RELATIONSHIPS BETWEEN STREAMSIDE MANAGEMENT ZONE WIDTH AND BIOTIC COMMUNITIES OF HEADWATER STREAMS IN WEST VIRGINIA

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ABSTRACT

The importance of streamside management zones (SMZ) in minimizing the impact of non-point source pollution from silvicultural operations is recognized by the forestry Best Management Practices of most states. However, research concerning the SMZ width and harvesting intensity required to maintain water quality and biotic communities is limited. The goal of this study is to evaluate the efficacy of different SMZ widths and forest harvesting intensities within SMZs, in maintaining the water quality and biotic communities of 22 headwater streams located in the mountains of East-central West Virginia. Streams were organized in four blocks and randomly assigned one of six silvicultural treatments involving variation of SMZ width and harvesting intensity within the SMZ; 30.5 m SMZ with no residual harvest, 30.5 m SMZ with 50% residual harvest, 15.3 m SMZ with no residual harvest, 15.3 m SMZ with a 50% residual harvest, 4.5 m SMZ and control (no harvest within the watershed). Stream water chemistry parameters (in particular, NO_3, NH_4, Ca, Mg, conductivity and total dissolved solids) as well as aquatic macroinvertebrate communities were monitored from June 2003 through March 2005. Average nitrate concentration in streams harvested with a 4.5 m SMZ was more than 4 times as high as that of control streams. Average summer and fall stream temperatures were inversely related to SMZ width. Mean values for a number of macroinvertebrate community metrics were indicative of poorer water quality in

streams harvested with a 4.5 m SMZ. During this short-term study SMZs of at least 15.3 m appeared to be sufficient to maintain water quality. However, harvesting was restricted to one side of the stream and logging induced stream disturbances were observed even with SMZs of 30.5 m. For these reasons SMZs of at least 30.5 m are recommended as a cautionary measure to minimize the potential for impacts to biotic communities. In addition, residual harvest of up to 50% of the basal area within the SMZ did not appear to impact water quality during the temporal scope of the study.

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Chapter I. Introduction

Rationale

Silvicultural practices such as road building, harvesting, log skidding and site preparation have the potential to degrade water quality. The potential for nonpoint source pollution from forestry activities was addressed in section 208 of the Clean Water Act (PL 92-500) and section 319 of the 1987 amendments to the Clean Water Act (PL 100-1). Clean Water Act legislation required states to develop water quality management plans and identify forestry best management practices (BMPs) to control nonpoint source pollution from silvicultural practices. BMPs are minimum requirements and specifications regulating silviculture and are designed to reduce erosion and prevent or control water pollution (Virginia Department of Forestry, 1997).

An important element of forestry BMPs is closely managing streams and the areas directly adjacent to them. BMPs of this type are often referred to as streamside management zones (SMZs). Riparian buffer strips, filter strips and riparian management zones are all SMZ correlates. A SMZ can be defined as a designated area that consists of the stream itself and an adjacent area of varying width where management that might affect water quality and aquatic habitat is modified (Phillips et al., 2000). All state BMP programs recognize the importance of SMZs and usually specify the width of the SMZ and harvesting intensity that can occur within the SMZ.

Forested areas directly adjacent to streams, termed riparian zones, maintain many important physical, biological and ecological functions and

important social benefits (Klapproth and Johnson, 2000a). Riparian forests have been shown to mitigate many of the impacts associated with silvicultural practices. Forested riparian areas reduce sediment inputs to streams by slowing runoff, which allows sediments to settle out before entering the stream. Nutrients attached to the retained sediment particles (Klapproth and Johnson, 2000b), are subsequently taken up by the vegetation are prevented from entering the stream. Leaving a forested SMZ helps maintain a streams preharvest flow regime (Kochenderfer et al., 1997) as a result of the vegetation slowing runoff and providing for evapotranspiration, reducing average and peak stream discharge. Maintenance of a forest canopy to shade streams minimizes stream temperature fluctuations (Patric, 1980; Sweeney, 1993).

Degraded water quality resulting from silvicultural practices has the potential to impact stream biota (Gregory et al., 1987). Macroinvertebrate community structure is intricately tied to water quality and stream habitat and therefore, macroinvertebrates are sensitive to silviculture and other activities that disturb the riparian area (Vowell, 2001). SMZs help to mitigate the chemical and physical impacts of silvicultural operations on streams. Indeed, the presence or absence of riparian forest may be the single most important human altered factor affecting the structure and function of stream macroinvertebrate communities (Sweeney, 1993).

Many studies regarding the effectiveness of BMPs and SMZs in maintaining pre-harvest water quality have been conducted (Phillips, 1989; Lynch and Corbett, 1990; Kochenderfer et al., 1997; Arthur et al., 1998; Wynn et al.,

2000; Vowel, 2001). However, surprisingly few studies investigating the efficacy of BMP programs in limiting biotic change in streams have been published (Hutchens et al., in press). Debate also centers around the width of SMZs and the harvest intensity that should be allowed within SMZs (Phillips et al., 2000). Much of this debate stems from the fact that SMZ requirements typically have been established by political expedience, rather than scientific merit (Castelle et al., 1994). SMZ dimensions should be adequate to protect water quality and stream biota, but should not be excessive as to exacerbate economic loss for landowners and the forest industry. The present study is part of a larger, long-term, investigation of physical, chemical, and biological impacts on streams, resulting from forest harvest with different SMZ widths and harvesting intensities within the SMZs. In addition to the study streams located in the mountains of Randolph Co., WV a similar number of sites are located in the piedmont near Appomattox, VA. West Virginia sites are characterized by Northern hardwoods while Piedmont sites are located on loblolly pine *Pinus taeda* plantations. The present study focuses on the sites in WV and aims to elucidate scientific knowledge of SMZ widths and harvesting intensities required to minimize the short term impacts of silvicultural operations on headwater streams in the Central Appalachians.

Objectives

The goal of this study is to evaluate the efficacy of different SMZ widths, and harvesting intensities within the SMZs, in maintaining the water quality and biotic communities of 22 headwater streams located in the mountains of East-

central West Virginia. The study objectives include an investigation of the impact of different SMZ widths, and harvesting intensities within the SMZs, on macroinvertebrate community structure, and to monitor changes to physical and chemical stream parameters resulting from logging with variable SMZ width and harvesting intensity that may impact stream biota.

Chapter II. Literature Review

Riparian Areas

During the past two decades, scientists around the world have begun to recognize the important role that riparian areas play in maintaining water quality (Klapproth and Johnson, 2000a). This sentiment is confirmed by the explosion in literature examining the structure and function of riparian communities and the value of riparian areas in water quality protection (Pert, 2000). Whether the focus is forest preservation/protection or anthropogenic value, healthy riparian forests are vital as filters and nutrient attenuators to protect water quality for drinking, fisheries and recreation (Archey, 2000).

Quality of stream water is dependent on a variety of parameters, including sediment loads, organic compounds, inorganic compounds, and temperature. Silvicultural practices such as road building, harvesting, log skidding and site preparation have the potential to degrade water quality and aquatic habitat through increased sedimentation, nutrient inputs, altered hydrology and increased stream temperature fluctuations (Golden et al., 1984; Binkley and Brown, 1993; Phillips et al., 2000; Wynn et al., 2000). Forestry BMPs have been developed to prevent or minimize water quality impacts from silvicultural activities while permitting intended forest management to occur (Phillips et al., 2000). All state BMP programs recognize the importance of retaining a SMZ in which management is modified to protect riparian habitat and water quality. Maintaining healthy riparian areas through the use of SMZs can mitigate water quality impacts from silvicultural operations.

Forestry Effects on Water Quality

Silvicultural practices such as road building, harvesting, log skidding and site preparation have the potential to degrade water quality and aquatic habitat. Primary concerns related to forestry activities include increased sedimentation, altered hydrology, nutrient inputs, and increased stream temperature fluctuations (Golden et al., 1984; Binkley and Brown, 1993; Phillips et al., 2000; Wynn et al., 2000).

Sediment

Sediment is considered to be the greatest water quality concern related to forestry (Binkley and Brown, 1993) and the relationship between sediment and forestry practices has been widely discussed in past publications (Trimble and Sartz, 1957; Patric, 1976; Patric, 1978; Yoho, 1980; Patric et al., 1984; Everest et al., 1987; Meehan, 1991; Waters, 1995). Mature forest provides the maximum protection against erosion (Patric, 1978), however the process of erosion occurs naturally even in old growth forest (Waters, 1995). Eroded soil particles become sediment once they enter a stream or other water body. Since overland flow is rare in an undisturbed forest most sediment results from channel erosion (Yoho, 1980; Waters, 1995) or natural soil disturbances such as windthrow mounds and landslides. Suspended sediment concentrations often increase following forest management activities such as road construction, harvest and site preparation, however BMPs may effectively prevent this increase (Binkley and Brown, 1993). Highest rates of erosion usually occur on logging roads and skid trails where soil

compaction and disturbance lead to overland flow (Trimble and Sartz, 1957; Patric, 1978; Waters, 1995) or where the stream bank or channel is directly disturbed by equipment (Yoho, 1980). Patric (1978) suggested that minimizing disturbance in the vicinity of the stream channel is the key to keeping levels of soil loss close to natural geological rates and maintained that soil eroded from roads would not reach streams that were more than 100 feet away. SMZs can reduce erosion and subsequent sedimentation by serving as a barrier to the transport of sediment in overland flow, stabilizing stream banks and promoting infiltration.

Studies investigating forestry induced sedimentation are numerous and the results variable. Most studies found increased sediment yields following harvest, however SMZs of 50 to 100 ft. in width were effective in minimizing sediment yields in several cases. Martin and Hornbeck (1994) reported 10 to 30 fold increases in sediment yields from clear-cut watersheds of the Hubbard Brook Experimental Forest in New Hampshire. Wynn et al. (2000) determined that average yearly sediment yield from a Coastal Plain watershed harvested without BMPs (no SMZ) increased 2.6 times following harvest, while sediment yields from a watershed harvested using BMPs including a 50 ft SMZ did not increase post-harvest. Similarly, Keim and Schoenholtz (1999) studied effects of harvesting in highly erosive loessal bluff forests and found that streams harvested without a SMZ had three times the sediment concentrations of streams harvested with a 30 m SMZ. On the other hand, Arthur et al. (1998) found that suspended sediment was 30 times greater than that of a control watershed for a

watershed cut without BMPs, and 14 times greater for a watershed cut with BMPs (50 ft SMZ). Lynch and Corbett (1990) also found suspended sediment and turbidity increases following a clear-cut with a 100 ft. SMZ and these increases were still evident 12 years post harvest. Swank et al. (2001) determined that large increases in sediment yield following road construction prior to harvest of a southern Appalachian watershed led to a 50% increase in sediment yield at the base of the catchment that lasted for at least 15 years post harvest. This illustrates the long-term effects that a pulse disturbance can have on a stream. However, most studies have shown that logging induced increases in sediment and turbidity return to preharvest levels within 2 to 6 years (Patric, 1980; Lynch and Corbett, 1990; Martin and Hornbeck, 1994; Kochenderfer et al., 1997; Arthur et al., 1998).

Hydrology

Removal of trees reduces evapotranspiration and results in corresponding gains in stream volume (Aubertin and Patric, 1974; Patric, 1978). Streamflow increases are approximately proportional to the severity and scale of forest cutting (Patric, 1978). Increased streamflow can exacerbate channel erosion and lead to a subsequent increase in sediment yields. Streamflow and water yield increases are generally short lived, returning to post harvest levels after 1-4 years (Lynch and Corbett, 1990; Hornbeck et al., 1997; Martin et al., 2000; Swank et al., 2001).

Due to the relationship between tree removal and amplified water yield SMZs may not be able to preclude increased streamflow, but may have protective effects by stabilizing stream banks and preventing an increase in channel erosion. Most studies support this and report augmented water yields irrespective of SMZ presence. Arthur et al. (1998) found stream base flow increases of 123% and 138% for watersheds cut with BMPs (50 ft SMZ) and without BMPs, respectively. Lynch and Corbett (1990) reported a significant increase in water yield three out of the first four years following a Pennsylvania clear-cut in which a 100 ft buffer was employed. Patrick (1980) reported a 38% increase in water yield following a clear-cut with a 20 m SMZ, however stormflow did not increase. Similarly, Swank et al. (2001) reported a 28% increase in streamflow for a southern Appalachian stream following a clear-cut without BMPs.

Conversely, Martin et al. (2000) found that a progressive strip cut, in which a clear cut was administered in thirds and a 15-25 m SMZ employed, increased water yield by only 4-9% during and for the first 3 years following harvest, while whole block clear cutting without an SMZ resulted in a 23% increase in water yield the year following harvest. Whole block clear cutting increased peak flows by up to 60% 1-2 years post harvest and also advanced snowmelt and snowmelt runoff by up to 17 days due to the absence of stem and branch shade following harvest (Hornbeck et al., 1997).

Nutrients

Harvesting trees temporarily accelerates nutrient release from the forest floor, which can result in enrichment of streams (Patric, 1978). Stream nutrient levels can increase when decomposition exceeds uptake by plants or when nutrients attached to eroded soil particles are washed into a stream. SMZs remove excess nutrients by filtering runoff water and providing for subsequent uptake of nutrients by plants (Castelle et al., 1994). The degree to which measured nutrient concentrations are affected by forestry operations is influenced by any alterations in streamflow since the concentration is based on the proportion of a nutrient to the amount of water. Excess nutrient loss following logging is not only a water quality concern but may have silvicultural implications due to loss of site productivity (Sopper, 1975).

