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Effectiveness of best management practices for sediment reduction at operational forest stream crossings

Laura R. Wear [a], W. Michael Aust [a,*], M. Chad Bolding [a], Brian D. Strahm [a], C. Andrew Dolloff [b]

[a] Department of Forest Resources & Environmental Conservation (0324), 228 Cheatham Hall, Virginia Tech, Blacksburg, VA 24061, USA
[b] USDA Forest Service Coldwater Fisheries Unit, Department of Fish and Wildlife Conservation (0321), 350 Latham Hall, Virginia Tech, Blacksburg, VA 24061, USA

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ABSTRACT

Temporary skid trail stream crossings have repeatedly been identified as having considerable potential to introduce sediment to streams. Forestry Best Management Practices (BMPs) have proven to be effective for controlling erosion and subsequent sedimentation, yet few studies have quantified sedimentation associated with various levels of BMPs for skidder stream crossings. Three skid trail stream crossing BMP treatments were installed and replicated three times to quantify BMP efficacy for reducing sedimentation. BMP treatments were: (1) slash, (2) mulch and grass seed, and (3) mulch, grass seed, and silt fence. Water samples were collected daily both upstream and downstream from operational skidder stream crossings for one year following timber harvesting and BMP treatment installation. Samples were evaluated for total suspended solids (TSSs). Results indicate that both slash and mulch treatments effectively reduced TSS following harvesting. Slash could be the preferred method of stream crossing closure, due to lower cost, especially if application is 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, seed, and silt fence treatment was the most expensive treatment and led to increased TSS, probably due to silt fence installation disturbances near the streams. Thus, silt fences should not be installed directly adjacent to streambanks, if other alternatives exist.

1. Introduction

Sediment is the leading cause of impaired waters in the United States (US Environmental Protection Agency, 2000). Increased sediment levels have been associated with impaired fish habitat (Elliot et al., 1994; Judy et al., 1982), decreased primary productivity, diversity, and abundance of macroinvertebrates and fish (Cheong et al., 1995; Wood and Armitage, 1997), and negative alteration of community structure, density, growth, and rates of reproduction and mortality in aquatic biota (Henley et al., 2000). Sediment also increases stream turbidity, which reduces light penetration, thereby decreasing photosynthesis in aquatic plants (Kirk, 1985; Ryan, 1991). Sediment can also fill harbors, reservoirs, and navigable streams and increase the cost of water treatment for human consumption (Crowder, 1987; Moore and McCarl, 1987; Holmes, 1988).

Accelerated sedimentation in streams is generally associated with anthropogenic activities within contributing watersheds, including urbanization, agriculture, silviculture, and mining (Marcus and Kearney, 1991). Land use disturbances may increase bare soil, decrease infiltration rates, increase runoff, and alter drainage pathways, thus increasing the potential for non-point source pollution (NPSP). Although forest silvicultural operations generally cause relatively low and ephemeral increases in sediment as compared to alternative land uses (Neary et al., 1989), forest roads and skid trails have significant potential to increase erosion and sedimentation (Patric, 1976; Swift and Burns, 1999; Aust and Blinn, 2004; Grace, 2005). Forest roads can alter hillslope hydrology by creating compact and less permeable surfaces (Megahan, 1972), decreasing infiltration (Grace, 2005), and increasing drainage networks with road surfaces and ditches (Wemple et al., 1996; Croke et al., 2001; Croke and Mockler, 2001; Jackson et al., 2005), thus resulting in increased overland flow, erosion, and sedimentation during rain events. Erosion rates have repeatedly been shown to be higher from roads, bladed (Wade et al., 2012a) or overland (Sawyers et al., 2012) skid trails, and log landings, compared to adjacent harvested and undisturbed areas (Yoho, 1980; Rothwell, 1983; Arthur et al., 1998; Worrell et al., 2011). Corbett et al. (1978) found that timber harvesting, if considered independently of roads, has minimal effects on stream sediment. 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

* Corresponding author. Tel.: +1 540 231 4523.
E-mail addresses: lauraw6@vt.edu (L.R. Wear), waust@vt.edu (W.M. Aust), bolding@vt.edu (M.C. Bolding), brian.strahm@vt.edu (B.D. Strahm), adoll@vt.edu (C.A. Dolloff).

