Effects of Forested Streamside Management Zone Widths and Thinning on Carbon Dynamics and Benthic Macroinvertebrates for Pine Plantations in the Piedmont of Virginia

Erica Fritz Wadl

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W. Michael Aust, Co-Chairman
John R. Seiler, Co-Chairman
W. Lee Daniels, Committee Member
C. Andrew Dolloff, Committee Member

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ABSTRACT

To protect the integrity of the United State's waters, the Federal Water Pollution Control Act of 1972 promoted the establishment of Best Management Practices (BMPs) for forestry. A commonly used BMP is the reservation of Streamside Management Zones (SMZs). In this study the effectiveness of three different SMZ widths, 30.5 m (100 ft), 15.3 m (50 ft), and 7.6 m (25 ft), as well as thinning in 15.3 m SMZs were studied. The objectives of the study were to determine the effects these SMZ treatments had on carbon pools, carbon fluxes and environmental conditions in the SMZ. The benthic macroinvertebrate populations present within the stream were also examined because of their relationship to ecosystem carbon dynamics. Carbon storage in plant communities, litter layer, soil (upper 10 cm), and total organic carbon present (TOC) within streams were measured and quantified. Total CO_2 efflux and the major environmental drivers of soil CO_2 efflux, soil moisture and soil temperature, were monitored along a single transect within each SMZ. This study showed that carbon dynamics and stream biota (benthic macroinvertebrates) were not adversely effected by more narrow SMZ width and thinning within the SMZ. SMZ width did affect soil temperature, one of the environmental drivers affecting soil respiration. Based on these short-term results a 15.3 m SMZ with thinning or without thinning appears adequate to prevent changes in ecosystem function and water quality for forest applications.

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Introduction

Study Justification

Silvicultural activities can impair water quality by increasing sediment and nutrient loading to streams or watersheds (Binkley and Brown 1993). Forestry best management practices (BMPs) have been implemented in the United States in an attempt to improve or maintain water quality as required by the Federal Water Quality Control Act of 1972 (Aust and Blinn 2004). One BMP recommendation that specifically targets the protection of streams and other bodies of water is the implementation of streamside management zones (SMZs) (Castelle *et al.* 1994).

SMZs are defined as "area[s] of reduced management activity on both sides of the banks of perennial and intermittent streams and bodies of open water where additional caution is used in carrying out forest practices in order to protect bank edges and water quality" (Virginia Department of Forestry 2002). SMZs are of general interest to natural resources managers as well as environmental agencies due to their important influence on water quality, habitat and carbon storage/transfers. SMZs have many potential functions including the trapping of sediment and nutrients, stream bank stabilization, stream temperature control, and protection of the forest floor (Governo *et al.* 2004). In addition to the positive effects on the physical and chemical properties of stream water, SMZs can also benefit terrestrial and aquatic organisms. SMZs maintain areas of mature habitat and species diversity, provide linear passages between habitat types, provide stream habitat diversity with the occasional addition of large woody debris, and provide detritus (carbon) inputs to streams to support aquatic food chains (Lemly and Hildebrand 2000). Benthic macroinvertebrate populations are used as an index of the biological integrity and health of the stream due to their acute response to both pollution and habitat alteration (Vowell 2001). Therefore, benthic populations can be used to measure the overall effectiveness of SMZs.

SMZs may also have different carbon storage and efflux patterns than the adjacent and more intensively managed landscape. With the recent increased focus in forest carbon sequestration and carbon credits, carbon dynamics within riparian zones are also of increasing interest (Cao and Woodward 1998).

Objectives

The overarching goal of this project was to determine the influence of SMZ width and thinning regimes within the SMZ on protecting stream water quality in harvested areas in the Piedmont region of Virginia. Previous studies have been conducted in this study region, focusing on erosion, sediment, and nutrient fluxes as influenced by SMZ treatments (Lakel *et al.* 2006, Walker Easterbrook *et al. 2003*). In this study, we examined the influence of various SMZ treatments (varying widths, thinning intensity) on carbon fluxes, carbon pools, and benthic organisms. These criteria were then used to assess changes in overall SMZ function.

Specifically I examined the influence of SMZ width and harvest level on:

1. aboveground biomass, litter layer, soil carbon (10 cm depth), TOC present within the stream, and total soil CO_2 efflux,

2. soil moisture and temperature, and

3. benthic macroinvertebrates.

Literature Review

Riparian Forests

Implementation of the Federal Water Pollution Control Act (FWPCA) in 1972 and associated state best management practices (BMPs) have increased interest in and appreciation for the important roles riparian areas play in forest ecosystems (Virginia Department of Forestry 2002). Protected riparian areas are effective non-point source pollution controls for silvicultural operations. Streamside forests protect water bodies from non-point source pollution and erosion, as well as provide a functional habitat for a variety of organisms (Welsch 1996). Riparian areas are defined as "being a three dimensional entity that includes the canopy, both terrestrial and aquatic ecosystems, as well as the area that drains into the water" (Illhardt *et al.* 2000).

Streamside Management Zone Regulations

Following passage of the FWPCA in 1972, BMPs were implemented by many state forestry organizations and land managers to comply with the new water quality standards. Within the forest industry, SMZs were a common BMP adopted to protect water quality (Alabama Forestry Commission 1993, Georgia Forestry Commission 1999, North Carolina Division of Forest Resources 2006, South Carolina Forestry Commission 1994, Virginia Department of Forestry 2002). Throughout the United States, individual state forestry agencies developed criteria for SMZ width and amount of harvesting suggested within a SMZ. Recommended SMZ widths range from 9 to 49 m and nearly 80% of the jurisdictions within the United States allow some degree of harvesting within the buffer zone (Lee *et al.* 2004). In some Piedmont states (Georgia and South Carolina), the width requirement or recommendation is based upon the slope of the riparian area (Table 1).

In Virginia, a minimum 15 m SMZ is recommended along intermittent and perennial and on certain ephemeral streams (Virginia Department of Forestry 2002). SMZs are required in all the counties immediately adjacent to the Chesapeake Bay. In addition, harvesting up to 50% of the basal area or up to 50% canopy removal can occur within a SMZ.

Table 1. SMZ recommendations in five states in the Piedmont region (adapted from: Alabama Forestry Commission 1993, Georgia Forestry Commission 1999, North Carolina Division of Forest Resources 2006, South Carolina Forestry Commission 1994, Virginia Department of Forestry 2002).

State	Minimum width (m)	Maximum harvesting level
Alabama	9.1	50% canopy cover
Georgia	12.2-30.5 *	11.5 m^2/ha or 50% canopy cover
North Carolina	15.3	< 20% bare ground following harvest
South Carolina	12.2-48.8 *	11.5 m^2/ha remains following harvest
Virginia	15.3	50% basal area or 50% canopy cover

* slope class dependent

Streamside Management Zone Functions

Habitat

Riparian zones provide a continuum between various terrestrial and aquatic environments. Certain individuals and species preferentially inhabit riparian areas as opposed to adjacent upland habitat areas due to the variety of woody vegetation and large woody debris present. This variety of vegetation provides a source of both food and cover for animals seeking refuge. In addition, these riparian areas serve as a corridor for migrating animals (Naiman and Decamps 1997). Within the northeastern United States, habitat for fifty-one species of birds and mammals are closely tied to riparian areas (DeGraaf and Yamasaki 2000). Additionally, many amphibians and reptiles rely on riparian forests for a portion of their life cycle. A study conducted in western North Carolina by Petranka and Smith (2005) found that eleven of the nineteen species of salamanders studied preferred to inhabit areas directly adjacent to a stream.

Riparian areas also influence fish populations. Jones *et al.* (1999) showed that the removal of riparian forests decreased habitat diversity as ripples and pools became filled with fine sediment. This increase in sedimentation due to the absence of riparian forests along southern Appalachian streams caused a shift in fish populations. Fish that either do not guard their eggs or rely on swift moving waters declined, while fish that guard their young increased. Riparian forests also provide large woody debris (LWD), a material preferred by some species of fish for cover. Flebbe (1999) evaluated trout streams in the mountains of North Carolina and reported that trout favored pools with at least two pieces of LWD while pools lacking a piece of LWD had low numbers of trout, apparently due to the lack of cover.

Macroinvertebrates, a food source to fish and other animals, favor areas with woody debris (Drury and Kelso 2000). This is because LWD impacts the pool:riffle ratio, specifically, causing more pools. In addition to modifying habitat, logging changes the energy source of the stream from allochthonous (outside the stream) to autochthonous (within the stream) (Wallace *et al.* 1998). This habitat modification results in a shift in the macroinvertebrate community to one dominated by collectors-gathers rather than shredders and scrapers (Lemly and Hilderbrand 2000). Because many animals feed upon macroinvertebrates, the shift in this community structure can result in a larger community change within the entire riparian area. Though the addition of LWD can be beneficial to the stream ecosystem, too much organic material can be detrimental to aquatic life. During the breakdown of this organic material, oxygen is used, resulting in lower dissolved oxygen (DO) levels. Levels below 5.5 mg/l can cause stress to aquatic life (Lakel 2008).

In addition to providing LWD, riparian forests also provide a particular light and temperature environment that is preferred by certain animals. Riparian vegetation protects the stream from sunlight, assisting in regulating stream temperature. Temperature is an important physical characteristic since many fish and other aquatic organisms are sensitive to temperature (Richards and Hollingsworth 2000). Dignan and Bren (2003) noted this and determined that light penetration is negatively affected after logging due to the edge effect. Detectable changes in light penetration existed 70-100 m into the riparian area (Dignan and Bren 2003). Salamanders, which are considered bioindicators of environmental quality (Vitt *et al.* 1990) and a dominant predator in headwater streams (Petranka and Smith 2005), typically require a riparian area during at least one of their life cycle stages. Clear-cutting can cause surface drying and increased water temperature, which in turn, increases physiological stress of salamanders (Ash 1995).