Many studies have observed increases in nutrient concentrations following forest harvest. Nitrate levels seem particularly susceptible to augmentation following silvicultural operations. However, most nitrate increases are slight and remain well within drinking water standards (Binkley and Brown, 1993). Aubertin and Patric (1974) found increased nitrate levels following a West Virginia clear-cut that employed a 10 to 20 m SMZ. Wynn et al. (2000) reported that harvesting and site preparation without the use of BMPs lead to nitrogen yield values that were two to three times pre-harvest levels. Wynn et al. also found that average post harvest total phosphorus levels increased three times when no BMPs were used, but remained unchanged when BMPs including a 50 ft SMZ were employed. These increases in total phosphorus were tied to sedimentation in

that much of the phosphorus entering the stream was sediment-bound. Lynch et al. (1985) reported much higher nitrate and potassium levels from a watershed clear-cut without a SMZ as opposed to a watershed cut using BMPs including a 30 m SMZ.

Stream Temperature

High stream temperatures can negatively affect aquatic organisms (Sopper, 1975; Beschta et al., 1987; Kochenderfer et al., 1997). Clearing of stream channels increases summer stream temperature until re-growth restores shading (Patric, 1978, Beschta et al., 1987). Stream temperature increases are the result of solar radiation reaching the stream surface and can be minimized by leaving a protective buffer strip along the channel (Beschta et al., 1987, Lynch and Corbett, 1990). Energy gained from streamside canopy removal is not readily dissipated even if the stream reenters a shaded corridor. Thus, increases in temperature of small headwater streams can lead to additive effects on the temperatures of higher order streams (Beschta et al., 1987). Also, stream temperatures during the winter months may decrease when streamside vegetation is removed leaving no canopy to prevent energy losses from the stream (Beschta et al., 1987).

Studies show that SMZs of 10-30 m can be effective in moderating stream temperatures, whereas harvesting without a buffer or with a very narrow buffer often leads to considerable stream temperature alterations. Lynch et al. (1985) reported that a 30 m (98 ft.) buffer maintained water temperatures within one 1

°C of pre-harvest values, whereas harvesting without a buffer led to an average monthly maximum temperature increase of 4.4 °C. Lynch et al. (1985) also showed the importance of an SMZ in moderating winter extremes as winter temperatures were as much as 4 °C cooler for the stream cut without a SMZ. Patric (1980) found that a 20 m SMZ was sufficient to prevent an increase in stream temperature, but subsequent removal of the SMZ resulted in stream temperature increases of almost 8 °C. Similarly, Aubertin and Patric (1974) determined that a 10 to 20 m SMZ prevented an increase in stream temperature following a clear-cut of a West Virginia watershed in the Fernow Experimental Forest. Conversely, Hewlett and Fortson (1982) found that a 20 ft buffer on each side of a clear-cut stream in Georgia was not sufficient to moderate temperatures as both summer and winter temperatures were far more extreme than those of a control stream.

Effects on Invertebrates

Logging can impact aquatic communities, especially macroinvertebrates, by altering water chemistry, energy sources, stream habitat, and temperature. Changes in stream nutrient concentration can alter the diversity and productivity of stream organisms (Wagner, 2001). Macroinvertebrate taxa have varying tolerances to water chemistry parameters such as suspended solids, total nitrogen, and reduced dissolved oxygen (Hilsenhoff, 1977). Thus, logging induced changes in water chemistry may alter macroinvertebrate communities.

Logging may affect stream biota by changing the streams energy base from one driven by leaf litter inputs, allochthonous, to one driven by primary production within the stream, autochthonous. Increases in primary production following streamside vegetation removal are most pronounced in small headwater streams (Gregory et al., 1987). Forest harvest also alters the quantity and quality of allochthonous inputs, which results in subsequent shifts in the density and functional feeding group structure of the macroinvertebrate community (Silsbee and Larson, 1983; Golladay et al., 1989; Griffith and Perry, 1991; Stout et al., 1993). Stone and Wallace (1998) reported that leaf litter inputs to a Southern Appalachian stream were only 1.6% of pre-disturbance levels during the first year following a clear-cut without a buffer. Wagner (2001) found that Southern Appalachian streams in virgin forests had nearly 400 times the woody debris of streams that had been logged 75 years previous to the study. However, total quantity of organic matter alone does not control stream productivity, as nutritional quality of the organic matter controls how fast energy can be extracted by the aquatic community (Murphy and Meehan, 1991).

Poorly conducted silvicultural operations may also degrade aquatic habitat (Gregory et al., 1987). Murphy et al. (1986) found that habitat in buffered Alaskan streams did not differ from that of old growth, whereas unbuffered streams in clear-cuts had lower channel stability and less canopy cover, pool volume, large woody debris, and undercut banks. Increased stream temperature extremes following deforestation also impact macroinvertebrate communities by

greatly altering important life history characteristics including growth rate, survivorship, fecundity, and timing of reproduction (Sweeney, 1993).

Sponseller et al. (2001) investigated the impact of different spatial scales on physico-chemical features and macroinvertebrate communities of Virginia headwater streams. They found that macroinvertebrate communities were most closely related to land-cover patterns within 200 m of the stream suggesting that riparian forest may have an ameliorative effect on watershed disturbance. The relationship between stream communities and the disturbance of the riparian zone is a strong argument for the use of SMZs. However, further investigation is needed to establish the required width of these specialized management zones.

Many studies have reported increased macroinvertebrate densities, biomass or production following clear-cut logging operations (Newbold et al., 1980; Hawkins et al., 1982; Silsbee and Larson, 1983; Duncan and Brusven, 1985; Noel et al., 1986; Carlson et al., 1990; Arthur et al., 1997; Stone and Wallace, 1998; Kedzierski and Smock, 2001). These increases in invertebrate abundance are most often attributed to increased primary production following the removal of riparian trees, although increased stream temperature and nutrient loading have also been suggested. Increases in invertebrate densities or biomass are often driven by a few taxa that commonly experience population explosions following logging; one of these taxa is the mayfly genus *Baetis* (Newbold et al., 1980; Noel et al., 1986; Wallace and Gurtz, 1986).

Other research has concluded that logging has little or no effect on macroinvertebrates. Stone and Wallace (1998) reported that clear-cutting

caused shifts in the predominance of invertebrate taxa, but few taxa were lost from the community. Boschung and O'Neil (1981) also found that clear-cutting in a steep Alabama watershed did not affect stream macroinvertebrate diversity or richness. Similarly, Williams et al. (2002) concluded that logging did not impact macroinvertebrate communities of streams in the Ouachita Mountains of Arkansas.

Conversely, some research shows that forestry may have long lasting influence on stream invertebrate communities. Growns and Davis (1991) reported differences in the macroinvertebrate community composition of clear-cut streams that were evident eight years post harvest. Silsbee and Larson (1983) reported differences in macroinvertebrate density, richness and functional feeding groups between logged and undisturbed catchments that were evident more than 50 years after clear-cut logging. Hawkins et al. (1982) reported that streams with an open canopy showed a greater degree of taxa dominance than shaded streams, however uncommon and rare taxa were unaffected by canopy status.

Several studies have investigated the impact of logging with SMZs on stream macroinvertebrates. Newbold et al. (1980) determined that a 30 m SMZ prevented any measurable effects of logging on the macrobenthos of California streams, conversely streams logged without a buffer had reduced diversity and altered species composition. Buffers of less than 30 m were also studied and the authors concluded that they were not fully effective in preventing logging effects. A similar study by Davies and Nelson (1994) found that 30 m buffers prevented

logging impacts on Australian macroinvertebrate communities, while streams with buffers of less than 30 m had reduced abundance of stoneflies and some mayflies. Both Newbold et al. (1980) and Davies and Nelson (1994) determined that logging effects were determined by riparian buffer width and were not significantly influenced by watershed characteristics such as slope or soil type. Vowell (2001) found 10-60 m SMZs were sufficient to maintain pre-harvest biotic index values for Florida streams. Noel et al. (1986) found that 8-9 m buffers were not sufficient to prevent a post logging change in the invertebrate communities of New England streams. Likewise, Arthur et al. (1997) reported decreased invertebrate diversity in small intermittent streams following clear-cutting with a 10 m SMZ.

Bioassessment

Biological assessment is the systematic use of living organisms or their responses to determine the quality of the aquatic environment (Rosenberg and Resh, 1996). Benthic macroinvertebrate communities are constant integrators of physical and chemical water quality parameters, and thus are indicators of past and present environmental conditions. Aquatic macroinvertebrates are widely used in bioassessment studies for several reasons: (1) they are common in most aquatic habitats; (2) the large number of taxa exhibit a range of responses to disturbance; (3) their sedentary nature allows for the determination of the spatial extent of environmental stress; (4) their relatively long life cycles allow macroinvertebrate communities to be representative of past perturbations not just

present conditions (Rosenberg and Resh, 1996). Stream macroinvertebrates have been shown to effectively monitor water quality impacts from forestry operations (Adams et al., 1995; Vowell, 2001) and can be applied over a wide range of stream types and physiographic regions (Adams et al., 1995).

Macroinvertebrates may exhibit a range of responses to environmental stress making it difficult to settle on what constitutes a negative impact on the community. Some studies have considered that logging induced increases in macroinvertebrate biomass may be beneficial to fish communities (Duncan and Brusven, 1985; Carlson et al., 1990). However, a conservative management approach aims to maintain streams in as close to their natural state as possible (Newbold et al., 1980) and therefore, any significant difference between the macroinvertebrate communities of logged streams and those of control streams can be seen as a silvicultural impact that should be avoided.

Chapter III. Methods and Materials

Study Area

The spatial scope of the study includes 22 watersheds located in Randolph County, West Virginia, in the area classified as the Allegheny Mountain and Plateau Physiographic province (Figure III-1). Study sites are industrial forests found on four MeadWestvaco tracts: Beaver Creek, Rocky Run, Wuchner and Rich Mountain. Individual streams were selected to minimize the influence of confounding factors. All study sites are located on high gradient, first order headwater streams. Management history is similar in the small watersheds drained by the streams and there is minimal anthropogenic disturbance upstream of the study sites. The streams are organized in four blocks based on proximity, soils and geology and treatments were randomly assigned to the streams within each block.

The history of the study watersheds is heavily tied to forestry. All timber in the area was completely clearcut in 1900, with light diameter limit cutting continuing until the 1970's (Keyser et. al, 2005). The result is a forest that is multi-aged, with abundant roads, skid trails, and haul roads. Study site elevations range from 670 m to 1000 m above sea level and the watersheds are characterized by steep side slopes, broad ridges, and narrow valleys with steep streams (Keyser et. al 2005). Small (< 2.5 ha), localized slumps and landslides are common in the study watersheds (Smith, 1995).

The average air temperature in Randolph County is 10° C. The average winter temperature is –0.5° C, with an average daily minimum of –6° C, while the

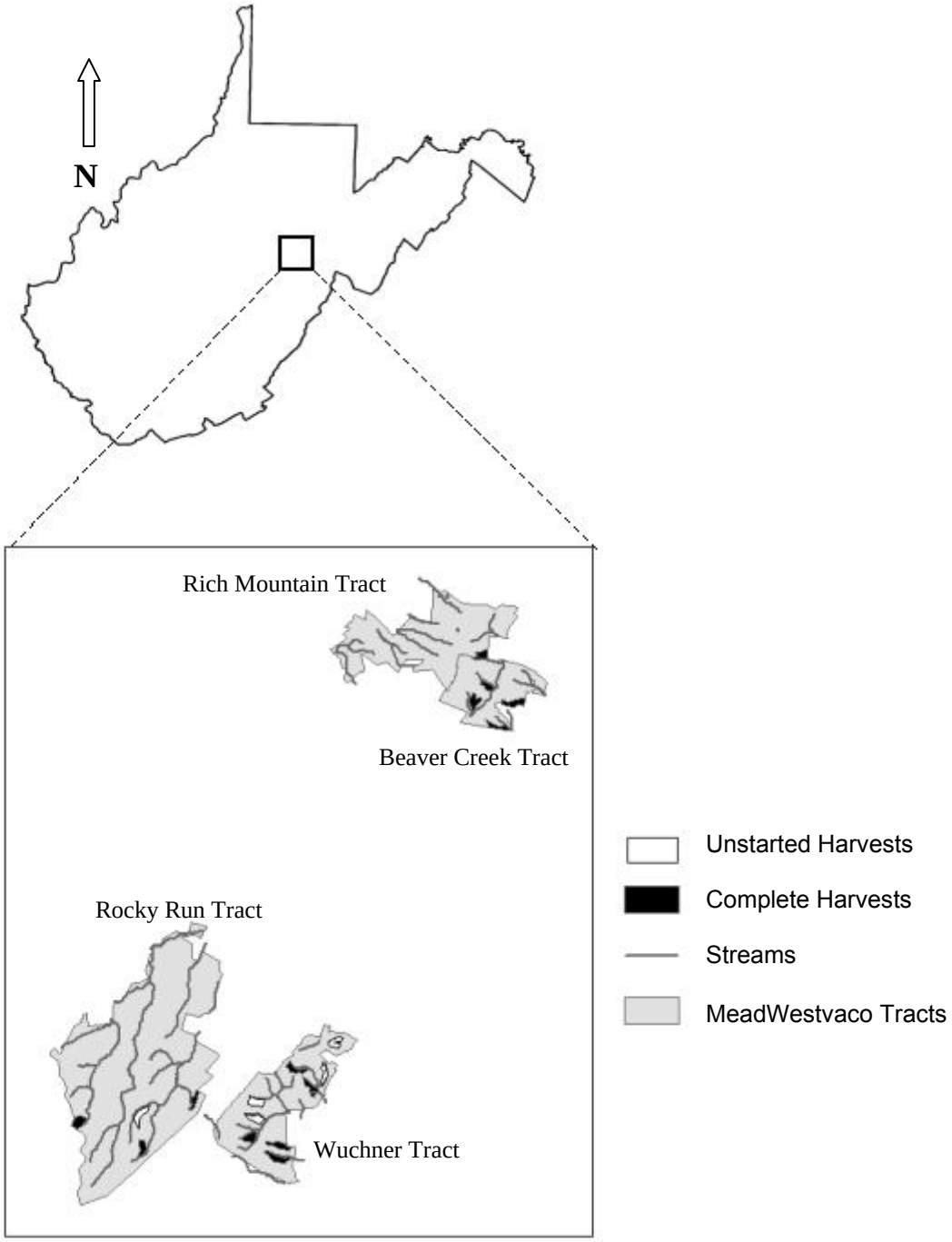


Figure III - 1. Location of study sites in Randolph County, West Virginia. Including MeadWestvaco tracts, completed and unstarted harvests, and streams.

average summer temperature is 19° C, with an average daily maximum of 26° C. In Randolph County, the daily minimum temperature is greater than freezing 118 days per year in nine out of ten years. Rainfall is evenly distributed throughout the year. Average total precipitation is 1070 mm per year. The predominant winds are from the northwest, which causes notably higher precipitation on windward facing slopes than in valleys.

