Skid Trail Stream Crossing Closure Techniques for Protecting Water Quality

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Abstract

The impact of forest roads and skid trails on stream health is being increasingly scrutinized. Forest roads and skid trails have repeatedly been identified as forest operations having the greatest potential to produce sediment by way of non-point source pollution. The stream crossing portion of a skid trail is where sediment delivery is most likely to occur. Forestry Best Management Practices (BMPs) have been developed by most states to reduce both erosion and sedimentation. In general, BMPs have been proven to be effective. Few studies have quantified the impact of various levels of BMPs on sedimentation. In this study, three replications of three skid trail stream crossing BMP treatments were monitored following skidder bridge removal to determine their efficacy in reducing sedimentation: slash, mulch, and mulch plus silt fence. Water samples were collected upstream and downstream of each crossing daily for one year following timber harvesting. Samples were evaluated for total suspended solids. Results indicate that both slash and mulch treatments applied to the stream crossing approach after skidder bridge removal are effective at reducing stream sedimentation after harvest. The mulch plus silt fence treatment allowed the most sediment to enter the stream at the approach, perhaps due to silt fence installation disturbances. We do not recommend using silt fences directly adjacent to a stream bank, if other alternatives exist.

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1. Introduction

Sediment is generally accepted as the major pollutant in forestry (Neary et al. 1989). It is also considered the pollutant that is most harmful to water quality (US Environmental Protection Agency 2003). Forest roads and skid trails are the component of forest operations that have the greatest potential to degrade water quality by way of erosion and sedimentation (Patric 1976, Swift and Burns 1999, Aust and Blinn 2004, Grace 2005). Forest roads and skid trails often comprise 2-10% of a harvested area (Kochenderfer 1977) and may account for 25% of ground area disturbance (Jackson et al. 2002), thus their impact can be significant.

The primary location for sediment to enter streams is at road-stream crossings (Rothwell 1983, MacDonald and Coe 2008). Erosion becomes sediment after entering the stream (Yoho 1980). Spikes in sediment concentrations in streams typically occur during large rainfall events. Rainfall events can produce overland flow on forest roads, which transports the sediment. Beasley (1979) concluded that large storm events can produce 70% of total sediment yield. Bare soil exposed at stream crossings increases the likelihood of erosion being deposited into the stream during a storm (Taylor et al. 1999). Sedimentation in streams can alter the physical and chemical properties of water, including wildlife habitat (Elliot et al., 1994).

Forestry Best Management Practices (BMPs) or other forest practice statutes have been developed by most states to reduce both erosion and sedimentation (Shepard 2006). Several studies have proven the efficacy of various BMPs. However, stream crossings have been identified as important contributors of sediment and where more attention should be focused (Taylor et al. 1999, Schuler and Briggs 2000, Grace 2005).

2. Erosion

Since intensive European agricultural development accelerated in the United States over 200 years ago, erosion has been an increasing problem. For the past 50 years, almost one-third of the world's topsoil has been lost due to erosion (Pimental et al. 1995). When erosion occurs, more water becomes surface runoff and less water enters the soil, creating a shortage of water and nutrients, including nitrogen, phosphorous, potassium, and calcium that are available for plants (Pimentel et al. 1995). This degradation and loss of soil not only decreases site productivity, but also has the potential for introducing sediment into waterways, causing damage to aquatic habitats and commercial waterways..

Erosion can be defined as the detachment, displacement, and deposition of soil particles (Yoho 1980). It occurs when the energy used for detachment and transport of soil particles is greater than the forces that bind soil together (Douglass 1975). Often, detachment and transport are caused by raindrops, but can also be caused by wind, snow, or the freezing and thawing cycles of soil (Douglass 1975). Erosion that is deposited in streams is considered sedimentation, as well as non-point source pollution (NPSP). NPSP is of great concern because erosion can also carry with it pesticides and nutrients such as phosphates, thus contributing to eutrophication of streams and lakes (Virginia Department of Forestry 2002).

Erosion occurs as a part of the earth's natural processes. Undisturbed forests produce minimal amounts of erosion and sedimentation (Yoho 1980, Pimentel et al. 1995, McBroom et al. 2008). Water quality in undisturbed forests is among the highest quality in the nation (NCASI 1994). Precipitation that occurs in an undisturbed forest experiences interception by way of vegetation and leaf litter, which decrease the rate of impact with the soil (Cook 1936, Douglass 1975). The presence of vegetation and leaf litter mean that runoff and erosion rarely occur in an undisturbed forest (Yoho 1980, Hartanto et al. 2003). Erosion rates depend on a variety of factors, including particle size, soil structure, moisture content, bulk density, and the physical features of a site (Cook 1936, Pimentel et al. 1995). The most commonly used method of estimating erosion in forestry is the Universal Soil Loss Equation (USLE) (Lane et al. 1992). The equation is A=RKLSCP, where (A) is the total soil loss per unit area per year, (R) is the rainfall and runoff factor, (K) is the soil erodibility factor, (L) is the slope length factor, (S) is the slope steepness factor, (C) is the cover and management factor, and (P) is the support practice factor (Dissmeyer and Foster, 1984). The total soil loss that is estimated with the USLE is the total sheet erosion which is lost from an area; not necessarily the amount that will reach a stream (Douglass 1975).

Forest operations can disturb the vegetation and forest floor sufficiently to accelerate sedimentation (Grace 2002, Hartanto et al. 2003). Forest roads have a higher potential for erosion than most forest operations (Patric 1976). Although timber harvests are associated with erosion by the general public, infiltration rates on recently harvested areas still remain high and overland flow is rare (Douglass 1975). Litter and logging residue protect the harvest area from raindrops and sites quickly become revegetated (Douglass 1975). However, the coverage provided by litter is removed by construction of forest roads and skid trails. Roads are highly compacted, have bare soil, and have low infiltration rates (Reid and Dunne 1984). Surface erosion on forest roads originates from rainsplash erosion, sheetwash, and rilling (MacDonald and Coe 2008). Unpaved forest roads have been shown to have erosion rates that are one to two orders higher than undisturbed forests (Megahan et al. 2001, MacDonald and Coe 2007). Roads affect the hydrology and sediment movement in a watershed by altering stream networks and hydroperiods, sediment loading, sediment transport and deposition, channel morphology, channel stability, stream temperatures, water quality, and riparian conditions (Wemple et al. 1996).

3. Sedimentation and the Environment

Erosion becomes sedimentation once it enters a waterbody (Yoho 1980). Like erosion, sedimentation is a natural process. However, practices including agriculture, mining, construction, and forestry can accelerate sedimentation rates in streams and rivers (Wood and Armitage 1997). Forest streams and wildlife habitat are typically degraded when sedimentation occurs (Elliot et al. 1994). Sedimentation is dynamic; particles are constantly being dislodged, transported, and deposited in other areas (Lemly 1982). Sedimentation occurs in more than 46% of US streams and rivers (Judy et al. 1984).

Sediment from forest roads can impair surface water physically, chemically, biologically, and ecologically (Great Lakes Environmental Center 2008). High levels of sediment may cause permanent alterations in community structure, diversity, density, biomass, growth, and rates of reproduction and mortality in aquatic organisms (Henley et al. 2000). These disturbances to water quality are first reflected in stream invertebrates, and eventually impair fish as well. Chemically, sedimentation can lead to increases in stream pH, thus limiting pH-sensitive fish (Lemly 1982). Reduced levels of dissolved oxygen can result from sedimentation, causing impaired respiratory functions of fish (Waters 1995). Physically, sediment particles can blanket and smother invertebrates, causing respiratory difficulties as well (Lemly 1982). Once sediment settles to the streambed, empty space becomes filled, and can eventually decrease the total area available for macroinvertebrates' habitat (Lenat et al. 1981) and limit filter feeders (Lemly

1982). Sediment can also clog spawning beds, shorten the life of reservoirs, as well as degrade drinking water (Grace et al. 1998). Biologically, suspended sediment in streams can decrease light penetration by increasing turbidity, thus decreasing the photosynthetic potential by aquatic plants (Kirk 1994; Hoetzel and Croome 1994). Turbidity can be defined as the properties of water that cause light to be scattered and absorbed (American Public Health Association 1992). It is caused by dissolved substances in water, such as organic and inorganic particulates and suspended solids (Henley et al. 2000). Although sediment is one contributor to turbidity, the two are not always correlated because of factors such as the physical and optical properties of the particles (Gippel 1989). Ecologically, turbidity decreases available food for herbivores, creating a reduction in phytoplankton, as well as zooplankton (Lloyd et al. 1987). Anthropogenic sedimentation and turbidity have been identified as the leading causes of habitat degradation and decreases in aquatic organisms (Ritcher et al. 1997). These negative effects caused by sedimentation and turbidity are first seen at the primary trophic level and eventually affect organisms in higher trophic levels (Henley et al. 2000). The prevention of sediment entry into streams and rivers is one way to address the problem at its source; options include revegetation, the use of buffer strips, as well as other effective best management practices (BMPs) (Wood and Armitage 1997).

The economic costs of sedimentation are high, both on-site and off-site (Moore and McCarl 1987). The off-site costs include the reduction of water storage capacities in reservoirs, dredging of reservoirs, increased water purification costs for drinking water, construction of sediment settling ponds, interference with navigation, and ditch cleaning (Crowder 1987, Moore and McCarl 1987, Holmes 1988). Dredging, which is needed to keep reservoirs functioning properly, only occurs in small reservoirs because of the high cost of disposal (Crowder 1987). In

1980, the estimated costs of dredging amounted to $50 million per year, and the net damage cost of erosion-related pollutants were estimated at $6 billion (Clark 1985). In 1997, erosion on agricultural lands was estimated at 640 million tons less than the 1982 rates (Hansen and Hellerstein 2007). This reduction in erosion resulted in $154 million conserved in reservoir benefits (Hansen and Hellerstein 2007). Although erosion rates have decreased since the 1980s, it is still occurring and costs are still high.

4. Best Management Practices

Forestry Best Management Practices (BMPs) or other low-impact forestry practice statutes are practices used in forest operations intended to prevent, mitigate, or reduce non-point source pollution (NPSP) such as erosion. BMPs are developed by each state for major land uses and act to protect water quality by reducing erosion and sedimentation from forest operations. (Virginia Department of Forestry 2002). The US Environmental Protection Agency (Rey 1980) has defined BMPs as:

> "…a practice or combination of practices, that are determined by a state, or designated area-wide planning agency, after problem assessment, examination of alternative practices, and appropriate public, to be the most effective, practicable (including technological, economic, and institutional considerations) means of preventing or reducing the amount of pollution generated by nonpoint sources to a level compatible with water quality goals."

The Clean Water Act permits each state to develop forestry BMP regulations, and charged them to ensure their implementation (Lucier and Shepard 1997). Forestry BMPs exist in

43 states, 29 of which have monitoring programs for compliance (Archey 2004). Although some states' BMP programs are voluntary (or "nonregulatory"), clean water standards can still be enforced through additional sediment related legislation (Aust and Blinn 2004, Ice et al. 1997). The basic guidelines for BMPs concerning roads, skid trails, and logging decks include proper planning and location, control of grade, control of water, surfacing, and road or trail closure (Swift 1985, Grace 2005). The location of a road or trail is of particular importance because avoiding steep grades and stream crossings can decrease the likelihood of sedimentation (Kochenderfer 1977). Erosion rates increase as slope increases (Pimentel et al. 1995) and several studies have shown that the closer a road or skid trail is to a stream, the higher the rates of sediment reaching the stream (Haupt and Kidd 1965, Swift 1986).