Water quality

SMZs may protect water quality in a variety of ways. They provide shade, stream bank stability, and mitigate runoff from the surrounding area. By slowing runoff, riparian forests regulate the amount of nutrients, sediments, and particulate organic matter that reach the stream (Palik *et al.* 2000). Specifically, the vegetation in forested riparian areas assists in trapping sediment and in the accumulation of organic matter (Klapproth and Johnson 2000). Fertilizer applications are a common practice in silvicultural operations. Forested riparian areas assist in the transformation and reduction of these nutrients, such as nitrogen, by providing areas of anoxic conditions enhancing denitrification (Fisher and Binkley 2000). Also, the vegetation present within SMZs uptake these nutrients. This uptake of nutrients is important to the entire system as it is thought to reduce diffuse-source pollution (Peterjohn and Correll 1984).

Reduction of nutrients reaching the stream also assists in the prevention of eutrophication in downstream bodies of water such as the Chesapeake Bay. Eutrophication occurs when nutrients such as phosphorus and nitrogen increase algal growth causing algal blooms that form a mat. This mat of algae prevents sunlight from reaching the stream bottom therefore killing any oxygen producing vegetation leading to an oxygen depleted water body deprived of many life forms (Welsch 1996). This increase in algae also leads to an increase in biological oxygen demand (BOD) after the algae dies and begins decomposition.

Another mechanism by which SMZs enhance water quality is by supplying the stream with organic matter. Organic matter occurs in many forms: as leaves, twigs, whole trees, and multiple levels of decomposed matter. When large woody debris occurs in streams, an organic debris dam can form. Organic debris dams play an active role in regulating the export of particulate organic matter from the watershed (Bilby 1981). Bilby noted that once a dam is removed the export of dissolved organic carbon, fine particulate matter, and coarse particulate matter increased. Organic matter provides energy, nutrients and habitat to streams (Dolloff and Webster 2000).

Benthic Biomonitoring

Benthic macroinvertebrates tend to be sensitive to disturbances and hence are often used as indices of stream health. Stone and Wallace (1998) found that logging could potentially alter a site by changing the energy balance of the stream from allochthonous to autochthonous. This change in energy could cause a modification in benthic macroinvertebrate community structure as defined by feeding preferences.

The Ephemeroptera (Mayflies) + Plecoptera (Stoneflies) + Trichoptera (Caddisflies) (EPT) index is often used in bioassessment because these three orders of insects are particularly sensitive to environmental disturbances (Wallace *et al.* 1996) Another common index is the Hilsenhoff Biotic Index (HBI) (Hilsenhoff 1987). The HBI is used to assess low dissolved oxygen caused by organic loading in streams. The HBI assigns a tolerance value of 0-10 to particular orders of arthropods, with 0 being assigned to those that are the most intolerant to organic stream loads (Hilsenhoff 1987). Similar to the EPT, the HBI uses benthic macroinvertebrates to evaluate water quality. Corrao (2005) used these and other indices to evaluate the influence of SMZ width on biotic communities demonstrating a 30.5 m SMZ maintained higher water quality as determined by biotic indices than did a 4.5 m SMZ.

Carbon Pools and Fluxes

Carbon is the essential building block for all organic molecules and all living things. Carbon moves between various sinks and sources through photosynthesis and respiration, composing the carbon cycle (Molles 1999). Some carbon is readily available (source), while some carbon is trapped for an extended period of time (sink). Within the terrestrial ecosystem, carbon moves through the vegetation, soil, and litter layer via photosynthesis, respiration, and decomposition (Cao and Woodward 1998). Carbon affects many of the activities within the riparian area and can be used as an index of both water quality and SMZ function. According to Giese *et al.* (2003), riparian areas have high potential to store carbon due to their relatively high rates of productivity and soil water saturation. Studies have been developed to determine total carbon within an area (Giese *et al.* 2003, Governo *et al.* 2004, and Trettin *et al.* 1999), but little research has been conducted to determine how silvicultural management practices may impact the flow of carbon through SMZs. A study conducted by Governo *et al.* (2004) compared three different harvest treatments within a 15 m SMZ; no harvest, partial harvest (50% basal removal of low-grade hardwoods and pines), and clear-cut. The Litterfall, leaf litter decomposition, understory vegetation, soil temperature and water chemistry were measured 9-14 months post harvest. This study showed that SMZs with either no harvest or partial harvest ensured that sufficient carbon inputs from litter layer were maintained to support stream health functions by maintaining adequate carbon inputs into the intermittent stream (Governo *et al.* 2004).

Aboveground Carbon

According to Turner *et al.* (1995), aboveground biomass is the second largest carbon sink present in forest ecosystems within the United States, following soil carbon, which is the largest sink. Photosynthesis is the mechanism through which plants remove CO_2 from the atmosphere and utilize this carbon in the building of their own biomass. Consequently, net primary productivity is a measure of the total amount of carbon or energy that plants incorporate into their tissues (Solomon *et al.* 1999).

Photosynthesis and net primary productivity are so closely linked that some researchers predict an increase in forest biomass due to the current increase in atmospheric carbon dioxide (Solomon and Kirilenko 1997). In an attempt to quantify this carbon pool and flux, researchers measure plant biomass. Ryan (1991) found a method through which a carbon budget can be estimated simply by determining plant biomass. Though photosynthesis is an important driver of biomass accumulation, other factors affect the amount of biomass accumulated, including the amount of available nutrients and water.

Litter Layer

The litter layer is a key component in the forest ecosystem as litter is one of the main contributors of aboveground carbon and nutrients to the forest floor. Litterfall quantity can also be used to predict both soil respiration and root respiration (Raich and Nadelhoffer 1989). This relationship indicates that the same environmental factors that influence aboveground carbon allocation impact belowground carbon allocation. Litterfall fluctuates with the seasons; Day (1979) found that the litter layer pool declines from December to August. This seasonal fluctuation is explained by the correlation between aboveground biomass allocation and litter layer accumulation (Trettin *et al.* 1999). A study conducted in Norway spruce stands by Vesterdal *et al.* (1995) found that both the litter layer and humus layer were greatest in unthinned plots when compared to plots that had undergone a thinning regime. It is believed that this decrease in litter layer in thinned plots was a result of a more open canopy which then resulted in more favorable moisture and temperature characteristics for decomposition. The litter layer may be a source of atmospheric CO_2 if litter decomposition exceeds accumulation rates. Any change in this equilibrium also indicates a perturbation to the system. This study also concluded that when litter accumulation is adversely affected by harvesting, the accumulation of carbon in both the litter layer and soil horizons decreases. A study by Roig *et al*. (2005) also found that litter fall was significantly less in heavily thinned pine stands as compared to unthinned stands. However, this particular study found that the thinning effect on litter fall disappeared five years after the thinning. Another study that occurred in Norway spruce stands, did not find a thinning effect on litter layer approximately two years after the thinning occurred (Slodicak *et al*. 2005)

Litter also moves within, into, and out of riparian areas. McDowell and Fisher (1976) found that within hardwood ecosystems, litter that is blown into the stream accounted for 21% of the total litter entering the stream. Most of this movement of litter occurred during the fall months, after peak litter fall. According to France (1995), this lateral transport of litter increases with slope, exposure to precipitation, and absence of ground cover. This same study found that deciduous leaves transport more readily than evergreen needles/leaves. Giese (2001) found that during flood events more litter was deposited into riparian areas than moved out of the riparian area and into the stream.

Soil Carbon

The soil carbon pool is the largest terrestrial carbon pool or sink and accounts for nearly 50% of the carbon found in United States timberlands (Turner *et al.* 1995). Aboveground litterfall/biomass decomposition and decomposing roots contribute to this pool. Consequently, a major factor influencing soil carbon is the rate of decomposition, or the rate at which carbon is recycled back into the soil. Many factors affect decomposition, including the microbial properties of the soil and other nutrients present within the organic matter.

A study by Marquez *et al.* (1999) involving riparian areas focused on changes in soil organic matter after a vegetation buffer had been established in row crops. Their study showed that after five growing seasons, the amount of soil organic matter in the upper 35 cm of soil increased by 8.5 percent after a buffer of poplars had been established in an area that had been historically used for row crops including the riparian area. This study illustrates the importance of the presence of vegetation as a contributor to the soil carbon pool. Another study by Selig *et al.* (2008) that occurred in loblolly pine plantations found that soil carbon concentrations were greatest in thinned stands when compared to stands 14 years after thinning. It is believed that this difference can be attributed to the contribution of organic matter and the decomposition of roots from harvested trees. Understory growth of shrub species may occur after thinning as well, adding organic matter and consequently increasing soil carbon concentration (Wetzel and Burgess 2001). This research is examining how soil CO_2 efflux (F_s), of which one component is microbial respiration (see below), is influenced by SMZ treatments.

Soil Respiration

The soil serves as both a sink and a source for carbon. Carbon is released from the soil in the form of CO_2; this process is often referred to as soil respiration or soil-CO_2 efflux (F_s). Microbial respiration and root respiration are the main contributors that drive F_s. Soil respiration rate is positively correlated with mean annual air temperatures as well as with mean annual precipitation (Raich and Schlesinger 1992). In addition, in a study done in the Virginia Piedmont by Wiseman and Seiler (2004), soil temperature, soil moisture, stand age, and proximity to the tree were the main factors influencing soil respiration.