To better characterize study areas, slope, watershed area, percent of watershed to be harvested, and stream length to be harvested were examined (Sharp, 2003). The average slope of the watersheds is 36%. The average watershed area is 40 ha. and the average percentage of watersheds that will be harvested is 28%. For more detailed watershed information, see Appendix A.

The study sites are located in two major soil associations: the Berks-Calvin-Weikert association and the Gilpin-Dekalb-Buchanan association (United States Department of Agriculture, Soil Conservation Service, and Forest Service, 1982). Both associations are characterized by steep, well-drained, acid soils for which the erosion risk is moderate to severe. Information regarding the taxonomic class of these soil series is available in Appendix B.

Treatments

Six treatments involving variation of SMZ width and harvesting intensity within the SMZ were applied to study watersheds, these treatments were: 30.5 m SMZ with no residual harvest, 30.5 m SMZ with a 50% residual harvest, 15.25 m SMZ with no residual harvest, 15.25 m SMZ with a 50% residual harvest, 4.5

m SMZ and control (no harvest within the watershed). Watersheds treated with a 30.5 m SMZ with no residual harvest served as reference to present West Virginia BMP recommendations (West Virginia Division of Forestry, 2001). SMZ width was measured from the center of the stream. Harvests outside the SMZ were commercial clearcuts and were carried out by loggers contracted by MeadWestvaco. Logging occurred on only one side of the stream leaving at least one half of the watershed unharvested. The original study was set up as a randomized complete block design. There were four blocks of five streams with each of the five treatments randomly assigned to one of the streams in each block. Reference streams (30.5 m SMZ) were included in two blocks. Harvesting was scheduled to take place during the spring, summer and fall of 2003, however extremely wet weather resulted in slow progress and logger labor issues and only 10 of the 18 harvests were completed during this time. Three additional harvests were completed in 2004 and one was completed in February of 2005. The topography of some of the watersheds was too extreme for the randomly assigned treatment necessitating the reassignment of some treatments based on practicability. In addition, contracted loggers applied several of the treatments incorrectly. Information on proposed and applied treatments and harvest completion is contained in Appendix C.

Field Methods

Water Chemistry

Stream water quality was monitored quarterly starting in June of 2003 and continuing until March of 2005. Grab samples were collected once each season and frozen until laboratory analysis could be performed. In addition, a Horiba U-22 multiparameter water meter was used to measure pH, conductivity, turbidity, dissolved oxygen, temperature, and total dissolved solids (TDS).

Stream water temperature was monitored via Hobo continuous temperature recording gauges. Temperature readings were recorded every thirty minutes. Water temperature was monitored from June of 2003 through March of 2005. Temperature gauges were downloaded quarterly to prevent data wrapping. High streamflow events resulted in some Hobos being washed onto the streambank. When this occurred, it was not feasible to determine the time temperature gauges were washed out of the stream. Therefore, it was assumed that the difference between streamwater and air temperature directly adjacent to the stream was negligible, and data were included for analysis (Lewis et al., 2000).

Macroinvertebrates

Aquatic Macroinvertebrate samples were collected during August of 2003 and 2004, and during March of 2004 and 2005. Samples were collected at the downstream boundary of the harvest stand on each watershed. When a road formed the downstream boundary of the stand being harvested the sampling reach began 50 m upstream of the disturbed area adjacent to the road.

Collection took place along a 50 m reach of each stream and subsamples were collected every 2 m within the reach. Subsamples were collected using a D-frame macroinvertebrate sampling net. To ensure samples were collected randomly a starting point of the left, center, or right part of the stream was selected randomly and subsequent subsamples were collected from left to right along the upstream progression. Dividing the stream width into three sampling regions was sufficient because the study streams were rarely more than 1 meter (approximately 3 D-frame net widths) wide. Each subsample was combined in a wash bucket with a 500 μm mesh bottom forming one homogenous sample. Any cobble, large gravel or large woody debris was scrubbed to retain any invertebrates clinging to these substrates and then discarded from the sample. Samples were then transferred to 1 liter plastic bottles and preserved with 75% ethyl alcohol.

Lab Methods

Water Chemistry

Grab samples were analyzed for calcium, magnesium, nitrate, and ammonium concentrations. Samples collected from June through November 2003 were sent to a MeadWestvaco Laboratory in Summerville, SC for calcium and magnesium analyses. Samples were processed using a Perkin-Elmer Optima 300 inductively coupled plasma emission spectrophotometer (ICP-OES). Calcium and magnesium analyses for samples collected from March 2004 through March 2005 were performed at the Virginia Tech Soils Testing

Laboratory using a SpectroFlame Modula Tabletop ICP with autosampler. Nitrate and ammonium analyses were processed using a Bran and Luebbe TRAACS 2000 Continuous Flow Analyzer, which utilizes a colorimetric procedure.

Macroinvertebrates

Macroinvertebrate samples were re-preserved in 95% ethyl alcohol within two days of returning to the lab. Immediately prior to sorting samples were rinsed in a 500 micron mesh net to remove the fine sediments. Samples were then transferred to a 15 x 12 white plastic pan, elutriated with approximately 2 cm of water, and sorted to completion to avoid potential biases associated with subsorting. Decapods, amphipods, isopods and aquatic insects, with the exception of individuals from the family Chironomidae, were identified to the genus level using keys found in Meritt and Cummins (1996), and Smith (2001). Chironomids were not identified beyond the family taxonomic level. All specimens were retained in labeled vials and are stored at the USDA Forest Service Center for Aquatic Technology Transfer in Blacksburg, VA.

Calculations

Stream Temperature

Stream temperature data were analyzed by season. Hobo data loggers recorded stream temperature twice per hour. Hourly averages as well as overall averages were calculated for each study stream during each season. Daily

temperature difference was calculated by subtracting the lowest hourly average for a given watershed and season from the highest hourly average for that watershed and season.

Macroinvertebrates

Raw macroinvertebrate data was used to calculate a number of community metrics. In addition to the individual metrics two multimetric indices were calculated; the West Virginia Stream Condition Index (WV SCI) and the Macroinvertebrate Aggregate Index for Streams (MAIS). These multimetric indices incorporate a number of individual metrics that are calibrated to a scale and transformed to a unitless score prior to being aggregated into the multimetric index. Multimetric indices can be more useful in assessing biological condition than individual metrics alone. A list of macroinvertebrate community metrics that were calculated from the samples, and some information about how they are calculated and the expected response to disturbance are provided in Appendix D.

Statistical Methods

Incomplete and incorrectly assigned treatments required that blocks be reconstructed for analyses. The original study design consisted of 4 blocks of 5 treatments, this design was condensed to form 2 blocks with 6 treatments. Care was given to reassign watersheds to the two new blocks based on proximity,

water chemistry and geology. In general, streams in block 1 were characterized by near neutral pH and watersheds with a west facing aspect, while streams in block 2 were highly acidic (pH between 4 and 5.5) and have a north facing aspect. Differences in stream pH were driven by the presence of the Belmont soil series, a soil with limestone parent material, found in the Wuchner and Beaver Creek Tracts, but absent from the Rocky Run Tract. A list of block assignments used in statistical analyses is included in Appendix E.

Analysis of variance (ANOVA) was used to detect statistical differences between treatments and blocks for response variables. Tukey's Highly Significant Difference was used to determine statistically significant differences between treatments for response variables (alpha = 0.05). Different sampling dates were treated as repeated measures in the ANOVA model. In some cases, linear regression was used to test for a trend between SMZ width and a given parameter. JMP Start Statistics (Sall et al., 2001) a program by the SAS Institute was used for all statistical analyses.

Chapter IV. Results

Water Chemistry

Post harvest nitrate levels were significantly higher ($p < 0.001$) in streams harvested with a 4.5 m SMZ than those harvested with wider SMZs. This trend was observed during all post harvest seasons (Figure IV-1). Average nitrate values for other treatments were higher than those of control watersheds, however this difference was not significant (Table IV-1). Nitrate values were slightly significantly different between the two blocks ($p < 0.03$). Post harvest ammonium concentrations did not vary significantly between blocks or treatments (Table IV-2).

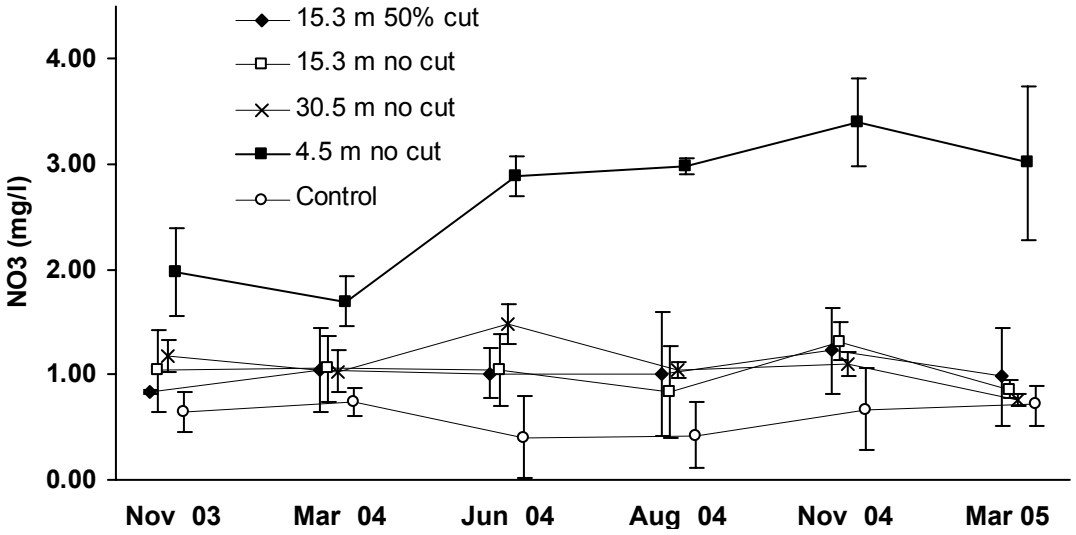


Figure IV - 1. Seasonal Nitrate concentrations for streams draining watersheds harvested with different SMZ widths. Error bars represent the range of nitrate concentrations for a given treatment and sample date.

Table IV - 1. Average stream water nitrate concentration (mg/L), standard error, and range by block and treatment. Letters indicate a significant difference at alpha=0.05 between blocks or treatments.

Block	n	Average NO_3 Concentration (mg/L)	Standard Error	Range
Block 1	30	1.41 [a]	0.137	0.52 – 3.82
Block 2	30	1.15 [b]	0.169	0.01 – 3.75
Treatment				
15.3 m 50% cut	12	1.02 [b]	0.124	0.42 – 1.64
15.3 m no cut	12	1.03 [b]	0.107	0.39 – 1.50
30.5 m no cut	12	1.10 [b]	0.077	0.70 – 1.67
4.5 m no cut	12	2.66 [a]	0.220	1.47 – 3.82
Control	12	0.60 [b]	0.096	0.01 – 1.06

Table IV - 2. Average stream water ammonium concentration (mg/L), standard error, and range by block and treatment. Letters indicate a significant difference at alpha=0.05 between blocks or treatments.

Block	n	Average NH_4 Concentration (mg/L)	Standard Error	Range
Block 1	30	0.21 [a]	0.019	0.08 – 0.48
Block 2	30	0.20 [a]	0.016	0.08 – 0.34
Treatment				
15.3 m 50% cut	12	0.21 [a]	0.026	0.08 – 0.34
15.3 m no cut	12	0.21 [a]	0.030	0.08 – 0.40
30.5 m no cut	12	0.22 [a]	0.035	0.08 – 0.48
4.5 m no cut	12	0.21 [a]	0.027	0.09 – 0.33
Control	12	0.19 [a]	0.016	0.08 – 0.28

Post harvest calcium concentrations were significantly higher than control streams for all treatments except 15.3 m, 50% cut (Table IV-3). In addition, block one calcium concentrations were significantly greater than those of block two ($p < 0.001$). Similarly, block one magnesium concentrations were significantly greater than those of block two ($p < 0.001$). Significant differences in

magnesium concentration were observed between treatments, however these differences did not appear to be related to SMZ width (Table IV-4).

Table IV - 3. Average stream water calcium concentration (mg/L), standard error, and range by block and treatment. Letters indicate a significant difference at alpha=0.05 between blocks or treatments.

Block	n	Average Ca Concentration (mg/L)	Standard Error	Range
Block 1	20	5.79 [a]	0.902	2.02 – 16.07
Block 2	20	2.21 [b]	0.313	0.63 – 4.43
Treatment				
15.3 m 50% cut	8	2.62 [b]	0.507	0.74 – 4.52
15.3 m no cut	8	8.37 [a]	1.854	3.08 – 16.07
30.5 m no cut	8	4.03 [ab]	0.227	2.94 – 5.05
4.5 m no cut	8	3.35 [ab]	0.904	0.63 – 6.53
Control	8	1.65 [bc]	0.236	0.88 – 2.48

Table IV - 4. Average stream water magnesium concentration (mg/L), standard error, and range by block and treatment. Letters indicate a significant difference at alpha=0.05 between blocks or treatments.