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content, and permeability (Geeves et al., 2000). A majority of Forestry Best Management Practices (BMPs) are specifically developed to address erosion associated with the silvicultural access network, including roads, decks, skid trails, and stream crossings (Aust and Blinn, 2004; Anderson and Lockaby, 2011).

Forestry BMPs have been developed by many countries for reduction of erosion and protection of water bodies from sedimentation (Croke et al., 1999a; Caruso, 2000; Nisbet, 2001; Shepard, 2006; Cubbage et al., 2007; Putz et al., 2008). Within the United States, forestry BMPs exist in 43 states, 29 of which have monitoring programs for compliance (Archey, 2004). Some states have mandatory BMPs, while others have voluntary programs, and some have 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 compromise streamside management zones and provide more direct conduits for sediment to enter streams (Rothwell, 1983; Swift, 1985; Grace, 2005; MacDonald and Coe, 2007; Witmer et al., 2009; Aust et al., 2011). Sediment concentrations are often increased downstream from 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 water-bar frequency and surface roughness on skid trails (e.g., litter, logging slash, and woody debris) in order to minimize the amount of sediment deposited into nearby streams. Schoenholtz (2004) concluded 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). Aust et al. (2011) evaluated 23 forest stream crossings used for harvesting operations and concluded that the approach design and BMPs on approaches were more influential on stream water quality than the type of crossing, but generally found that temporary panel bridges were less detrimental than reinforced fords or culverts.

Stream crossings should be minimized by pre-harvest planning (Virginia Department of Forestry, 2011). However, 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 (Aust et al., 2011). 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 construction standards than haul roads (Grushecky et al., 2009) and have less elaborate water control structures than permanent roads. Forest roads and stream crossings can add significantly to operational costs (Conrad et al., 2012). For example, temporary skidder bridges currently cost as much as \$8000–\$16,000 in 2010 (McKee et al., 2012).

Swift and Burns (1999) suggested that stream crossings should be "restored to a stable, non-eroding 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, and 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 itself (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 et al., 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; Boston, 2012). This decision may eventually require a National Pollutant Discharge Elimination System (NPDES) permit from the EPA to construct a forest road (Schilling et al., 2007; Boston, 2012). 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 and is scheduled to go before the U.S. Supreme Court in the later part of 2012 (Boston, 2012). However, this recent litigation emphasizes the need for quantification of the effectiveness of specific BMPs on forest roads and skid trails.

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) suggested 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 is needed to assist these managers in decision making. The primary objective of this research was to evaluate three levels of skid trail stream crossing closure BMPs (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 were:

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 predicted 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). Site were located between 36°32′39″N to 38°02′32″N latitude and 78°14′02″W to 79°49′44″W longitude. All sites had temporary stream crossings and used portable metal bridges for skidding with rubber-tired grapple skidders. All sites were located on forest industry (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.

One harvest had four stream crossings (crossing numbers 2–5) that were located on the same stream (Table 1). Another harvest operation had two crossings (crossing numbers 8 and 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 area represents the amount of timber harvested and skidded across the stream towards the landing area. 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 served a minimum of 24 skidding cycles. Maximum stream crossing approach slopes ranged from 6% to 18%. Detailed physical characteristics of each stream are presented in Table 2. Manning–Chezy values were also 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.

Stands were clearcut harvested using rubber-tired feller bunchers and grapple skidders. Average total harvest area was 65 ha. Stream crossing locations were identified prior to harvesting by a professional forester in order to minimize the number of crossings needed. Stream crossing structures were steel paneled bridges varying from 7.3 to 9.7 m in length.

Three 1-m-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 minimum size 15-m streamside management zone (SMZ) was intended for each side of the streams, but actual SMZs ranged from 13 to 45 m.

Study sites had mean annual precipitation values ranging from 1070 to 1140 mm year^{-1}, and a mean air temperature between 18 and 22 °C during the growing season and between 5 and 8 °C during the dormant season (USDA NRCS, 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 ha above the crossing points. Each of the five sites also had similar soil types, being hapudults and ultic hapludalfs (USDA NRCS, 2011). As is typical for the Piedmont region, 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, many 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 (Trimble, 1974).