4.1 Roads and Skid Trails

Forest roads and skid trails are primary hydraulic flow pathways for erosion (Wemple et al. 1996), thus much BMP technology is specific to roads. Often water control structures, such as waterbars and water turnouts, are used to divert channelized water from roads into forest litter for redistribution (Swift and Burns 1999). Waterbars are the principle current road and trail closure BMP regulation used in Virginia. Seeding and mulching are other recommended BMPs that reduce erosion by stabilizing the soil (Swift 1985, Grace 2002). Both water bars and vegetation establishment decrease the velocity of overland flow and deposit the erosion before it reaches a stream (Swift and Burns 1999). Closure of skid trails after harvesting is of particular importance because skid trails are often built to lower standards than permanent roads, and pose a greater risk for soil erosion (Garland 1983, Grushecky et al. 2009). When a stream is present on a harvest area, skidder stream crossings increase the opportunity for sediment to enter directly into waterways. Therefore closure of the skid trail near the stream crossing approaches is critically important. Skid trails can be closed (or stabilized) in a variety of ways to minimize erosion, including the installation of waterbars, seeding and mulching, and applying logging slash (Schuler and Briggs 2000, Sawyers et al. 2012, Wade et al. 2013).

Residual limbs, tops, and other non-merchantable portions of trees remaining after harvesting are commonly referred to as slash, laps, or logging debris. Slash application to skid trails during closure has been shown to decrease erosion rates when compared with waterbars only (McBroom et al. 2008, Wade et al. 2013). Wade et al. (2013) compared five skid trail closure techniques and found that both slash and grass with mulch closure treatments were the most effective erosion control methods. The waterbar only treatment in the study averaged 137.7 tonnes/ha/yr of erosion. Seed-only treatments produced 31.5 tonnes/ha/yr of erosion, hardwood slash produced 8.9 tonnes/ha/yr, pine slash produced 5.9 tonnes/ha/yr, and mulch treatments produced 3.0 tonnes/ha/yr of erosion. Mulch was the most effective treatment in reducing erosion, while slash was a close second, and is potentially less expensive to apply.

Rothwell (1983) compared two closure treatments at stream crossing approaches; both containing grass and fertilizer, with one mulched, and one unmulched. Suspended sediment was measured upstream and downstream of each stream crossing. The unmulched sites generated slightly more sediment than the mulched sites in the stream, site inspections indicated that there was much less erosion at the mulched crossings. The mulch acted to trap sediment moving via overland flow before reaching the downhill stream (Rothwell 1983).

Lyons and Day (2009) conducted a field trial on logging roads to determine if logging residue (chipped on-site) would outperform a soil-only road. They concluded that the chipped wood cover was superior to bare soil (Lyons and Day 2009). Both slash and mulch act to intercept the velocity of rain before it hits the bare soil (Pimentel et al. 1995). It also acts to decrease the rate of overland flow, as well as to divert overland flow before it reaches a stream. Slash and mulch are superior to seeding because of their immediate coverage and stabilization of soil (Rothwell 1983, Wade et al. 2013).

The use of silt fences on closed skid trails can also decrease sedimentation rates. For several decades, silt fences have been used in the construction industry to control erosion. A silt fence is made of a synthetic fabric with small openings that allow water, but not sand and silt, to pass through (Robichaud and Brown 2002). Its application in forestry is to trap erosion before it reaches the downhill waterway, thus reducing sedimentation levels in streams. Erosion-trapping studies have found that properly installed silt fences are 68-98% effective in trapping erosion. Silt fences can be used on an array of slopes, from 3 to 70% (Robichaud and Brown 2002).

4.2 Streamside Management Zones

Streamside management zones (SMZs) are an additional BMP guideline used for overall erosion control near streams throughout the harvesting process (Lakel et al. 2010). Also referred to as riparian forests, filter strips, or buffer strips, SMZs are natural forested areas adjacent to streams (Castelle et al. 1994). SMZs act to slow the velocity of surface flow, thus allowing sediment to drop from suspension or be trapped by litter, resulting in decreased amounts of sediment which could reach the stream (Castelle et al. 1994, Croke and Hairsine 2006). They are the first areas to reach saturation so infiltration may not be increased. SMZs not only filter sediment and nutrients, but also to provide streambank stabilization, maintain natural stream temperatures through shading, provide wildlife habitat, and provide aesthetic value (Welsch 1991, Rivenbark and Jackson 2004). They have been proven to be effective in many settings (Castelle et al. 1994), and have even been found to result in higher number and richness of macroinvertebrates in streams (Henley et al. 2000). However, SMZs are not currently 100%

effective at trapping sediment; Ward and Jackson (2004) found efficacies of 71 to 99% below site prepared stands in the Georgia Piedmont, and Lakel et al. (2010) found efficiencies of 85 to 97% below harvested sites in the Virginia Piedmont.

Rivenbark and Jackson (2004) studied the efficiency of SMZs and the characteristics of SMZs where “breakthroughs” occurred (or where sediment had obviously traveled through the SMZ). They found that the breakthroughs often occurred in areas with large contributing areas, large amounts of bare soil (minimal leaf litter present), and steep slopes. They recommended using special management techniques in these areas, including stacking slash along the SMZ boundaries (Rivenbark and Jackson 2004). Another sensitive part of a SMZ is at a stream crossing. When a stream crossing is used, the SMZ is bisected by the skid trail. This compromise of the SMZ integrity increases the likelihood of sedimentation in the stream because of the physical intersection of the road and the stream (Grace 2005, Aust et al. 2011).

4.3 Stream Crossings

The stream crossing portion of the forest road is where sedimentation typically occurs (Rothwell 1983, Swift 1985, Grace 2005, MacDonald and Coe 2007, Witmer et al. 2009). Stream crossings should be minimized during proper planning of logging operations (Virginia Department of Forestry 2011). Sometimes stream crossings are unavoidable due to factors such as topography, practicality, and feasibility (Grace 2005). When crossings are necessary, appropriate BMP guidelines should be followed.

Stream crossing options for use on skid trails during active harvesting operations include culverts, fords, and bridges. Culverts and fords are more commonly found on permanent haul road stream crossings, whereas portable timber or metal bridges are more commonly used on skid trail crossings (Aust et al. 2011). A recent survey of Virginia loggers showed that portable bridges (both wooden and metal) are the most commonly used skidder crossing type in the Coastal Plain and Piedmont regions of Virginia (McKee et al. 2012). Temporary metal bridges have been shown to have the least amount of water quality impacts associated with them than other crossing types (culverts, fords, pole bridges) (Aust et al. 2011). They are thought to have the lowest water quality impacts because they do not usually impede water flow, nor do they require fill like culverts (Taylor et al. 1999). Portable bridges are typically the most expensive stream crossing option for one use (Taylor et al. 1999, Aust et al. 2003), although after portable bridges are used many times, they become more cost-effective due to longevity (McKee et al. 2012).

During installation, portable bridges generally have minimal disturbance and water quality impacts. In a study which occurred during the installation of a temporary timber bridge, almost no sedimentation was detected because no equipment had to enter the steam and stream channels were virtually untouched (Thompson et al. 1995). Another study which evaluated sediment during the installation of a temporary metal bridge found no significant differences between upstream and downstream sediment concentrations during the installation period (Tornatore 1995). Aust et al. (2011) found that bridges require fewer alterations to the stream channel compared to other crossing types. Although portable bridges have the lowest impact on water quality compared to other stream crossing options, it is important to recognize that the physical properties of the stream crossing approach have a larger influence on sedimentation rates than the specific crossing type (Aust et al. 2011).

4.4 BMP Effectiveness and Implementation

Wynn et al. (2000) evaluated sediment concentrations within watersheds treated with BMPs and without BMPs in the Virginia Coastal Plain. The watershed with BMPs consisted of a pre-harvest plan, an SMZ, waterbars on skid trails, and seeded landings. Total suspended solids found in the streams had a median concentration in the pre-harvest period of 400 mg L^{-1} and the post-harvest period displayed a median of 3300 mg L^{-1} in the non-BMP watershed. The watershed with BMPs had an average annual sediment yield that remained relatively constant during the study. This research indicates that BMPs were effective in reducing suspended sediment levels in clearcuts in the coastal plain region.

Arthur et al. (1998) studied the effect of BMP implementation on stream water quality in a clearcut harvest in the Allegheny Plateau of Kentucky. BMPs consisted of a 15.2 m buffer strip on either side of the stream, road grades of less than 10%, and roads, skid trails, and landings were seeded after closure. The paired watershed method was used and data showed that BMPs reduced the amount of sediment exported from the watershed compared to the non-BMP watershed. During harvest, the non-BMP watershed had 16 times more suspended sediment than the BMP watershed. During the 17 months after harvest, the non-BMP watershed had 2.5 times more suspended sediment than the BMP watershed. BMPs also minimally reduced the amount of nutrients, including nitrogen, potassium, and calcium that were exported into the streams.

As of 2010, forestry BMPs in the United States are being implemented at an 89% rate and the southern states at an 87% rate (Ice et al. 2010). This rate has been increasing for the past 35 years, and now many wood products facilities only accept timber from BMP-trained loggers (Ice et al. 2010). Numerous studies have shown that in general, forestry BMPs help to decrease soil loss and sedimentation in streams (Arthur et al. 1998, Schuler and Briggs 2000, Wynn et al. 2000, Aust and Blinn 2004, Grace 2005). However, more research is needed regarding the effectiveness, benefits, and costs of specific BMPs (Aust and Blinn 2004, Anderson and Lockaby

2011). Such data could be used to maximize the efficiency of BMP implementation and reduce implementation costs (Thompson et al. 2010, Anderson and Lockaby 2011).

5. Recent Litigation

Clinton and Vose (2003) emphasized the resurgence of interest regarding impacts of forest roads on stream characteristics and health. They noted the pressures placed upon land managers, organizations, and regulatory personnel to protect both terrestrial and aquatic systems that might be negatively affected by management operations. Additional information regarding forest operations and water quality impacts are needed to assist these managers in their decision making process.

Recently, the U.S. ninth circuit court of appeals ruled that runoff caused by forest roads is point-source pollution (Boston and Thompson 2009), which would require a National Pollutant Discharge Elimination System (NPDES) permit from the Environmental Protection Agency (EPA) (Schilling et al. 2007). Historically, sediments from forest roads have been considered non-point source pollution and were therefore exempt from federal Clean Water Act standards (Boston and Thompson 2009). The controversial issue is scheduled to go before the US Supreme Court for final interpretation. Regardless of the outcome, this recent litigation indicates the necessity for more research regarding the effectiveness of specific BMPs on forest roads. Anderson and Lockaby (2011) outlined current research needs regarding stream sediment and forest management. Although there is much literature suggesting that properly implemented BMPs protect water quality in general, they point out the need to quantify the effectiveness of specific BMPs. Grace (2005) mentions that although much research has been conducted on forest road erosion rates, more research is needed regarding the connection between road erosion and sediment delivery to streams. BMPs may need to be refined in order to improve silvicultural management (Aust and Blinn 2004). The EPA has expressed the need for additional research on the extent to which runoff from forest roads has degraded water quality, and the effectiveness of state forestry BMP programs in preventing water quality degradation (GLEC 2008).

