar in 2002 because they contained greater than 15%

Table 3.1. Mean stream habitat parameters at time of invertebrate sampling in 2002 and 2003 streams for low-order headwater streams of North Central Mississippi.

	Treatment	Streambank cover[1]				Canopy cover (%)				Number of streams
		Mean	(S.D.)[2]	Max	Min	Mean	(S.D.)	Max	Min	sampled
2002	**Reference**									
	Upstream	2.5	1.3	4	1	71	44	95	5	2
	Downstream	2.4	1.1	4	1.5	55	10	70	50	2
	SMZ									
	Upstream	5.0	2.2	7.5	2	64	25	90	25	3
	Downstream	3.2	2.8	8	1	73	18	90	40	4
	No SMZ									
	Upstream	4.7	2.1	7	1	52	31	100	0	5
	Downstream	5.2	2.9	8.5	1.5	27	33	80	0	5
2003	**Reference**									
	Upstream	2.5	1.8	10	3.5	66	23	95	40	2
	Downstream	2.9	1.4	5	2	60	16	80	40	2
	SMZ									
	Upstream	4.2	3.1	8	1	54	25	70	10	3
	Downstream	1.4	0.9	3	0	61	23	85	20	5
	No SMZ									
	Upstream	5.1	2.9	9	1.5	35	32	90	0	6
	Downstream	5.3	3.2	10	1.5	38	39	90	0	6

[1] Streambank cover on a scale of 1-10, 1=poor cover 10=excellent cover (Barbour et al. 1999; Plafkin et al. 1999).

[2] Standard deviation.

Table 3.2. Mean water quality parameters at time of invertebrate sampling in 2002 and 2003 streams for low-order headwater streams of North Central Mississippi.

Treatment	pH				Temperature (°C)				DO (mg/L)				Conductivity (µS/cm)				Number of streams sampled
	Mean	(S.D.)[1]	Max	Min	Mean	(S.D.)	Max	Min	Mean	(S.D.)	Max	Min	Mean	(S.D.)	Max	Min	
2002 Reference																	
Upstream	6.8	0.1	6.9	6.7	21.1	0.1	21.2	21.0	6.2	0.3	6.4	6.0	55	28	75	35	2
Downstream	7.6	0.9	8.3	7.0	21.2	0.4	21.4	20.9	7.0	1.2	7.8	6.2	12	3	14	10	2
SMZ																	
Upstream	6.3	1.3	7.6	5.0	20.8	2.0	22.9	18.9	6.6	1.7	8.5	5.5	108	126	253	32	3
Downstream	6.8	1.2	7.7	5.0	20.5	1.1	21.5	19.0	7.1	1.4	8.1	5.1	65	95	207	10	4
No SMZ																	
Upstream	6.5	0.8	7.5	5.2	22.0	2.1	24.3	19.3	5.4	3.0	8.1	1.6	50	27	93	27	5
Downstream	6.5	0.9	7.6	5.1	21.6	1.7	23.4	18.9	5.8	3.0	8.2	1.3	44	31	100	27	5
2003 Reference																	
Upstream	6.3	0.1	6.3	6.2	21.2	0.9	21.9	20.6	7.4	0.1	7.5	7.2	45	35	70	20	2
Downstream	6.3	0.2	6.5	6.2	21.2	1.4	22.2	20.3	7.4	0.1	7.4	7.3	61	40	89	33	2
SMZ																	
Upstream	6.3	1.1	7.5	5.3	19.4	0.8	20.0	18.5	7.3	2.6	8.6	5.5	95	111	223	30	3
Downstream	6.3	0.6	7.3	5.9	19.8	0.9	20.7	18.4	6.7	2.5	8.5	2.5	68	85	219	12	5
No SMZ																	
Upstream	6.2	0.4	6.6	5.7	20.1	1.9	23.2	17.8	8.0	0.7	9.1	7.3	37	13	55	23	6
Downstream	5.7	0.7	6.8	5.1	19.8	2.3	23.4	17.3	8.0	1.1	9.0	6.5	32	15	58	21	6

[1] Standard deviation.

Table 3.3. Mean sediment size at time of invertebrate sampling in 2002 and 2003 streams for low-order headwater streams of North Central Mississippi .

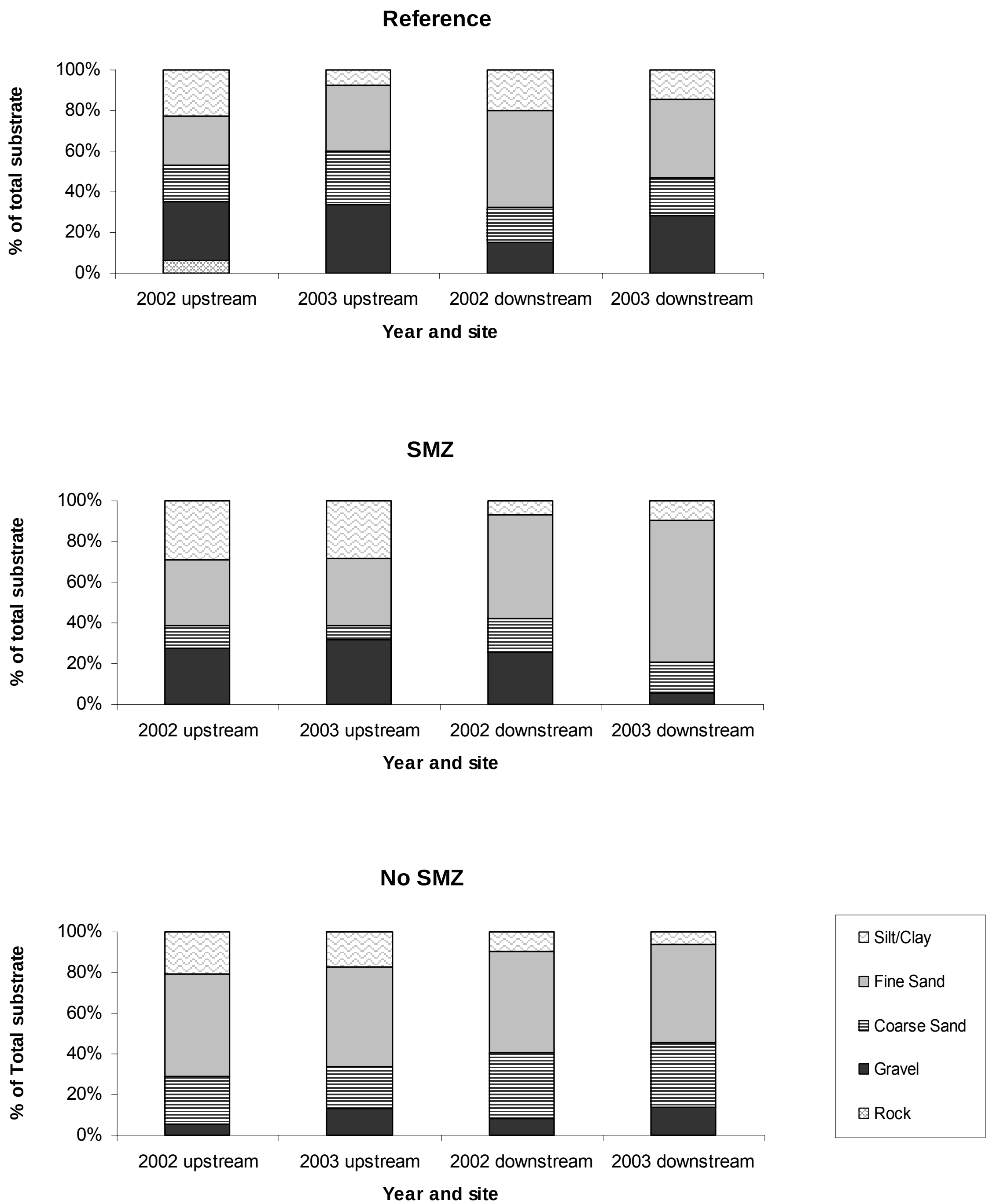
	% Rock[1]				% Gravel				% Coarse Sand				% Fine Sand				% Silt/Clay				Number of streams sampled
	Mean	(S.D.)[2]	Max	Min	Mean	(S.D.)	Max	Min	Mean	(S.D.)	Max	Min	Mean	(S.D.)	Max	Min	Mean	(S.D.)	Max	Min	
2002 Reference																					
Upstream	[3]6.4	12.8	25.5	0	28.8	11.6	40.5	10.6	18.0	11.6	34.3	7.8	24.1	6.4	16.7	30.9	22.6	25.6	60.4	5.4	2
Downstream					15.2	20.2	45.3	3.2	17.4	3.0	20.6	14.1	47.5	22.5	24.4	70.9	19.9	22.7	53.4	5.5	2
Change					- 13.6				- 0.6				+ 23.5				- 2.7				
SMZ																					
Upstream					27.4	24.4	60.5	0.0	10.9	6.4	21.2	4.1	32.4	26.4	77.5	13.8	29.2	21.8	57.9	7.1	3
Downstream					29.1	35.8	71.2	0.1	18.4	18.8	54.8	4.6	45.6	31.2	87.5	11.0	6.8	3.9	10.6	1.3	4
Change					+ 1.7				+ 7.5				+ 13.2				- 22.4				
No SMZ																					
Upstream					5.1	5.5	17.7	0.0	23.7	21.6	62.4	0.3	50.4	19.9	88.7	24.3	20.8	22.5	63.7	0.5	5
Downstream					8.5	15.7	46.7	0.0	32.4	19.4	62.3	6.8	49.3	13.7	76.8	30.6	9.713	66	21.8	0.5	5
Change					+ 3.4				+ 8.8				- 1.1				- 11.1				
2003 Reference																					
Upstream					37.5	26.9	74.8	11.7	[3]27.4 **a**	11.3	34.8	10.6	29.0	14.9	46.3	11.3	6.1	13.6	11.4	3.2	2
Downstream					24.8	26.9	62.1	2.8	14.1	10.6	29.4	6.1	43.9	13.2	53.5	25.0	17.1	16.5	41.2	5.0	2
Change					- 12.7				+ 13.3				+ [3]15.0 **a**				+ 11.0				
SMZ																					
Upstream					22.0	37.0	85.3	0.0	[3]3.7 **b**	4.6	10.1	0.0	38.6	37.6	90.6	5.2	35.7	31.7	77.4	2.3	3
Downstream					4.7	5.1	16.3	0.0	13.7	16.8	56.7	0.0	72.4	19.9	94.7	41.3	9.2	8.2	27.1	2.1	5
Change					- 17.3				- 10.0				+ [3]33.8 **a**				- 26.5				
No SMZ																					
Upstream					12.9	19.1	55.3	0.0	[3]21.0 **a**	19.3	57.0	0.0	46.7	24.3	94.6	17.6	19.4	21.3	57.5	0.3	6
Downstream					13.8	19.0	57.0	0.0	34.3	21.7	59.6	2.5	47.0	23.5	91.3	21.5	4.8	5.0	17.5	0.4	6
Change					+ 0.9				+ 13.3				+ [3]0.2 **b**				- 14.6				