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

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.

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

2.2. Study design

After harvesting, skidder bridges were removed and three BMP closure treatments were randomly applied to the nine stream crossings (18 crossing 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; i.e., BMP treatments were the same on each side of the stream. Virginia's forestry BMPs recommend waterbars and seed as a minimal level of closure, thus we did not include a "worst case" level of treatments. The stream crossing closure treatments are provided below and displayed in Fig. 1.

(1) Slash – A rubber-tired grapple skidder removed logging slash (tree limbs and tops) from decking areas 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 (Fig. 1a).
(2) Mulch – Grass seed (fescue), 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 straw mulch, equating to 20 bales per crossing (Fig. 1b).
(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 burial of the silt 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. As in the Mulch treatment, each approach was covered with 10 bales of mulch, equating to 20 bales per crossing (Fig. 1c).

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., Lincoln, NE) or Sigma 900MAX (Hach Company, Loveland, CO), were installed. One automated sampler was positioned approximately 10 m upstream and the second was positioned 10 m downstream from the crossing in a similar fashion to the stream crossing evaluations by Taylor et al. (1999) (Fig. 2).

The water samplers were installed after harvesting (for equipment safety and logistical reasons), but before the BMP 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 stream beds with landscaping staples. The streams ranged from 5 to 20 cm in depth during base flow conditions. All automated water samplers were programmed to collect one 500 mL sample per day at 10:00 am.

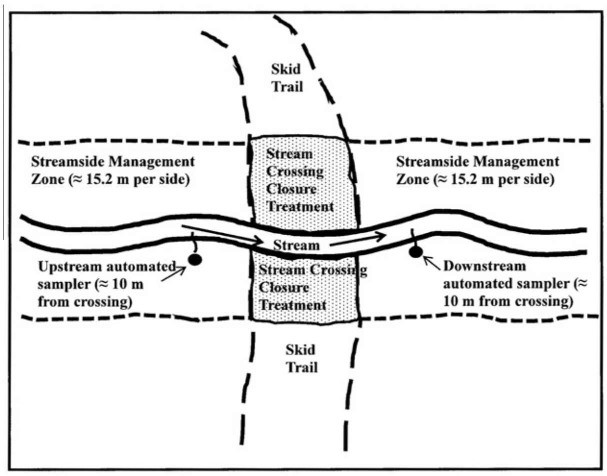

Fig. 2. Idealized diagram of study sites. Not to scale. Water samplers were placed approximately 10 m upstream and 10 m downstream from the stream crossing. The stream crossing closure treatments were placed on the skid trail within the SMZ.

After the sample was pumped to the housing and dispensed into it is designated bottle, the tubing was purged of water. Each sampler held 24 water samples, thus the retrieval of 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 (TSSs) 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.

Although 18 water samplers were available, only three stream discharge monitors were available. Thus, limited direct measures of stream discharge data were available. Therefore, stream surveys were conducted at each water sampler location for stream cross-sectional area, width, wetted perimeter, and stream slope for full bank conditions (Harrelson et al., 1994). Manning roughness coefficients were selected based on stream bedload, sinuosity, and vegetation and stream channel characteristics were applied to the Manning equation so that relative estimates of full bank water yields, similar to the methods used by Riedel et al. (2005).

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 using 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

Fig. 1. Representative stream crossing approaches that were closed with slash treatments (a), mulch treatments (b), and mulch + silt fence treatments (c).

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 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. (2012b). These USLE data were used for comparison with other similar studies which used the same treatments to close out skid trails.

The physical features of the approaches used as inputs to the USLE were also used for correlation with in-stream TSS loading values. Sediment 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 due to 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 tonnes $year^{-1}$.

Treatment costs were recorded and 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. Costs were reported as averages 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 daily rainfall data into quartiles above zero, and then combining the lowest category with the days with no rain. The categories were as follows: low = 0.00–1.0 mm; medium = 1.1–4.0 mm; high = 4.1–10.00 mm; and maximum > 10.0 mm. A daily TSS percent change value was calculated for analysis using the following equation

$$\text{Daily TSS percent change} = [(\text{Downstream TSS} - \text{Upstream TSS})/\text{Upstream TSS}] \times 100$$

Data were analyzed for statistical significance using JMP Statistical Discovery Software (JMP Version 9., 2010). 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 as conducted by Clinton and Vose (2003). The physical features of the stream crossing approaches were also measured and analyzed for significance with a Pearson's correlation matrix.

