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The effect of increasing gravel cover on forest roads for reduced sediment delivery to stream crossings

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Abstract:

Direct sediment inputs from forest roads at stream crossings are a major concern for water quality and aquatic habitat. Legacy road–stream crossing approaches, or the section of road leading to the stream, may have poor water and grade control upon reopening, thus increasing the potential for negative impacts to water quality. Rainfall simulation experiments were conducted on the entire running surface area associated with six reopened stream crossing approaches in the south-western Virginia Piedmont physiographic region, USA. Event-based surface run-off and associated total suspended solid (TSS) concentrations were compared among a succession of gravel surfacing treatments that represented increasing intensities of best management practice (BMP) implementation. The three treatments were no gravel (10–19% cover), low gravel (34–60% cover), and high gravel (50–99% cover). Increased field hydraulic conductivity was associated with maximized surface cover and ranged from 7.2 to 41.6, 11.9 to 46.3, and 16.0 to 58.6 mm h^{-1} respectively for the no gravel, low gravel, and high gravel treatments. Median TSS concentration of surface run-off for the no gravel treatment (2.84 g l^{-1}) was greater than low gravel (1.10 g l^{-1}) and high gravel (0.82 g l^{-1}) by factors of 2.6 and 3.5 respectively. Stream crossing approaches with 90–99% surface cover had TSS concentrations below 1 g l^{-1}. Reducing the length of road segments that drain directly to the stream can reduce the costs associated with gravel surfacing. This research demonstrates that judicious and low-cost BMPs can ameliorate poor water control and soil erosion associated with reopening legacy roads. Copyright © 2014 John Wiley & Sons, Ltd.

KEY WORDS forest roads; stream crossing approaches; rainfall simulation; sediment delivery; BMP cost-effectiveness; non-point-source pollution

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INTRODUCTION

Forest roads are the dominant sediment sources associated with forested land use (Croke *et al*., 1999a). The potential for sediment delivery to waterways is greatest where the road meets the riparian area because forest road–stream crossings penetrate riparian vegetation and represent relatively unimpeded paths for overland flow to the stream channel. The direct nature of sediment delivery to streams at road–stream crossings has sparked a series of recent legislative debates in the USA about Federal Clean Water Act permits for forest roads [i.e. *NEDC vs Brown* (Boston, 2012)], as well as their non-point-source pollution status [i.e. *NEDC vs Decker* (U.S. Court of Appeals for the Ninth Circuit, 2013)].

Major sediment inputs to streams occur during stream crossing installation (Lane and Sheridan, 2002; Wang *et al*., 2013) and removal (Foltz *et al*., 2008b), but stream crossing approaches can also represent significant sediment sources (Lane and Sheridan, 2002; Aust *et al*., 2011; Brown *et al*., 2013). The potential for road–stream crossings to negatively impact water quality relates to crossing type, traffic type, vehicle weight, traffic frequency, surface cover, road approach length and slope, and the effectiveness of water-control structures (Reid and Dunne, 1984; Sheridan and Noske, 2007). In addition, reopening abandoned 'legacy' roads that were originally constructed prior to the best management practice (BMP) era in the USA (circa 1977) raises concerns about their potential impacts to water quality. Common water quality problem areas associated with

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legacy roads include poor road location, as well as inadequate water and grade control. The effects of forest roads on water quality and aquatic habitat may be more pronounced for small streams ($<10\,\mathrm{l\,s^{-1}}$), while the impacts of road-derived sediment may be diluted on larger streams ($>500\,\mathrm{l\,s^{-1}}$; Sheridan and Noske, 2007).

Few studies have quantified the efficacy of BMPs to reduce soil erosion and sediment delivery, as well as BMP implementation costs (Brown *et al.*, 1993; Sawyers *et al.*, 2012; Wear *et al.*, 2013), but both are critical to justify BMP implementation. In particular, how effectively sediment is reduced by different intensities of BMP implementation or how effective BMPs are under the influence of individual rain events with different characteristics is not known. Rainfall simulation experiments at stream crossing approaches can be used to provide insights about both of these factors.