6. Objective

The objective is to evaluate three levels of skid trail stream crossing closure techniques on measured sedimentation levels. The treatments applied to the stream crossing approaches were: 1) logging slash (Slash), 2) seed, straw mulch, and fertilizer (Mulch), and 3) seed, straw mulch, fertilizer, and silt fence (Mulch + Silt fence).

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Chapter 2 – Effectiveness of Operational Best Management Practices for Sediment Reduction at Skidder Stream Crossings

Abstract

Forest roads and skid trails have repeatedly been identified as forest operations having great potential to produce non-point source pollutants such as sediment. Stream crossings associated with skid trails are zones where sediment delivery potential is high. Forestry Best Management Practices (BMPs) have been developed by most states to reduce both erosion and sedimentation. In general, BMPs have been proven to be effective. However, few studies have quantified the impact of various levels of BMPs on sedimentation. In this study, three skid trail stream crossing BMP treatments were installed and replicated to determine their efficacy for reducing sedimentation. Treatments were: 1) slash, 2) mulch, and 3) mulch plus silt fence. Water samples were collected daily both upstream and downstream from operational stream crossings for one year following timber harvesting and BMP installation. Samples were evaluated for total suspended solids. Results indicate that both slash and mulch treatments applied to the stream crossing approaches after removal of temporary skidder bridges were effective at reducing the amount of sediment entering the stream after harvest. If available on site, slash could be the preferred method of stream crossing closure, because of lower cost, especially if incorporated into logging operations. However, if slash was being utilized for biomass and was not available, seed and mulch is a viable option for stream crossing closure. The mulch plus silt fence treatment was the most expensive treatment, yet it allowed more sediment to enter the stream at the approach. This reduction in efficiency was perhaps due to silt fence installation disturbances. Thus silt fences should not be installed directly adjacent to a stream bank, if other alternatives

exist. Overall, all BMP closure techniques minimized the effects of stream crossings on sediment inputs.

1. Introduction

Soil forms slowly, at a rate of about 300 Mg ha^{-1} $year^{-1}$ of topsoil per 300-1,000 years, yet it can erode at a rate greater than 100 Mg ha^{-1} $year^{-1}$ if not carefully managed (Bennett 1955, Pimentel et al. 1976). Since intensive agricultural operations began in the United States over 200 years ago, anthropogenic erosion has been a recognized problem. During the past 50 years, almost one-third of the world's topsoil has been displaced due to erosion (Pimental et al. 1995). Erosion of soil not only decreases site productivity (Lal 2001, Quinton 2010), but also has the potential for introducing sediment into waterways, causing damage to aquatic habitats (Elliot et al. 1994) and water bodies. According to the United States Environmental Protection Agency, sediment is the leading cause of the nation's impaired waters (US Environmental Protection Agency 2000). Forest streams and wildlife habitat are degraded when sedimentation occurs (Elliot et al. 1994). Sedimentation is considered the central factor limiting fish habitat in about 46% of US streams and rivers (Judy et al. 1982). High levels of sediment in streams have been shown to decrease primary productivity, diversity, and abundance of macroinvertebrates and fish (Wood and Armitage 1997). In addition, sediment may cause permanent changes in community structure, density, growth, and rates of reproduction and mortality in aquatic biota (Henley et al. 2000). Sediment also leads to increased turbidity in streams, reducing levels of light penetration, thus decreasing the amount of photosynthesis that can occur in aquatic plants (Kirk 1985, Ryan 1991). It can also have negative physical effects on harbors, reservoirs, navigable streams, and the cost of water treatment (Crowder 1987, Moore and McCarl 1987, Holmes 1988).

Accelerated sedimentation in streams often occurs following anthropogenic activities including urbanization, agriculture, silviculture, and mining (Marcus and Kearney 1991). Land use activities often disrupt soil and alter drainage pathways, leading to non-point source pollution (NPSP). It is well documented that forest roads and skid trails are the component of forest operations that have the greatest potential to degrade water quality by way of erosion and sedimentation (Patric 1976, Swift and Burns 1999, Aust and Blinn 2004, Grace 2005). Unpaved forest roads can alter hillslope hydrology by creating a more impermeable surface (Megahan 1972) and decreasing infiltration (Grace 2005), resulting in overland flow during rain events. The combination of increased overland flow (concentrated on the road surface) and exposed soil particles (intrinsic to forest roads) increases the potential for surface erosion. Erosion rates have repeatedly been shown to be higher on roads, skid trails, and log landings, compared to adjacent harvested and undisturbed areas (Yoho 1980, Rothwell 1983, Arthur et al. 1998). Corbett et al. (1978) found that timber harvesting, if considered independently of roads, has minimal effects on stream turbidity. However, road and skid trail construction associated with harvesting were the major sources of sediment from harvesting. Factors affecting surface erosion on a forest road include slope steepness (Pimentel et al. 1995), traffic volume, and the time since construction (Fu et al. 2010). The erosiveness of the road surface depends on factors including cohesiveness, particle size distribution, organic matter content, and permeability (Geeves et al. 2000).

Erosion includes three processes: detachment of soil particles, transportation (by rain, wind, snow, etc.), and final deposition (Yoho 1980). The Universal Soil Loss Equation (USLE) is an empirical model used for the prediction of erosion. The USLE was modified in 1984 by Dissmeyer and Foster (1984) for forestry use. The predicted erosion by the USLE is defined as “the amount of soil delivered to the toe of the slope where either deposition begins or where runoff becomes concentrated" (Dissmeyer and Foster 1984). The USLE is the most commonly used method of predicting erosion in forestry (Lane et al. 1992). It is important to recognize that erosion rates do not necessarily equal the amount of sediment being deposited into streams (Grace 2005). Erosion becomes sediment only when it enters a stream (Yoho 1980). Sediment delivery can be defined as the delivery of eroded sediment from areas such as road features to stream networks (Fu et al. 2010). Only a fraction of erosion will become sediment because of deposition and temporary or permanent storage downslope, prior to entering a water body (Walling 1983). Surface runoff delivers sediment, thus the rainfall amount, rainfall intensity, and slope percentage are contributing factors that determine the quantity of erosion that reaches the stream and becomes sediment (Croke and Hairsine 2006).

The stream crossing portion of the forest road is where the road intersects the stream and where sedimentation potential is greatest (Rothwell 1983, Swift 1985, Grace 2005, MacDonald and Coe 2007, Witmer et al. 2009). Sediment concentrations are often increased downstream of road-stream crossings, where the majority of sediment generated is delivered to a stream (Lane and Sheridan 2002, Croke et al. 2005). Litschert and MacDonald (2009) found that 83% of erosion features (i.e., sediment delivery pathways) that were connected to the stream channel originated from skid trails. They recommended increasing waterbar frequency and surface roughness on skid trails (by way of litter, logging slash, and woody debris) in order to minimize the amount of sediment deposited into nearby streams. Schoenholtz (2004) stated that sediment yield in streams is proportional to the relative road density in a particular watershed. Sediment yield to nearby streams has also been noted as being inversely proportional to the recovery time since road construction (Luce and Black 1999, Schoenholtz 2004). It has been well documented that following forest harvesting operations, water table levels and water yield both increase as a result of reduced evapotranspiration and interception rates (Arthur et al. 1998, Wynn et al. 2000, Sun et al. 2004). Since suspended sediment concentration is positively correlated with water discharge (Cheong et al. 1995), sediment concentrations in streams may increase following harvest-induced increases in water yield. These levels have the ability to persist until hydrologic recovery occurs (Lewis et al. 2001). The length of time it takes for hydrologic recovery can vary by site, and is defined as forest regeneration where hydrologic conditions return to pre-harvest levels (Macdonald et al. 2003). Although sediment yield is site specific, Haupt and Kidd (1965) found that increased sediment movement persisted for three years after a timber harvest. Aust and Blinn (2004) reviewed harvesting effects research in the eastern US and concluded that water yield and sediment yields generally became similar to non-harvested conditions between 2 to 7 years after harvest.

Forestry Best Management Practices (BMPs) or other forest practice statutes have been developed by most states to reduce both erosion and sedimentation (Shepard 2006). Forestry BMPs exist in 43 states, 29 of which have monitoring programs for compliance (Archey 2004). Some states have mandatory BMPs, while others are voluntary, and some are a combination of both (Aust and Blinn 2004). Although some states' BMP programs are voluntary (or "nonregulatory"), clean water standards are still enforced (Ice et al. 1997). The basic guidelines for BMPs concerning roads, skid trails, and logging decks include proper planning and location, use of streamside management zones (SMZs) or buffer strips, control of grade, control of water, surfacing, road or trail closure to minimize soil disturbance, and revegetation following harvesting (Swift 1985, Aust and Blinn 2004, Grace 2005). In general, numerous studies have shown that the aggregation of forestry BMPs decrease sedimentation in streams (Arthur et al. 1998, Schuler and Briggs 2000, Wynn et al. 2000, Aust and Blinn 2004). In 2010, the southern states employed forestry BMPs at an 87% implementation rate and the national rate was 89% (Ice et al. 2010). Although overall implementation rates are high, additional research is needed regarding the efficacies of specific forestry BMPs in order to maximize efficiency and potentially reduce costs (Anderson and Lockaby 2011).

Stream crossings should be minimized by pre-harvest planning (Virginia Department of Forestry 2011). Stream crossings may be unavoidable due to factors such as topography, practicality, and feasibility (Grace 2005). In situations where stream crossings are necessary, BMPs should be employed to minimize the potential effects. Stream crossing options include fords, culverts, and a variety of bridges such as pole bridges, metal bridges, and wooden stringer bridges. Forest stream crossings can be either permanent or temporary. Permanent crossings are primarily located on truck haul roads intended for long-term use, while temporary stream crossings are often those employed on temporary roads or skid trails. Skid trails may have more potential for erosion than haul roads because they have lower standards than haul roads (Grushecky et al. 2009) and have less elaborate water control structures than permanent roads. Stream crossings can add significantly to operational costs, e.g., temporary skidder bridges currently cost as much as $8,000 to $16,000 in 2010 (Conrad et al. 2012, McKee et al. 2012).

All stream crossing options have demonstrated the potential to degrade water quality, especially during installation and harvesting (Aust et al. 2011). Culverts and fords are more commonly found on permanent haul road stream crossings, whereas a mix of culverts, timber bridges, and portable metal bridges are more commonly used on skid trail crossings (Aust et al. 2011, McKee et al. 2012). A recent survey of Virginia loggers indicated that more skid trail stream crossings were installed than haul road crossings in all regions of Virginia in 2009 (McKee et al. 2012). For skidder crossings, culverts were most commonly used in the Virginia

Mountains, while steel and wooden bridges were preferred in both the Piedmont and Coastal Plain regions (McKee et al. 2012). Portable bridges typically have the least water quality impacts because they do not usually impede water flow or require fill material to be placed in the channel (Taylor et al. 1999).