Block	n	Average Mg Concentration (mg/L)	Standard Error	Range
Block 1	20	1.56 [a]	0.075	0.96 – 2.24
Block 2	20	0.90 [b]	0.127	0.19 – 1.81
Treatment				
15.3 m 50% cut	8	1.10 [b]	0.130	0.48 – 1.61
15.3 m no cut	8	1.55 [a]	0.132	0.96 – 2.19
30.5 m no cut	8	1.53 [a]	0.075	1.22 – 1.84
4.5 m no cut	8	1.12 [b]	0.304	0.19 – 2.24
Control	8	1.50 [b]	0.213	0.30 – 3.05

Several seasons of turbidity measurements were removed from the data set when it was discovered after the fact that the meter was malfunctioning and not providing accurate data. Additional turbidity data was unfit for analyses due to the fact that some sites were sampled prior to a rain event, while others were sampled during or immediately following the rain event. For these reasons

turbidity data were not analyzed and are not included in this document. Post harvest conductivity levels were higher than control streams for all treatments and were significantly higher for 15.3 m no cut and 4.5 m no cut treatments (Table IV-5). Block one conductivity levels were significantly greater than those of block two ($p < 0.001$). TDS results were similar to those of conductivity in that TDS levels were higher than control streams for all treatments and were significantly higher for 15.3 m no cut and 4.5 m no cut treatments (Table IV-6). In addition, block one conductivity levels were significantly greater than those of block two ($p < 0.001$).

Table IV - 5. Average stream water conductivity(μS) , standard error, and range by block and treatment. Letters indicate a significant difference at alpha=0.05 between blocks or treatments.

Block	n	Average Conductivity (μS)	Standard Error	Range
Block 1	30	6.08 [a]	0.825	0 – 16.4
Block 2	30	3.80 [b]	0.326	0 – 8.3
Treatment				
15.3 m 50% cut	12	3.38 [bc]	0.500	0 – 6.6
15.3 m no cut	12	9.57 [a]	1.314	4.1 – 16.4
30.5 m no cut	12	4.33 [bc]	0.621	0 – 6.7
4.5 m no cut	12	4.91 [b]	0.727	0 – 8.2
Control	12	2.53 [c]	0.394	0 – 4.6

Table IV - 6. Average stream water TDS (g/L), standard error, and range by block and treatment. Letters indicate a significant difference at alpha=0.05 between blocks or treatments.

Block	n	Average TDS (g/L)	Standard Error	Range
Block 1	30	0.042 [a]	0.005	0 – 0.11
Block 2	30	0.024 [b]	0.001	0 – 0.04
Treatment				
15.3 m 50% cut	12	0.021 [c]	0.004	0 – 0.04
15.3 m no cut	12	0.062 [a]	0.009	0.03 – 0.11
30.5 m no cut	12	0.030 [bc]	0.003	0 – 0.04
4.5 m no cut	12	0.033 [b]	0.004	0 – 0.05
Control	12	0.021 [c]	0.003	0 – 0.04

Stream Temperature

During the course of the study a number of hobo temperature loggers were broken or lost resulting in missing data. As expected stream water temperature fluctuated seasonally, with the highest temperatures occurring in the summer and lowest temperatures in the winter (Figure IV-2). Temperatures of the study streams rarely exceeded 20 ˚C. However, the maximum temperature recorded was over 30 ˚C, and was recorded following the 15.3 m 50% cut treatment applied to B3W1. When all seasons were included SMZ width did not significantly impact average stream temperature. However, when the seasons were considered individually with linear regression an inverse relationship between SMZ width and summer and fall stream water temperatures was observed (Figure IV-3). In addition, an inverse relationship between SMZ width and average daily stream temperature difference ($^{\Delta}$T) was detected during the summer (Figure IV-4). SMZ width did not appear to impact stream water temperature during winter and spring.

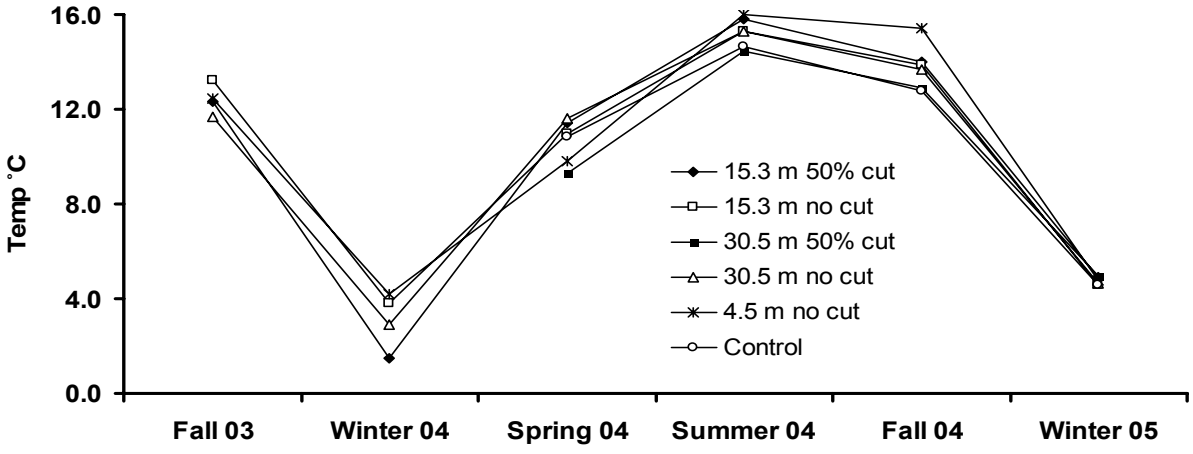
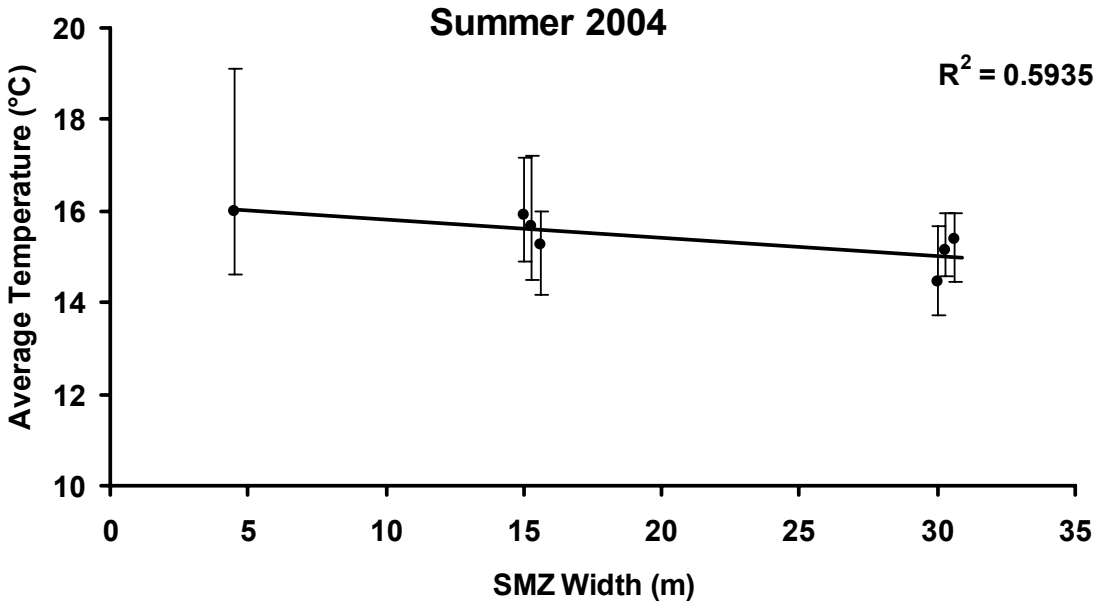


Figure IV - 2. Seasonal temperature variation for streams draining watersheds harvested with different SMZ widths and harvesting intensities.



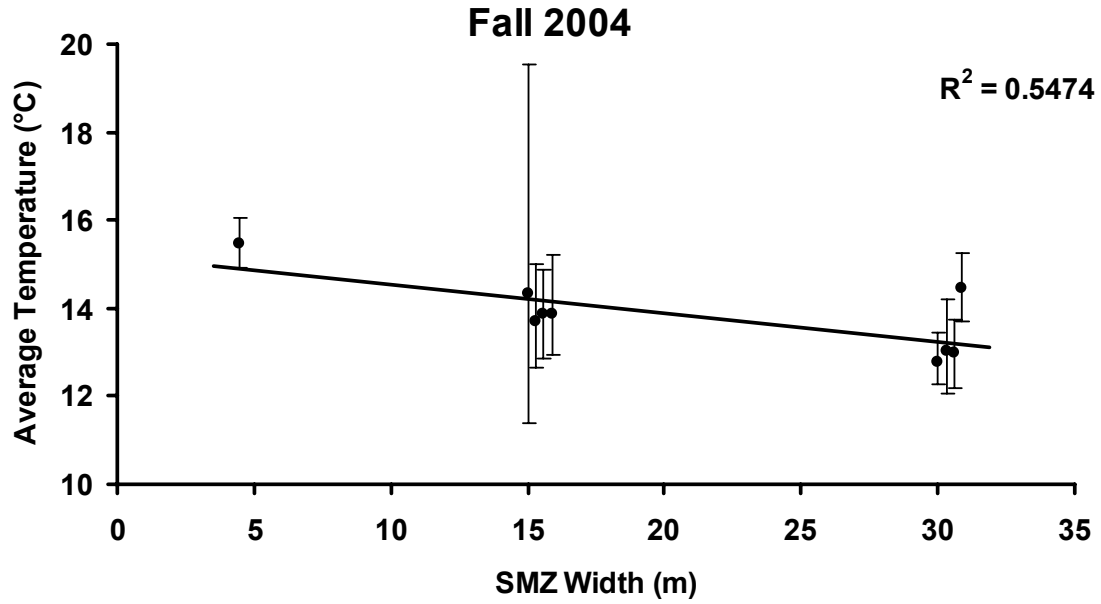


Figure IV - 3. Average 2004 summer and fall stream temperatures for streams draining watersheds harvested with different SMZ widths. Error bars represent minimum and maximum average hourly temperatures for each watershed.

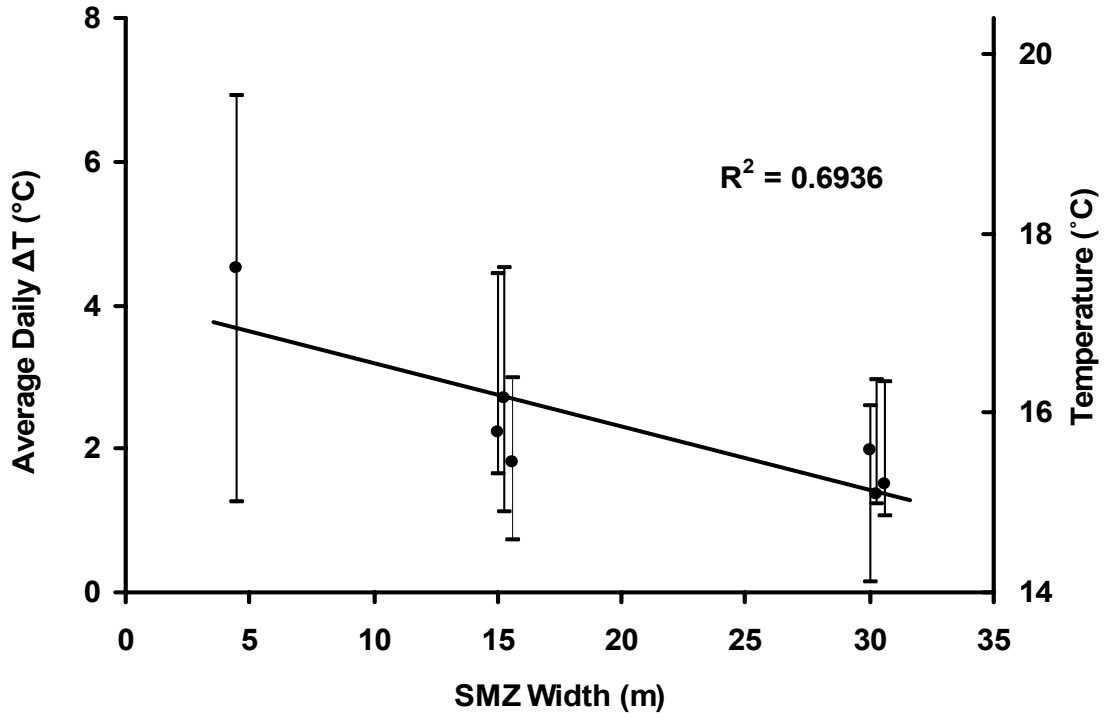


Figure IV - 4. Average summer daily $^{\Delta}$T for streams draining watersheds harvested with different SMZ widths. Error bars represent maximum and minimum hourly averages and are plotted on the temperature axis.

Macroinvertebrates

A total of 60 macroinvertebrate families and 109 genera were identified from the study streams (Appendix F). EPT (Ephemeroptera, Plecoptera, and Trichoptera) taxa comprised 28 of the families and 60 of the genera collected from the study streams.