Microorganisms (i.e. bacteria, fungi, mold, and algae) and soil fauna (arthropods) inhabit the soil and decompose plant material. The majority of microorganisms and soil fauna are located in the upper 60 cm of the soil and require moisture to live (Plaster 1997). Both arthropods and microorganisms contribute to soil respiration. A study by Yi and Moldenke (2008) illustrated that the abundance of arthropods decreased with thinning intensity. This decrease in arthropods correlated with decreasing litter moisture. According to Borken *et al.* (2003), microbial respiration of the 0 horizon decreases as soil moisture decreases, illustrating the effects of soil moisture and seasonal variation. Borken *et al.* (2003) hypothesized that the decomposition rate of the litter layer is influenced by moisture content. During this same study, soil respiration spiked immediately following a rain event, illustrating a wetting/drying effect. Other studies have shown that soil moisture may or may not be related to soil respiration.

Bolstad *et al.* (2004); Fang and Moncrieff (2001); and Coleman *et al.* (2002) showed no relationship between soil moisture and F_s. However, Bouma and Bryla (2000) estimated that a change in soil moisture can affect F_s, especially in fine textured soils.

Seasonal trends have also been found with F_s; Euskirchen *et al.* (2003) found higher values of F_s in late summer than in the fall, correlating with soil temperature.

In addition to seasonal variation, harvesting also affects F_s. On clear-cut plots, soil respiration increased by 16 percent when compared to uncut plots (Lytle and Cronan 1998). This increase in F_s might be attributed to the change in microclimate. A clearcut can cause more favorable conditions for decomposition by increasing soil temperature and soil moisture content. Selig *et al.* (2008) concluded that thinning will increase soil temperature and consequently F_s. But, Striegl and Wickland (1998) demonstrated that F_s rates decreased after clear-cutting probably because of the death of tree roots along with the disruption in the soil layer. Santantonio and Santantonio (1987) found that both live fine roots (<1mm) and live small roots (1-5mm) decreased by nearly 50% two years following a thinning. In a review of soil respiration studies, Hanson *et al.* (2000) illustrated that as much as 90 percent of soil respiration can be attributed to root respiration, illustrating that root death after silvicultural activities could be the cause of reduced soil respiration. Consequently, Wiseman and Seiler (2004) showed that soil F_s decreases with increasing distance from a tree.

Materials and Methods

Study Site

I examined the effects of width and thinning in SMZs on riparian carbon dynamics, environmental conditions, amount of instream carbon, and stream benthic communities in 16 watersheds in the Virginia Piedmont. The watersheds contained first order headwater streams that were ephemeral or intermittent and watershed sizes averaged 27 ha and ranged in size from approximately 6 to 72 ha in size. All were located on industrial forest land owned by MeadWestvaco and were located in Buckingham County (Figure 1). Overstory vegetation within the SMZs predominantly consisted of planted 23-25 year old loblolly pine (*Pinus taeda)* and native hardwoods such as *Acer rubrum* (red maple)*,Quercus alba* (white oak), *Quercus prinus* (chestnut oak) and *Carya glabra* (pignut hickory). Buckingham County averages 106.7 cm of precipitation per year with average temperatures ranging from 3.3˚C to 20.7˚C (Wiseman 2001).

The dominant soil series mapped in the riparian zones of the study area is the Chewacla series (Fine-loamy, mixed, active, thermic Fluvaquentic Dystrudepts) consisting of very deep, moderately permeable, somewhat poorly drained soils on flood plains. The dominant soil series of the upland areas consisted of well drained Cecil and Appling series (fine, kaolinitic, thermic Typic Kanhapludults).

This study was part of a long term study with pre-harvest data collected and analyzed in 2002 (Easterbrook et al. 2003). SMZ treatments were installed and adjacent timber was clear-cut in the summer and fall of 2003, and in 2004, The majority of the watersheds outside of the SMZ treatments consisted of two year old loblolly pine plantations that were established following clearcut harvest and subsequent drum chopping, prescribed burning, and herbicide applications.

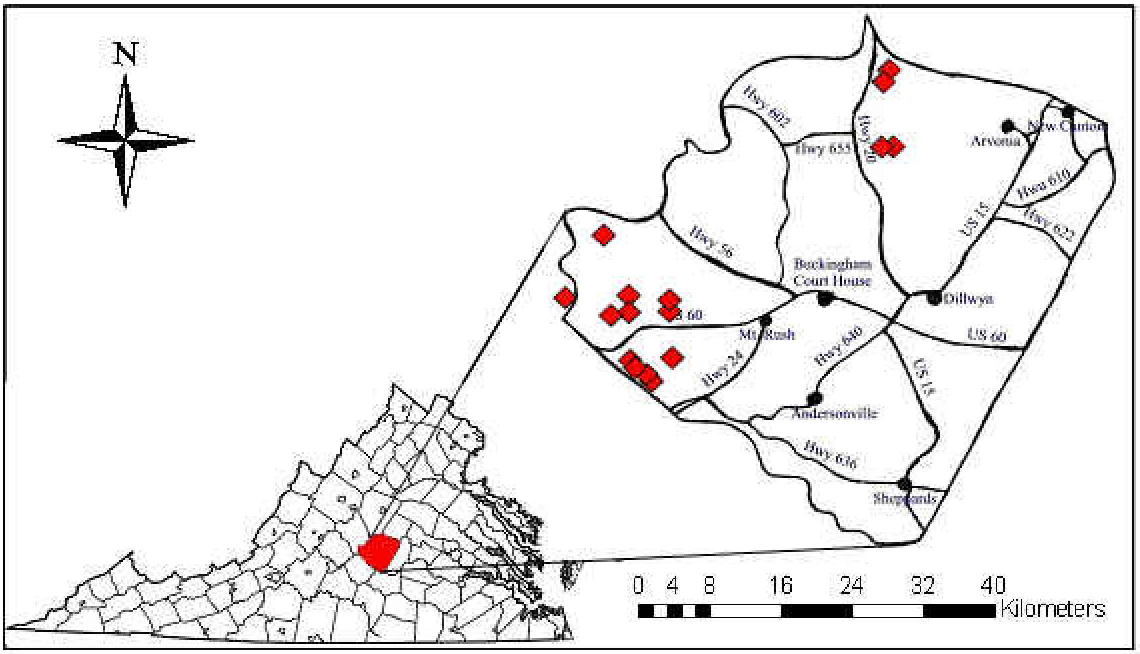

Figure 1. Map of study area showing the approximate distribution of the 16 study sites (◆)and major highways of Buckingham County, Virginia (adapted from public domain http://commons.wikimedia.org/wiki/File:Map_of_Virginia_highlighting_Buckingham_County.svg)

Treatments

Four SMZ treatments were installed: a 30.5 m (100 ft) no harvest SMZ, two 15.3 m (50 ft) SMZs, one with 50% thinning and one with no thinning, and a 7.6 m (25ft) SMZ with no harvest. Thinning was defined as the removal of approximately 50% basal area. The watersheds were blocked based on geological and site characteristics. Not all treatments were installed properly when commercially harvested which resulted in unequal replication due to logger decisions. The number or replications for each treatment is shown in Table 2.

Table 2. Number of replications per treatment for the entire study.

Treatment	Replications
7.6 meter no thin	3
15.3 meter no thin	6
15.3 meter thin	3
30.5 meter no thin	4

Field Work

I sampled the aboveground biomass, litter layer, and soil carbon pools, soil respiration and stream total organic carbon fluxes. The original study was designed to include three transects along the entire stream length, for this study only one transect was sampled. This single transect was selected for ease of accessibility. Systematic subplot samples were collected at the edge of the clearcut, middle of the SMZ, and at the creek side to capture the range of variability (Figure 2). The order in which the subplot samples within the transect were collected was randomized for each treatment to eliminate any variability created by sampling order. In addition carbon, benthic sampling was conducted in those streams that contained sufficient water for sampling.