Swift and Burns (1999) have suggested that stream crossings should be "restored to a stable, noneroding condition" in order to be resilient during storm events. Channel stabilization after bridge removal is considered to be the most important aspect of protecting water quality by the Virginia Department of Forestry (Virginia Department of Forestry 2011). However, methods of stabilization are not explained in the Virginia BMP manual. Closure techniques are not specified for stream crossings in many of the state BMP manuals in the South. During annual BMP audits, the Virginia Department of Forestry identified stream crossings as an area where BMP compliance could be improved (Virginia Department of Forestry 2008).

BMPs used on the stream approaches may be more important to stream water quality than the stream crossing type (Aust et al. 2011). Stream crossing approaches vary by length, slope, percent bare soil, and BMPs implemented. The slope of the stream crossing approach can affect erosion potentials as steeper slopes can have higher runoff energy, thus increasing erosion potential (Grace 1998). Aust et al. (2011) suggested that the high rates of total dissolved solids could potentially be improved with enhanced BMP stream approach closure techniques. After temporary crossings are removed, skid trails with exposed bare soil often remain, thus creating the pathway for erosion to travel directly into the stream during overland flow. During the critical closure phase, enhanced BMPs could potentially decrease the amount of sediment entering the stream.

A recent decision from the U.S. ninth circuit court of appeals ruled that runoff caused by forest roads is point-source pollution rather than non-point source pollution (Boston and Thompson 2009). This decision may eventually require a National Pollutant Discharge Elimination System (NPDES) permit from the EPA to construct (Schilling et al. 2007). Historically, sediment from forest roads has been considered non-point source pollution and therefore exempt from federal Clean Water Act standards (Boston and Thompson 2009). The issue has yet to be resolved. However, this recent litigation emphasizes the need for quantification of the effectiveness of specific BMPs on forest roads.

Anderson and Lockaby (2011) outlined current research needs regarding stream sediment and forest management. Although much literature suggests that properly implemented BMPs protect water quality in general, they point out the need to quantify the effectiveness of specific BMPs. Grace (2005) mentions that although much research has been conducted on forest road erosion rates, more research is needed regarding the connection between road erosion and sediment delivery to streams. Jackson et al. (2005) concluded that watershed improvement efforts should focus on the reduction of sediment delivery from unpaved roads. Clinton and Vose (2003) emphasized the resurgence of interest regarding impacts of forest roads on stream characteristics and health. They noted the pressures placed upon land managers, organizations, and regulatory personnel to protect both terrestrial and aquatic systems that might be negatively affected by forest management operations.

Additional information regarding forest operations and water quality impacts are needed to assist these managers in their decision making process. The objective of this research was to evaluate three levels of skid trail stream crossing closure techniques (slash, mulch, and mulch + silt fence) on stream sediment levels. A secondary objective was to quantify the costs of the

BMP treatments. This information will provide land managers and regulators with options for erosion control at stream crossings. Specific hypotheses addressed include:

Ho_1: Three levels of stream crossing approach BMPs will not result in significant differences between percent change of upstream and downstream TSS levels.

Ho_2: Three levels of stream crossing approach BMPs will not result in significantly different BMP efficiencies.

Ho_3: USLE estimates will not result in different levels of erosion between the three stream crossing approach closure treatments.

Ho_4: Each of the three treatments will have similar costs.

2. Materials and Methods

2.1 Study Sites

Nine operational stream crossings were located on five harvest sites in the Piedmont physiographic region of Virginia (Nelson, Pittsylvania, Amherst, Appomattox, and Buckingham counties) (Figure 1). All sites had temporary stream crossings and used portable metal bridges for skidding. All sites were located on MWV property and were harvested in the fall of 2010 or spring of 2011. Stands were managed loblolly pine (*Pinus taeda*) plantations ranging from 18 to 25 years old.

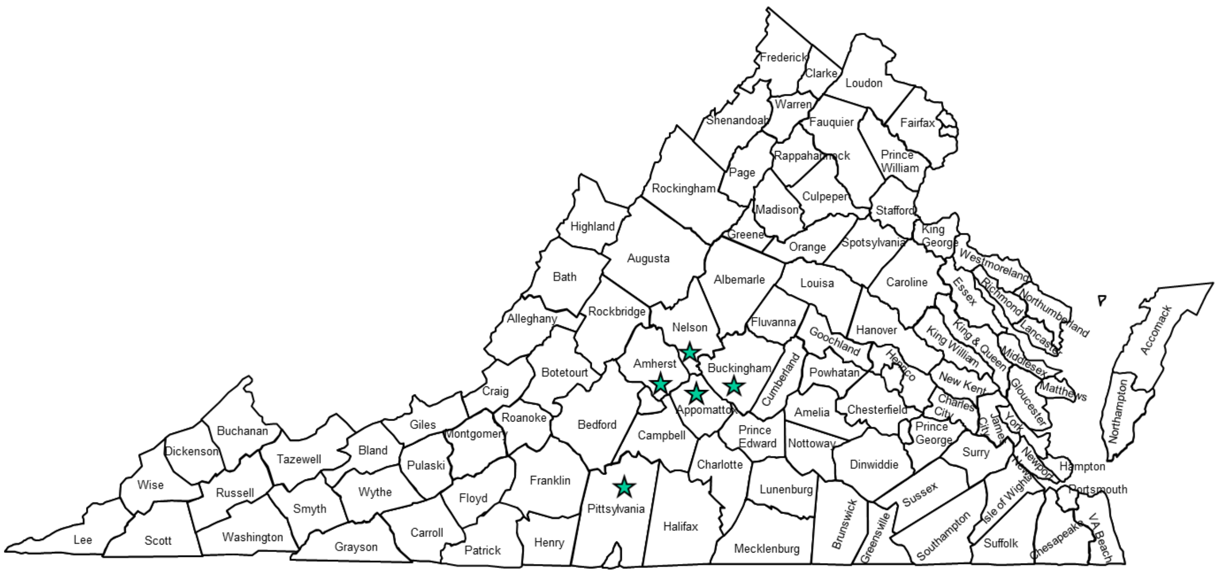

Figure 1. Study site locations in the Piedmont physiographic region of Virginia. Counties are marked with a star and include Nelson, Pittsylvania, Amherst, Appomattox, and Buckingham.

One harvest had four stream crossings (crossing numbers 2-5) that were each located on the same stream (Table 1). Another operation had two crossings (crossing numbers 8, 9) located on the same timber tract, but the crossings were on separate streams. The area of harvest on the far side of each crossing ranged from 0.17 to 7.29 ha. This acreage represents the amount of timber harvested and skidded across the stream towards the landing. Four of the stream crossings served small areas (<1 ha) because of the proximity of boundary lines across the streams. Although some stream crossings served small areas, each crossing had at least 24 skidder passes. Maximum stream crossing approach slopes ranged from 6-18%. Soil K values (which express soil susceptibility to erosion) were similar, with values ranging from 0.24 to 0.32.

Table 1. Site specifications for each stream crossing. Crossings 2 through 5 were located on the same stream. Crossings 8 and 9 were located on the same timber tract, but separate streams.

Crossing	Treatment	Hectares harvested (that used crossing)	Tonnes harvested (that used crossing)	Average approach slope (%)	Maximum approach slope (%)	Soil texture	Soil series	Soil K value
1	Mulch	1.60	506.43	8	11	Silt loam	Elioak	0.32
2	Slash	0.24	70.65	11	18	Silt loam	Spears Mountain	0.32
3	Mulch+silt	0.67	194.31	14	18	Silt loam	Spears Mountain	0.32
4	Slash	0.30	88.32	13	13	Silt loam	Spears Mountain	0.32
5	Mulch	0.17	48.28	10	18	Silt loam	Spears Mountain	0.32
6	Slash	5.47	1617.47	5	6	Silt loam	Delanco-Elsinboro Complex	0.32
7	Mulch+silt	2.26	613.48	8.5	12	Sandy loam	Mayodan	0.24
8	Mulch+silt	4.86	575.76	9	14	Clay loam	Mecklenberg-Poindexter Complex	0.28
9	Mulch	7.29	863.65	6	8	Clay loam	Mecklenberg-Poindexter Complex	0.28

Physical characteristics of each stream are presented in Table 2. Manning-Chezy values were measured at each stream crossing in order to determine relative flow rates (Ward and Trimble 2004). The stream that crossing 6 was located on had the highest flow rates compared to all others. Crossings 7 and 9 had very deep stream channels, with unstable, cut banks present.

Stands were clear-cut harvested using rubber-tired feller bunchers and grapple skidders. Average total harvest area was 65 ha. Skidder stream crossing locations were identified prior to harvest by a professional forester in order to minimize the number of crossings while accessing timber. Stream crossings were steel paneled bridges varying from 7.3 to 9.7 meters in length.

Table 2. Physical characteristics of each stream. Crossings 2 through 5 were located on the same stream. Crossings 8 and 9 were located on the same timber tract, but separate streams.

County	Crossing #	Stream bed material	Height of bank	Width of channel	Manning-Chezy total discharge/year (in millions)
Nelson	1	Gravel, cobble, silt/clay	1.2 m	1.0 m	14.3 m^3/year
Buckingham	2,3,4,5	Gravel, cobble	0.6 m	1.2 m	21.1 m^3/year
Amherst	6	Gravel, cobble	0.3 m	1.8 m	40.9 m^3/year
Pittsylvania	7	Gravel, cobble, few boulders	1.9 m	0.8 m	13.5 m^3/year
Appomattox (stream 1)	8	Gravel, silt	0.3 m	1.0 m	6.9 m^3/year
Appomattox (stream 2)	9	Gravel, silt	2.0 m	0.6 m	9.0 m^3/year

Three 1-meter wide panels (3 m wide total) were used on each crossing. Panels were installed and removed with rubber tired grapple skidders, thus some stream bank disturbance occurred. A 15 m streamside management zone (SMZ) was intended for each side of the streams, but SMZs ranged from 13-45 m.

Study sites had mean annual precipitation values ranging from 1070 to 1140 mm yr^{-1}, and a mean annual air temperature between 18 and 22°C during the growing season and between 5 and 8°C during the dormant season (USDA Natural Resource Conservation Service 2011). Topography was rolling with average sideslopes of 15% and maximum sideslopes of 30%. Stream crossings were on intermittent streams having watershed sizes from 3 to 39 hectares above the crossing points. Each of the five sites also had similar soil types, being either a typic alfisol or typic ultisol (USDA Natural Resource Conservation Service 2011). As is typical for the Piedmont, all sites had a history of prior agricultural disturbance, abandonment, and establishment of old field forests prior to industrial forest management (Nutter and Douglass

1978). During the agricultural period excessive erosion and gullying occurred and as a result over 60 cm of soil is believed to be lost. Thus, the soils of the Piedmont have low productivity and sediment originating from the past disturbance is still present in the streams (Trimble 1974, Nutter and Douglass 1978, Jackson et al. 2005). Many sites are dominated by legacy erosion gullies that are still visible and present potential problems (Trimble 1974).

2.2 Study Design

After harvests, skidder bridges were removed and three BMP closure treatments were randomly applied to nine stream crossings (18 approaches). The approaches were defined as the skid trail area on either side of the stream and within the SMZ. Each treatment was replicated three times for a total of nine stream crossings having 18 approaches. The stream crossing closure treatments are provided below (Figure 2).

Figure 2. Representative stream crossing approaches that were closed with slash treatments (a), mulch treatments (b), and mulch + silt fence treatments (c).