Mean macroinvertebrate metric values by block, season, and treatment and the corresponding standard error values are included in Table IV-7. Metric values by watershed and sampling date are included in Appendix G. Mean values for the two blocks were significantly different for EPT taxa, EPT %, % mayflies, Simpsons Diversity Index, % Chironomids, % haptobenthos, % scraper,

% shredder, West Virginia Stream Condition index (WV SCI) and Macroinvertebrate Aggregated Index for Streams (MAIS). Mean metric values for early spring samples and summer samples were significantly different for EPT taxa, Hilsenhoff Family Biotic Index (HBI), % 5 dominant taxa, % intolerant, and % shredder. Significant differences between treatment means were also detected for a number of the metrics (Table IV-7). Streams harvested with a 4.5 m SMZ had significantly lower EPT Taxa, EPT %, % mayflies, % haptobenthos, % scraper, and WV SCI values than streams harvested with a 30.5 m SMZ with no residual harvest. Streams harvested with a 4.5 m SMZ also had a significantly higher % of Chironomids than streams harvested with a 30.5 m SMZ with no residual harvest. Control streams had significantly lower HBI values than streams with 4.5 m, 15.3 m no cut, and 30.5 m no cut treatments and higher % shredders than 15.3 m no cut and 30.5 m no cut treatments.

Table IV - 7. Macroinvertebrate metric means and standard errors by block, season, and treatment. Letters indicate a significant difference at alpha=0.05 between blocks, seasons, or treatments.

EPT Taxa				EPT %			
Block	n	Mean	Std. Error	Block	n	Mean	Std. Error
Block 1	15	13.8[a]	0.518	Block 1	15	83.9[a]	2.784
Block 2	15	12.4[b]	0.689	Block 2	15	73.0[b]	3.662
Season	n	Mean	Std. Error	Season	n	Mean	Std. Error
Early Spring	10	13.9[a]	0.437	Early Spring	10	80.9[a]	2.649
Summer	20	11.6[b]	0.846	Summer	20	73.5[a]	5.047
Treatment	n	Mean	Std. Error	Treatment	n	Mean	Std. Error
4.5 m no cut	6	10.8[b]	1.302	4.5 m no cut	6	70.1[b]	7.985
15.25 m, 50% cut	6	13.8[ab]	0.792	15.25 m, 50% cut	6	80.1[ab]	4.225
15.25 m no cut	6	12.7[ab]	0.494	15.25 m no cut	6	73.5[ab]	5.634
30.5 m no cut	6	15.0[a]	0.577	30.5 m no cut	6	90.1[a]	2.413
Control	6	13.2[ab]	0.946	Control	6	78.5[ab]	3.473
Taxa Richness				**% Mayflies**			
Block	n	Mean	Std. Error	Block	n	Mean	Std. Error
Block 1	15	29.3[a]	0.848	Block 1	15	49.9[a]	3.703
Block 2	15	27.5[a]	1.770	Block 2	15	23.7[b]	5.776
Season	n	Mean	Std. Error	Season	n	Mean	Std. Error
Early Spring	10	29.7[a]	1.215	Early Spring	10	37.2[a]	5.508
Summer	20	25.9[a]	1.402	Summer	20	35.8[a]	6.234
Treatment	n	Mean	Std. Error	Treatment	n	Mean	Std. Error
4.5 m no cut	6	23.7[a]	2.801	4.5 m no cut	6	31.7[b]	13.563
15.25 m, 50% cut	6	30.5[a]	2.291	15.25 m, 50% cut	6	32.6[b]	6.506
15.25 m no cut	6	28.2[a]	1.515	15.25 m no cut	6	35.9[ab]	6.907
30.5 m no cut	6	30.7[a]	1.174	30.5 m no cut	6	60.6[a]	3.769
Control	6	29.0[a]	2.113	Control	6	23.1[b]	7.277
HBI				**Simpson's Diversity Index**			
Block	n	Mean	Std. Error	Block	n	Mean	Std. Error
Block 1	15	3.1[a]	0.156	Block 1	15	0.85[a]	0.011
Block 2	15	2.9[a]	0.182	Block 2	15	0.78[b]	0.021
Season	n	Mean	Std. Error	Season	n	Mean	Std. Error
Early Spring	10	2.8[a]	0.134	Early Spring	10	0.80[a]	0.016
Summer	20	3.3[b]	0.203	Summer	20	0.84[a]	0.022
Treatment	n	Mean	Std. Error	Treatment	n	Mean	Std. Error
4.5 m no cut	6	3.1[a]	0.114	4.5 m no cut	6	0.78[a]	0.031
15.25 m, 50% cut	6	2.8[ab]	0.247	15.25 m, 50% cut	6	0.84[a]	0.021
15.25 m no cut	6	3.4[a]	0.340	15.25 m no cut	6	0.87[a]	0.021
30.5 m no cut	6	3.3[a]	0.145	30.5 m no cut	6	0.80[a]	0.028
Control	6	2.3[b]	0.226	Control	6	0.80[a]	0.040

Table IV-7 (continued). Macroinvertebrate metric means and standard errors by block, season, and treatment. Letters indicate a significant difference at alpha=0.05 between blocks, seasons, or treatments.

% 5 Dominant Taxa				% Intolerant			
Block	n	Mean	Std. Error	Block	n	Mean	Std. Error
Block 1	15	49.9[a]	9.414	Block 1	15	67.6[a]	3.136
Block 2	15	53.7[a]	10.232	Block 2	15	65.4[a]	3.099
Season	n	Mean	Std. Error	Season	n	Mean	Std. Error
Early Spring	10	39.4[b]	8.963	Early Spring	10	69.7[a]	2.352
Summer	20	76.6[a]	3.209	Summer	20	60.1[b]	3.928
Treatment	n	Mean	Std. Error	Treatment	n	Mean	Std. Error
4.5 m no cut	6	57.0[a]	18.011	4.5 m no cut	6	61.2[a]	4.969
15.25 m, 50% cut	6	47.8[a]	15.158	15.25 m, 50% cut	6	68.4[a]	4.572
15.25 m no cut	6	50.2[a]	15.713	15.25 m no cut	6	58.7[a]	5.889
30.5 m no cut	6	52.1[a]	16.406	30.5 m no cut	6	69.5[a]	3.390
Control	6	51.8[a]	16.659	Control	6	74.7[a]	3.429
% Chironomidae				**% Haptobenthos**			
Block	n	Mean	Std. Error	Block	n	Mean	Std. Error
Block 1	15	6.4[b]	0.865	Block 1	15	89.2[a]	1.201
Block 2	15	16.9[a]	4.129	Block 2	15	76.0[b]	3.880
Season	n	Mean	Std. Error	Season	n	Mean	Std. Error
Early Spring	10	12.3[a]	2.745	Early Spring	10	82.1[a]	2.859
Summer	20	10.5[a]	4.323	Summer	20	83.7[a]	4.287
Treatment	n	Mean	Std. Error	Treatment	n	Mean	Std. Error
4.5 m no cut	6	24.5[a]	8.563	4.5 m no cut	6	71.5[b]	8.572
15.25 m, 50% cut	6	13.2[ab]	4.511	15.25 m, 50% cut	6	82.0[ab]	4.508
15.25 m no cut	6	6.9[b]	1.248	15.25 m no cut	6	88.1[a]	1.540
30.5 m no cut	6	4.0[b]	0.815	30.5 m no cut	6	91.0[a]	2.255
Control	6	9.9[ab]	2.451	Control	6	80.5[ab]	3.487
% Scraper				**% Shredder**			
Block	n	Mean	Std. Error	Block	n	Mean	Std. Error
Block 1	15	27.6[a]	2.286	Block 1	15	19.5[b]	3.475
Block 2	15	12.9[b]	4.190	Block 2	15	39.8[a]	5.815
Season	n	Mean	Std. Error	Season	n	Mean	Std. Error
Early Spring	10	19.5[a]	3.461	Early Spring	10	33.5[a]	5.124
Summer	20	21.9[a]	4.483	Summer	20	22.0[b]	4.539
Treatment	n	Mean	Std. Error	Treatment	n	Mean	Std. Error
4.5 m no cut	6	14.3[b]	6.281	4.5 m no cut	6	28.5[ab]	6.653
15.25 m, 50% cut	6	18.3[b]	6.183	15.25 m, 50% cut	6	33.5[ab]	7.616
15.25 m no cut	6	22.0[ab]	4.097	15.25 m no cut	6	25.7[b]	8.700
30.5 m no cut	6	36.0[a]	4.185	30.5 m no cut	6	10.6[b]	2.407
Control	6	10.8[b]	5.060	Control	6	50.0[a]	8.835

Table IV-7 (continued). Macroinvertebrate metric means and standard errors by block, season, and treatment. Letters indicate a significant difference at alpha=0.05 between blocks, seasons, or treatments.

WV Stream Condition Index				MAIS			
Block	n	Mean	Std. Error	Block	n	Mean	Std. Error
Block 1	15	94.4[a]	1.324	Block 1	15	17.2[a]	0.223
Block 2	15	85.4[b]	3.288	Block 2	15	13.9[b]	0.892
Season	n	Mean	Std. Error	Season	n	Mean	Std. Error
Early Spring	10	91.2[a]	2.238	Early Spring	10	15.7[a]	0.567
Summer	20	87.4[a]	3.734	Summer	20	15.3[a]	1.221
Treatment	n	Mean	Std. Error	Treatment	n	Mean	Std. Error
4.5 m no cut	6	79.5[b]	7.113	4.5 m no cut	6	13.7[a]	2.076
15.25 m, 50% cut	6	93.2[ab]	2.973	15.25 m, 50% cut	6	15.7[a]	1.022
15.25 m no cut	6	90.4[ab]	2.656	15.25 m no cut	6	17.2[a]	0.543
30.5 m no cut	6	94.7[a]	0.946	30.5 m no cut	6	16.8[a]	0.307
Control	6	91.9[ab]	3.424	Control	6	14.5[a]	1.025

Chapter V. Discussion

Throughout the discussion consideration must be given to the underlying fact that harvesting was restricted to one side of the stream, whereas most forest harvest studies have involved harvesting on both sides of the stream. Operationally, the practice of harvesting only one side of a stream is common, indeed these were actual sale units. Harvesting only one half of the watershed is likely to mitigate water quality impacts from forestry because at least half of the trees remain for nutrient and water uptake, stream shading and the supply of carbon and coarse woody debris inputs. Still, statistically significant differences were observed for many of the parameters tracked during the study.

In addition to statistically significant trends that were identified by the methods employed, a number of important disturbances were observed outside of the study design. On two watersheds the primary skid trail leading to the log deck crossed the stream. Pole bridges with logs stacked in the stream were used as stream crossings and were intended to maintain water flow. In both cases the spaces between the logs became entrained with sediment impeding the flow of water and essentially creating a dam on the stream. In addition to stream morphology and flow alterations from the damming effects, the poorly designed stream crossings were likely substantial sources of sediment during periods of high flow. The use of pole bridges is generally not recommended except for non-flowing ephemeral streams (Virginia Department of Forestry, 1997). The use of temporary bridges or culverts could have minimized stream disturbance (Taylor et al., 2002).

Two watersheds also experienced substantial logging induced hillside slumps. In one case the slump occurred during logging operations, while the other occurred post logging. In both cases large amounts of sediment and debris was carried down the hillside and into the stream. Exposed soil resulting from the slumps appeared to be a considerable source of sediment through the duration of this study.

On several watersheds vegetation remained absent from skid trails and log decks more than a year after harvesting was complete. The bare soil on skid trails and log decks is a potential source of sediment if any runoff from these highly disturbed areas enters the stream. Additional observations included an intermittent channel flowing across a log deck and sediment laden runoff from poorly drained skid trails entering one of the study streams. A detailed watershed by watershed discussion of these observations is contained in Appendix H.

Water Chemistry

Average nitrate concentration for control streams was 0.60 mg/L. This value is similar to the highest preharvest nitrate concentration of 0.49 mg/L measured by Sharp (2003) during the spring, summer and fall of 2002. However, the 30.5 m no cut, 15.3 m no cut and 15.3 m 50% cut treatments exhibited mean nitrate concentrations twice the highest pre-harvest nitrate concentration, while streams harvested with a 4.5 m no cut treatment had average nitrate concentrations that were five times this value. Increased post harvest nitrate levels are a typical finding of forest harvest studies (Aubertin and Patric, 1974;

Lynch et al., 1985; Kochenderfer et al., 1997; Wynn et al., 2000; Swank et al., 2001). Elevated nitrate levels were likely due to decomposing slash left behind in harvested areas. Complete vegetation removal during clearcut logging greatly reduces nutrient uptake resulting in nitrate export from the watershed. This trend lasts until nutrient uptake by regrowth catches up with nutrients made available by decomposing slash. SMZs of at least 15.3 m appeared to mitigate nitrate export as increases in these streams were small compared to those of streams harvested with 4.5 m SMZs. Removal of 50% of the basal area within the SMZ did not appear to impact stream nitrate concentrations as streams harvested with 15.3 m 50% harvest treatments had similar nitrate concentrations to those harvested with 15.3 m and 30.5 m no cut treatments.

Increased nitrate levels were evident during all seasons and on all sampling dates. Nitrate concentrations for harvested watersheds remained elevated more than a year and a half following the harvests with no trend towards a return to baseline levels. Nitrate concentrations may remain elevated for up to seven years following forest harvest (Kochenderfer et al., 1997), however a quicker return to baseline level is often observed (Arthur et al., 1998).

Nitrate increases observed during this study remained well within drinking water standards, which supports the findings of most forest harvest studies (Binkley and Brown, 1993). Although streams harvested with a 4.5 m no cut treatment did exhibit mean nitrate concentrations that were nearly five times those of control streams, the nitrate levels were still quite low and the increase may not have been biologically important. However, it should be noted that

although stream flow was not monitored during this study, increases in stream flow typically follow forest harvest (Hornbeck et al., 1997; Arthur et al., 1998; Martin et al., 2000; Swank et al., 2001 and may have diluted nitrate concentrations masking an even greater increase in total nitrate export from the watershed (Lynch and Corbett, 1990).