[1] Rock-sized substrate found in only one reference stream. All other values for rock are zero.

[2] Standard deviation.

[3] For upstream, downstream, or change within a given year and substrate class, a significant treatment effect exists at alpha=0.10. Corresponding letters indicate significantly similar samples.

Figure 3.1. Mean relative percentages of substrate classes for each SMZ treatment by year and site streams (upstream of or downstream within the harvest unit) for low-order headwater streams of North Central Mississippi.



gravel and more fine sand than coarse sand. Downstream sites in 2003 contained similar substrate composition for SMZ and No-SMZ sites due to similar amounts of total sand and less than 15% gravel. The Reference sites were not as similar because of the greater amount of gravel (24.8%).

Sediment composition was analyzed to determine if there were trends within treatment groups. Analysis of variance was used to test for differences between all upstream sites, all downstream sites, and the difference between down and upstream relative percentages by substrate size class for 2002 and 2003 (Table 3.3). Significant differences among SMZ treatments were found for 2003 upstream percent coarse sand. Reference and No-SMZ streams had similar amounts of coarse sand (27.4% and 21.0%, respectively) and SMZ treatments had the least (3.7%). For 2003, the change in fine sand between upstream and downstream sites was significantly different with the SMZ and Reference treatments having the greatest change (33.0%, 15.0%, respectively) and No-SMZ treatments having almost no change (-0.02%).

3.2. Invertebrate Composition

The invertebrates found in the 92 streambed cores collected in 2002 and 2003 represented 9 orders of aquatic insects and 7 orders of non-insect invertebrates (Table 3.4). The most abundant taxa as a percent total of taxa counted for 2002 and 2003 cores were Chironomidae (65.1%, 69.6%, respectively), Copepoda (8.2%, 7.9%, respectively), and Oligicheata (7.3% and 8.6%, respectively) (Table 3.4). There was wide variation in the composition of each core as shown by the range of taxa richness from 2 to 12 taxa/sample and Shannon-Wiener diversity, a measure of the number of taxa of equal abundance, ranging from 0.58 to 0.90 (Table 3.5). However, mean values for Shannon-Wiener diversity among treatments and years was less variable, ranging from 0.72 to 0.89. It is of note that mean taxa richness in 2002 and 2003 was numerically higher in downstream locations of No-SMZ streams than in downstream locations of Reference streams (Table 3.5).

Table 3.4 Invertebrate composition from stream channel cores collected in low-order headwater streams in North Central Mississippi in 2002 and 2003.

Order	Family	Total counted 2002	% Total 2002	Total counted 2003	% Total 2003
Diptera	Chironomidae	2801	65.1	3547	69.6
Copepoda	Unknown	352	8.2	402	7.9
Oligochaeta	Unknown	312	7.3	436	8.6
Diptera	Ceratopogonidae	280	6.5	264	5.2
Nematoda	Unknown	220	5.1	217	4.3
Ephemeroptera	Unknown	123	2.9	24	0.5
Plecoptera	Unknown	44	1	42	0.8
Diptera	Unknown	39	0.9	11	0.2
Hydracarnia	Unknown	32	0.7	5	0.1
Coleoptera	Elmidae	25	0.6	30	0.6
Cladocera	Unknown	10	0.2	0	0
Odonata	Unknown	9	0.2	1	<0.1
Crustacea	Unknown	8	0.2	0	0
Mollusk	Unknown	8	0.2	0	0
Ephemeroptera	Caenidae	6	0.1	7	0.1
Diptera	Tabanidae	6	0.1	8	0.2
Ephemeroptera	Baetidae	5	0.1	4	0.1
Trichoptera	Unknown	5	0.1	8	0.2
Coleoptera	Istomidae	4	0.1	0	0
Diptera	Tipulidae	4	0.1	20	0.4
Coleoptera	Unknown	2	<0.1	3	0.1
Diptera	Culicidae	2	<0.1	0	0
Amphipoda	Unknown	1	<0.1	0	0
Odonata	Calopterygidae	1	<0.1	0	0
Odonata	Cordulidae	1	<0.1	0	0
Coleoptera	Dytiscidae	1	<0.1	0	0
Coleoptera	Hydrophilidae	1	<0.1	0	0
Megaloptera	Sialidae	1	<0.1	5	0.1
Colembola	Unknown	0	0	2	<0.1
Odonata	Aeshnidae	0	0	1	<0.1
Odonata	Cordulagastridae	0	0	9	0.2
Hemiptera	Corixidae	0	0	1	<0.1
Odonata	Corydalidae	0	0	1	<0.1
Diptera	Dixidae	0	0	1	<0.1
Odonata	Gomphidae	0	0	11	0.2
Ephemeroptera	Heptageniidae	0	0	2	<0.1
Coleoptera	Hydroptilidae	0	0	5	0.1
Ephemeroptera	Leptophlebiidae	0	0	28	0.5
	Total:	4303	100.0%	5095	100.0%

Table 3.5. Average aquatic invertebrate taxa richness and diversity (H’) in low-order in low-order headwater streams of North Central Mississippi.