1. Slash – A grapple skidder removed logging slash (tree tops and limbs) from decks and slash piles and placed it on the skid trail approaches (not in the stream). Slash was piled to depths ranging from 0.25 to 1 m (Figure 2a).

2. Mulch – Grass seed, fertilizer, lime (to promote grass establishment), and straw mulch were spread on the approaches (not in the stream), with the mulch providing 100% coverage of bare soil. Each approach was covered with 10 bales of mulch, equating to 20 bales per crossing (Figure 2b).

3. Mulch + silt fence – Silt fences were installed <1 m from the stream bank, parallel to the stream on both sides of the stream channel. Installation included recommended burial of the fence into a trench in order to effectively trap sediment carried by overland flow. In addition, grass seed, fertilizer, lime, and straw mulch were spread on the approaches (not in the stream), with the mulch providing 100% coverage of bare soil. Each approach was covered with 10 bales of mulch, equating to 20 bales per crossing (Figure 2c).

The streams in the study were first or second order intermittent streams. At each stream crossing, two automated water samplers, either ISCO 3700 (Teledyne Isco, Inc., 4700 Superior St. Lincoln, NE 68504) or Sigma 900MAX (Hach Company, P.O. Box 389, Loveland, CO 80539) were installed. One automated sampler was positioned approximately 10 meters upstream and the second was positioned 10 meters downstream from the crossing in a similar fashion to the stream crossings evaluations in Taylor et al. (1999) (Figure 3).

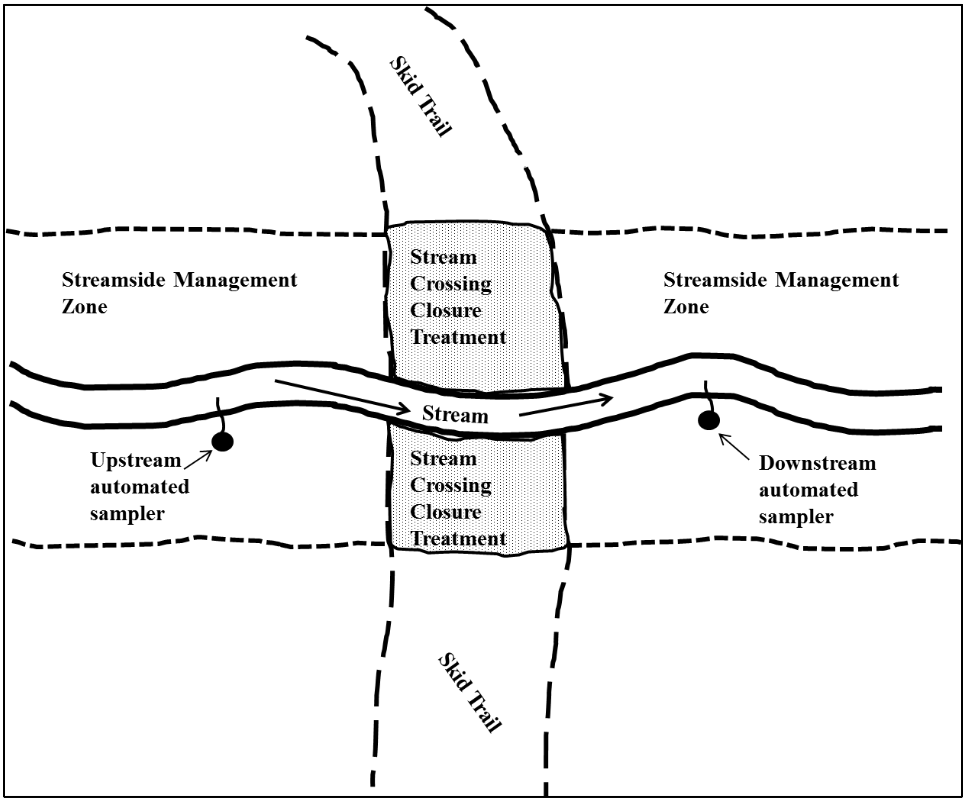

Figure 3. Idealized diagram of study sites. Not to scale. Water samplers were placed approximately 10 meters upstream and 10 meters downstream from the stream crossing. The stream crossing closure treatments were placed on the skid trail within the SMZ.

The water samplers were installed after harvest (for equipment safety and logistical reasons), but before the closure treatments were applied (which ranged from a period of 1-10 days depending on the location). The water samplers were placed uphill from the streambanks and were powered with 12-volt marine batteries. Vinyl tubing connected the sampling pump to the intake filter. The weighted intake filters were positioned in riffle sections of the streams and were attached to the gravel streambeds with landscaping staples. The streams ranged from 5 to 20 cm in depth during baseflow conditions. All automated water samplers were programmed to collect one 500 mL sample per day at 10 AM. After the sample was pumped to the housing and dispensed into its designated bottle, the tubing was purged of water. Each sampler holds 24 water samples, thus the retrieval of the samples occurred every three weeks and samples were taken to the lab for analysis. Water quality was evaluated by analyzing the samples for total suspended solids (TSS) using the method outlined by Eaton et al. (2005). Filters used for TSS extraction were 47 mm in diameter and had pore sizes of 1.5 μm. Data collection continued for one year following harvesting. Daily precipitation data were collected from National Oceanic and Atmospheric Administration (NOAA) weather stations that were closest to each tract. Each day a rainfall value was noted, and upstream and downstream TSS values were collected.

The Universal Soil Loss Equation (USLE) was used to predict potential erosion rates from each stream crossing approach after the closure treatments were applied. Variables included slope steepness and length (LS), cover (C), rainfall (R), management practice (P), and soil erodibility (K). The USLE equation is: A = RKLSCP where A equals the average annual soil loss in tonnes/hectare (Dissmeyer and Foster 1984). This modeling method was shown to provide satisfactory erosion estimates for Piedmont skid trails by Wade et al. (2013). These USLE data were used for comparison with other similar studies which used the same treatments to close out skid trails.

The raw data (physical features of the approaches) used for the calculation of the USLE were also used for correlation with in-stream TSS loading values. Loading values were calculated by correlating the flow data from these streams with daily rainfall values, since rainfall values were available for each day, whereas the flow data was sporadic as a result of equipment malfunctions. The average flow rate for the year was used to calculate TSS loading for each sample. Flow was recorded in $cm^3\ sec^{-1}$ and TSS was recorded in $mg\ L^{-1}$. After converting the units, the final loading value was expressed as $Mg\ yr^{-1}$.

Treatment costs were reported by the loggers responsible for installation. Costs included both materials and labor. The slash treatment did not require a material cost, so costs were based on labor and machine time only. Reported costs were reported on average for each treatment.

2.3 Statistical Analysis

Statistical analyses were based on the methods from a similar road surface and sediment generation study by Clinton and Vose (2003), which used rain events as statistical blocks in order to control TSS variation at different rainfall intensities. In this study, four rainfall categories were established by dividing the rainfall data into quartiles above zero, and then combining the lowest category with the days with no rain. The categories were as follows: low = 0.00 - 0.10 cm, medium = 0.11 - 0.40 cm, high = 0.41 - 1.00 cm, and maximum > 1.00 cm. A daily TSS percent change value was calculated for analysis using the following equation:

Daily TSS percent change = [(Downstream TSS – Upstream TSS)/Upstream TSS] x 100

Data were analyzed for statistical significance using JMP Statistical Discovery Software (JMP, Version 9). Data were not normally distributed, thus, non-parametric tests were used. Both the Kruskal-Wallis test (Ott and Longnecker, 2010a) and the Wilcoxon test (Ott and Longnecker, 2010b) were used to detect treatment differences. Rainfall categories were analyzed separately creating statistical blocks.

Limited post-harvest, pre-closure TSS data were collected prior to the installation of the treatments. These pre-treatment data were compared at the rainfall categories greater than 0.10 cm (medium, high, and maximum) as a separate control treatment. The low rainfall category was omitted because of the assumption that it is base flow. The BMP efficiency of each treatment was evaluated by calculating percent change in TSS, compared with the pre-treatment data using the following equation, adapted from Edwards and Williard (2010):

$$\%\ BMP_{efficiency} = [(Pre\text{-}treatment - Treatment)/Pre\text{-}treatment] \times 100$$

Where treatment is the mean percent change in TSS for the respective treatment at the rainfall category being evaluated, and pre-treatment is the mean percent change in TSS of the pre-treatment values at that rainfall category.

The physical features of the stream crossing approaches were measured and analyzed for significance with a Pearson's correlation matrix using JMP Statistical Discovery Software (JMP, Version 9).

3. Results and Discussion

3.1 Total suspended solids

Results from the Kruskal-Wallis statistical test indicated the rank in which the treatments performed (Table 3). Higher scores (score mean values) indicate higher sediment values downstream, compared to upstream values. The results of the Wilcoxon test shows the treatment differences between each paired treatment at each rainfall category (Table 4). The rainfall categories which displayed significant differences between treatments were low, medium, and high (in the Kruskal-Wallis test). The maximum rainfall category had a p-value of 0.1212, which at an alpha level of 0.15, would be considered significant. This higher alpha level is commonly used in studies such as this with a natural and operational component. The letters were inserted into the score mean column of the Kruskal Wallis test after evaluating the specific treatment

Table 3. Results of Kruskal-Wallis Test. The score mean values show the rank in which the treatments performed. Higher scores (score mean values) indicate a higher percentage of sediment downstream, compared to other treatments. The asterisk (*) in the P-value column denotes significant differences between treatments at the respective rainfall category, at $\alpha = 0.05$. Score means not connected by the same letter are significantly different, according to the Wilcoxon test.

Daily Rainfall Category	Chi Square	P-value	Treatment	N	Score Mean
Low 0.00 – 0.10 cm	14.9433	0.0006*	Slash	245	193.27 a
			Mulch	96	231.95 b
			Mulch + Silt fence	83	246.77 b
Medium 0.11 – 0.40 cm	9.0407	0.0109*	Slash	27	24.14 a
			Mulch	16	26.25 a
			Mulch + Silt fence	13	40.30 b
High 0.41 – 1.00 cm	11.7111	0.0029*	Slash	37	38.00 a
			Mulch	31	43.90 a
			Mulch + Silt fence	23	61.69 b
Maximum > 1 cm	4.2202	0.1212	Slash	43	42.25 a
			Mulch	24	40.95 a
			Mulch + Silt fence	22	54.77 b

Table 4. Results from Wilcoxon test. Each treatment was compared with all other treatments within each rainfall category. The asterisk (*) in the P-value column denotes significant differences between the two treatments being compared at $\alpha = 0.10$. Score mean difference is the difference between the score means from the Kruskal-Wallis test, with the standard error difference factored in.