Ammonium concentrations were not impacted by forest harvest or SMZ width as mean values for treatment streams were not significantly different from those of control streams. Post harvest ammonium concentrations were lower than the preharvest average of 0.47 mg/L measured by Sharp (2003). However, this difference is likely due to natural variation as logged streams had similar ammonium concentrations to control streams and those of streams that were preharvest during the study at hand. Patric (1980), Golladay et al. (1992) and Swank et al. (2001) also found no effect on ammonium concentration following logging.

Calcium concentrations were significantly higher in harvested streams than control streams, except for those harvested with a 15.3 m 50% harvest treatment, suggesting that calcium concentrations were impacted by forest harvest. However, considerable preharvest variability in calcium concentration was observed as limestone deposits in areas of the Wuchner and Beaver Creek tracts led to some streams having very high baseline calcium levels while streams in watersheds without limestone deposits had very low calcium concentrations. When preharvest calcium data collected by Sharp (2003) were considered no trend was discovered as calcium levels in most streams remained

consistent between pre and post harvest periods. This suggesting that the increased calcium levels observed for harvested streams in this study were due to underlying geology and not treatment effects. Similarly, significant differences between control and treatment stream magnesium concentrations were likely driven by geologic variation not logging treatments. Patric (1980) found no change in stream calcium and magnesium concentrations following logging, whereas other investigators have reported both increases (Kochenderfer et al., 1997; Arthur et al., 1998; Swank et al., 2001) and decreases (Lynch and Corbett, 1990) in post harvest calcium and magnesium levels.

Conductivity and TDS were very low for both control and treatment streams. However, mean levels for 4.5 m no cut and 15.3 m no cut treatment streams were significantly higher than those of control streams for both parameters. Similar results are expected for the two parameters because both are related to the amount of dissolved ions in the stream. Conductivity and TDS were highest in streams harvested with 15.3 m no cut treatments, which was likely due to high average calcium concentrations in these streams. Elevated conductivity and TDS in streams harvested with a 4.5 m SMZ may have been related to high levels of nitrate following harvest. The increased conductivity in treatment streams supports the findings of Growns and Davis (1991) and Arthur et al. (1998), but is contrary to those of Patric (1980) and Lynch and Corbett (1990) who saw an initial decrease in conductivity following forest harvest.

Although turbidity data were not included for reasons described in the results section, several important observations were made during fieldwork.

Stream turbidity and suspended sediment during rain and snow melt events appeared to be heavily influenced by maintained logging roads, more so than skid trails or log decks. During rain and snowfall events, runoff from maintained logging roads was often observed entering streams just downstream of culvert crossings. In many cases water upstream of the road had high clarity while downstream from the road the water had very low visibility and carried high sediment loads. An exception to road dominated sediment inputs is discussed in Appendix H under the B4W2 observations where runoff from a skid trail was so severe that turbidity readings exceeded the limit of a 999 NTU turbidity meter.

Stream Temperature

Summer and fall stream temperatures were inversely related to SMZ width. Harvesting of adjacent trees increases direct solar radiation reaching a stream resulting in increased water temperatures (Patric, 1980; Beschta et al., 1987). Increased summer water temperatures were expected as Sharp (2003) found an inverse relationship between summer water temps and in stream canopy cover. Narrow SMZs decrease canopy cover and therefore were likely to influence summer water temps. Other investigator have shown that buffer strips of 20 m (Patric, 1980) to 30 m (Lynch et al., 1985) can mitigate water temperature increases associated with forest harvest. However, this was the first study to consider multiple buffer widths.

Pre harvest data from Sharp, 2003 showed that B4W1 had the lowest overall and summer average stream temperature. However, following harvest

with a 4.5 m SMZ summer and fall temperatures were the highest of all study watersheds. Preharvest data also indicated that B4W1 had the highest average instream canopy cover of all study watersheds (93%) therefore it is reasonable that this stream would be greatly affected by complete tree removal to within 4.5 m of the center of the stream.

Beschta et al. (1987) found that stream temperatures during the winter months may decrease when streamside vegetation was removed leaving no canopy to prevent energy losses from the stream. However, winter temperatures did not vary between harvested and control streams in the study at hand, nor was there a trend between winter stream temperature and SMZ width.

SMZ width appeared to be more important than partial tree removal within the SMZ. This is evident by the fact that streams harvested with a 4.5 m SMZ showed greater temperature increases than those in which 50% of the basal area was removed from a 15.3 m or 30.5 m SMZ. Similarly, Kochenderfer et al. (1997) reported that removal of 44% of the basal area adjacent to a stream did not lead to significant temperature increases.

Macroinvertebrates

Mean values for several metrics (EPT taxa, EPT %, % Mayflies, % Chironomids, % haptobenthos, % scraper, and WV SCI) were significantly different between streams harvested with a 4.5 m SMZ and those harvested with a 30.5 m no cut treatment. In all cases the difference suggests that streams harvested with a 30.5 m SMZ and no residual harvest maintained higher water

quality, as indicated by macroinvertebrate community metrics, than streams harvested with a 4.5 m SMZ. Mean values for these metrics were not significantly different between streams harvested with a 4.5 m SMZ and control streams. However, substantial variability in macroinvertebrate communities between streams was not controlled by blocking and may have masked differences between control and treatment streams.

The one metric that did show significant differences between control and harvest streams was the HBI. Mean HBI values were significantly lower (indicative of higher water quality) for control streams than streams harvested with 4.5 no cut, 15.25 no cut, and 30.5 no cut treatments. HBI was the only metric that showed a significant difference between 30.5 m no cut streams and control streams suggesting that buffers of at least 30.5 m are sufficient to prevent logging induced impacts on stream macroinvertebrate communities. Newbold et al. (1980), Growns and Davis (1991), and Davies and Nelson (1994) also concluded that buffers of 30 m or wider prevent logging effects on macroinvertebrate communities.

Although significant differences were detected for several metrics, logging with SMZs as narrow as 4.5 m did not appear to greatly alter the macroinvertebrate communities of the study streams. This suggests that these communities are resistant to logging disturbance. The history of the study area is heavily tied to logging and so it is reasonable to believe that the macroinvertebrate communities would be adapted to logging disturbance.

In addition, the study streams were characterized by extremely high water quality pre harvest (Sharp, 2003). These, high gradient streams resist sediment deposition so it is likely that eroded sediment from roads, log decks, and skid trails was quickly washed out of the study reaches and did not impact benthic habitat within the reaches. Baseline water temperatures and nutrient levels were very low so small increases in these parameters may not have been important. Perhaps slight logging induced degradation of this high water quality was not enough to greatly impact the macroinvertebrate communities. However, warmer water temperatures, higher nutrient loads, and fine sediment addition may be compounded downstream exacerbating water quality problems at higher stream orders.

Forest harvest and SMZ width did not influence macroinvertebrate taxa richness. No difference in taxa richness supports the findings of Growns and Davis (1991) and Noel et al. (1986). Other investigators found that logged streams had higher taxa richness (Silsbee and Larson, 1983).

Seasonal differences were only significant for 5 of the 14 metrics reported in the results. This suggests that the metrics used were relatively robust to seasonal differences in macroinvertebrate communities of headwater Appalachian streams. The significant seasonal difference for HBI supports the findings of Hilsenhoff (1977). EPT taxa and EPT % were significantly higher during early spring sampling, likely because of increased stonefly presence. Increased early spring stonefly presence is due to several common early spring stonefly taxa being in egg or nymphal diapause stages during the summer.

Acid precipitation has resulted in extremely low pH (pH < 5) for many streams draining unbuffered Appalachian watersheds. Three of the study streams, B4W1, B4W3 and CN4, were found to have extremely low pH (mean values of 4.4, 4.57, and 4.2, respectively). B4W3 was not included in any analyses because the harvest on this watershed was not completed until February of 2005. B4W1 was harvested with a 4.5 m no cut treatment and CN4 was an unharvested control stream, both were included in macroinvertebrate analyses. Many mayfly taxa are sensitive to low pH and highly acidic streams have been shown to have low relative abundance of mayflies (Kouszewski and Perry, 1993). Mayflies are important in the calculation of many metrics, especially EPT metrics and multimetrics. Macroinvertebrate community metrics calculated from streams with extremely low pH may indicate reduced water quality relative to metrics calculated for streams with higher pH regardless of logging status. In an attempt to control this variability, streams with low pH were blocked together. However, a stream with low pH was not available for all treatments so blocking was not able to adequately control variability in macroinvertebrate communities driven by low pH. Inadequate control of this variation may have masked logging effects on the macroinvertebrate communities of treatment streams.

Newbold et al. (1980) and Noel et al. (1986) found that forest harvest led to increased abundance of several tolerant taxa including the mayfly genus *Baetis* and the dipteran family Chironomidae. Stone and Wallace (1998) also reported increased percent *Baetis* following forest harvest. *Baetis* feeds by

grazing on periphyton which is often more abundant following logging (Noel et al., 1986; Davies and Nelson, 1994) due to increased sunlight reaching the stream and allowing for augmented primary production. In the present study, relative abundance of *Baetis* did not show a response to logging. On the other hand, Chironomid relative abundance did show a response to SMZ width, as Chironomid larvae made up a significantly greater percent of taxa in streams harvest with a 4.5 m SMZ (24.5%) than those harvested with 15.3 m or 30.5 m SMZs (6.9 and 4.0%, respectively). Control stream mean % Chironomids (9.9%) was much lower than that of streams harvested with a 4.5 m SMZ, but this difference was not significant. Generally Chironomids prefer fine sediments, therefore logging induced sedimentation may lead to an increase in Chironomid relative abundance.

Chapter VI. Conclusions

During this short term study SMZs of at least 15.3 m appeared to be sufficient to protect the water quality and macroinvertebrate communities of headwater streams located in the Allegheny Mountain and Plateau Physiographic province. However, harvesting was restricted to one side of the stream so harvesting on both sides of the stream may require wider SMZs. SMZs of 4.5 m resulted in increased nitrate concentration, elevated stream temperatures, and altered macroinvertebrate communities. These impacts were not observed for 15.3 m SMZs, however considerable uncontrolled variability in baseline water chemistry and macroinvertebrate communities may have masked logging effects.

The small streams involved in this study are the headwaters of quality trout streams and brook trout *Salvelinus fontinalis* par were observed in several streams. Forested headwater streams generally have very high water quality and therefore tend to be resistant to small changes in nutrient concentrations, water temperature and sediment inputs. However, the physical, chemical, and biological changes brought on by logging headwater streams with an insufficient SMZ could be magnified downstream in areas where water quality may already be compromised. With the potential for impact to downstream biological communities in mind, leaving a SMZ of at least 30.5 m seems prudent.

Regardless of SMZ width, residual harvest of up to 50% did not seem to increase the impact of forest harvest on stream water quality and macroinvertebrate communities. Sharp (2003) reported that for the same study sites nearly 90% of the stumpage value contained within an SMZ can be attained

through the removal of 50% of the basal area. This has significant economic implications for landowners as it greatly reduces the monetary loss associated with leaving unharvested trees adjacent to streams.

Throughout the study it was evident that SMZ width alone did not determine the effects of logging on water quality. Nor did leaving a substantial SMZ of 30.5 m guarantee that disturbance from logging would be avoided (see observations in Appendix H). Questionable decisions by loggers may lead to a breach of the SMZ regardless of width, leading to subsequent water quality impacts. A breach can be described as any disturbance of the SMZ that directly connects the stream to the harvest site. Breaches include improper or poorly designed stream crossings, and areas where concentrated runoff penetrates the SMZ and flows directly into the stream. SMZ width alone is not an adequate predictor of whether or not a breach will occur. In addition to a sufficiently wide SMZ, care must be taken when locating log decks, constructing skid trails, and designing and implementing stream crossings to ensure water quality protection.

Chapter VII. Recommendations

- At two of the study sites the log deck was located across the stream from the harvest requiring the primary skid trail to cross the stream. Large quantities of sediment were observed entering the stream from these inadequate stream crossings. Skidding across streams should be avoided unless it is the only way to carry out a harvest.
- When it is necessary for a skid trail or temporary road to cross a stream care must be taken to minimize disturbance of the stream bank and channel. Portable bridges or culverts are recommended for temporary stream crossings as they can greatly reduce the amount of sediment entering the stream.
- Whenever possible stream crossings on roads should be located at a high point so runoff drains away from the stream crossing. When it is necessary for a stream crossing to be located at a low point efforts should be made to drain water from the road prior to the runoff reaching the stream.
- Road maintenance is very important. Some road sections were extremely soft during wet conditions resulting in erosion and the potential for excessive sedimentation if runoff water reached a stream. Adding gravel or crushed rock to these areas would stabilize the road and reduce sediment loss during rain events.
- Skid trails at most of the harvests remained exposed soil through the duration of the study. Post harvest revegetation of skid trails is important

and should be a priority to minimize erosion and reduce the potential for eroded sediment to reach the stream.

- Logging induced slumps occurred on two of the study watersheds due to machinery being operated too close to the stream. These slumps resulted in large amounts of sediment and debris sliding into the stream impeding the natural flow. Both watersheds are characterized by exceedingly steep terrain, with slopes of greater than 70% within and beyond the SMZ. Increased SMZ width is recommended when harvesting watersheds in which areas directly adjacent to the stream are extremely steep.

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Appendices

Appendix A: Table of study watershed characteristics (Sharp, 2003).

Watershed	Slope (%)	Watershed Area (ha)	Percent of Watershed Harvested	Stream Length to be Cut (km)
B1W1	29	27	50	0.67
B1W2	45	39	31	0.63
B1W3	30	31	22	0.58
B1W4	27	42	18	0.83
B2W1	22	42	33	0.48
B2W2	30	22	41	0.58
B2W3	42	40	40	0.64
B2W4	48	25	40	0.92
B3W1	54	61	17	0.36
B3W2	43	64	14	0.56
B3W3	39	63	11	0.54
B3W4	50	42	31	0.39
B4W1	21	18	43	0.33
B4W2	30	28	13	0.52
B4W3	29	52	30	0.85
B4W4	23	20	25	0.48
C2R2	44	57	12	0.52
C1R1	42	42	40	0.84

Appendix B: Soil series and taxonomic class (Soil Survey Division, Natural Resources Conservation Service, United States Department of Agriculture, 2001).