	Treatment	Taxa Richness[1]				Taxa Shannon-Wiener Diversity[2]				Number of streams sampled
		Mean	(S.D.)[3]	Max	Min	Mean	(S.D.)	Max	Min	
2002	**Reference**									
	Upstream	9.3	2.2	11	6	0.89	0.03	0.90	0.87	2
	Downstream	6.0	2.4	9	3	0.81	0.08	0.86	0.75	2
	SMZ									
	Upstream	5.5	2.9	10	2	0.76	0.07	0.84	0.70	3
	Downstream	7.4	1.8	9	4	0.85	0.03	0.88	0.81	4
	No SMZ									
	Upstream	7.0	1.7	9	4	0.85	0.04	0.88	0.79	5
	Downstream	7.0	1.7	10	5	0.84	0.02	0.88	0.82	5
2003	**Reference**									
	Upstream	5.5	1.7	8	4	0.72	0.19	0.85	0.58	2
	Downstream	5.5	3.5	9	2	0.73	0.21	0.88	0.58	2
	SMZ									
	Upstream	7.4	3.6	12	2	0.86	0.01	0.88	0.85	3
	Downstream	4.6	1.8	8	2	0.75	0.09	0.85	0.65	5
	No SMZ									
	Upstream	6.3	2.5	11	3	0.76	0.08	0.89	0.68	6
	Downstream	5.8	2.7	11	2	0.81	0.08	0.87	0.67	6

[1] Mean number of taxa/sample.

[2] Shannon-Wiener diversity $H = -\sum_{i=1}^{s} p_i (\ln p_i)$

p_i = Proportion of total sample belonging to ith species. S = number of taxa.

[3] Standard deviation.

3.3. Invertebrate-Habitat Relationships

Forty significant relationships were found between percent invertebrate taxa and water quality or habitat metrics of which two relationships had correlation coefficients greater than 0.20 (Table 3.6). Copepoda abundance in 2003 was negatively correlated to DO with a correlation coefficient of 0.32 (Figure 3.2). Ceratopogonidae abundance in 2002 was positively correlated to streambank cover with a correlation coefficient of 0.21. Percent Oligichaeta was negatively correlated (r^2=0.07, 0.08, respectively) and Tabanidae was positively correlated (r^2=0.08, 0.06, respectively) to streambank cover for both years. For both years relative abundances of Chironomidae (r^2=0.07, 0.18, respectively) and Ephemeroptera (r^2=0.08, 0.07, respectively) were negatively correlated and Oligichaeta (r^2=0.10, 0.19, respectively) was positively correlated to the presence of large wood (Table 3.6).

Of 29 significant relationships found between percent invertebrate taxa and percent substrate, two relationships had coefficients greater than 0.20 (Table 3.7). Ephemeroptera and gravel and Sialidae and silt/clay were positively correlated (Figure 3.3, 3.4) (r^2=0.29, 0.47, respectively). Positive relationships were found in both years for percent Elmidae and gravel (r^2=0.17, 0.12, respectively) and percent Oligichaeta and fine sand (r^2=0.17, 0.13, respectively).

The relationship between 2003 taxa richness and silt/clay was the strongest, yet still had a correlation coefficient of 0.26 and the relationship was not significant in 2002 ($p < 0.10$) (Figure 3.5). Other relationships were weaker and did not suggest the presence of detectable influences in invertebrate taxa richness.

3.4. Cluster Analysis

Using hierarchical agglomerative (HA) cluster analysis for the streamwater pH, temperature, conductivity, dissolved oxygen, relative

Table 3.6. Simple linear regression correlation coefficients between relative abundance of invertebrate taxa and water quality or habitat metrics in low-order headwater streams in North-Central Mississippi.[1]

	pH		Temperature		Dissolved Oxygen		Conductivity		Streamside cover		Canopy cover		Small wood present		Large wood present	
Taxa	2002	2003	2002	2003	2002	2003	2002	2003	2002	2003	2002	2003	2002	2003	2002	2003
Caenidae																
Ceratopogonidae					+0.09				+0.21			-0.09				+0.06
Chironomidae		+0.06		+0.09	+0.06								-0.12		-0.07	-0.18
Cladocera					-0.09											
Coleoptera																
Copepoda		-0.06				-0.32							+0.06			
Cordulagastridae																
Crustacean																
Diptera																
Elmidae																-0.06
Ephemeroptera				+0.07			+0.18				+0.07				-0.08	-0.07
Gomphidae																
Hydracarina																
Istomidae																
Leptophlebiidae												+0.06		+0.06		
Mollusca																
Nematoda						+0.13			-0.07			-0.07				
Odonata											-0.09					
Oligichaeta			-0.14				-0.1		-0.07	-0.08					+0.1	+0.19
Plecoptera									-0.07							
Sialidae		-0.1				-0.09				+0.11						
Tabanidae									+0.08	+0.06		+0.08				
Tipulidae																
Trichoptera							+0.11			+0.1		+0.1		-0.06		

[1] Values included are for regressions with $p<0.10$.

Table 3.7. Simple linear regression correlation coefficients between relative abundance of invertebrate taxa and % substrate in low-order headwater streams in North Central Mississippi.[1]

Taxa	Gravel		Coarse Sand		Fine Sand		Silt/Clay	
	2002	2003	2002	2003	2002	2003	2002	2003
Caenidae					-0.06			
Ceratopogonidae			+0.13					
Chironomidae		-0.07			-0.07			
Cladocera								
Coleoptera							+0.13	
Copepoda				-0.09				+0.17
Cordulagstridae								
Crustacean							+0.09	
Diptera				+0.11				
Elmidae	+0.17	+0.12			-0.07			
Ephemeroptera	+0.29		-0.06					
Gomphidae								
Hydracarina	+0.08				-0.09			
Istomidae								
Leptophlebiidae				-0.06				
Mollusca					+0.14			
Nematoda		-0.07		+0.05				
Odonata								
Oligichaeta	-0.09				+0.17	+0.13		
Plecoptera		+0.15				-0.06		
Sialidae						-0.06		+0.5
Tabanidae			+0.08					
Tipulidae								
Trichoptera								

[1] Values included are for regressions with p<0.10.

Figure 3.2. Simple linear regression between Copepoda and Dissolved Oxygen (DO) in 2003 streams for low-order headwater streams of North Central Mississippi.

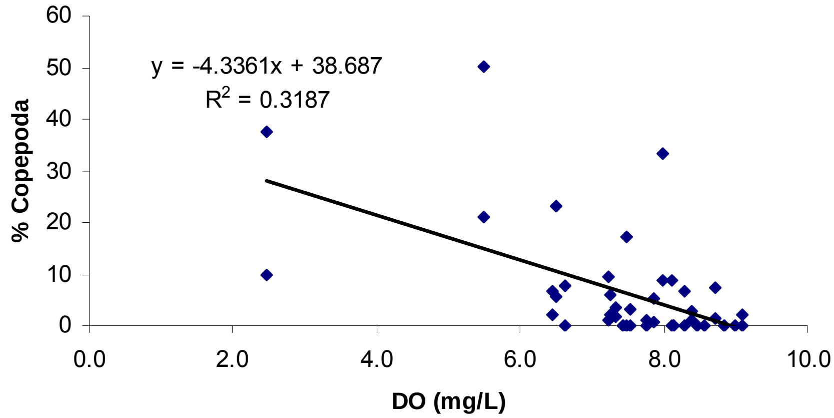


Figure 3.3. Simple linear regression between Ephemeroptera and Gravel in 2002 streams for low-order headwater streams of North Central Mississippi.

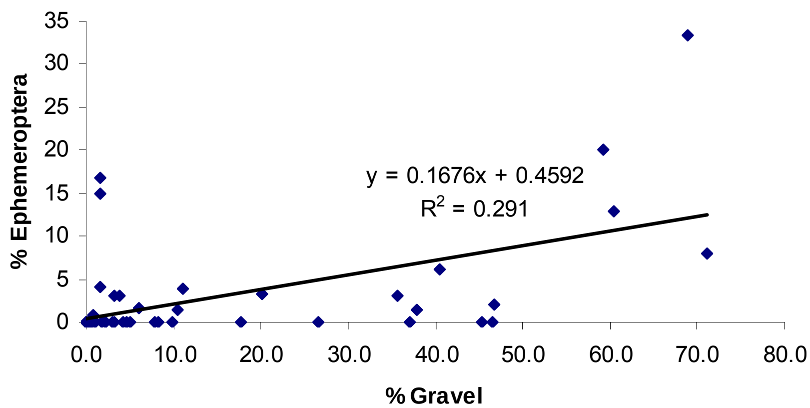


Figure 3.4. Simple linear regression between Sialidae and Silt/Clay in 2003 streams for low-order headwater streams of North Central Mississippi .