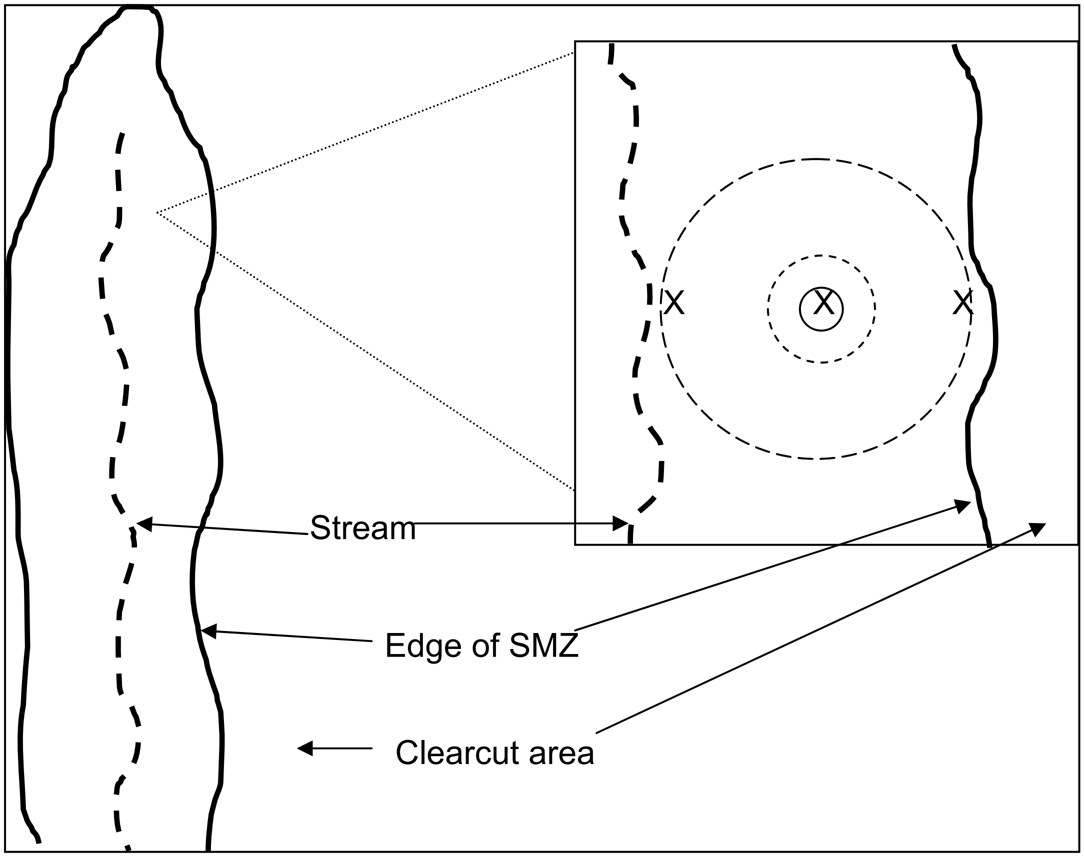

Figure 2. Schematic of sample transect, illustrating the nested overstory (large circle) midstory (medium circle), groundcover (small circle), and soil respiration-temperature-litter samples (x's) used for estimation of soil carbon pools and fluxes.

Aboveground Biomass

Standing Biomass

Aboveground biomass sampling was performed for three strata: overstory, shrubs, and groundcover (Figure 2). The center of the sampling plots was positioned in the middle of one side of the SMZ. For the 7.6 m wide treatment, the center of the plot was placed in the stream to ensure that the plot did not extend outside of the SMZ. Overstory vegetation included trees with a diameter greater than 9.1 cm (3.6 inches) at 1.3 m aboveground (DBH). A 1/50 hectare (8.01 m radius) circular plot was established, and all vegetation meeting these requirements were measured for total height and DBH. Shrubs were defined as any vegetation with a diameter less than 9.1 cm at DBH. For shrubs, a circular plot with a 2.5 m radius (1/500 hectare) was established along the same location within the transect and all species were measured for height and DBH. A 1/2500 hectare (1.1 m radius) ground cover plot was established. All vegetation within the plot was clipped to the ground line and transported to the lab for dry weight determination.

Data collected in the field for both the over story and shrub strata were used in conjunction with existing biomass equations to estimate aboveground biomass within each treatment (Clark *et al.* 1986 and Hauser 1992). On average, carbon accounts for about 50 percent of tree dry weight (Birdsey 1992). Consequently, biomass estimates were converted to carbon by multiplying by fifty percent. The groundcover samples were brought back to the lab and dried at 85º C in an oven and then weighed to determine biomass and carbon content.

Litter Layer

The litter layer was destructively collected twice during the year, once in late summer (September 2005) and again following litter fall in the winter (January 2006) at three locations (creekside, middle of SMZ, and upper edge of SMZ). Sampling periods were chosen in an effort to capture seasonal variability. Litter samples were taken at the same time and location as a measure of total CO_2 efflux. A soil CO_2 efflux (F_s) measurement was taken (see below) and the litter from the area (506.5 cm^2) underneath the efflux chamber was removed. The entire O horizon (litter, fragmented, and humus layers) was collected and placed in a paper bag.

Litter samples were brought to the lab and dried at 85º C in an oven to a constant weight and then weighed. These dry samples were then placed in muffle furnace at 380º C for 24 hours to correct for mineral soil contamination. Litter weight was calculated as oven dried weight minus ash weight obtained from the muffle furnace (Wisemen 2001).

Belowground Biomass

Soil

In November 2005, ten push tube soil samples were taken to a depth of ten centimeters at each of the three sub-sample locations (creekside, middle, and upper edge) along each transect. These ten samples were taken along a 3.05 m linear transect running perpendicular to the original sub- sample location. These samples were mixed together and a single composite sample was brought to the lab to determine carbon content. A bulk density soil sample was also taken to a depth of 10 cm using a double-cylinder bulk density corer (Blake and Hartge 1986).

Soil samples were brought back to the lab and dried at 105º C for 24 hours and weighed in order to determine bulk density (Blake and Hartge 1986). These samples were then ground using a mortar and pestle and passed through a 0.64 cm mesh to separate out the coarse fragments (Chen and Xu 2004). These coarse fragments were then weighed in order to determine percent coarse fragment content. The separated soil was analyzed for percent carbon content using an Elmentar varioMax CNS analyzer (Elementar American Inc., Mt. Laurel, NJ).

Carbon Fluxes

Soil Respiration

From May 2005 through January 2006, total soil CO_2 efflux (F_s), soil temperature, and soil moisture were measured on five different dates in an attempt to capture seasonal variation. Measurements were taken along the same transects and locations as the litter measurements. The sampling order of these three locations within the transect was randomized. A 15 cm Digi-Sense® thermocouple thermometer (Cole-Parmer, Vernon Hills, IL) was utilized to determine soil temperature down to a depth of 15 cm. Soil moisture was measured using a HydroSense portable TDR system (Campbell Scientific, Inc., Logan UT) to a depth of 13 cm.

Total F_s was measured using a Li-Cor 6200 infrared gas analyzer (Li-Cor Inc., Lincoln, Nebraska) with a dynamic closed cuvette chamber system. The system was constructed of PVC pipe walls, plexi-glass top, and a stainless steel edge on the bottom to ensure a tight seal once placed on top of the sampling location. The internal diameter of the chamber was 25.5 cm and the height was 13.5 cm. Total volume equals 6744 cm^3. Plastic tubing (0.32 cm diameter) was used to attach the cuvette chamber to the infrared gas analyzer. Air enters the cuvette chamber through an input hose on the side of the chamber. This air is then diffused through the chamber via a perforated hose that runs along the interior wall of the chamber. Air then leaves the system through a hose that is located at the top of the chamber through the plexi-glass top (Selig *et al.* 2008).

Before each sampling date, the system was calibrated in the lab by running a known CO_2 concentration through the system. Care was taken to be sure no living plant material was in the chamber and that CO_2 concentrations were at or near ambient levels near the ground. After CO_2 concentrations were found to be steadily rising (typically occurring in less than 1 minute) a measurement period of 30 seconds was used to estimate total F_s. Rates were calculated by the LiCor 6200 using the following equation:

$$F_s = (\Delta C/\Delta t)(PV_t/RT)/\text{Area}$$

Where C = CO_2 , t = time (30 s), P = atmospheric pressure, V_t = system volume, R = universal gas constant, and T = temperature.

Total Organic Carbon (TOC) in Stream Water

From August 2004 through February 2006, stream water grab samples were taken on five different dates in an attempt to capture seasonal variation. Samples were taken downstream of the sampling transect but still within the SMZ treatment area. The grab samples were located further downstream in order to ensure the presence of water. Three 25 ml bottles were filled, by being placed in the stream with the opening facing upstream to capture flowing water. The bottles were then placed on ice and brought back to the lab where they were placed in the freezer until further analysis could be performed.

The water samples were analyzed for total organic carbon (TOC) using Persulfate-Ultraviolet Oxidation (Siever DC 80 TOC analyzer, Boulder, CO). Sampleswere purged prior to analysis using oxygen to remove inorganic carbonates.

Bethic Macroinvertebrate Sampling

Benthic macroinvertebrate sampling occurred in January 2006 2-3 days after rainfall to ensure water was present during sampling. A D-frame macro invertebrate sampling net was utilized to sample a 50 m stretch of stream. Along this 50 m stretch, a sub-sample was collected every 2 m. The benthic samples were then washed in a 500μm mesh bottom wash bucket and preserved in 90% ethyl alcohol.

The macroinvertebrate samples were sorted and separated from gravel and woody debris pieces that may have been collected. These samples were then stored in 95% Ethanol (EtOH). These sample specimens were identified to family by Stephen Hiner of Virginia Polytechnic Institute and State University. Samples specimens were then analyzed using ten macroinvertebrate indices; 1. Tricoptera (caddisfly) abundance, 2. Plectopera (stonefly) abundance, 3. Ephemeroptera (mayfly) abundance, 4. % mayflies present, 5. Ephemeroptera, Plecoptera, and Tricoptera (EPT) abundance, 6. % EPT, 7. EPT richness, 8. overall abundance of macroinvertebrates, 9. taxa richness, and 10. Hilsenhoff's Biotic Index (HBI). The first seven of these metrics assesses a stream based on the presence of three orders of benthic macroinvertebrates that are sensitive to both pollution and any environmental disturbance (Wallace *et al.* 1996). The HBI index involves rating species based upon their toleration to low dissolved oxygen that is caused by organic loading in streams (Hilsenhoff 1987). All ten metrics are used to assess water quality based on the presence or absence of specific benthic macroinvertebrates.

Statistical Analyses

An Incomplete Block Design was used for analysis of variance (ANOVA) to determine if differences existed between treatment data (SAS v. 9.1, Cary, NC). Differences in means were determined using Tukey's studentized range test (HSD) at the alpha (α)= 0.1 significance level. The three different biomass strata were analyzed separately. For data that was not normally distributed, a log transformation was performed and the back transformed means are reported. For F_s, soil moisture, soil temperature, and TOC data, a repeated measures analysis was used to evaluate significant differences through time caused by the various treatments.