Daily Rainfall Category	Treatment vs.	Treatment	Score Mean Difference	Standard Error Difference	Z	P-value
Low 0.00 – 0.10 cm	Mulch	Slash	30.567	11.870	2.575	0.0100*
	Mulch + silt fence	Slash	41.969	12.044	3.485	0.0005*
	Mulch + silt fence	Mulch	5.425	7.766	0.699	0.4848
Medium 0.11 – 0.40 cm	Mulch + silt fence	Slash	10.826	3.946	2.743	0.0061*
	Mulch + silt fence	Mulch	8.016	3.179	2.521	0.0117*
	Mulch	Slash	2.140	3.961	0.540	0.5891
High 0.41 – 1.00 cm	Mulch + silt fence	Slash	15.440	4.637	3.329	0.0009*
	Mulch + silt fence	Mulch	10.678	4.329	2.466	0.0136*
	Mulch	Slash	4.505	4.814	0.935	0.3494
Maximum > 1 cm	Mulch + silt fence	Slash	9.378	4.956	1.892	0.0584*
	Mulch + silt fence	Mulch	6.751	3.961	1.704	0.0883*
	Mulch	Slash	-1.201	4.964	-0.241	0.8088

comparisons from the Wilcoxon test. The low rainfall category showed that slash was significantly different than the mulch and mulch + silt fence treatments, with regards to sediment levels in the stream. The lower score mean value indicated that the slash performed better than the other two treatments with regards to sediment reduction at the low rainfall category. However, the medium, high, and maximum rainfall categories showed a different trend, but one that was identical throughout all three categories. They indicated that the slash and mulch treatments were statistically the same, while they both were different than the mulch + silt fence treatment. Because the slash and mulch treatments had lower score mean values compared to the mulch + silt fence treatment, it can be concluded that the slash and mulch treatments performed better than the mulch + silt fence treatment.

Median TSS percent difference values are provided in Figure 4. The graph of the median TSS values explains the biological significance of the treatments by displaying which treatments had positive impacts and which ones had negative impacts on stream sediment levels. These results cannot be seen using the non-parametric tests. Positive median values indicate that new sediment entered the stream at the stream crossing, and negative median values imply that no new sediment entered the stream at the stream crossing. The slash treatment had negative median values within all rainfall categories, which indicates there was more sediment upstream than downstream of the stream crossing, and that no new sediment entered the stream at the crossing. The mulch treatment displayed the same trend, except for the low rainfall category. The mulch + silt fence treatment had positive sediment values at all rainfall categories, indicating that it allowed new sediment to enter the stream at the stream crossing approach. Overall, the slash and mulch treatments did not allow new sediment to enter the stream at the stream crossing, while the mulch + silt fence treatment did.

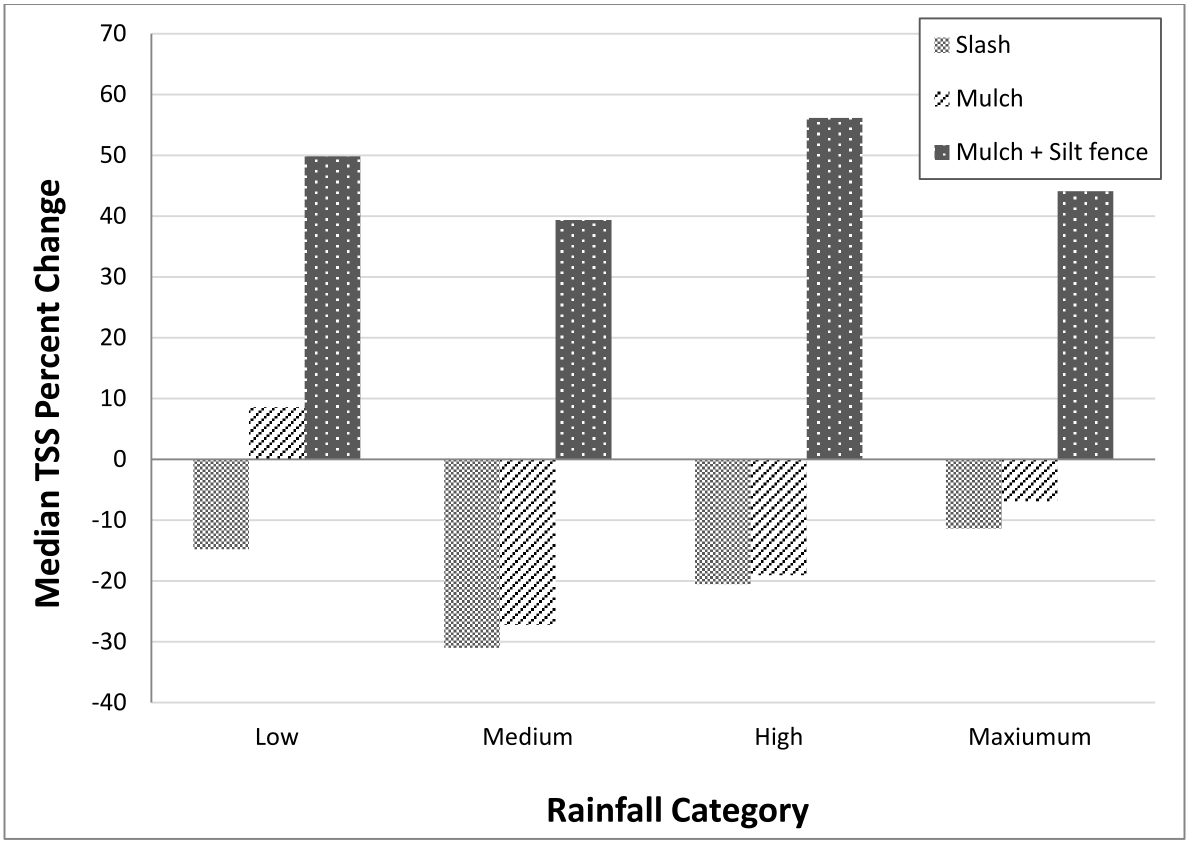

Figure 4. Median TSS percent change values at each treatment and rainfall category. Negative values indicate that sediment was trapped at the stream crossing, and positive values indicate that new sediment entered the stream at the stream crossing. Rainfall categories are based on total daily precipitation: low = 0.00 - 0.10 cm, medium = 0.11 - 0.40 cm, high = 0.41 - 1.00 cm, maximum > 1.00 cm.

The mulch + silt fence treatment resulted in the greatest downstream sediment increase. Silt fence is a proven BMP for reducing silt-size and larger sediment (Robichaud and Brown 2002), but its installation requires disturbance. The silt fence was installed immediately adjacent to the streams (< 1 m from the stream bank), thus the installation disturbances were unfortunately positioned to introduce sediment. Another explanation of silt fence failure could be the high clay content commonly found in the Piedmont of Virginia. Clay soil particles are smaller than silt particles and therefore have the ability to pass through silt fence. These results show the need to minimize disturbances within the riparian zone even while installing BMPs designed to reduce sedimentation.

Average total suspended solids (TSS) values for each rainfall category and treatment are presented in Table 5 and Figure 5. Although the data are not normally distributed, and therefore the averages were not evaluated for significance, it is valuable to compare average TSS values with other studies. Other stream crossing studies have shown a general trend of higher sediment values downstream from stream crossings, even with BMPs installed. Aust et al. (2011) found TSS increases of 144 to 252 mg L^{-1} downstream of stream crossings with a variety of crossing types. During storm events, Rothwell (1983) found on average a 41% increase in TSS downstream at a mulched approach (from 20 mg L^{-1} upstream to 34 mg L^{-1} downstream), and a 76% increase at a bare approach (from 28 mg L^{-1} upstream to 118 mg L^{-1} downstream). During our greatest storm events (maximum rainfall category), the slash

Table 5. Average TSS values (mg L^{-1}) upstream, downstream, and differences at each treatment and rainfall category. In the average difference column, positive values indicate higher TSS levels downstream of the crossing, while negative values indicate higher TSS values upstream of the crossing compared to downstream.

Daily Rainfall Category	**Treatment**	**Average Upstream TSS (mg L^{-1})**	**Std Error**	**Average Downstream TSS (mg L^{-1})**	**Std Error**	**Difference (Down – Up) TSS (mg L^{-1})**	**Std Error**
Low 0.00 – 0.10 cm	Slash	30.74	4.91	26.45	10.02	-4.29	10.60
	Mulch	85.49	8.41	136.98	16.24	51.49	16.84
	Mulch + Silt Fence	11.93	7.52	21.05	17.92	9.12	18.21
Medium 0.11 – 0.40 cm	Slash	51.83	16.45	40.84	30.05	-10.99	26.46
	Mulch	90.47	22.89	115.55	38.79	25.08	34.37
	Mulch + Silt Fence	11.22	21.58	30.23	41.90	19.01	38.13
High 0.41 – 1.00 cm	Slash	54.31	16.81	49.05	25.62	-5.26	21.04
	Mulch	132.93	19.33	164.73	30.38	31.80	22.99
	Mulch + Silt Fence	12.60	20.34	23.69	34.37	11.09	26.68
Maximum > 1 cm	Slash	140.92	34.86	127.49	31.79	-13.43	28.44
	Mulch	235.26	50.29	227.57	47.89	-7.69	38.06
	Mulch + Silt Fence	36.62	49.31	49.46	47.89	12.84	39.75

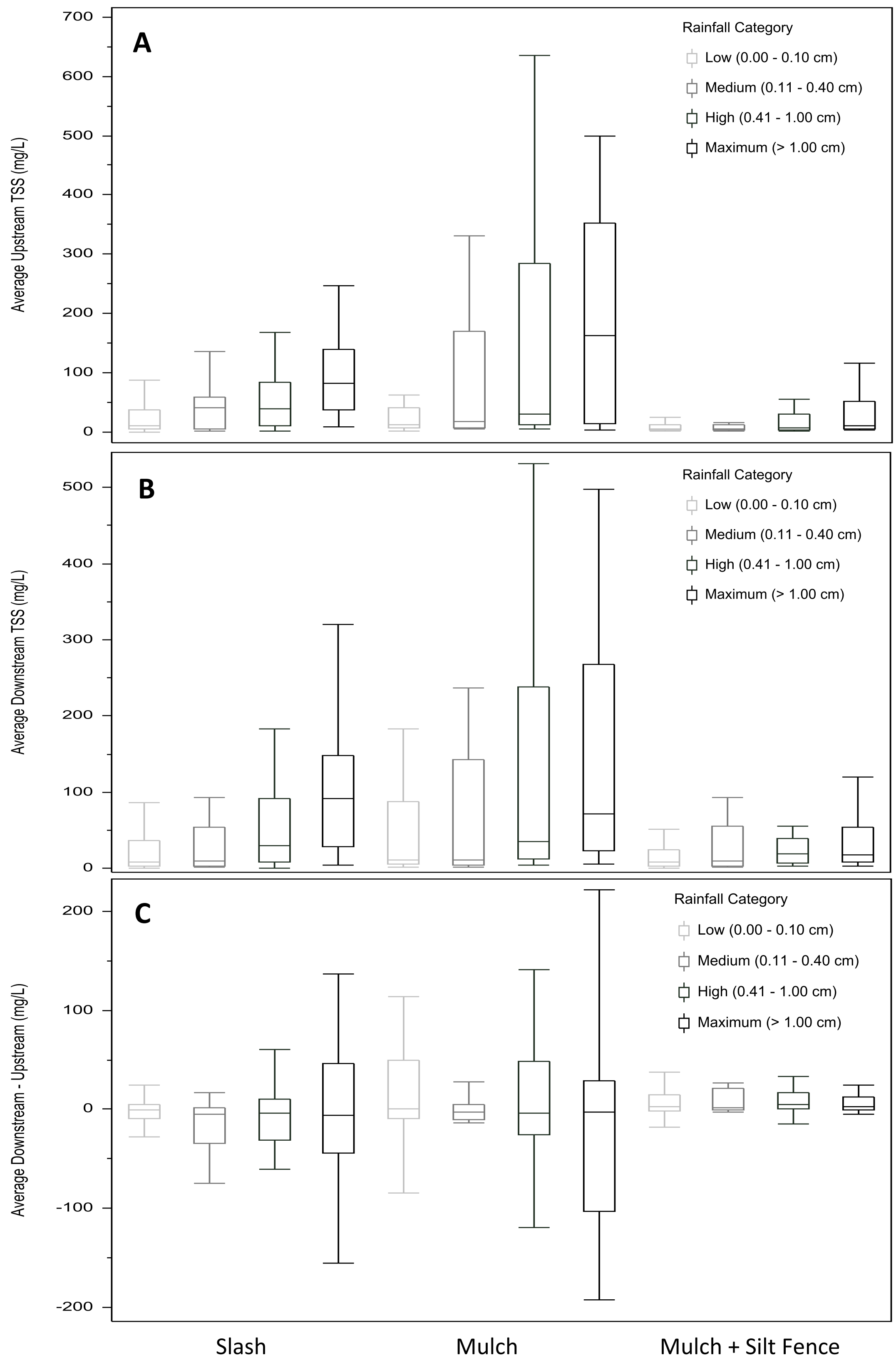

Figure 5. TSS averages upstream (A), downstream (B), and downstream – upstream (C).