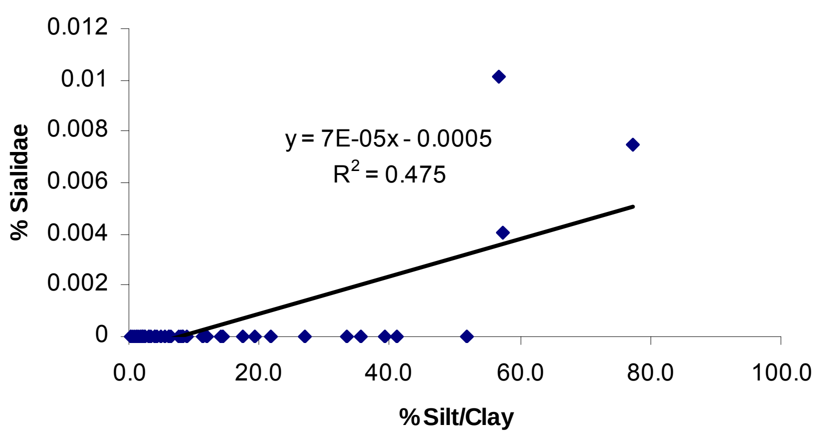
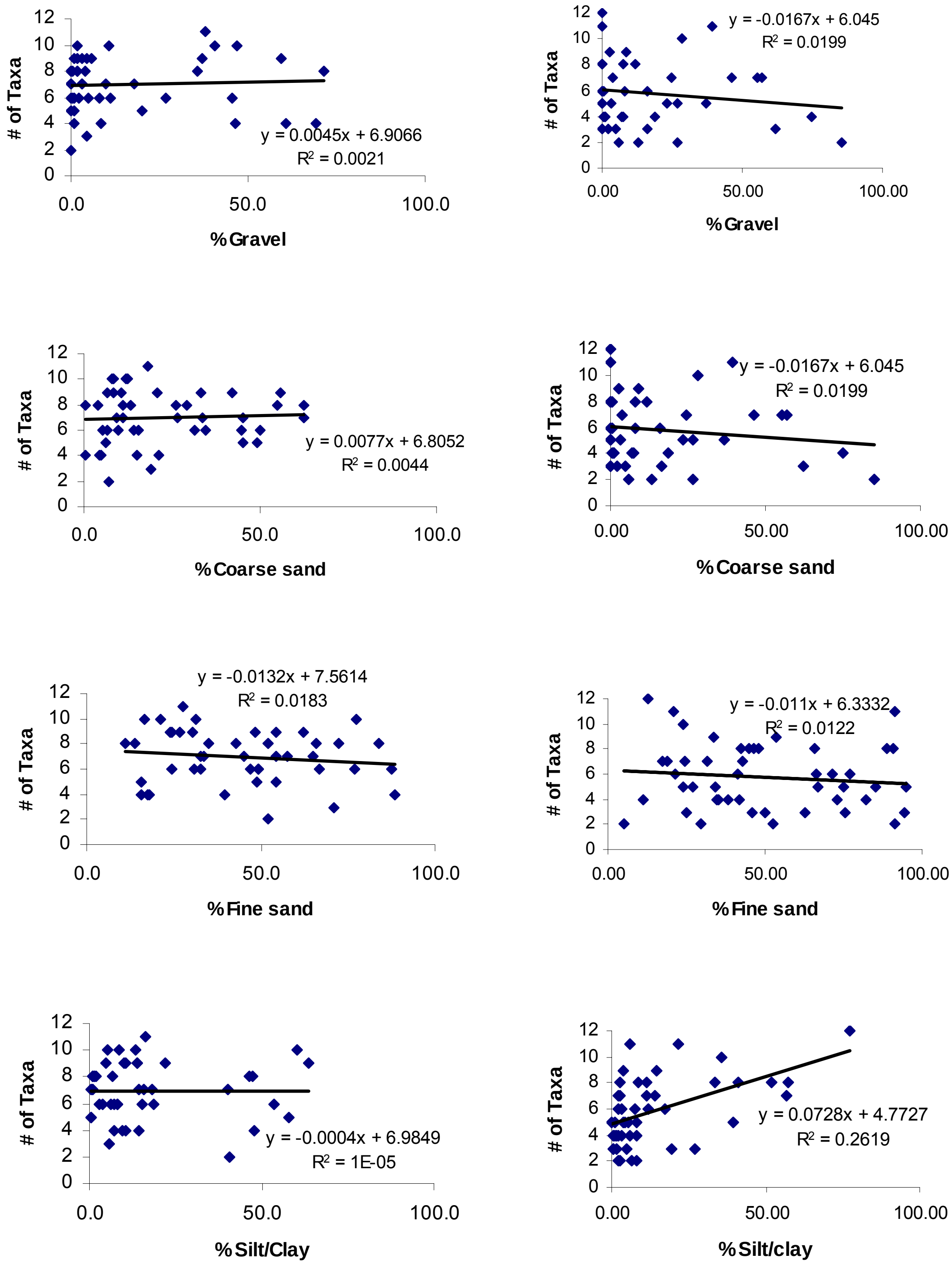


Figure 3.5.Simple linear regressions between invertebrate taxa richness and % substrate class for low-order headwater streams of North Central Mississippi.

2002 2003



amounts of substrate, streambank, and canopy cover, the 2002 data formed three groups after pruning the dendrogram at 80% information remaining (Figure 3.6a). The 2003 data formed three groups after pruning the dendrogram at 60% information remaining (Figure 3.6b). The three levels of grouping provided a compromise between loss of information by including too many samples within a group and providing an interpretable summary of ecological relationships among measured habitat variables. Group 1 of both dendrograms was a mixture of all SMZ treatment types. Approximately 85% of Group 2 for each year was comprised of No-smz sites. This corresponds to about half of the total No-smz sites. The third group in each dendrogram corresponds to samples taken in one SMZ stream characterized by very high specific conductance (range 2002: 206-253 μS/cm, 2003: 219-223 μS/cm).

HA cluster analysis was also run using only relative amounts of substrate from each core. Four groups were differentiated in the resulting dendrograms representing gravel, coarse sand, fine sand, and silt/clay (Figure 3.7 a, 3.7 b). Sample units in each group found by HA cluster analysis were coded with symbols to overlay onto the ordination graphs.

3.5. Ordinations

Sample units used in the ordinations were the presence/absence invertebrate data from each streambed core. The ordination from the 2002 presence/absence invertebrate data was not rotated and had two axes that correlated 71.0 % of the distance in the original multi-dimensional space to the ordination distances. Sample units close together on the ordination are more similar than sample units farther apart. Coding the sample units based on the harvest type showed no separation of samples based on the harvest treatment (Figure 3.8 a).

The variables used in the HA water quality and cover clustering and also taxa richness were overlaid as a joint plot on the ordination but no patterns were apparent for 2002 data (Figure 3.8 b). Substrate groups from the cluster analysis were also overlaid on the ordination, but no patterns in

Figure 3.6 a. 2002 cluster analysis of habitat variables including selected water quality parameters, substrate, streamside cover, and canopy cover streams for low-order headwater streams of North Central Mississippi .

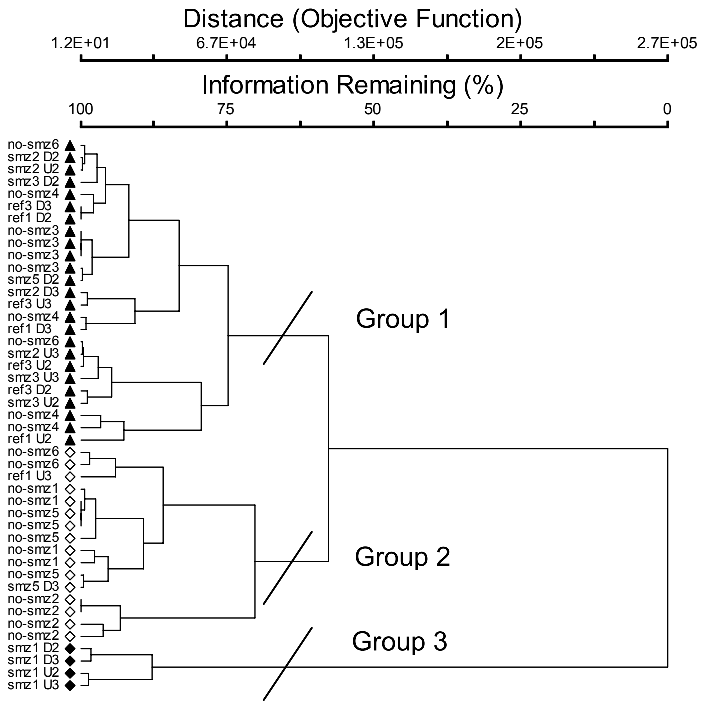


Figure 3.6 b. 2003 cluster analysis of habitat variables including selected water quality parameters, substrate, streamside cover, and canopy cover.