Results

Aboveground Carbon

Standing Biomass

No statistically significant differences in total standing biomass carbon were found between the various SMZ treatments (Figure 3, p=.30). This was likely due to the large variation associated with the uneven and low sample size (n=3 for 15.3T and 7.6NT). However, it is interesting to note that the 15.3 m thinned treatment did have approximately half the biomass of the 15.3 m unthinned treatment and the lowest overall total standing biomass carbon.

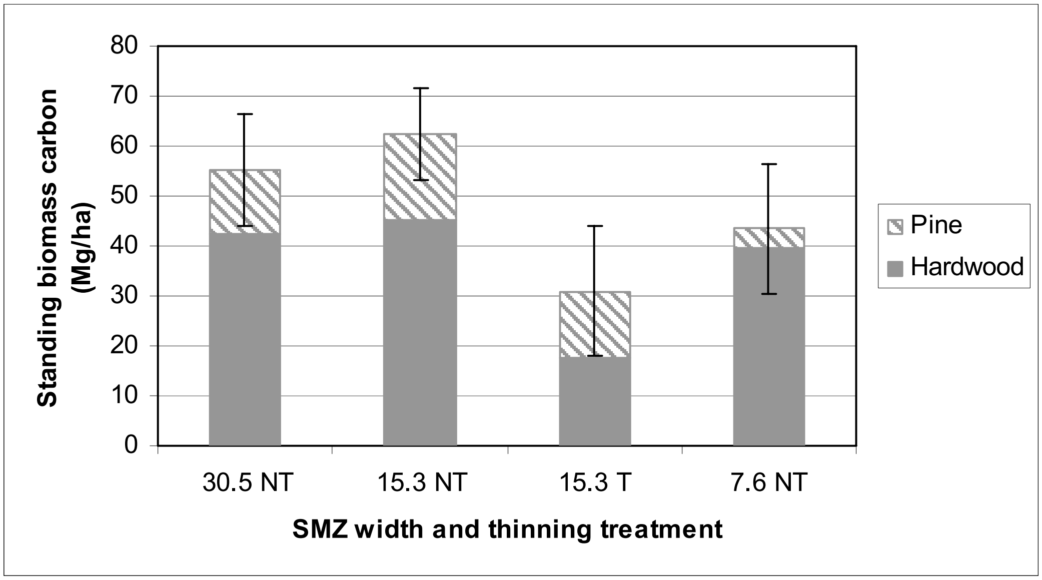

Figure 3. Aboveground standing carbon, pine and hardwood, as influenced by streamside management zone treatment (numbers (e.g. 30.5) are SMZ width in meters, NT=no thinning, T=thinning) on the Piedmont of Virginia. Vertical lines represent ± one standard error.

No statistically significant differences in total groundcover carbon nor shrub carbon were found between the various SMZ treatments (Figure 4, p=.43 for groundcover, p=.26 for shrubs).

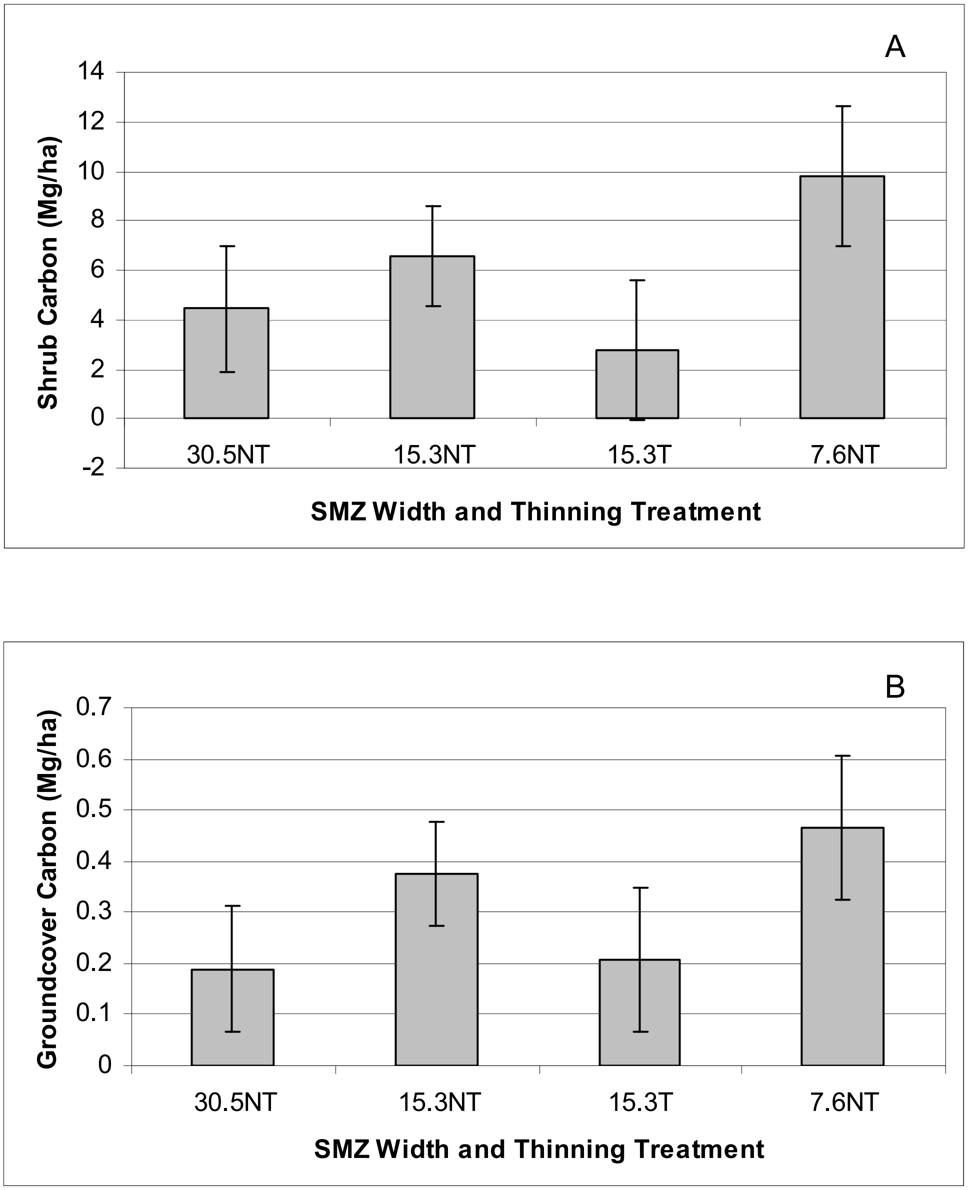

Figure 4. Aboveground shrub (A) and groundcover (B) carbon content as influenced by streamside management treatment (numbers (e.g. 30.5) are SMZ width in meters, NT=no thinning, T=thinning) on the Piedmont of Virginia. Vertical lines represent ± one standard error.

Litter Layer

There were no statistically significant differences in litter layer carbon between SMZ treatments (p=.25, Figure 5). The 7.6 m treatment, however, did show a sharp drop in litter layer carbon.

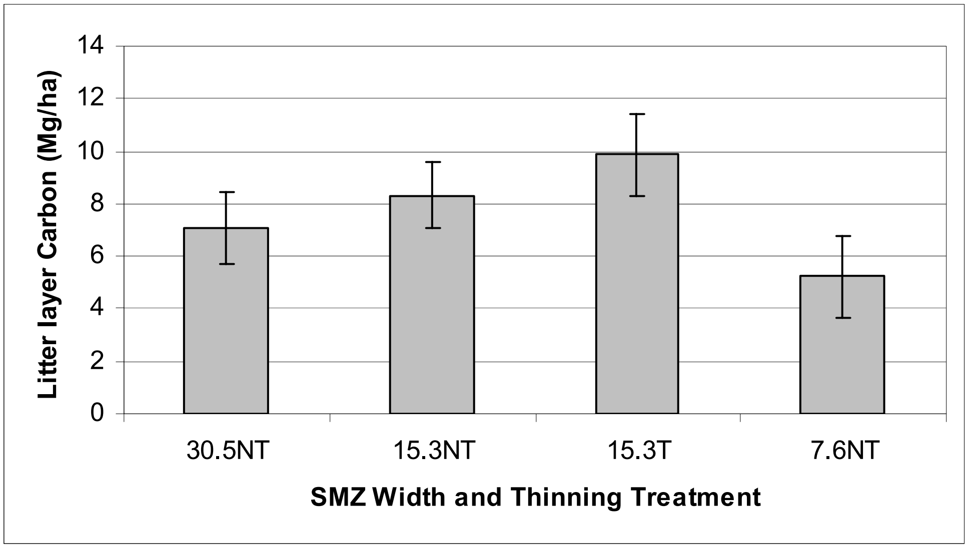

Figure 5. Mean carbon amounts of litter layer samples as influenced by streamside management treatment (numbers (e.g. 30.5) are SMZ width in meters, NT=no thinning, T=thinning) on the Piedmont of Virginia. Vertical lines represent ± one standard error.

Belowground Carbon

Soil

There were no statistically significant differences found for soil carbon between the various SMZ treatments (p=0.24, Table 3), with numbers ranging from 13.7 Mg/ha in the 7.6 m SMZ to 23.3 Mg/ha in the 15.3 m thinned SMZ.

Table 3. Mean carbon amounts and sample size for soil samples (0-10 cm) as influenced by streamside management treatment (numbers (e.g. 30.5) are SMZ width in meters, NT=no thinning, T=thinning) on the Piedmont of Virginia.