This study focuses on three research questions in the Piedmont physiographic region of south-western VA, USA: (1) How does gravel cover affect road surface run-off?, (2) what are the total suspended solid (TSS) concentrations in road surface run-off during rainfall events for reopened stream crossing approaches?, and (3) what sediment reduction can be achieved using different graveling intensities on stream crossing approaches and what is the cost of reducing sediment by graveling? Rainfall simulation experiments were conducted on the entire running surface area associated with six reopened stream crossing approaches. Event-based road surface run-off and associated TSS concentrations were compared among a succession of treatments that represented increasing intensities of BMP implementation. The three treatments were (1) no gravel: existing conditions following reopening by bulldozer blading (10–19% cover), (2) low gravel: gravel application beginning at the stream and continuing uphill for 9.8 m (34–60% cover), and (3) high gravel: gravel application that doubled the length of the low gravel treatment (50–99% cover). The cost-effectiveness of the gravel treatments was evaluated by relating the costs associated with BMP implementation to reductions in sediment yield.

MATERIALS AND METHODS

Study area

Six road approaches associated with three unimproved ford stream crossings on a legacy forest road at the Reynolds Homestead Forest Resources Research Center (RHFRRC), located in Critz, VA (Patrick County), USA, were selected for study of event-based surface run-off and associated TSS concentrations (Figure 1). Topography is characterized by rolling hills, with side slopes generally ranging from 8 to 25%, and a mean elevation of approximately 335 m above mean sea level (NRCS, 2011). As is typical of the Piedmont region, old agricultural gullies are common because most of the contemporary forested watersheds were in agriculture during the 1800s (Trimble, 1974). Mean annual rainfall is 1250 mm, with a mean snow contribution of 270 mm to the total precipitation. Mean air temperature ranges from a low of −1.8 °C in January to a high of 29.7 °C in July (Sawyers *et al.*, 2012). The predominant soil series is Fairview sandy clay loam (fine, kaolinitic, mesic typic Kanhapludults). Soil parent material is residuum from mica schist and mica gneiss. The severe erosion hazard rating for forest roads and trails at RHFRRC (NRCS, 2011) underscores the importance of controlling road grade, water, and surface cover to reduce erosion and sediment delivery.

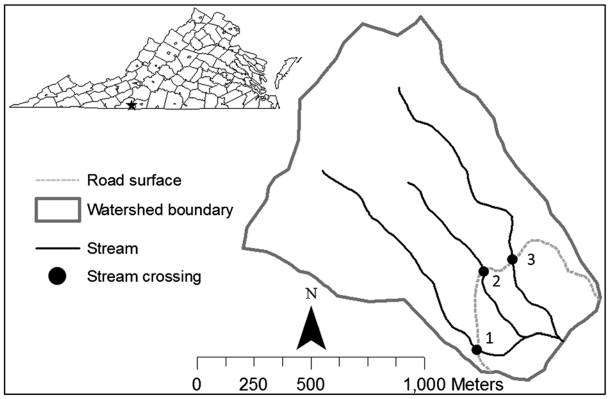

Figure 1. Location map of the Reynolds Homestead Forest Resources Research Center in Critz, VA (Patrick County), USA, and a schematic showing the road location within the second-order watershed that contains the three unimproved ford stream crossings. This 135-ha watershed drains to the south-east from No Business Mountain and is part of the western headwaters of Mill Creek

Field methods

Prior to road reopening, a Sokkia model SET-520 total station was used to measure the length of the stream crossing approach study plots, as well as approach slope and mean running surface width (Table I). Length was defined as the distance between the nearest water-control structure (i.e. water bar/turnout) and the stream. In late July 2011, six stream crossing approaches were reopened by bulldozer blading, creating initial conditions of approximately 100% bare soil on the approach running surfaces. The road profiles of each approach study plot were cut-and-fill. Bulldozer blading created a slight cross slope in the running surface, which directed surface run-off and suspended sediment towards the cut slope bank *en route* to the bottom of the plot, which ended just before the stream (Figure 2). Outboard-edge berm installation was also used to prevent surface run-off from travelling away from the running surface and down the fill slope bank. Open-top box culverts

Table I. Physical characteristics of the running surface component of the road–stream crossing approach study sites at the Reynolds Homestead Forest Resources Research Center, Critz, VA (Patrick County), USA