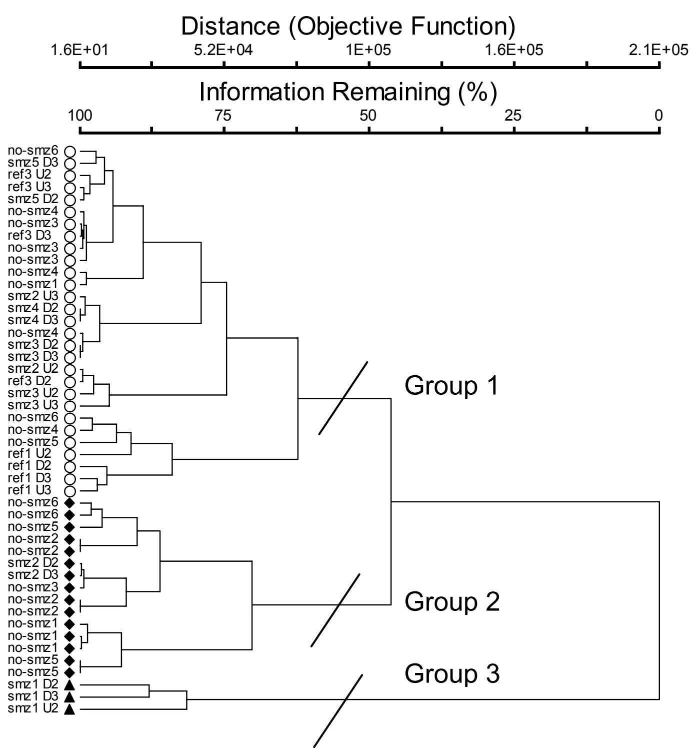


Figure 3.7a. 2002 cluster analysis of substrate size classes streams for low-order headwater streams of North Central Mississippi .

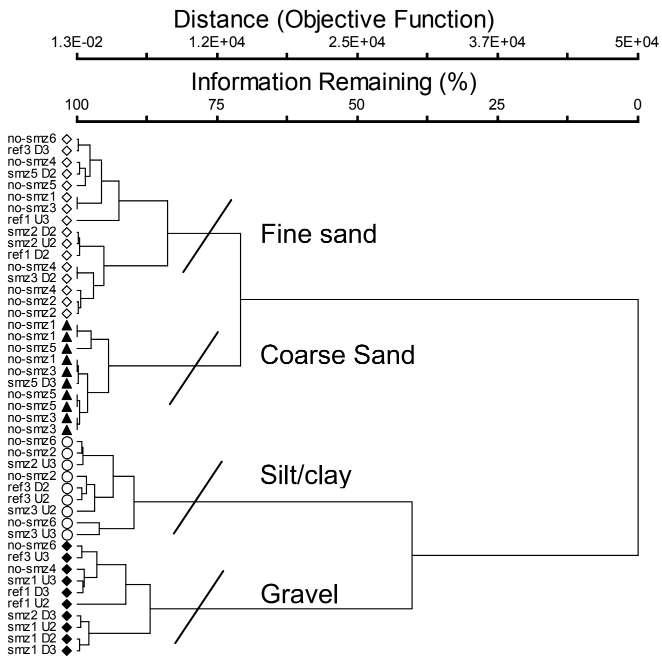


Figure 3.7b. 2003 cluster analysis of substrate size classes.

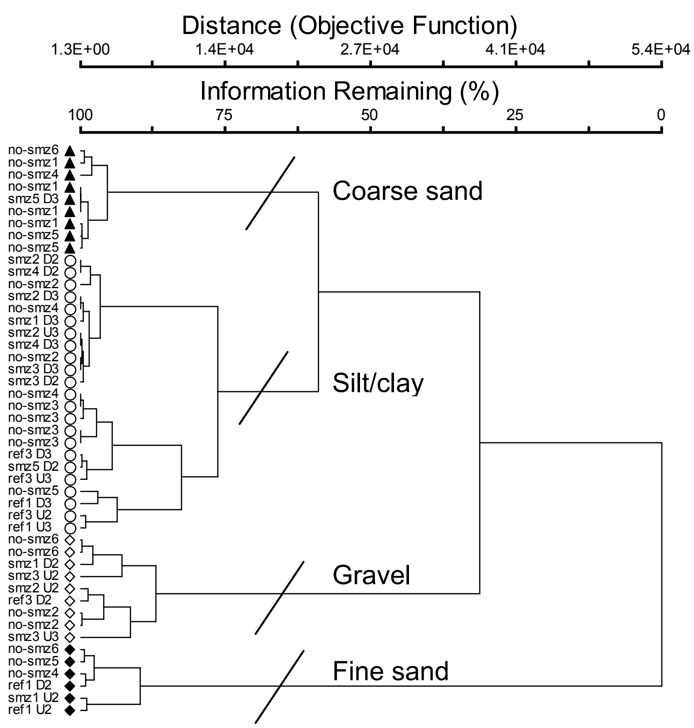
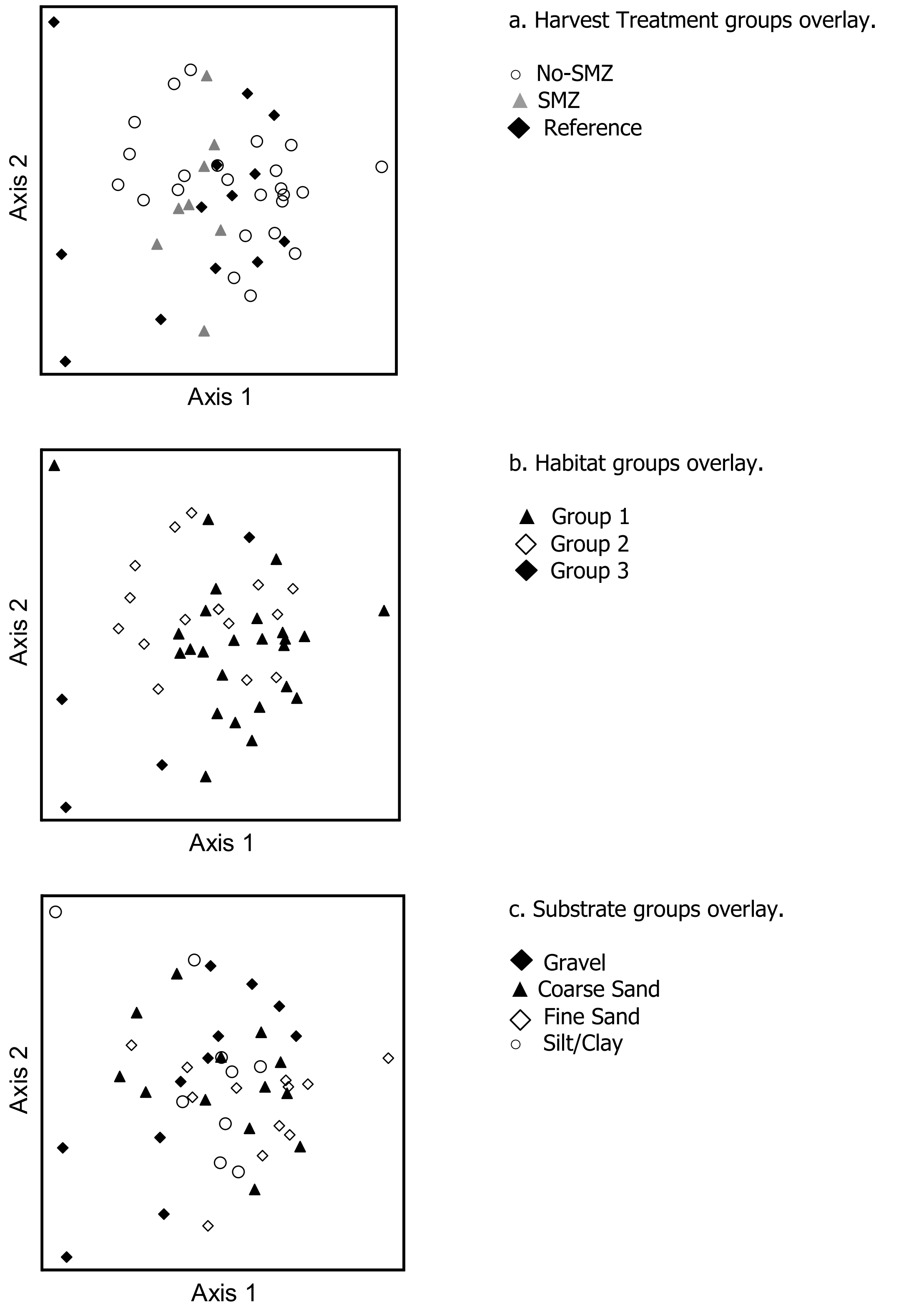


Figure 3.8 2002 Invertebrate ordination and overlays streams for low-order headwater streams of North Central Mississippi .



distribution were apparent (Figure 3.8 c).