Treatment	Mean Carbon (Mg/ha)	Sample Size (n)
30.5m No Thin	16.45a± 7.4	12
15.3m No Thin	19.40a± 6.4	18
15.3m Thin	23.17a± 8.5	9
7.6m No Thin	13.70a± 8.5	9

Carbon Fluxes

Soil Respiration

Averaged over all measurement dates, there were no differences found in F_s rates between treatments (p=0.13, Table 4).

Table 4. Seasonal means for soil temperature, soil moisture, and soil CO_2 efflux (F_s) as influenced by the SMZ treatments.

SMZ Treatment	Soil temperature (°C)	Soil moisture (%)	Soil CO_2 efflux (uMol/m^2 s)
30.5 m no thin	15.6ab	12.4a	3.4a
15.3m no thin	15.4b	16.8a	4.2a
15.3m thinned	15.7ab	17.3a	4.1a
7.6m no thin	16.0a	23.4a	4.8a

*Means in a column followed by the same letter do not differ significantly (p=0.1)

Figure 6 shows seasonal trends in F_s for the various treatments. Overall F_s decreased from May 2005 through January 2006 with the 15.3 m thinned treatment generally having the lowest F_s rate except for a sharp increase during the July 2005 sampling date. The 30.5 m treatment also tended to have low F_s rates which can also be seen in the overall seasonal average (Table 4).

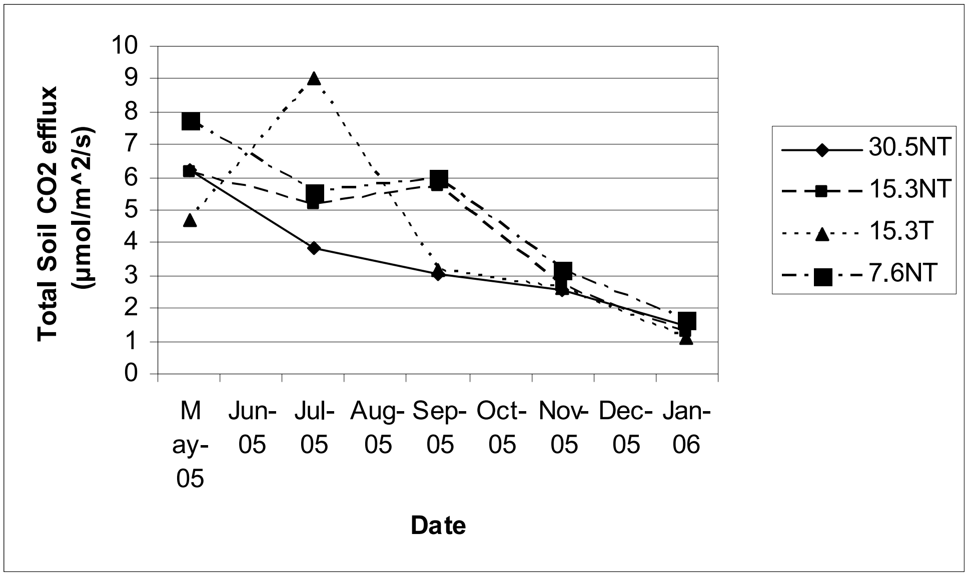

Figure 6. Seasonal trends in total soil CO_2 efflux as influenced by streamside management treatment (numbers (e.g. 30.5) are SMZ width in meters, NT=no thinning, T=thinning) on the Piedmont of Virginia.

Soil Moisture

We found no differences in soil moisture between treatments (p=.19). We did find that sampling date was highly significant (p<0.0001) on soil moisture content (Figure 7).

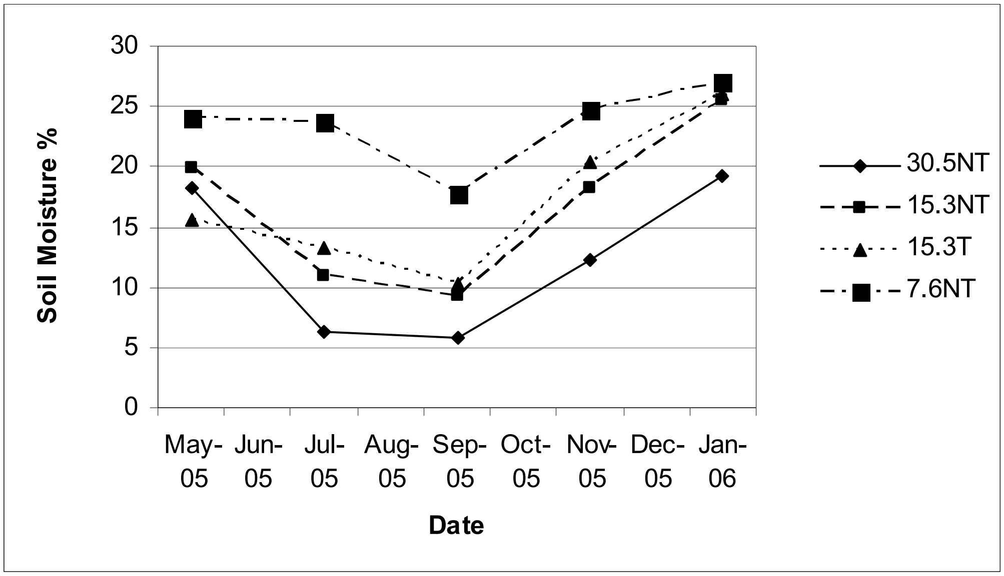

Figure 7. Seasonal trends in soil moisture content as influenced by streamside management treatment (numbers (e.g. 30.5) are SMZ width in meters, NT=no thinning, T=thinning) on the Piedmont of Virginia.

Soil Temperature

There was a statistically significant difference between soil temperature in the various SMZ treatments (p=.08). The 7.6 m no thin SMZ had a slightly higher soil temperature than the other three treatments (Table 4). Date also had a statistically significant effect on soil temperature (p<.0001), with soil temperature being higher during the summer months (Figure 8).

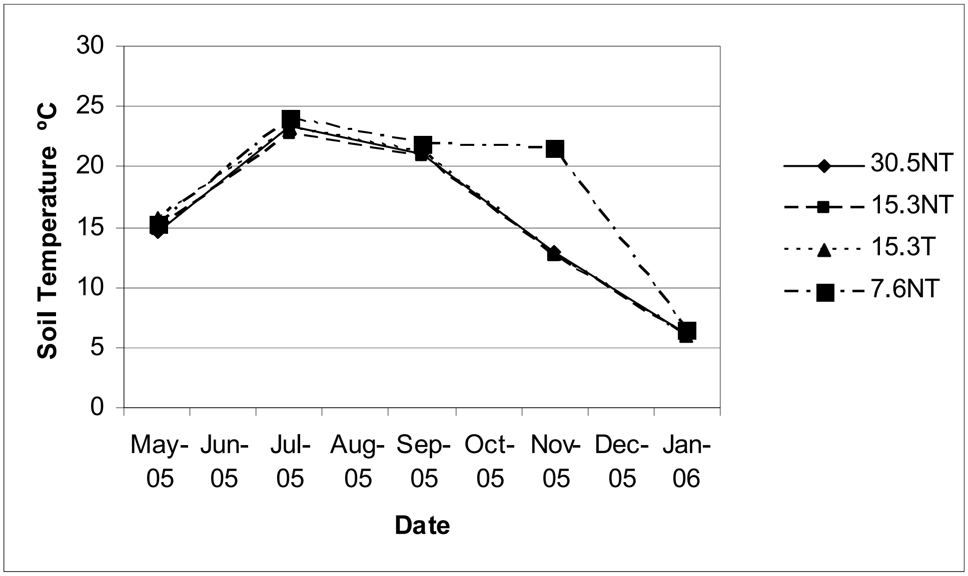

Figure 8. Seasonal trends in soil temperature as influenced by streamside management treatment (numbers (e.g. 30.5) are SMZ width in meters, NT=no thinning, T=thinning) on the Piedmont of Virginia.

Water

Grab sample collections were obtained in all sixteen streams that had water on all five dates. Grab samples were not collected from one of the watersheds due to the ephemeral nature of the watershed. In addition, grab samples were collected in one other watershed only once and this collection occurred after remnants of a hurricane passed through the area. This one collection during high flow instead of base flow resulted in date having a highly statistically significant effect ($p=<0.0001$). No differences were detected between the various SMZ treatments ($p=0.84$) or interaction between treatment and date ($p=0.99$). Though no treatment effect was detected, the 15.3 m thinned treatment consistently had the highest TOC during all five sampling dates (Figure 9).