Site ID	Approach length (m)	Mean width (m)	Surface area (m^2)	Mean slope (%)	Bulk density ($g\,cm^{-3}$)	Soil texture*	Road vertical curvature	Running surface profile
1	25.8	3.1	79.4	9.4	1.61	SCL	S-shaped	Insloped
2	41.3	2.5	102.6	4.7	1.55	SCL	Linear	Insloped
3	19.2	3.2	61.3	13.7	1.51	SCL	Convex	Insloped
4	35.3	2.5	86.8	15.4	1.46	SCL	S-shaped	Insloped
5	39.6	2.9	115.7	16.1	1.37	SCL	S-shaped	Insloped
6	23.5	2.6	61.3	6.4	1.47	CL	Linear	Insloped

*SCL and CL denote sandy clay loam and clay loam soil textures respectively.

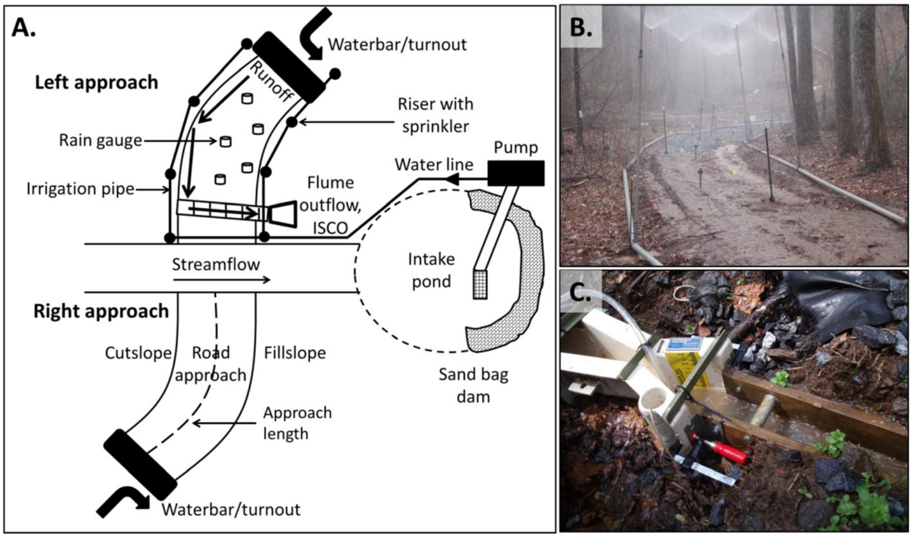

Figure 2. Plan view of two idealized stream crossing approaches with rainfall simulator equipment and monitoring instrumentation. Open-top box culverts were located at the bottom of the plot, immediately upslope of the stream (A). Photographs depicting a rainfall simulation experiment on 12 March 2012 (B) and the equipment used to measure surface run-off quantity and quality (C)

(Trimble and Sartz, 1957) were installed at the lower plot boundary to redirect storm water run-off away from the stream and towards a flume, where run-off volumes were measured. In October 2011, Kadak (2012, unpublished data) used double-ring infiltrometers to estimate infiltration rates of the reopened stream crossing approaches. Infiltration rates ranged from 0.6 to 7.2 mm h^{-1}. Bulk density samples (n = 4 to 7 per site) were obtained from the running surface via the soil extraction method (SSSA, 1986).