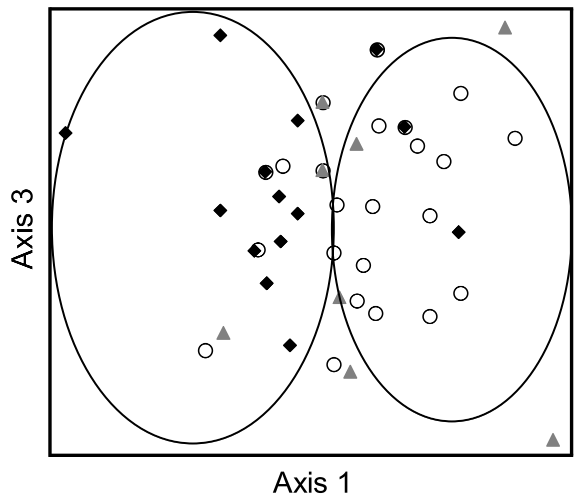
The 2003 invertebrate presence/absence data had more structure than the 2002 data based on the number of significant axes and a greater percent variation explained by the ordination. The ordination was rotated 320 degrees to express patterns more clearly and had three axes that explained 83.5% of the variation in the data. The first and third axes explained 34.8 and 30.1% variation, respectively. Overlaying harvest type on the 2003 invertebrate ordination showed a separation of No-SMZ and Reference sites, which is related to the first axis (Figure 3.9 a). The SMZ sites did not clearly form a group on this ordination. HA cluster analysis overlays showed no patterns on the invertebrate ordination for either habitat clusters or substrate clusters (Figure 3.9 b, c). The only significant variables were taxa richness which negatively correlated strongly to axis 3, and coarse sand which correlated to the first axis. Correlation coefficients are not given for relationships to non-continuous variables such as presence/absence data.

3.6. Stream Channel Cross-Sections

Examining four years of stream channel cross-section data from the nine streams treated in 2000 revealed that patterns of cross-sectional changes among all three SMZ treatments show alterations between aggradation and degradation regardless of treatment (Figure 3.10). The most aggradation, or filling in of the channel was 25% for the REF 3 stream when measured in fall 2001. The most degradation, or loss of sediment, was in SMZ 3, with a loss of 25% of the stream cross section at the second downstream transect when measured in spring 2002.

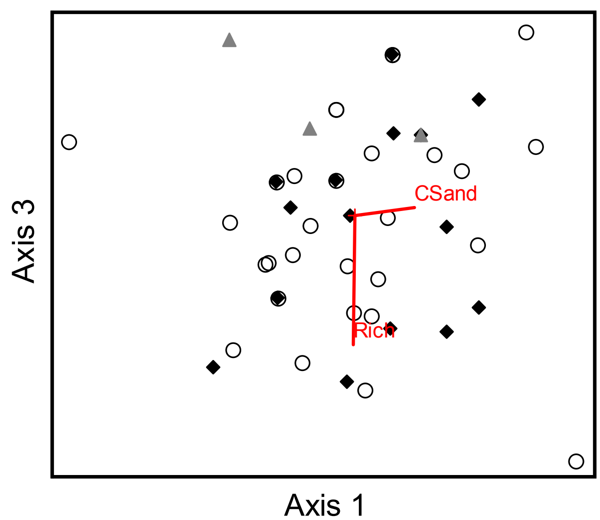
Repeated measures ANOVA using a mixed model (SAS 2000) indicated no significant changes among SMZ treatments in stream channel cross-sectional area at approximately 12 and 24 months after the first cross-section

Figure 3.9 2003 Invertebrate ordination with overlays for low-order headwater streams of North Central Mississippi. Ordinations rotated 320°.



a. Harvest Treatment groups overlay. Ellipses represent groups of smz treatments

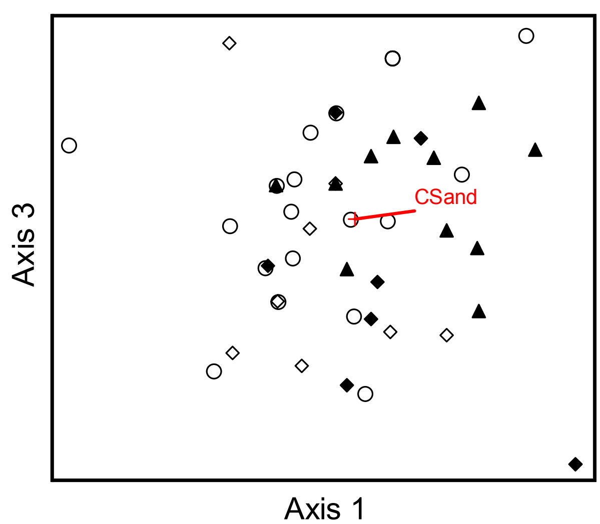
○ No-SMZ
▲ SMZ
◆ Reference



b. Habitat groups overlay. Vectors indicate variable correlated to the axis it aligns to.

Csand = coarse sand
Rich = taxa richness

○ Group 1
◆ Group 2
▲ Group 3

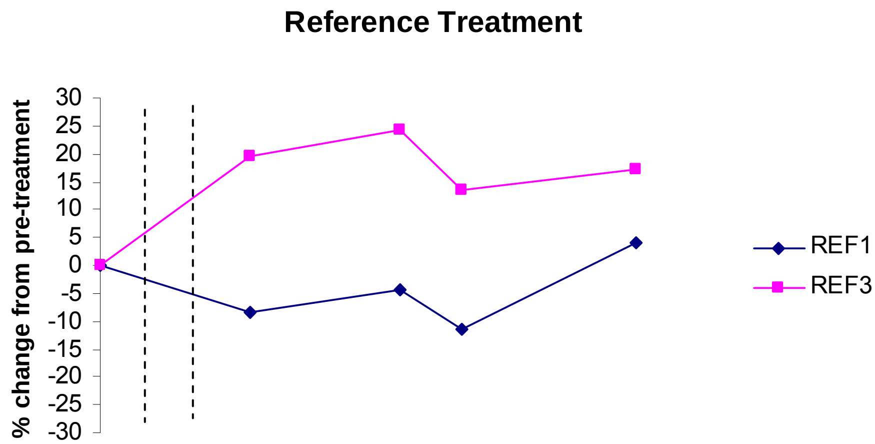


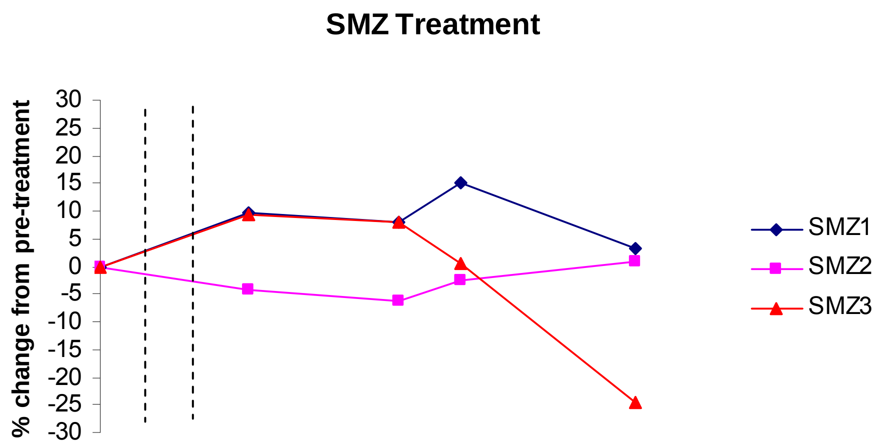
c. Sediment groups overlay. Vector indicates variable correlated to the axis it aligns to.