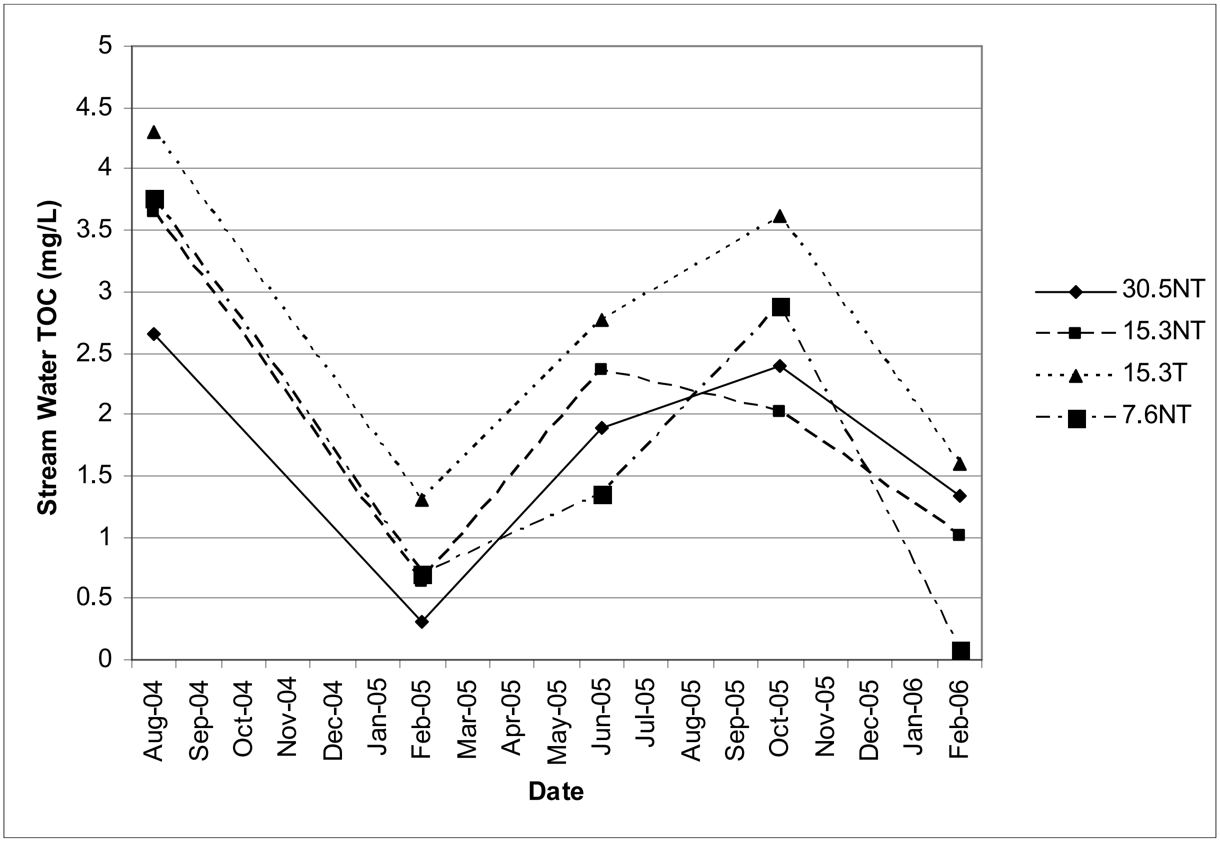

Figure 9. Seasonal trends in stream total organic carbon (TOC) as influenced by streamside management treatment (numbers (e.g. 30.5) are SMZ width in meters, NT=no thinning, T=thinning) on the Piedmont of Virginia.

Benthic Samples

Only thirteen of the sixteen streams were sampled for benthic macroinvertebrates due to insufficient water in three of the streams. A total of twelve macroinvertebrate orders and thirty-one families were identified. One genus that was found is rare, *Anisocentropus*. In addition, there were several families found in only one stream.

Table 5. Macroinvertebrate means for each index as influenced by SMZ treatment in the Piedmont of Virginia.

Abundance			**Taxa Richness**		
Treatment	n	Mean	Treatment	n	Mean
7.6m no thin	3	20.7± 8.2	7.6m no thin	3	9.0± 2.6
15.3m no thin	5	22.4± 6.3	15.3m no thin	5	7.4± 2.0
15.3m thin	2	15.5± 10.0	15.3m thin	2	7.0± 3.2
30.5m no thin	3	9.0± 8.2	30.5m no thin	3	3.6± 2.6
Caddisfly Abundance			**Stonefly Abundance**		
Treatment	n	Mean	Treatment	n	Mean
7.6m no thin	3	2.0± 1.2	7.6m no thin	3	4.0± 1.0
15.3m no thin	5	3.4± 0.9	15.3m no thin	5	1.4± 0.8
15.3m thin	2	1.5± 1.4	15.3m thin	2	0.5± 1.2
30.5m no thin	3	1.3± 1.2	30.5m no thin	3	0.0± 1.0
Mayfly Abundance			**% Mayfly**		
Treatment	n	Mean	Treatment	n	Mean
7.6m no thin	3	3.3± 2.1	7.6m no thin	3	18.6± 14.2
15.3m no thin	5	5.2± 1.6	15.3m no thin	5	27.3± 11.0
15.3m thin	2	2.5± 2.6	15.3m thin	2	18.3± 17.4
30.5m no thin	3	0.3± 2.1	30.5m no thin	3	8.3± 14.2
EPT Abundance			**% EPT**		
Treatment	n	Mean	Treatment	n	Mean
7.6m no thin	3	9.3± 3.3	7.6m no thin	3	47.1± 16.7
15.3m no thin	5	10.0± 2.6	15.3m no thin	5	64.1± 13.0
15.3m thin	2	4.5± 4.0	15.3m thin	2	32.5± 20.5
30.5m no thin	3	1.7± 3.3	30.5m no thin	3	15.0± 16.7
EPT Richness			**HBI Value**		
Treatment	n	Mean	Treatment	n	Mean
7.6m no thin	3	4.4± 1.4	7.6m no thin	3	4.0± 1.0
15.3m no thin	5	4.4± 1.1	15.3m no thin	5	3.7± 0.7
15.3m thin	2	3.5± 1.7	15.3m thin	2	4.3± 1.2
30.5m no thin	3	1.0± 1.4	30.5m no thin	3	3.8± 1.0

Although no differences were found between the various SMZ treatments, the 15.3 m no thin treatment had the best value in seven of the ten tests performed Table 5). The purpose of the HBI is to measure both nutrient and organic pollution which cause lower levels of dissolved oxygen. Hilsenhoff (1987) assigned tolerance values to Arthropod genera were assigned tolerance values of 0-10 based upon their presence or absence in particular streams with known levels of both nutrient and organic pollution.

All four treatments scored between 3.5 and 4.5 on the HBI indicating very good water quality with possible slight organic pollution present (Hilsenhoff 1987). Three of the four treatments scored "fair" in the % EPT, only the 30.5 m no thin, was "poor." According to the Stream Condition Index (SCI) for Virginia a typical stream located in the piedmont of Virginia scores "fair" in % EPT (20-65%) (Tetra Tech, Inc 2003).

Discussion

Aboveground Biomass

Standing Biomass

No significant differences were detected in the three strata measured for aboveground biomass. However, sampling for this study occurred only two years following the harvest, therefore residual trees had not yet reoccupied the site fully and it appears that thinning has reduced aboveground standing biomass carbon storage. Although this study shows a possible decrease in carbon in the thinned SMZ, a study by Peterson et al. (1997) demonstrated that during the six years following a similar thinning treatment regime, the trees responded with both increased crown dimensions and bole diameter increases therefore quickly recovering the initial carbon loss from the thinning. The diameter increase is believed to be in direct result of the increased photosynthetic rate by the expanded crown (Ginn *et al.* 1991). Other studies have shown increased diameter growth resulting in increased basal area approximately five years after a thinning treatment removing 25-50% basal area (Roberts and Harrington 2008 and Oliver 1972). Roberts and Harrington observed a 26% increase in basal area growth in various evergreen species five years after 25% of the basal area was removed. Meadows and Goelz (2001) measured growth responses to thinning (40-45% basal area removal) in natural stands of bottomland hardwood stands. Five years after the thinning treatment, a steady recovery to full site occupancy was observed, it is estimated that the site will recover to 100% stocking 10-15 years following the thinning. Thysell and Carey (2000) observed an increase in both species richness and abundance of understory species in thinned tracts approximately 10 years after thinning, the herbaceous layer accounted for a large portion of this increase, contributing to a greater percentage of cover as well in the thinned tract. In addition, fewer but larger trees were present in the thinned tract 10 years after thinning. This same effect may occur in the small, 7.6 m SMZ, resulting in greater amounts of both shrubs and groundcover.

On average, approximately 13% of a watershed area is occupied by SMZs within the Piedmont Region, assuming a 15.3 m SMZ was installed as is now recommended by the Virginia Department of Forestry (Williams *et al.* 2004). Between 1985 and 1991, an average of 41,976 hectares was harvested annually across the Virginia Piedmont (Johnson 1991 and Thompson 1991). Table 6 shows the estimated total carbon (includes standing biomass, litter layer, and soil carbon) in the four studied SMZ treatments in the Virginia Piedmont. Across the Virginia Piedmont landscape, removing fifty percent basal area in SMZs could translate into a reduction of 169,018 Mg/ha of carbon. Live biomass carbon only accounts for 33% of the carbon found in United States timberland (Turner *et al.* 1995). Therefore, removing 50% of the basal area within the SMZ only results in a 33% reduction of total carbon storage.

Table 6. Annual average hectares and carbon found in SMZs in the VA piedmont in the various SMZ treatments.

Annual Harvesting in the Virginia Piedmont		
SMZ width	**Hectares in SMZ**	**Potential Carbon amount in SMZ (Mg)**
30.5m NT	10,914	874,964
15.3m NT	5,457	511,533
15.3m T	5,547	342,515
7.6m NT	2,728	193,760

* Based on 41,976 hectares harvested annually in the Virginia Piedmont (adapted from Johnson 1991 and Thompson 1991).

Litter Layer

Although there were no differences detected in the carbon content of the litter layer between the various treatments, actual values showed that the litter layer carbon pool was highest in the 15.3 m thinned SMZ treatment. Other studies have shown that thinning reduces the amount of litter fall. Roig *et al.* (2005) found that litter fall was reduced in thinned plots two years after the treatment was installed; however, this same study concluded that the effects of thinning disappeared within five years. A study by Versterdal *et al.* (1995) found that the accumulation of litter layer was lowest in the thinned plots versus the unthinned plots. Anther study, found no differences in litter fall within two years of thinning (Slodicak *et al.* 2005).