Soil Series	Taxonomic Class
Belmont	Fine-loamy, mixed, active, mesic Typic
Berks	Loamy-skeletal, mixed, active, mesic Typic Dystrudepts
Buchanan	Fine-loamy, mixed, semiactive, mesic Aquic Fragiudults
Calvin	Loamy-skeletal, mixed, active, mesic Typic Dystrudepts
Dekalb	Loamy-skeletal, siliceous, active, mesic Typic Dystrudepts
Gilpin	Fine-loamy, mixed, active, mesic Typic Hapludults
Weikert	Loamy-skeletal, mixed, active, mesic Lithic Dystrudepts

Appendix C: Proposed SMZ treatments, actual treatments that were applied and approximate date that harvests were completed.

Watershed	Proposed SMZ Treatment	SMZ Treatment Applied	Date Harvest Completed
B1W1	15' no cut	Not Harvested	
B1W2	50' no cut	50' no cut	September 2003
B1W3	50' 50% cut	50' no cut	July 2003
B1W4	100' 50% cut	Not Harvested	
B2W1	50' 50% cut	Not Harvested	
B2W2	15' no cut	15' no cut	September 2003
B2W3	100' 50% cut	100' no cut	June 2003
B2W4	50' no cut	Not Harvested	
B3W1	50' 50% cut	50' 50% cut	September 2003
B3W2	100' 50% cut	50' 50% cut	May 2004
B3W3	15' no cut	50' no cut	November 2003
B3W4	50' no cut	100' no cut	October 2004
B4W1	15' no cut	15' no cut	July 2003
B4W2	100' 50% cut	100' 50% cut	September 2003
B4W3	50' no cut	50' no cut	February 2005
B4W4	50' 50% cut	50' 50% cut	September 2003
C1R1	100' no cut	100' no cut	June 2003
C2R2	100' no cut	100' 50% cut	September 2004
CN1	Control (no harvest)	Control (no harvest)	
CN2	Control (no harvest)	Control (no harvest)	
CN3	Control (no harvest)	Control (no harvest)	
CN4	Control (no harvest)	Control (no harvest)	

Appendix D: List and description of macroinvertebrate metrics that will be calculated.

Category	Metric	Explanation	Expected Response to Disturbance
Richness measures	Taxa Richness	Number of families or genera	negative
	EPT Index	Number of mayfly (Ephemeroptera), stonefly (Plecoptera), and caddisfly (Trichoptera) families	negative
	# Mayfly Taxa	Number of mayfly families	negative
	# Stonefly Taxa	Number of stonefly families	negative
Composition measures	% EPT	Percent abundance of mayfly, stonefly, and caddisfly nymphs	negative
	% Mayflies	Percent abundance of mayfly nymphs	negative
	% Stoneflies	Percent abundance of stonefly nymphs	negative
Balance measures	% 5 Dominant Taxa	Percent abundance of the 5 most abundant taxa combined	positive
	% *Chironomids*	Percent abundance of nymphs in the dipteran family Chironomidae	positive
	Simpson Diversity Index	Integrates richness and evenness into a measure of general diversity	negative
Tolerance measures	HBI (Hilsenhoff Biotic Index)	Weighted sum of total taxa by pollution tolerance	positive
	% Intolerant taxa	Percent abundance of macroinvertebrate taxa with tolerance values of 5 or less	negative
Trophic measures	% Scrapers	Percent abundance of macroinvertebrates feeding upon periphyton	negative
	% Shredders	Percent abundance of macroinvertebrates feeding on coarse particulate organic matter	negative
Habit measures	% Haptobenthos	Percent abundance of macroinvertebrates requiring clean, coarse, firm substrates	negative
Multimetric measures	West Virginia Stream Condition Index	Incorporates taxa richness, EPT taxa, % EPT, % 2 dominant taxa, % Chironomids and HBI and rates streams from 0-100 with higher scores representing stream with macroinvertebrate communities similar to those of reference streams.	negative
	Macroinvertebrate Aggregated Index for Streams	Incorporates EPT Index, # Ephemeroptera Taxa, % Ephemeroptera, % 5 Dominant Taxa, Simpsons Diversity Index, HBI, # Intolerant Taxa, % Scrapers, and % Haptobenthos. Streams are scored from 0-18 and given qualitative rankings ranging from very poor to very good.	negative

Appendix E: Table of study watershed characteristics (Sharp, 2003).

Watershed	Block	Treatment
B2W1	1	4.5 m no cut
B1W3	1	15.3 m 50% cut
B3W1*	1	15.3 m no cut
C2R2**	1	30.5 m 50% cut
C1R1	1	30.5 m no cut
CN2	1	Control (no harvest)
B4W1	2	4.5 m no cut
B4W4	2	15.3 m 50% cut
B1W2	2	15.3 m no cut
B4W2	2	30.5 m 50% cut
B2W3	2	30.5 m no cut
CN4	2	Control (no harvest)

* B3W1 was not accessible during March of 2005 so B3W2 was substituted for analyses requiring blocking (B3W2 was harvested with the same SMZ treatment and was part of the same original block).

** The C2R2 harvest was not complete until August of 2004, therefore 30.5 m 50% cut treatment data collected prior are not included for analyses requiring blocking.

Appendix F: Macroinvertebrate Taxa List.

Order	Family	Genus	
Trichoptera	Hydropsychidae	*Diplectrona sp.*	
		Parapsyche sp.	
		Ceratopsyche sp.	
		Homoplectra sp.	
	Psychomiidae	*Lype sp.*	
	Rhyacophilidae	*Rhyacophila sp.*	
	Uenoidae	*Neophylax sp.*	
	Glossossomatidae	*Glossosoma sp.*	
		Agapetus sp.	
	Lepidostomatidae	*Lepidostoma sp.*	
	Ploycentropodidae	*Polycentropus sp.*	
	Limnephilidae	*Hydatophylax sp.*	
		Pycnopsyche	
	Philopotamidae	*Wormaldia sp.*	
		Dolophilodes sp.	
	Goeridae	*Goera sp.*	
	Molannidae	*Molanna sp.*	
	Phryganeidae	*Oligostomis sp.*	
	Hydroptilidae	*Palaeagapatus sp.*	
Totals		13	*19*
Ephemeroptera	Heptageniidae	*Epeorus sp.*	
		Stenonema sp.	
		Leucrocuta sp.	
		Cynygmula sp.	
	Ephemerellidae	*Ephemerella sp.*	
		Drunella sp.	
		Eurylophella sp.	
	Baetidae	*Baetis sp.*	
	Leptophlebiidae	*Paraleptophlebia sp.*	
	Ephemeridae	*Ephemera sp.*	
	Baetiscidae	*Baetisca sp.*	
Totals		6	*11*
Plecoptera	Perlodidae	*Yugus sp.*	
		Isoperla sp.	
		Malirekus sp.	
	Perlidae	*Acroneuria sp.*	
		Eccoptera sp.	
		Beloneuria sp.	
	Chloroperlidae	*Haploperla sp.*	
		Sweltsa sp.	
	Nemouridae	*Amphinemura sp.*	
		Nemoura sp.	
		Soyedina sp.	
		Podmosta sp.	
	Leuctridae	*Leuctra sp.*	
		Megaleuctra sp.	
	Peltoperlidae	*Peltoperla sp.*	
		Tallaperla sp.	
	Pteronarcyidae	*Pteronarcys sp.*	
	Capniidae	*Paracapnia sp.*	
	Taeniopterygidae	*Oemopteryx sp.*	
		Capnia sp.	
Totals		9	*20*

Appendix F (continued): Macroinvertebrate Taxa List.

Order	Family	Genus
Diptera	Tipulidae	*Hexatoma sp.*
		Dicranota sp.
		Pseudolimnephila sp.
		Limnophila sp.
		Limonia sp.
		Tipula sp.
		Ormosia sp.
		Molophilus sp.
		Pedicia sp.
		Lipsothrix sp.
		Antocha sp.
		Polymera sp.
		Pilaria sp.
		Brachypremna sp.
	Dixidae	*Dixa sp.*
	Chironomidae	
	Simuliidae	*Simulium sp.*
		Prosimulium sp.
	Tabanidae	*Hybomitra sp.*
		Chrysops sp.
	Empididae	*Oreogeton sp.*
		Chelifera sp.
		Clinocera sp.
	Dolichopodidae	
	Psychodidae	*Psychoda sp.*
		Pericoma sp.
	Ceratopogonidae	*Atrichopogon sp.*
		Probezzia sp.
		Ceratopogon sp.
		Dasyhelea sp.
		Forcipomyia sp.
	Pelerorhynchidae	*Glutops sp.*
	Scathophagidae	
	Phoridae	
	Stratiomyidae	*Allognosta sp.*
Totals	13	*35*
Megaloptera	Corydalidae	*Nigronia sp.*
	Sialidae	*Sialis sp.*
Odonata	Gomphidae	*Lanthus sp.*
	Cordulegastridae	*Cordulegaster sp.*
Coleoptera	Elmidae	*Oulimnius sp.*
		Dubiraphia sp.
		Promoresia sp.
	Hydrophilidae	
	Psephenidae	*Psephenus sp.*
		Ectopria sp.
	Hydrochidae	*Hydrochus sp.*
	Dytiscidae	*Hygrotus sp.*
	Carabidae	
	Lampyridae	
	Staphylinidae	

Appendix F (continued): Macroinvertebrate Taxa List.

Order	Family	Genus	
Coleoptera	Curculionidae		
Hemiptera	Veliidae	*Microvelia sp.*	
		Rhagovelia sp.	
	Gerridae	*Aquarius sp.*	
		Gerris sp.	
Amphipoda	Gammaridae	*Gammarus sp.*	
Decapoda	Cambaridae	*Cambarus sp.*	
Colembola	Entomobryidae		
Class: Oligochaeta			
Totals		19	24
Grand Totals		**60**	**109**

Appendix G: Macroinvertebrate metric data.
August 2003

Watershed	Treatment	EPT Taxa	% EPT	Taxa Richness	% Mayflies	Simpsons Diversity	HBI	% 5 Dominant Taxa	% Chiro-nomid	% Into-lerant	% Hapto-benthos	% Scraper	% Shredder	WV MM	MAIS
C1R1	30.5m no cut	13	87.2	23	42.6	0.87	3.51	67.6	5.4	60.8	90.5	29.1	16.2	91.7	18
C2R2	Not Started	13	83.4	31	32.7	0.87	3.35	70.6	6.0	62.3	88.3	29.4	21.5	97.3	18
B1W1	Not Started	10	64.1	26	36.6	0.90	3.73	65.8	11.6	56.5	82.7	26.6	14.3	85.7	16
B1W2	In Progress	12	59.1	29	19.7	0.91	3.96	63.2	19.8	47.5	75.6	11.4	18.8	87.8	17
B1W3	15.3m no cut	14	53.7	27	16.5	0.94	4.47	73.5	11.0	35.0	86.3	9.3	9.6	87.4	15
B1W4	Not Started	9	57.7	24	27.7	0.93	3.36	66.3	6.6	59.7	86.7	20.9	19.9	83.0	17
B2w1	Not Started	11	83.1	25	33.1	0.88	3.39	68.3	7.6	61.7	86.1	20.5	20.1	91.2	18
B2w2	In Progress	12	54.8	29	31.8	0.85	3.56	71.3	31.9	55.8	62.7	16.7	17.0	82.6	17
B2w3	30.5m no cut	14	67.4	37	32.2	0.85	4.39	76.9	25.5	34.3	70.8	11.2	10.1	92.3	16
B2w4	Not Started	15	61.5	32	31.9	0.88	3.7	68.5	21.1	56.9	73.6	21.8	17.1	94.2	17
B3W1	In Progress	12	79.1	25	52.7	0.84	3.15	79.1	13.8	72.4	81.9	33.5	17.3	90.6	16
B3W2	Not Started	13	76.7	30	31.3	0.87	3.75	74.9	11.9	48.9	82.2	20.1	23.3	93.0	17
B3W3	Not Started	13	72.0	24	30.6	0.89	3.34	66.9	18.3	64.6	75.9	17.9	28.4	92.4	16
B3W4	Not Started	12	84.2	23	28.7	0.86	3.06	78.4	10.3	59.2	85.3	20.8	26.3	91.6	18
B4W1	4.5m no cut	6	36.7	16	0.0	0.67	4.1	92.8	51.5	35.7	40	0.0	24.9	53.8	6
B4W2	In Progress	12	35.3	24	9.0	0.79	4.99	84.1	36.7	26.8	38.8	0.5	9.2	73.0	10
B4W3	Not Started	11	72.9	24	7.3	0.86	1.94	70.8	12.0	78.0	82	0.0	62.8	89.8	12
B4W4	Not Started	13	73.7	26	12.1	0.88	2.36	69.8	16.7	72.3	76.2	3.9	50.3	94.7	15
CN1	Control	13	55.1	29	23.0	0.93	4.55	77.4	5.4	35.2	91.2	12.4	6.8	86.8	17
CN2	Control	12	52.8	25	19.8	0.85	3.58	65.2	33.4	54.3	61.3	9.9	26.4	83.6	15
CN3	Control	13	73.8	32	30.0	0.91	3.42	57.6	7.6	57.3	87.6	22.7	18.9	98.4	18
CN4	Control	9	62.0	22	23.7	0.80	2.53	88.4	22.5	66.3	68.4	0.0	45.9	79.5	11