Csand = coarse sand

◆ Gravel
▲ Coarse Sand
◇ Fine Sand
○ Silt/Clay

Figure 3.10. Relative change from pre-harvest channel cross-section area from November 1999 through June 2003 for downstream reaches for low-order headwater streams of North Central Mississippi. Positive numbers indicate aggradation, or a loss of the channel area, negative numbers indicate degradation, or gain in channel area.





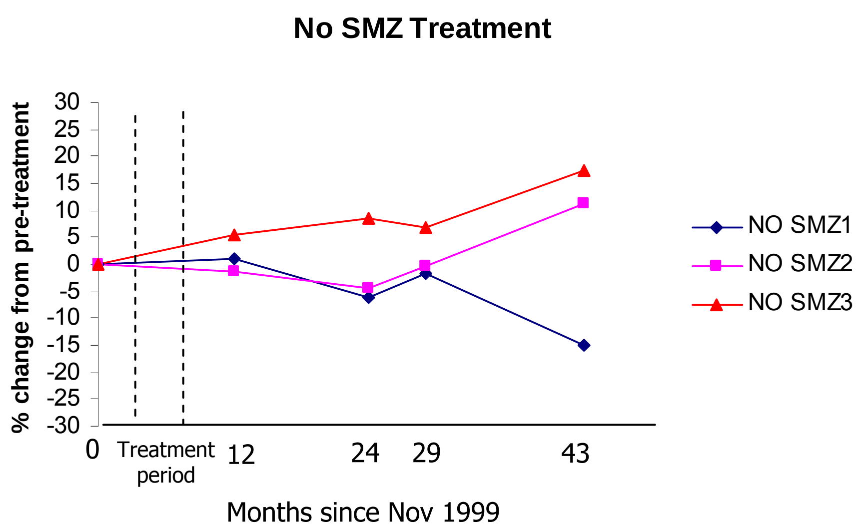


Table 3.8 a. Effects of SMZ treatments on percent change of stream channel cross sections for each time period in low-order headwater streams of North Central Mississippi. Different letters within each column correspond to significant differences (α=0.10) between treatments for each time period.

	Months since November 1999[1]			
Treatment	12	24	29	43
Reference	5.49a	2.68a	-6.11a	-1.75a
SMZ	8.22a	-2.18a	4.99b	-2.46a
No-SMZ	1.72a	-2.17a	1.41b	4.22a

[1] Time at which first measurements taken.

measurement, which were approximately six and 18 months after the treatments were applied (Table 3.8 a). However, between 24 and 29 months after the first measurement a significant treatment effect occurred with REF streams degrading, while SMZ and No-SMZ streams aggraded (Table 3.8). Thereafter, treatment effects on changes in cross-sectional area were not significant.

Intervals of mean aggradation and degradation were observed for all three SMZ treatments (Table 3.8 b). Significant mean degradation occurred in REF streams between 24 and 29 month measurements, whereas significant mean aggradation occurred during this interval in SMZ streams. Significant changes in mean cross-sectional areas between measurement periods were not observed in No-SMZ streams.

Table 3.8 b. Effects of SMZ treatments on percent change of stream channel cross sections among time periods. Different letters within each column correspond to significant differences (α=0.10) between dates for each SMZ treatment.

Months since November 1999[1]	Reference	SMZ	No-SMZ
12	5.49ab	8.22a	1.72a
24	2.68a	-2.18b	-2.17a
29	-6.11b	4.99a	1.41a
43	-1.75ab	-2.46ab	4.22a

[1] Time at which first measurements taken.

4. Discussion

One of the goals of this project was to sample aquatic macroinvertebrates within streams across a gradient of habitat and substrate compositions throughout a relatively small area in North Central Mississippi on industrial timberland sites. This objective was accomplished and the wide range of values for measured habitat and water quality parameters is evidence of this (Table 3.1, 3.2, 3.3). These low-order, often sandy-bottomed streams provided a wide variety of relatively unexplored conditions to study aquatic macroinvertebrate-habitat relationships, although the surrounding land use (i.e. industrial timberlands) was similar. The wide range of site conditions allowed me to determine if there were any area-wide relationships among attributes of the streams or between the invertebrates and the measured habitat conditions in which they live. Even in unharvested REF streams there were notable differences between variables measured only a few hundred meters apart. For example, mean taxa richness for REF sites in 2002 differed by 3.3 taxa between upstream and downstream sites (Table 3.5). This was the greatest difference between upstream and downstream taxa richness for any treatment during the two years of sampling (Table 3.5).

4.1. Habitat and Substrate Composition

The streams studied over the course of this project were surprisingly heterogeneous with regard to measured habitat and substrate, which contributed to the wide distribution within the data collected for water quality and habitat. Standard deviations were included with the means of each variable to illustrate the variance of the data. For example from 2003 data, the mean conductivity for the downstream SMZ treated streams was 67.8 µs/cm with a standard deviation of 85.3 µs/cm; but excluding SMZ 1, a site with very high conductivity, the mean is 30.0 µs/cm with a standard deviation of 12.0 µs/cm (Table 3.2). Means alone do not seem to fully describe data from these small streams because of relatively high inherent variation which may result from differences in geology and other landscape features including stream type and stream morphology as well as long-term effects of previous land uses, which are likely to have varied in timing and practice among the small basins I studied.

Substrate composition was also highly variable and not consistent within separate SMZ treatments (Table 3.3, Figure 3.1). This high degree of variability within and among SMZ treatments did not reveal statistically significant treatment effects on relative percentages of gravel, large and small sand grains, or silt/clay sized particles in downstream locations among the three SMZ treatments . Treatment differences would have to be very large for the ANOVA to detect significant treatment effects due to the wide range of data and large standard deviations. Among all study streams, the dominant substrate was sand-sized particles, with the most variability occurring in the gravel-sized fraction. Influxes of fine sediment to streambed substrate in response to logging has often been reported (Beschta 1978, Lewis 1998). Results of my study show no detectible increases in fine sediment relative to REF streams for either SMZ or No-SMZ treatments. Marion and Ursic (1993) reported that in southern forest streams, most of the sediment delivered to the streams came from erosion in minor channels developed during former land uses and not from the forested slopes.

The high natural variability of the sites may outweigh actual treatment effects on the physical habitat and render statistical differences undetectable. It is also possible that when I sampled 3-5 years after harvest, the effects of harvest had been mediated by natural processes and no real treatment effect on measured parameters exists. Both scenarios have been reported from other studies in the Southeastern U.S. on small, sandy streams in harvested sites (Marion and Ursic 1993, Young 2002, Vowell 2001).

4.2. Invertebrates

The most abundant taxa, the family Chironomidae, comprised 65.1 and 69.6% (2002, 2003, respectively) of the total numbers of invertebrates counted (Table 3.4). Overall, the distribution of invertebrates was clumped over the four most abundant taxa, which may help explain the weak relationships between the relative percent of invertebrate taxa found in each core and the measured habitat variables and substrate relative percentages (Table 3.6, 3.7). Even though measurements of habitat and water quality varied widely across the sites, there was a consistent pattern of Chironomidae numerically dominating the taxa found in the streambed cores.

The relatively high abundance of Chironomidae likely suppressed interpretable patterns of invertebrate associations with either measured habitat variables or substrate relative percentages. One solution for this issue might have been to identify all invertebrates to lower taxonomic levels, which includes taking the individuals in the family Chironomidae to the genus level of identification. This option was outside the time, financial, and expertise limits of this project but should be considered for future studies in small, warmwater sandy streams in the Southeast.