Limited (approximately 1 month) 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 1.0 mm (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)

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

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.

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 show the treatment differences between each paired treatment at each rainfall category (Table 4). The rainfall categories that 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. The letters were inserted into the score mean column of the Kruskal–Wallis test after evaluating the specific treatment comparisons from the Wilcoxon test. The low rainfall category showed that slash was significantly different than the mulch and mulch + silt fence treatments, with regard to sediment levels in the stream. The lower score mean value indicated that the slash performed better than the other two treatments with regard 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 Fig. 3. 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 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. These finding might be due to lab analysis variation in stream TSS, however, on site observations indicated that slash had been unintentionally deposited in the stream and this slash may have actually resulted in sediment trapping. 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

Table 3
Results of the 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 0.0–1.0 mm	14.9433	0.0006*	Slash	245	193.27 a
			Mulch	96	231.95 b
			Mulch + silt fence	83	246.77 b
Medium 1.11–4.0 mm	9.0407	0.0109*	Slash	27	24.14 a
			Mulch	16	26.25 a
			Mulch + silt fence	13	40.30 b
High 4.1–10.0 mm	11.7111	0.0029*	Slash	37	38.00 a
			Mulch	31	43.90 a
			Mulch + silt fence	23	61.69 b
Maximum > 10 mm	4.2202	0.1212	Slash	43	42.25 a
			Mulch	24	40.95 a
			Mulch + silt fence	22	54.77 a

Table 4
Results of the 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 0.00–1.0 mm	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 1.1–4.0 mm	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 4.1–10.0 mm	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 > 10.0 mm	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

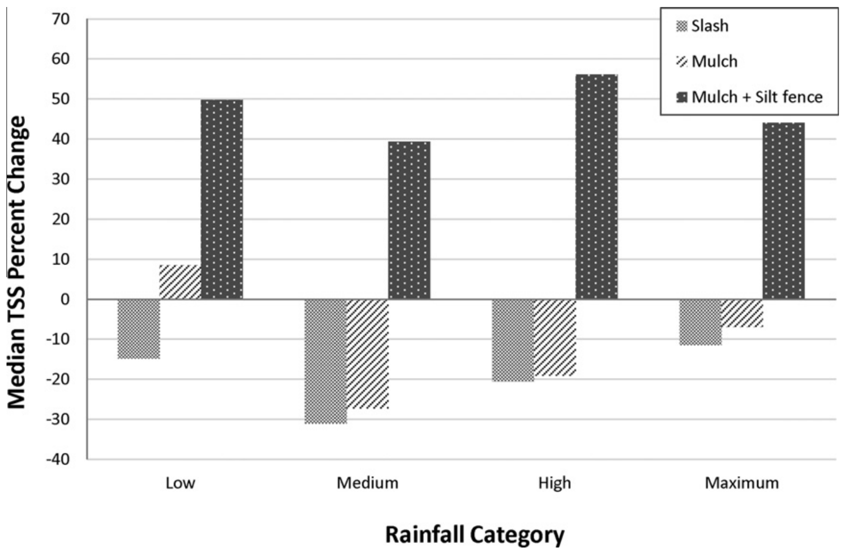

Fig. 3. Median TSS percent change values at each treatment and rainfall category. Negative values indicate that sediment was trapped at the stream crossing, and positive values indicate that new sediment entered the stream at the stream crossing. Rainfall categories are based on total daily precipitation: low = 0.0–0.1.0 mm, medium = 1.1–4.0 mm, high = 4.1–10.0 mm, maximum > 10.0 mm.

not allow new sediment to enter the stream at the stream crossing, while the mulch + silt fence treatment did.

The mulch + silt fence treatment resulted in the greatest downstream sediment increase. Silt fence installation is a proven BMP for reducing silt-sized and larger sediment (Robichaud and Brown, 2002), but it is 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.

3.2. BMP efficiency

BMP efficiency results are displayed in Table 5. The treatments were not significantly different with regard to TSS mean percent difference when analyzed using *t*-tests at an alpha level of 0.05. However, the calculation of BMP efficiency is beneficial for 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, 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 installation.