Appendix G (continued): Macroinvertebrate metric data.
March 2004

Watershed	Treatment	EPT Taxa	% EPT	Taxa Richness	% Mayflies	Simpsons Diversity	HBI	% 5 Dominant Taxa	% Chiro-nomid	% Into-lerant	% Hapto-benthos	% Scraper	% Shredder	WV MM	MAIS
C1R1	30.5m no cut	15	96.4	27	74.6	0.79	2.85	84.1	2.6	81.6	96.7	34.7	9.5	94.1	16
C2R2	Not Started	14	86.8	36	43.5	0.87	2.42	67.2	7.2	79.0	91.2	23.5	34.4	101.7	18
B1W1	Not Started	14	81.1	36	44.5	0.87	2.95	69.7	11.9	70.4	86.4	28.9	23.1	98.5	18
B1W2	15.3m no cut	14	83.2	35	32.3	0.87	2.77	73.5	8.7	69.7	87.9	19.1	43.0	99.2	18
B1W3	15.3m no cut	12	77.8	27	55.3	0.88	3.57	71.6	8.9	56.4	88.5	26.0	10.7	91.4	18
B1W4	Not Started	13	86.4	28	63.5	0.83	3.51	75.3	4.1	58.4	94.5	33.8	11.2	92.5	17
B2w1	Not Started	14	86.4	33	51.3	0.90	2.85	59.7	8.0	72.0	87.1	21.5	16.8	102.2	18
B2w2	4.5m no cut	13	86.1	32	62.1	0.86	2.91	77.2	8.7	71.1	89.0	19.1	15.9	98.7	18
B2w3	30.5m no cut	13	79.1	35	53.0	0.88	3.21	67.3	6.8	67.5	81.1	26.5	15.1	93.4	17
B2w4	Not Started	14	82.9	35	48.7	0.86	3.04	63.0	7.1	72.8	88.9	33.0	21.6	100.2	18
B3W1	15.3m, 50% cut	13	89.2	32	34.9	0.83	1.95	77.1	6.0	84.7	91.5	19.6	51.1	98.4	17
B3W2	Not Started	16	82.2	36	38.1	0.90	2.44	59.3	4.2	78.6	93.9	27.3	33.5	105.4	18
B3W3	30.5m no cut	12	88.3	32	39.4	0.84	2.35	78.3	7.0	86.3	91.5	29.2	41.3	95.0	18
B3W4	Not Started	14	90.2	33	35.5	0.84	2.04	73.2	6.1	84.2	91.8	18.7	44.6	99.3	18
B4W1	4.5m no cut	9	62.9	20	2.5	0.75	3.03	92.9	33.3	61.2	64.0	0.4	53.4	71.0	11
B4W2	30.5m, 50% cut	9	74.5	22	15.2	0.69	3.81	92.4	20.2	61.1	76.3	0.5	53.5	74.4	11
B4W3	Not Started	11	85.5	23	2.3	0.71	1.75	87.3	3.5	91.5	94.5	3.0	86.5	87.4	13
B4W4	15.3m, 50% cut	15	75.3	41	13.9	0.77	2.20	81.2	19.3	73.4	77.3	4.5	56.2	97.2	13
CN1	Control	13	85.1	30	60.1	0.84	3.54	73.6	5.7	56.2	93.3	28.6	11.1	93.0	18
CN2	Control	14	80.7	33	31.8	0.87	2.37	65.6	9.9	73.7	83.9	15.8	44.9	99.7	17
CN3	Control	13	81.7	29	32.7	0.88	2.38	67.3	8.5	80.6	89.7	17.6	26.0	97.5	18
CN4	Control	12	84.0	23	4.3	0.70	1.70	89.1	4.8	87.5	93.3	0.9	84.2	87.8	13

Appendix G (continued): Macroinvertebrate metric data.
August 2004

Watershed	Treatment	EPT Taxa	% EPT	Taxa Richness	% Mayflies	Simpsons Diversity	HBI	% 5 Dominant Taxa	% Chiro-nomid	% Into-lerant	% Hapto-benthos	% Scraper	% Shredder	WV MM	MAIS
C1R1	30.5m no cut	16	93.2	29	52.3	0.77	3.82	81.5	1.5	59.1	95.1	38.2	4.6	94.0	16
C2R2	Not Started	12	82.1	27	41.9	0.88	3.55	74.0	8.4	48.3	87.8	20.6	18.2	94.8	18
B1W1	Not Started	12	62.1	30	23.2	0.89	3.44	63.8	20.4	56.6	68.1	16.6	26.8	89.4	17
B1W2	15.3m no cut	11	62.7	27	29.7	0.91	3.89	73.0	7.5	51.8	86.4	24.0	16.7	86.6	18
B1W3	15.3m no cut	12	50.5	27	19.4	0.94	4.54	81.6	1.8	34.2	95.1	15.5	12.2	81.3	16
B1W4	Not Started	10	76.4	26	54.4	0.71	3.26	78.7	10.5	73.3	82.1	57.4	5.7	84.0	16
B2w1	Not Started	14	82.8	27	35.9	0.88	3.45	69.4	10.4	59.7	85.1	20.9	19.0	96.2	18
B2w2	4.5m no cut	10	86.8	26	58.3	0.84	2.80	75.8	1.5	75.5	91.7	32.8	21.5	92.1	18
B2w3	30.5m no cut	14	89.1	30	59.9	0.81	3.57	78.2	2.9	63.3	92.8	37.2	7.7	96.8	17
B2w4	Not Started	12	83.5	29	62.5	0.72	3.49	74.4	3.7	71.7	88.9	51.5	10.8	91.1	17
B3W1	15.3m, 50% cut	12	80.4	27	43.8	0.89	3.38	62.5	6.6	61.2	86.8	27.8	15.5	92.7	18
B3W2	Not Started	14	82.8	33	39.2	0.87	3.76	72.8	6.2	44.9	87.9	18.0	16.4	99.3	18
B3W3	30.5m no cut	11	87.9	26	34.9	0.87	3.45	73.2	3.0	56.7	93.0	20.5	23.5	93.7	18
B3W4	Not Started	11	81.0	23	38.0	0.87	3.25	70.2	12.5	64.9	82.7	26.8	18.8	88.1	17
B4W1	4.5m no cut	6	50.0	14	0.0	0.71	3.29	94.1	45.9	49.4	50.0	0.0	37.6	57.9	6
B4W2	30.5m, 50% cut	13	27.7	31	10.6	0.67	5.45	87.4	54.3	15.2	29.8	1.4	8.8	65.6	11
B4W3	Not Started	11	61.7	19	7.0	0.88	1.70	75.6	8.3	83.4	88.6	0.3	66.3	85.5	15
B4W4	15.3m, 50% cut	12	81.3	26	35.0	0.90	3.03	64.4	8.3	63.3	83.1	12.6	25.5	95.1	17
CN1	Control	10	47.9	26	18.3	0.81	5.04	78.3	4.2	37.9	51.8	15.9	13.9	77.5	14
CN2	Control	13	79.3	28	44.4	0.86	2.83	67.3	8.4	74.5	81.3	30.7	26.3	96.0	16
CN3	Control	11	79.2	25	50.8	0.86	3.76	70.7	12.4	45.3	84.0	16.6	11.7	87.3	18
CN4	Control	10	61.5	25	15.5	0.81	2.33	87.6	20.5	68.3	74.3	0.3	52.1	81.8	11

Appendix G (continued): Macroinvertebrate metric data.
March 2005

Watershed	Treatment	EPT Taxa	% EPT	Taxa Richness	% Mayflies	Simpsons Diversity	HBI	% 5 Dominant Taxa	% Chiro-nomid	% Into-lerant	% Hapto-benthos	% Scraper	% Shredder	WV MM	MAIS
C1R1	30.5m no cut	15	91.8	30	54.9	0.86	3.02	0.7	5.2	69.1	90.7	25.5	20.1	98.0	18
B1W1	Not Started	17	90.1	29	54.0	0.87	2.98	0.7	5.1	65.4	88.7	24.5	20.9	98.0	18
B1W2	15.3m no cut	14	83.3	29	20.6	0.80	2.18	0.7	4.6	71.6	84.0	9.2	61.0	95.9	15
B1W3	15.3m no cut	13	83.4	24	58.2	0.83	3.25	0.7	9.7	68.7	86.6	38.4	10.3	88.0	18
B1W4	Not Started	16	79.4	27	53.8	0.83	3.78	0.8	17.1	51.0	81.3	30.6	10.7	88.7	17
B2W2	4.5m no cut	15	88.7	30	65.4	0.83	3.31	0.8	8.2	65.7	88.7	31.0	8.8	93.6	18
B2W3	30.5m no cut	17	90.8	33	69.0	0.69	3.29	0.8	5.1	76.4	89.6	53.8	6.4	91.7	17
B2W4	Not Started	17	84.6	31	45.1	0.87	3.29	0.7	7.4	64.3	85.4	22.6	28.7	99.3	18
B3W2	15.3m, 50% cut	17	91.6	30	53.7	0.82	3.08	0.7	5.6	74.2	91.0	42.5	11.8	96.9	17
B4W1	4.5 m no cut	12	46.1	20	1.8	0.68	3.54	0.9	49.4	44.5	45.5	2.4	33.9	63.6	11
B4W2	30.5m, 50% cut	12	53.6	24	5.6	0.69	4.31	0.9	42.1	47.4	53.6	0.3	38.4	64.2	10
B4W3	15.3 m no cut	12	88.5	18	3.7	0.78	1.57	0.8	4.4	85.1	85.5	6.4	72.0	86.2	14
B4W4	15.3m, 50% cut	14	62.9	27	14.2	0.80	3.33	0.8	33.1	53.4	62.2	2.6	41.0	78.9	12
CN1	Control	16	73.4	30	46.4	0.84	3.67	0.7	20.0	55.8	77.3	31.4	10.4	90.5	17
CN2	Control	17	82.5	37	39.1	0.90	2.92	0.6	11.8	64.0	81.7	16.7	29.6	101.7	17
CN3	Control	15	95.4	28	60.3	0.86	2.53	0.7	1.6	78.4	97.2	14.8	18.6	97.7	18
CN4	Control	13	83.1	28	3.3	0.66	1.59	0.8	3.9	80.4	68.3	0.6	62.8	84.2	13

Appendix H: Observations by watershed.

B1W2

The main skid trail leading to the log deck crossed the stream channel (stream is intermittent at the crossing). The skid trail was still bare earth more than a year after logging was completed and was a considerable source of sediment during rain and snow melt events. In addition, the skid trail served as a sediment dam during wet periods when the intermittent upstream channel had water in it. This watershed was also plagued with several slumps following logging. The stream has extremely steep side slopes that likely contributed to the hillside slumps. The slumps were a considerable source of sediment to the stream.

B2W3

A significant slump occurred during logging in which approximately a quarter acre of hillside adjacent to the stream gave way and slid into the stream. The sediment carried into the stream from this event created a dam that formed a large pool of dead water in what was a high gradient riffle/run section of stream.

B3W4

The main skid trail leading to the log deck on this watershed also crossed the stream channel. There was evidence that the loggers tried to construct a pole bridge over the stream that would allow them to skid over the stream while permitting the stream to flow through large diameter logs stacked parallel to the stream flow. Following logging large amounts of sediment filled the spaces between the logs halting the flow of water through the bridge. This sediment dam altered the stream from a high gradient riffle/run reach with a maximum width of approximately 0.5 m to a large pool approximately 30 m long, 15 m wide and 1.5 m deep at its deepest point.

All of the watersheds involved in this study showed signs of the areas logging legacy in the form of old stumps and grown over but still evident skid trails. In addition to these ubiquitous signs of past disturbance B3W4 exhibited highly eroded unstable banks and the remains of an old cross tie and rail system running down the stream channel.

B4W2

During logging operations on August 12, 2003 the stream was observed flowing saturated with sediment. The suspended sediment in the stream maxed out the 999 maximum NTU meter used to measure turbidity during this study. It was not raining at the time and other streams measured on the same day did not exhibit elevated turbidity levels. At the time the source of the sediment could not be determined due to the danger of entering an area being actively logged. Upon later return to the watershed it appeared that runoff from previous rain events had created a channel that flowed across a logging road and down a long section of skid trail before being diverted off the skid trail by a water bar. The water bar served its purpose but in this case the accumulated runoff was enough to cut a

channel through the organic layer of the forest floor allowing sediment laden water to flow into the stream.
Streams in block 4 typically had very low pH. In an effort to raise pH and improve conditions for brook trout *Salvelinus fontinalis* MeadWestvaco dumped piles of crushed limestone next to many of these streams at the upper most road crossing. During rain events the lime sand was washed into the streams and pH was increased with varying degrees of success. In addition to the desired effect of buffered pH levels the addition of lime sand creates a streambed dominated by fine sediments and increases the imbeddedness of coarse substrates. A high bedload of fine sediment drastically alters benthic habitat can impact macroinvertebrate communities (Waters, 1995). B4W2 was located downstream of the upper most road crossing and may have been impacted by liming. All other study sites in block 4 were located upstream of the upper most road crossing and therefore were not impact by lime dumping.

B4W4
Following logging an intermittent channel formed that flowed across the log deck and into the study stream. This channel appeared to be a significant source of sediment during wet periods as the log deck remained bare soil more than a year and a half after logging was complete. The confluence of the intermittent channel and the study stream was downstream of study site and therefore any impacts related to high sediment loads from the intermittent channel would not be represented in the data.

Vita

Jason James Corrao was born on May 3, 1980 in Prairie du Sac, Wisconsin to Ruthann Corrao and James Vincent Corrao. He graduated from Sauk Prairie High School in 1998. Jason completed his undergraduate degrees in Fisheries and Limnology and Biology from the University of Wisconsin Stevens Point in December of 2002. Immediately following his undergraduate work he entered the masters program at Virginia Polytechnic Institute and State University and completed his MS in Fisheries and Wildlife Sciences in August, 2005.