treatment tended to decrease sediment levels downstream (from 140 mg L^{-1} upstream to 127 mg L^{-1} downstream), and the mulch also tended to reduce sediment levels downstream (from 235 mg L^{-1} upstream to 227 mg L^{-1}downstream). However, the mulch + silt fence treatment tended to add new sediment to the stream at the crossing (from 36 mg L^{-1} upstream to 49 mg L^{-1} downstream). The mulch + silt fence treatment added new sediment to the stream at all rainfall categories (according to both the mean and median data), perhaps due to silt fence installation disturbances.

3.2 BMP efficiency

BMP efficiency results are displayed in Table 6. The treatments were not significantly different with regards to TSS mean percent difference when analyzed using the student's t-test at an alpha level of 0.05. However, the calculation of BMP efficiency is beneficial for

Table 6. BMP efficiency measured in percent reduction in TSS (+ efficiency) or percent increase in TSS (- efficiency) after closure treatments were applied. The no rain and low rainfall categories were omitted because of the assumption that they are base flow stream measurements. The "control" treatment represents pre-treatment values. For each rainfall category, the slash was the most effective treatment, with the highest percent reduction in TSS.

Rainfall Category	**Treatment**	**TSS Mean Percent Difference**	**Mean Standard Error**	**BMP Efficiency**
Medium (0.1 – 0.4 cm)	Pre-treatment	76.56 **a**	57.8	
	Slash	2.11 **a**	35.7	+97.2%
	Mulch	0.31 **a**	18.0	+99.6%
	Mulch + Silt fence	456.70 **a**	211.7	-496.9%
High (0.4 – 1.0 cm)	Pre-treatment	97.7 **a**	39.6	
	Slash	31.5 **a**	24.6	+67.7%
	Mulch	127.0 **a**	55.6	-29.9%
	Mulch + Silt fence	475.7 **a**	229.7	-386.0%
Maximum (> 1.0 cm)	Pre-treatment	100.6 **a**	45.4	
	Slash	37.5 **a**	18.3	+62.7%
	Mulch	84.7 **a**	45.9	+15.8%
	Mulch + Silt fence	89.9 **a**	29.5	+10.6%

understanding the change in sediment levels that occurred after each treatment was applied. Some treatments increased sediment levels following installation, while others helped to reduce stream sediment levels following their installation. For the medium rainfall category, both the slash and the mulch treatments reduced TSS values by 97.2% and 99.6%, respectively. However, the mulch + silt fence treatment increased TSS values by 496.9%. At the high rainfall category, slash was the only treatment that effectively reduced TSS values, with a reduction of 67.7%. The mulch and mulch + silt fence treatments increased sediment downstream at the high rainfall category. At the maximum rainfall category, all three treatments were effective in reducing TSS values. The most effective was the slash treatment (62.7% reduction), followed by the mulch treatment (15.8% reduction), and finally the mulch + silt fence treatment (10.5% reduction). Overall, the slash and mulch were both more effective at reducing sediment levels compared to pre-treatment levels than the mulch + silt fence treatment. The mulch + silt fence tended to add new sediment to the stream following its installation.

3.3 USLE Estimates and Physical Features

USLE erosion estimates for the stream crossing approaches ranged from 0.011 to 38.304 Mg ha^{-1} $year^{-1}$ (Table 7). Crossing 1 had the largest USLE prediction due to an extremely long stream crossing approach as a result of two SMZs intersecting in a “y” shape. Crossing 5 also had a high USLE prediction due to a steep slope and a long approach as a result of nearby boundary lines. However, percent cover values were comparable to the other crossing approaches of the same treatment. Therefore, because the treatments were randomly applied, the two mulched crossings were coincidently on longer approach lengths, thus generating much larger USLE estimates than the others. The generally low USLE erosion predictions indicate

Table 7. USLE estimates taken at each stream crossing approach after closure treatments were applied.

Crossing	Treatment	USLE estimate Approach 1	USLE estimate Approach 2	USLE Average by Crossing	USLE Average by Treatment
		………………	Mg ha^{-1} year^{-1}	……………………	……………
2	Slash	0.011	1.501	0.756	
4	Slash	0.605	0.112	0.358	
6	Slash	0.224	0.022	0.123	0.412
1	Mulch	0.907	38.304	19.605	
5	Mulch	0.045	16.755	8.400	
9	Mulch	0.918	0.694	0.806	9.604
3	Mulch+silt	1.075	0.717	0.896	
7	Mulch+silt	0.175	0.336	0.271	
8	Mulch+silt	0.246	0.045	0.146	0.437

that all of the closure treatments provided substantial cover to the stream crossing approaches (Table 7).

The physical features of the stream crossing approaches that were used for the calculation of the USLE were compared with the TSS change loading values (Mg year^{-1}) at each crossing. Although we know that the features are correlated with erosion rates, this analysis was used to assess their significance with regards to in-stream sediment values. The TSS loading values were used because they normalized the data for various flow rates. The loading values and physical features were analyzed for significance using Pearson's correlation matrix (Table 8). Slope length and area of approach were expectedly correlated because length was one of the two factors involved in the calculation of area. The increases in total suspended solids (between the upstream and downstream sampling locations) were positively correlated with slope length and

Table 8. Pearson's correlation matrix of the physical features of the stream crossing approaches and TSS loading values.

	Area of Approach (acres)	**Slope Length (m)**	**Slope Percent**	**Percent Bare Soil**	**TSS Loading (Mg year^{-1})**
Area of Approach (acres)	1.0000	**0.9025**	0.0048	-0.2690	**0.3980**
Slope Length (m)	**0.9025**	1.0000	0.0473	-0.0837	**0.5871**
Slope Percent	0.0048	0.0473	1.0000	-0.0156	-0.1966
Percent Bare Soil	-0.2690	-0.0837	-0.0156	1.0000	-0.2303
TSS Loading (Mg year^{-1})	**0.3980**	**0.5871**	-0.1966	-0.2303	1.0000

slope percent. The stream crossing approach was defined as the area of the skid trail on both sides of a stream crossing, within the SMZ. This correlation is expected because stream crossing approaches having greater area would have greater erosion potential. It has been shown that decreasing the length of road that drains directly into streams at road-stream crossings can effectively reduce sediment delivery (McGreer et al. 1998). In order to decrease stream crossing approach length, water turnouts and wing ditches should be implemented along the skid trail, leading to the approach, which is also recommended by Croke et al. (1999). Reducing total approach area will reduce the amount of sediment which could potentially be introduced to the stream. Other studies have also concluded that soil movement and sediment delivery could be reduced by minimizing the quantity and size of skid trails (McBroom et al., 2008) and contributing disturbed areas (Croke et al. 1999). Aust et al. (2011) found that area of SMZ disturbance was positively related to downstream sediment.

Slope percent and percent bare soil were not correlated with TSS increases in the streams. The slope values on the study sites had little variability (6-18%), which might have contributed to them not being correlated with stream sediment levels. It was apparent that the length of the stream crossing approach was more of a contributing factor to stream sediment than the slope percent. All approach treatments provided substantial coverage of bare soil, thus also having little variability. Complete bare soil coverage of the stream crossing approach can substantially reduce the sediment which could otherwise enter the stream.

Erosion estimates were compared with recent studies which used similar treatments to close out skid trails (Table 9). The two studies used for comparison (Sawyers et al. 2012; Wade et al. 2013) assessed several methods of skid trail closure, as well as the accuracy of USLE

Table 9. Erosion control comparison with similar BMP closure studies. "Pine" slash was averaged together with "hardwood" slash on both comparison studies, for a single "slash" value. Since we did not collect pre-treatment USLE data on the stream crossing approaches, the control, or "bare soil" value was generated from the adjacent skid trail to each crossing.

	Stream crossing approaches (USLE estimate) (Mg ha^{-1} year^{-1})	**Wade et al. 2013 – bladed skid trails (measured erosion) (Mg ha^{-1} year^{-1})**	**Sawyers et al. 2012 – overland skid trails (measured erosion) (Mg ha^{-1} year^{-1})**
Bare soil	39.55	137.70	24.24
Slash (% change from bare)	0.41 (+98.9%)	7.40 (+94.6%)	5.24 (+78.3%)
Mulch and seed (% change from bare)	9.60 (+75.7%)	3.00 (+97.8%)	3.29 (+86.4%)

predictions with actual measured erosion using geotextile sediment traps. Wade et al. (2013) found that the USLE was an adequate predictor of actual erosion. Sawyers et al. (2012) found that the USLE was an acceptable predictor with regards to ranking different cover types accurately. The data used for comparison were from their actual measured erosion (since it was available and more accurate than USLE), while the data from this study was a USLE estimate.

Both Sawyers et al. (2012) and Wade et al. (2013) found that using straw mulch (with seed) as a closure treatment on skid trails was slightly more effective at reducing erosion compared to slash, although slash was still effective. However, they both suggested that over time, the slash could outperform the mulch after the mulch decomposes. Slash will take more time to decompose and could provide better coverage over a long period of time. The USLE estimates from the stream crossing approaches suggested that slash works better than mulch in terms of reducing potential erosion; the mulch treatment reduced erosion by 75.7%, while the slash treatment reduced erosion by 98.9%. However, since the percent bare soil (one of the variables used in calculating the USLE) was comparable for all of the treatments, and the main difference triggering a higher USLE estimate at the mulch treatment was the percent slope and slope length, the slash and mulch treatments could be considered equal in terms of providing cover to bare soil. Slash also has the advantage of providing some closure potential for minimizing ATV traffic. Christopher and Visser (2007) found that post-harvest ATV traffic was a significant cause of BMP failure in Virginia.

3.4 Cost

The itemized costs of each treatment were reported by the logging contractors and are presented in Table 10. The slash treatment was the least expensive option, at $120 per stream crossing, assuming that logging slash is available on site, and that it is moved after harvest has been completed. The cost consists of 2 hours of operator and machine time for slash application. This cost would further be reduced if slash was spread on stream crossing approaches during

Table 10. Treatment costs per stream crossing as reported by the logging contractors.