4.3. Invertebrate-Habitat Relationships

Many significant linear relationships were found between invertebrates and the habitat in which they live, but very few relationships had appreciable predictive power because of the often low correlation coefficients. Even in the case of the highest correlation coefficient for the relationship of Sialidae to the relative percent of silt/clay sized particles, the predictive power is misleading. The correlation coefficient value is due to almost all sites being devoid of Sialidae, while the one site with this family was silt/clay dominated. The streambed cores taken from this site had a large proportion of organic material including roots and sticks which were not quantified in my laboratory analyses and may also contribute to the presence of the family Sialidae. Other invertebrate-habitat relations may have been better explained by a more thorough description of the substrate.

Because of the lack of strong relationships between measured habitat variables and invertebrates, cluster analysis and ordinations were explored as alternatives to detect presence of patterns among the invertebrate samples.

Cluster analysis results did not place harvest treatments into separate groups for either year (Figure 3.6 a, b). Invertebrate ordinations for 2003 showed a moderate separation based on presence of invertebrate taxa between REF and No-SMZ treatments along the first axis of the ordination (Figure 3.8 a). The cause of this separation is unknown because no measured or calculated variables correlated to this axis. This suggests that treatment effects may linger for several years after the harvest, but may only be expressed in the biotic community and not in physical stream attributes that were selected for this study. Ordination plots revealed little more information, except for the presence of weakly detectable relationships between the first axis and coarse sand, and between taxa richness and axis three in 2003 (Figure 3.8).

The separation between harvest types on the ordination described above was not observed for the 2002 data and there were no correlations between the axes and measured habitat and substrate variables (Figure 3.7). Laboratory methods for the two years were different and may contribute to the lack of detectable patterns in 2002. Furthermore, use of presence/absence data because of the prevalence of Chironomidae means that taxa representing only a small proportion of the total are given equal weight as the most abundant taxa. Invertebrate samples from 2002 were subsampled extensively, not randomly, and we assume that some taxa were not represented in the final counts. It is difficult to speculate as to why so few relationships exist and why those that are detected (Tables 3.6, 3.7, Figure 3.8) have such limited explanatory power. Results of the ordination of 2003 samples suggest that the invertebrate distribution has detectable structure, but unmeasured environmental variables are likely to drive the distribution.

Mauger (2001) had similar problems in detecting which environmental factors controlled the patterns of invertebrate distribution in springs in southern Oregon. Gradients of the measured environmental variables correlated to the axes of the ordination plots, but groups of sites with similar habitats did not group together on the invertebrate ordination as expected. She found that topographic sub-basin was the most influential factor with regard to the invertebrate distribution in her springs, which was originally an unmeasured environmental variable.

Organic material of various sizes and distributions was shown to exert control over the macroinvertebrate populations in sandy-substrate-dominated streams (Kedzierski and Smock 2001, Metzler and Smock 1989, Strommer and Smock 1989). The organic debris dams, amount of surface organic matter, aquatic macrophytes, and subsurface organic matter in a stream all provide food and habitat for macroinvertebrates (Batzer et al. 2000, Smock et al. 1989, Strommmer and Smock 1989). In the absence of rocks, cobbles, and boulders, the organic material may also provide refuge for macroinvertebrates during spates (Smock et al. 1989). A sandy stream can be thought of as having three major storage units for organic material: the surface, the subsurface, and the floodplain. The relative amounts of material stored in each compartment stay in a dynamic equilibrium unless disrupted by a major scour event (Metzler and Smock 1989, Smock et al. 1989). As part of my sediment analysis, I attempted to quantify the amount of organic matter contained in each streambed core, but my methods were ineffective. This unmeasured habitat variable may have added to the explanation of the patterns in the data and the ordinations.

4.4. Stream Channel Cross-Sections

Because increased erosion, sedimentation, and subsequent changes in stream channel morphology are often associated with logging activities, particularly if best management practices are not used (e.g., Keim and Schoenholtz 1999, Prud'homme and Greis 2002), I examined stream channel cross-sections in the interest of detecting potential SMZ treatment effects on a larger scale than individual sediment cores. Some of the most notable differences among the three SMZ treatments were between the trends of stream channel aggradation and degradation measured at intervals of approximately 12, 24, 29, and 43 months after harvesting treatments were established. For each period of sampling, changes in cross-sectional area were not significantly different among the three treatments except for changes between month 24 and month 29 (Table 3.8 a). During this interval, the REF treatments had degraded, while the SMZ and No-SMZ treatments had aggraded sufficiently to result in significant differences between the harvested treatments and the REF treatment. This indicates that at all other intervals between sampling, streams subjected to any of the treatments were not degrading or aggrading with a significant degree of difference.

Comparing cross-section data among intervals within each harvest treatment revealed different trends than were observed above. REF and SMZ treatments changed significantly between time periods, whereas the No-SMZ streams had similar amounts of cross section change throughout the study (Table 3.8 b). This suggests that the REF and SMZ treatments may have had less stable channel configurations as exhibited by larger changes in aggradation and degradation between each time period. The REF streams were expected to be the most stable, thereby having the least change in channel cross-section, and thus provide a benchmark for comparison with the other treatments. However, these streams had similar degrees of channel change when compared to the streams adjacent to harvesting. As observed in results from stream substrate and invertebrates, lack of detectable treatment effect on stream channel may be related to 1) the actual absence of a treatment effect, 2) the relatively small consequences of the harvesting treatments on stream channels relative to inherent heterogeneity caused by natural variability among streams, or 3) variability remaining from historical land use in the region.

4.5. Summary and Conclusions

Although the high natural variability in measured habitat parameters within and among streams observed in this study created conditions in which SMZ treatment effects were generally not detectable, this same inherent gradient of habitat conditions was expected to provide a range of habitats within which to explore key factors contributing to the composition of aquatic macroinvertebrate communities in these small headwater streams of north central Mississippi. Results of Young (2002) and Carroll et al. (2004) corroborate the high degree of heterogeneity in water quality and biotic communities within these streams, and, in general, report few detectable effects of harvesting treatments on measured parameters. All of the streams in this study are linked to a past characterized by repeated human disturbance (e.g. clearing, farming, logging), which makes comparison of treated sites to undisturbed reference sites nearly impossible (Clouse 2003). Contributions of historical land use practices to high variability in ecological data must be considered in evaluations of contemporary management practices (Marion and Ursic 1993, Harding et al. 1998). For example, Harding et al. (1998) found that watershed use in the 1950s was the best predictor of fish and invertebrate diversity found in 24 catchments in western North Carolina. Streams in catchments with a high amount of agriculture in the 1950s, which were forested in the 1990s, had bioassessment scores more similar to catchments that were currently in agriculture. In a similar manner, assessment of the historic site conditions in north central Mississippi may also help explain the patterns found in the ordinations from my study.

The underlying goal of this project was to test the predictive ability of streambed sediment composition for estimating aquatic macroinvertebrate abundance and/or composition in the interest of contributing to more rapid and cost-effective methodologies to assess biotic conditions in small, seasonally warm, headwater streams. Results of my evaluations of relationships between sediment composition (as well as a limited number of other selected habitat variables) and invertebrate composition using simple linear regression and ordination techniques show that sediment composition may be a poor predictor of invertebrate communities in these streams as determined by my methodology. Several explanations for these results should be considered. First, invertebrate communities in these warmwater streams are dominated by Chironomidae. Further taxonomic resolution within this family could provide previously undiscovered patterns of distribution in relation to sediment composition. Second, sampling was restricted to warm summer conditions in June and July. Aquatic macroinvertebrate communities are dynamic and thus could potentially show considerable seasonal variation that could elucidate previously undetectable relationships. Third, there are a myriad of other habitat variables such as organic matter and detritus that were not measured in this study that may have stronger influences on invertebrate communities than sediment in the substrate. Legacies of previous land use might also be considered if information is available because of potential contributions to habitat conditions, particularly as these streams rework agriculturally derived sediments deposited in channels as a consequence of row-crop agriculture.

This study provides an initial exploration of sediment-invertebrate relationships in small warmwater streams in North Central Mississipi. Although strong relationships were not observed, the study was able to document that forest harvesting, either with or without SMZs did not create detectable shifts in stream sediment substrate, channel cross-sectional areas, or invertebrate taxa richness and diversity when compared to reference streams that were not subjected to forest harvesting. These response metrics were highly variable within and between treatment streams, suggesting that (1) treatment effects would have to be very large for statistical detection, and (2) high degrees of inherent variability in small headwater streams must be considered for monitoring and evaluative purposes when assessing effects of land use on aquatic resources.

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