The 7.6 m no thin SMZ had the least amount of litter layer, despite the presence of overstory vegetation that could potentially contribute litterfall. The 7.6 m no thin treatment had the highest soil moisture content and the warmest soil temperature. These two environmental factors have a positive correlation with decomposition rates (Raich and Schlesinger 1992). It is possible that the litter layer is decomposing at a higher rate in the 7.6 m no thin SMZ. Other studies have also shown that litter can be blown out of an SMZ. According to France (1995), this lateral transport increases with exposure to precipitation. The 7.6 m no thin treatment was comprised of mostly hardwoods. The same study by France (1995) concluded that hardwood leaves transport more readily than evergreen needles. In addition, Elliott *et al.* (1993) concluded that hardwood forest litter decomposes at a faster rate than evergreen forests.

Belowground Carbon

Soil

Though no statistically significant differences were detected between the various treatments, soil carbon content appeared to be greatest in the 15.3 m thinned treatment. This is what we would expect considering studies have shown that thinning actually alters the microclimate, creating more favorable conditions for decomposition (Della-Bianca and Dils 1960). Additionally, litter layer was also greatest in the thinned treatment translating into higher levels of carbon. Selig *et al.* (2008) observed higher soil carbon concentrations 14 years after thinning occurred. This increase was attributed to the additional inputs of organic matter caused by the thinning treatment in addition to the increase in both soil moisture and temperature.

Carbon Fluxes

Soil Respiration

SMZ treatment had little effect on CO_2 efflux (F_s), but the 7.6 m no thin treatment generally had the greatest respiration rates over the course of this study (Table 5). Due to the relative narrow widths of the actual floodplains, the 7.6 m no thin treatment tended to have the most gradual slope, resulting in a moisture environment. A study conducted on similar sites on the Piedmont of Virginia by Wiseman and Seiler (2004) indicated both soil temperature and soil moisture as two of the main driving factors influencing F_s. As was discussed earlier, these environmental factors greatly influence decomposition rates which in turn effect soil respiration (Lloyd and Taylor 1994, Davidson et al. 1998). Thinning did not appear to affect $F_{s.}$ Selig *et al.* (2008) observed increase F_s in thinned stands as well as higher soil temperature rates. While no treatment effect was found, sampling date was significant with soil respiration rates decreasing from summer up through winter. Previous studies have also found seasonal trends with soil respiration. Euskirchen *et al.* (2003) noted that these differences correlated with the changes in soil temperature. In this study higher values of soil respiration were detected during the summer months and began to decrease during the fall and into the winter. During this study, soil temperature was greatest during the summer months, possibly having a direct effect on soil respiration rates.

Previous studies estimated that a change in soil moisture can effect soil respiration (Bouma and Bryla 2000). In a study done by Broken *et al.* (2003), a strong correlation was found between soil moisture levels and soil respiration when concerning the O horizon. It is believed that this is due to the decomposition rate of the litter layer being affected by the moisture content. In this study, soil moisture content was lowest during the summer months.

Stream TOC

Our water samples showed that total organic carbon (TOC) amounts in stream water were consistently greater in the 15.3 m thinned treatment than in the other treatments. As was previously mentioned, thinning can increase the amount of large wood debris that is present within the riparian area therefore increasing the potential for introduction of carbon into the water system. The additional large woody debris can also form organic debris dams which assist in the control of exportation of particulate matter downstream (Bilby 1981). Samples that were taken from the 30.5 m no thin treatment had lower TOC than the other treatments during the post hurricane sampling and consequently during high flow. This suggests that the width of the SMZ may reduce the potential of carbon from entering the stream during storm events.

Benthic Macroinvertebrates

Though no discernable trend was detected when benthic populations were analyzed, the 15.3 m no thin treatment tended to have higher quality values. The only metric that yielded the highest score for the 15.3m thinned treatment was the HBI. This treatment also had some of the lowest carbon content in the various pools assessed. Though no statistical differences were detected, studies have shown that the introduction of LWD to streams improves habitat and consequently there are shifts in benthic populations (Lemly and Hilderbrand 2000). It is worth noting that a study carried out by Stone and Wallace (1998) indicated that the use of the EPT indices may not be useful in determining recovery from logging. In their study, no differences could be detected between the reference and disturbed streams over a 16 year study. In fact, their study showed that benthic macroinvertebrate abundance was greater in the disturbed stream when compared with the reference stream. It is believed that this initial increase in populations may be due to the increase in carbon inputs to the stream. In our study, the 30.5 m no thin treatment had the lowest numbers in the six metrics that involved the EPT species. Benthic sampling occurred in the winter of 2006. A study by Alden *et al.* compared benthic samples from all four seasons to determine which season, or combination of, was optimal to detect differences between stressed and reference sites (1997). For studies where one season sampling is used, summer is the optimal choice to detect differences between degraded sites and reference sites, while winter is the worst. The authors believe that this is due to the temperatures during these two seasons. Summer is the time of year when hypoxic stress might occur.

It is also important to note that our study area has been logged and farmed historically and therefore has had constant disturbance since the the late 1700's to early 1800's (Lakel 2008). Invertebrate community structures change depending upon the presence of a buffer zone or if the clear-cut reaches the water body (Quinn *et al.* 2004). For instance, mayflies seem to be the most sensitive macroinvertebrate to clear-cut logging (Quinn *et al.* 2004). These studies demonstrate that logging impacts are related to water temperature and associated with changes in stream lighting.

Nevertheless, seven of the ten indices indicated that the 15.3 m no thin treatment has the highest water quality in regards to benthic macroinvertebrate populations. Also, the HBI rates species that are most tolerant to the absence of dissolved oxygen caused by the presence of organic carbon in the stream. The 15.3m thinned treatment had the poorest score in the HBI metric indicating the possibility of low dissolved oxygen (DO) present within the stream. However, in another study conducted on these same watersheds, no differences were detected in DO levels between treatments/streams. DO levels below 5.5 mg/l puts aquatic life under stress and all treatments had DO levels greater than 8.0 mg/l (Lakel 2008). This correlates well with the highest present of TOC being found within this treatment as well.

Conclusions

As intended, thinning in SMZs reduces carbon storage in standing biomass and stand biomass has not recovered after two years. Both soil and litter layer carbon content were greatest in the thinned treatment, but is believed that this is a short-term effect caused by the actual event not by long-term ecological processes. Vesterdal (1995) reported a decrease in both litter layer and soil carbon content in areas that are thinned regularly for 30 years. According to the data trends, thinning did increase the amount of TOC present within the streams, potentially lowering the HBI score. The thinning treatment did not have an effect on soil respiration nor the environmental drivers that influence soil respiration.

SMZ width did not have a significant effect on carbon pools. The 7.6 m no thin SMZ did have the higher soil respiration rates along with the highest soil temperature and moisture content values, illustrating that width does affect the environmental factors that influence soil respiration. This suggests that in the future, soil carbon levels could fall and stream carbon levels may increase. In addition, SMZ width did not appear to affect benthic macroinvertebrates. The 15.3 m no thin treatment is acceptable for the benthic population as out of the 10 indices applied; this treatment scored the best in 8 of the indices. The two indices that did not score well were taxa richness and stonefly abundance. Taxa richness (total taxa found within community sampled) illustrates the health of a community through diversity and increases with increasing habitat diversity, suitability and water quality (Plakin *et al.* 1989). The family Plecoptera (stoneflies) is commonly the first group of organisms to disappear after a disturbance such as logging. Other studies have shown that logged areas have a higher density of benthic macroinvertebrates compared to non-logged areas (Stone and Wallace 1998). Kedzierski and Smock (2001) found that taxa richness was greatest in logged (clear-cut) areas three years after the event.

The 30.5 m and the 15.3 m no thin treatments were the best choices in terms of carbon storage as illustrated in Table 6. The 30.5 m no thin is an obvious choice due to its size. However, the 30.5 m no thin treatment did not have as much understory or diversity when compared with the 15.3 m no thin treatment.

Table 7 summarizes the effects that all four treatments had on all ten variables measured within this study and ranks these effects from best to worst in terms of overall SMZ function. Overall the 15.3 m no thin SMZ was the most effective in functioning as a riparian zone, consequently preventing changes in water quality.

Table 7. Ranking of overall effects (1=best, 4=worst) on carbon pools and fluxes as influenced by four streamside management treatments (numbers (e.g. 30.5) are SMZ width in meters) on the Piedmont of Virginia.

Variable	**7.6 m no thin**	**15.3 m thin**	**15.3 m no thin**	**30.5 m no thin**
Overstory	3	4	2	1
Shrubs	1	4	2	3
Groundcover	1	3	2	4
Soil	4	1	2	3
Litter Layer	4	1	2	3
Soil Temp.	4	3	1	2
Soil Moist.	1	2	3	4
F_s	1	3	2	4
TOC	2	4	3	1
Benthics	2	3	1	4
Average Rank	**2.3**	**2.8**	**2.0**	**2.9**

Further studies should be conducted seven to ten years after the thinning operations to determine what long-term effects could occur, such as canopy closure. Canopy closure could alter the microclimate of the forest floor resulting in similar characteristics of the non thinned SMZs.

Within the Virginia piedmont, 13% of forestry land is taken out of production due to the installation of the recommended 15.3 m (50 ft) SMZ (Williams *et al*. 2004). The installation of SMZs increases the manager's costs by taking land out of production (foregone timber value) and by increasing the distance of skidder tracks to avoid these protected areas (Kluender *et al*. 2000). Continuing to allow thinning within the SMZ, reduces the total cost of BMP installation while continuing to support the forest ecosystem functions.

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