Rainfall simulation experiments were conducted for a succession of gravel surfacing treatments that represented increasing intensities of BMP implementation on the reopened stream crossing approaches. Rainfall simulation experiments began in February 2012. The unsurfaced approaches were trafficked with a bulldozer immediately before the first series of experiments to mimic newly disturbed conditions associated with road reopening. Following this treatment (‘no gravel’), the stream crossing approaches had 10–19% surface cover, which consisted of residual leaf litter and other debris. Following the no gravel treatment rainfall simulations, a dump truck was used to tailgate spread a mixture of size 3, 5, and 7 (ranging from 5.1 to 1.9-cm diameter) granite gravel beginning at the lower plot boundary and continuing uphill for a distance of 9.8 m (‘low gravel’ treatment). Mean gravel depth was approximately 0.08 m, and the width of gravel application extended across the width of the road between the outer edges of the running surfaces, which averaged 2.8 m. Gravel was not washed prior to application to the stream crossing approaches as is typical during forest road construction and graveling. The near-stream 9.8-m gravel section was chosen for the low gravel treatment because this length approximated half the distance of the shortest approach used in this study. The low gravel treatment resulted in different proportions of cover on the running surface area of the study plots primarily because approach lengths were all different, ranging from 19.2 to 41.3 m (Table I). For example, the low gravel treatment resulted in 60% surface cover for the shortest approach and 40% surface cover for the longest approach used in this study. Following the low gravel rainfall simulations, no additional gravel was applied to the initial 9.8 m long segment, but gravel was applied to the adjacent (uphill) 9.8-m section of the road approach ('high gravel'). This treatment effectively doubled the length of the first gravel application and resulted in an overall range of 50 to 99% surface cover on the approach running surfaces. The succession of treatments at each site (no gravel, low gravel, and high gravel) represented increasing intensities of BMP implementation and facilitated the evaluation of a wide range of surface cover on the stream crossing approaches (10–99%) in reducing sediment delivery to streams during simulated rainfall events.

Calculation of gravel cost

The volume of gravel (m^3) applied to each stream crossing approach was calculated as mean gravel depth (0.08 m) × mean running surface width (range = 2.5 to 3.2 m) × length of the gravelled section (i.e. initially 9.8 m for the low gravel treatment and then 19.6 m for the high gravel treatment). Gravel volume was converted to a mass (Mg) by assuming a gravel bulk density of 1.6 Mg m^{-3} (O'Neal *et al.*, 2006). Local gravel cost for Critz, VA, was \$27.56 Mg^{-1}. Mean gravel costs (i.e. the cost of the rock only) for the low gravel and high gravel treatments were \$91.77 and \$183.54 respectively. Mean gravel cost per metre of approach length was \$9.41 for the low gravel and high gravel treatments. Mean gravel cost per unit area on the running surface was \$3.34 m^{-2}. Calculation of gravel costs allowed for an evaluation of the cost-effectiveness of the treatments to reduce sediment delivery to streams during simulated rain events.

Rainfall simulation

A high-pressure pump with a 1.34×10^4-W (18-hp) gas-driven engine and a maximum flow rate of 37.9 l s^{-1} (600 gal min^{-1}) was used to pump water for the rainfall simulation experiments from temporary impoundments located approximately 50 m downstream of each stream crossing. Irrigation equipment included a 10.2-cm diameter intake hose and strainer, a 7.6-cm diameter outflow component, and fire hose to connect to 7.6-cm diameter aluminium irrigation pipelines that ran upslope along both sides of the stream crossing approach (Figure 2). A 3.0 m long irrigation pipe, and a pair of 90° angle couplings, joined the two parallel segments of pipe downslope of the lower plot boundary. The pipelines adjacent to the study plots consisted of 6.1 m long irrigation pipes that were coupled together for the entire length of the approach (Table I). A water-control structure (water bar/turnout) and the open-top box culvert served as the upper and lower plot boundaries respectively. Risers with sprinklers (Rain Jet 78C, Rain Jet Corp.) at a height of 3.4 m to attain near-terminal velocity of individual water drops (Ward, 1986) were located at 6.4-m intervals along the entire length of the study plots. This irrigation set-up was used previously in rainfall simulation experiments to test the effectiveness of agricultural BMPs in reducing soil erosion during rain events (Dillaha *et al.*, 1988) and has a designed rainfall application rate of 50.8 mm h^{-1}. At 50.8 mm h^{-1}, the Rain Jet 78C nozzle provides about 40% of the kinetic energy of natural rainfall (Renard, 1989).

A series of three applied rainfall events, ranging in duration from 10 to 50 min (median = 20 min), was performed for each treatment (no gravel, low gravel, and high gravel) at each of the six road approaches for a total of $3 \times 3 \times 6 = 54$ experiments (Table II). When stream discharge permitted sufficient water storage in the impoundments, rainfall was applied to mimic short, medium, and long events (usually 10, 20, and >30 min respectively) for a given site and treatment combination (e.g. Site 1: No Gravel 1, No Gravel 2, and No Gravel 3). When water availability was limited (i.e. dry periods in the summer months), rainfall was applied until the impoundments were pumped dry. Applied rainfall events for a given site and treatment combination were typically completed on the same day, with approximately 30 min between each experiment. Rainfall simulation experiments