Treatment	Materials	(cost)	Labor	(cost)	Total Cost per Stream Crossing
Slash	Logging slash	n/a	Skidder machine time (2 hrs)	$120	$120
Mulch	Straw mulch (20 bales) Lime Fertilizer and seed	$100 $5 $5	Dozer machine time Manual labor (2 hrs)	$90 $80	$280
Mulch + Silt fence	Straw mulch (20 bales) Lime Fertilizer and seed Silt fence	$100 $5 $5 $25	Dozer machine time Manual labor (3 hrs)	$90 $120	$345

normal logging operations, while the skidder returns empty, or while bridges are being removed. This method is also known as integrated slash (Sawyers et al. 2012). The mulch treatment was the option of intermediate costs, at $280 per stream crossing and includes material and labor. The most expensive treatment was the mulch + silt fence application, which cost $345 per stream crossing, including materials and labor.

In comparison, McKee et al. (2012) found that the estimated costs to install stream crossing BMPs ranged from $533 to $655 throughout the state of Virginia. However, the majority of these numbers include the installation of water bars on the skid trails, as well as different combinations of treatments including straw mulch, seeding, slash, water turnouts, staked bales, gravel, and silt fence.

3.5 Hypotheses Revisited

Hypothesis 1 stated that three levels of stream crossing approach BMPs will not result in significant differences between percent change of upstream and downstream TSS levels. Our findings indicated that the slash and mulch treatments were statistically the same with regards to sedimentation rates in the stream. The mulch + silt fence treatment was significantly different than both the slash and mulch treatments and tended to introduce the most sediment to the stream.

Hypothesis 2 stated that three levels of stream crossing approach BMPs will not result in significantly different BMP efficiencies. BMP efficiency was used to compare pre-treatment sediment data to post-treatment sediment data. The TSS percent change values that were used to calculate BMP efficiency were not significantly different. However, the BMP efficiency results could not be tested statistically because they did not involve repetition. The BMP efficiencies ranged from -497% to +99%, which is a difference of almost four orders of magnitude between sediment levels.

Hypothesis 3 stated that USLE estimates will not result in different levels of erosion between the three stream crossing approach closure treatments. Our results indicate that the USLE estimates did not result in different erosion estimates between treatments. The data did show two outliers with higher erosion rates, but they were a result of the physical features of the stream crossing approaches, not the amount of cover provided by the treatments. Overall, the three treatments provided similar cover to the bare soil.

Hypothesis 4 stated that each of the three treatments will have similar costs. Results indicate that the three treatments had different costs. The slash treatment cost $120 per stream crossing, the mulch treatment was $280 per stream crossing, and the mulch + silt fence was $345 per stream crossing.

4. Summary and Conclusions

The impact of forest roads and skid trails on stream health is clearly recognized. The stream crossing portion of a skid trail is where sediment delivery has the greatest potential to occur. Thus, the closure and reduction of bare soil at stream crossing approaches could significantly reduce sediment delivery to streams on harvested sites. Our results indicate that applications of slash or seed and mulch to the stream crossing approaches immediately following the removal of temporary bridges protects water quality. Slash was the least expensive option, and therefore would be more desirable. If slash is not available on site, mulch and seed is another viable, but potentially more expensive option. The immediate coverage provided with either slash or mulch at the stream crossing approach protects bare soil from erosive forces that would otherwise carry exposed sediment to the adjacent stream. Although these treatments are effective at reducing surface erosion, the complete elimination of sediment inputs to the stream is challenging because of other forms of erosion such as channel erosion and seepage erosion. These forms of erosion are complex and should be investigated further with regards to land use.

Slash and mulch treatments have proven effective on both overland (Sawyers et al. 2012) and bladed skid trails (Wade et al. 2013). Coverage of the skid trail effectively reduces surface erosion from occurring. The coverage adjacent to the stream not only prevents the generation of erosion at the stream crossing approach, but it also acts to intercept sediment already moving downhill along the skid trail (assuming little or no cover on the skid trail) before it is delivered to the stream. This study indicates that the nearly complete coverage provided by the slash or mulch treatments were more important than the slope of the approach (at slopes up to 18%). Silt fences should not be used in close proximity to streambanks if alternative options exist because the disturbance required during its installation is greater than its benefits.

Slash was the most cost effective option. Forest operations support a large portion of Virginia’s economy and cost of implementing BMPs can impact the overall cost to harvest timber. Harvesting costs are absorbed by loggers or reflected in lower stumpage values to landowners. Logging slash is commonly available on site in the form of tree tops and limbs, and if integrated into normal skidding operations the cost could be further reduced because additional machine operator time is not necessary (Sawyers et al. 2012). If slash is not available on site (e.g., due to biomass harvesting), applying straw mulch and seed to the stream crossing approach after bridge removal is the next best option in terms of cost.

Reducing erosion at its source not only maintains soil health and productivity, but also preserves stream health which is crucial for the survival and vitality of aquatic biota. This study provides land managers, landowners, and loggers with options for protecting water quality after silvicultural harvest activities are complete. Applying either slash or mulch with seed to the stream crossing approaches in a timely fashion will reduce the amount of sediment which could otherwise enter the stream at these sensitive areas. Skidder stream crossings can be effectively closed after use, as long as coverage of bare soil is completed immediately following (if not during) harvest. When stream crossing closure techniques are properly employed in combination with streamside management zones, minimal sedimentation and stream disturbance occurs after harvest.

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Chapter 3 – Summary and Conclusions Regarding the Effect of Skid Trail Closure Treatments

Study Objective

The objective of this study was to evaluate three levels of BMP stream crossing closure techniques on stream sediment levels. The treatments applied to the stream crossing approaches were: 1) Slash – consisting of logging slash, 2) Mulch – consisting of seed, straw mulch, and fertilizer, and 3) Mulch + silt fence – consisting of seed, straw mulch, fertilizer, and silt fence.

Summary

Water samples were collected upstream and downstream of the nine stream crossings daily, for one year following harvest. Samples were evaluated for total suspended solids (TSS). Treatment differences were detected using JMP statistical software (JMP, 2010). Non-parametric tests were used since the data were not normally distributed. Data were separated into four rainfall categories (low, medium, high, and maximum) as blocks in order to control for different sedimentation rates during rain events. According to both the Kruskall-Wallis and the Wilcoxon tests, the trend was the same: the slash and mulch treatments were the most effective treatments in regards to reducing the amount of downstream sediment, while the mulch + silt fence treatment was the least effective treatment. The mulch + silt fence treatment resulted in the greatest downstream sediment increase.

The slash and mulch treatments were effective at reducing erosion at the stream crossing approaches because they provided cover to soil which would have otherwise been left bare. This coverage protected the soil from rainsplash erosion, thus lowering the intensity of the impact of raindrops falling on bare soil. They also helped to stabilize the soil to protect it from sheet erosion caused by overland flow, which is common on forest roads and skid trails.

The cost of each of the treatments was also evaluated. The least-cost option was the slash treatment, because the materials were found on-site, as they often are, in the form of logging residue. The only cost incurred for the slash treatment was labor, at approximately $120. The second best cost option was the mulch treatment, which included seed, fertilizer, lime, and straw mulch, at approximately $280 per stream crossing. The most expensive stream crossing closure treatment was the mulch + silt fence treatment ($345 per crossing). This treatment included the components of the mulch treatment, with the addition of silt fence, which added additional material and labor costs. The mulch + silt fence treatment was not only the least effective closure treatment, but also the most costly. Therefore, we do not recommend using the mulch + silt fence treatment for stream crossing approach closure.

The slash treatment could be partially utilized during harvest operations, because repeated skidder travel would not be damaging to the treatment. Application of some slash during harvest would protect the soil near the streambanks when the most disturbance is occurring. Application after harvest will ensure that future erosion and sedimentation will be minimized, and the site will be "closed." However, slash is not always available on-site, as the practice of chipping it for biofuel is currently increasing (Conrad and Bolding 2011). Whichever treatment is better suited for the situation (e.g., slash availability, cost constraints), should be applied to stream crossing approaches directly following closure (if not during harvest). However, we do not recommend using silt fence directly adjacent to stream banks, if other alternatives exist, as its installation causes additional soil disturbance and potential sedimentation.

Further Applications

Off-road vehicles (ORVs) or all-terrain vehicles (ATVs) are another disturbance to consider at stream crossings, especially in our national forests (Ayala et al. 2005). Creation of trails for ORV use is unauthorized and unmanaged, with no road standards implemented. About 11 million visits to national forests annually involve the use of ORVs, which accounts for about 5% of all recreational visits (Great Lakes Environmental Center 2008). ORV trail damage is similar to skid trail damage because they compact the soil, decrease infiltration, and increase erosion rates, which can lead to increased sedimentation and stream impairment (Chin et al. 2004). They have been found to contribute large suspended sediment loads during storm events (Ayala et al. 2005). The crossing treatments evaluated here could also be applied to known ORV crossings to help reduce bare soil at the stream crossing approach, and reduce sedimentation. The slash would be a better option if trying to close out, or block the trails altogether, and mulch would be optimal if leaving the trails open.

Fire lines are another forest disturbance which could benefit from these stream crossing closure treatments. Fire lines are trails constructed to control the advancement of fire, built either with hand tools, bulldozers, or explosives (Backer et al. 2004). When fire lines are located near streams, they have an increased potential to generate sediment and turbidity, because they often have direct channels leading to a stream (Landsberg and Tiedemann 2000). Grass seeding is the most common method for reducing post-fire erosion on fire lines; however, it has had mixed results (Backer et al. 2004). The application of mulch on top of these seeded locations would further prevent erosion by reducing the exposure of bare soil, thus allowing seeds to germinate before they are washed away by rain. Slash would not be a typical choice on fire lines because it is not readily available on-site and it would negate future use of the fire line.

BMP treatments could also be applied to agricultural areas, particularly at cattle stream crossings. Agriculture has been identified by the USEPA as the predominant source of NPSP (US Environmental Protection Agency 2003). Often cattle producers use streams as a water source for their cattle, which increases the land impacts near streams. Streambank erosion results when cattle hooves repeatedly walk on the ground near a stream, as well as the reduction of riparian vegetation from grazing (Belsky et al. 1999). The treatment that would most likely work in this situation is the mulch treatment. Although seed is unlikely to become established when constantly trampled on, the coverage that the straw mulch would provide could reduce soil movement when bare soil is exposed, while still allowing cattle access. In this case, a different type of mulch should be used instead of straw mulch, as cows might prefer to eat it. Slash would only be feasible of it was available on-site, and if a certain area was intended to be excluded from grazing. Cattle would be limited in range if slash was applied over large areas. Therefore, mulch (perhaps pine straw mulch or wood chips) would be the best treatment to apply on streambanks where cattle often cross the stream, in order to reduce sedimentation resulting from streambank erosion.

Forest stream crossings, ORV trails, fire lines, and cattle crossings are all sensitive areas with one thing in common: a direct pathway of bare soil channeling directly into a stream. Any time that bare soil is adjacent to or leads directly into a stream, there is potential for sedimentation to occur. In order to reduce sedimentation that would otherwise pollute a stream, these areas should be properly managed with the application of either slash or mulch and seed. The appropriate closure technique will be condition-specific, depending on current land uses, future land uses, as well as cost limitations. This research provides landowners and land managers with complete assessments of three stream crossing closure treatments, in order to assist them in their BMP decision-making processes.

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