3.3. USLE estimates and physical features

USLE erosion estimates for the stream crossing approaches ranged from 0.011 to 38.304 tonnes ha^{-1} year^{-1} (Table 6). 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 coincidentally on longer approach lengths, thus generating much larger USLE estimates than the others. The generally low USLE erosion predictions indicate that all of the closure treatments provided substantial cover to the stream crossing approaches (Table 6).

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 (tonnes 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 regard 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 7). 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 slope percent. 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

Table 5
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. $\alpha = 0.05$.

Rainfall category	Treatment	TSS mean percent difference	Mean standard error	BMP efficiency (%)
Medium (1.1–4.0 mm)	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 (4.1–10.0 mm)	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 (>10.0 mm)	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

Table 6
USLE estimates taken at each stream crossing approach after closure treatments were applied.

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

Table 7
Pearson's correlation matrix of the physical features of the stream crossing approaches and TSS loading values.

	Area of approach (ha)	Slope length (m)	Slope percent	Percent bare soil	TSS loading (tonnes year^{-1})
Area of approach (ha)	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 (tonnes year^{-1})	**0.3980**	**0.5871**	−0.1966	−0.2303	1.0000

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. (1999a). Reducing total approach area will also reduce the amount of sediment that 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., 1999b). 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 their 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 that could otherwise enter the stream.

Erosion estimates were compared with recent studies that used similar treatments to close out skid trails (Table 8). The two studies used for comparison (Sawyers et al., 2012; Wade et al., 2012b) assessed several methods of skid trail closure, as well as the accuracy of USLE predictions with actual measured erosion using geotextile sediment traps. Wade et al. (2012b) found that the USLE was an adequate predictor of actual erosion. Sawyers et al. (2012) found that the USLE was an acceptable predictor with regard 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 USLE data from this study were estimated.

Both Sawyers et al. (2012) and Wade et al. (2012b) 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 longer 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 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 BMP treatment were reported by the logging contractors responsible for installing the treatments and are presented in Table 9. 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 h of operator and machine time for slash application. This cost would further be reduced if slash was spread on stream crossing approaches during normal logging

Table 8
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) (tonnes ha^{-1} year^{-1})	Wade et al. (2012b) – Bladed skid trails (measured erosion) (tonnes ha^{-1} year^{-1})	Sawyers et al. (2012) – Overland skid trails (measured erosion) (tonnes 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%)

Table 9
Treatment costs per stream crossing as reported by the logging contractors.

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

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 waterbars 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 of insufficient replication of the pretreatment data. The BMP efficiencies ranged from −497% to +99%, which is almost four orders of magnitude between sediment levels. It should be noted that these efficiencies are based on sediment concentrations as opposed to sediment loading values.

Hypothesis 3 stated that USLE estimates will not result in different levels of erosion among the three stream crossing approach closure treatments. Our results indicate that the USLE estimates did not result in different erosion estimates among 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 average 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. Conclusions

The potential impacts of forest roads and skid trails and associated BMPs on stream water quality and stream health have be recognized in North America (MacDonald et al., 2003; Shepard, 2006; McGinley et al., 2012), Europe (Nisbet, 2001), South America (Keller and Berry, 1989; Frederickson and Frederickson, 2004; Putz et al., 2008; McGinley et al., 2012), Africa (Horswell and Quinn, 2003) Oceania (Croke and Mockler, 2001; Cornish, 2001), and Asia (Keller and Berry, 1989; Croke and Hairsine, 2006; Chang et al., 2008; Putz et al., 2008). Practically all forestry best management practice recommendations recognize that stream crossing portions of skid trails are where sediment delivery has the greatest potential to occur. However, as pointed out by Anderson and Lockaby (2011) fewer studies have specifically addressed BMP efficacy for closing stream crossings. 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 regard to land use.

Slash and mulch treatments have proven effective on both overland (Sawyers et al., 2012) and bladed skid trails (Wade et al., 2012b). 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 was 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 their installation is greater than the 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 of harvesting timber (Bolding et al., 2010). 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 limbs and tops, 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 it is 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 that 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 occur after harvest.

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