Table II. Running surface cover on the stream crossing approaches and applied rainfall event characteristics for each site and treatment combination

Site ID	Surface cover (%)			Mean rainfall duration (min)			Mean rainfall intensity (mm h^{-1})		
	No gravel	Low gravel	High gravel	No gravel	Low gravel	High gravel	No gravel	Low gravel	High gravel
1	10	35	90	21	20	35	51	40	48
2	15	40	50	24	30	24	63	24	30
3	17	60	90	19	24	22	64	58	52
4	10	34	50	16	14	14	56	64	64
5	14	47	63	23	15	13	60	48	46
6	19	60	99	33	23	18	56	72	60

began within 1 to 2 days following treatment application at each site (i.e. first bulldozer trafficking, then low gravel application, then high gravel application). The time in between treatment applications at each site ranged from 2 to 4 weeks. Median intensity of applied rainfall was 49.8 mm h^{-1} and ranged from 20.4 to 73.0 mm h^{-1} (Table II). Event rainfall totals ranged from 6.4 to 41.6 mm, with a median of 16.6 mm. Simulated rainfall events had recurrence intervals of <1 to 5 years for Critz, VA, USA, (NWS, 2011).

Sediment collection

Rainfall amount and intensity were measured with six wedge collection-type rain gauges and five automatic tipping-bucket rain gauges (ECRN-50 Low-Resolution Rain Gauge, Decagon Devices, Inc.) that were set to log data at 1-min intervals. Surface run-off discharge was measured with a 2.5 × 45.7-cm cut-throat flume (Tracom Fiberglass Products) that was fitted to the outflow end of the open-top box culverts, located at the bottom of the plot (Figure 2). The flume stilling well was equipped with a pressure transducer (HOBO U20 Water Level Data Logger, Onset Computer Corporation) to measure water level at 1-min intervals. Manual water level observations were also made using a staff gage that was mounted to the wall of the flume at the inflow end. Water level was converted to discharge (l s^{-1}) using the stage–discharge equation for the specific dimensions of the cut-throat flume. An ISCO automatic storm water sampler (ISCO 3700 Series, Teledyne ISCO) was programmed to collect 500-ml storm water run-off samples at 2 to 5-min intervals. Storm water run-off samples were analysed in the laboratory for TSS concentration by way of vacuum filtration of a known amount of sample volume and mass of oven-dried sediment trapped by the filters (Eaton *et al.*, 2005). During rainfall simulation, cheesecloth was stretched over the intake strainer to filter suspended solids from the source water to limit carry-over; however, applied rainfall samples were collected during each experiment for laboratory analysis of TSS concentration and this was subtracted from the TSS concentration of road surface run-off. Event-based suspended-sediment loads were calculated by multiplying TSS concentration by surface run-off volume in 1-min intervals and then summing the 1-min sediment loads for the entire event duration.

Event hydrographs were used to identify the time intervals corresponding to nearly constant run-off conditions (i.e. soil was partially saturated), and during these times, field hydraulic conductivity of the road surface, K_s, was estimated as rainfall rate minus run-off rate (Reid and Dunne, 1984; Figure 3). The median K_s value was used to approximate road surface K_s. Event sediment yield was quantified by dividing event TSS load by the running surface area of the stream crossing approach, which accounted for differences in approach length [i.e. the approach running surface widths did not differ substantially (Table I)]. Also, to account for variability in applied rainfall amounts and intensities (Table II), event sediment yield was standardized per unit rainfall (mg m^{-2} mm^{-1}) according to the methods of Sheridan and Noske (2007) and Sheridan *et al.* (2008):

$$TSS_{\text{yield (standardized)}} = TSS_{\text{yield}} / rain_{\text{total}} \quad (1)$$

where TSS_{yield} is the event TSS transport per unit area (mg m^{-2}) and $rain_{\text{total}}$ is the total event rainfall (mm).

Statistical analysis

The dependent variables included TSS concentration of road surface run-off, sediment yield per unit rainfall, and K_s. TSS concentration and sediment yield per unit rainfall were natural-log transformed. Each dependent variable was analysed with a full-factorial design using JMP 10

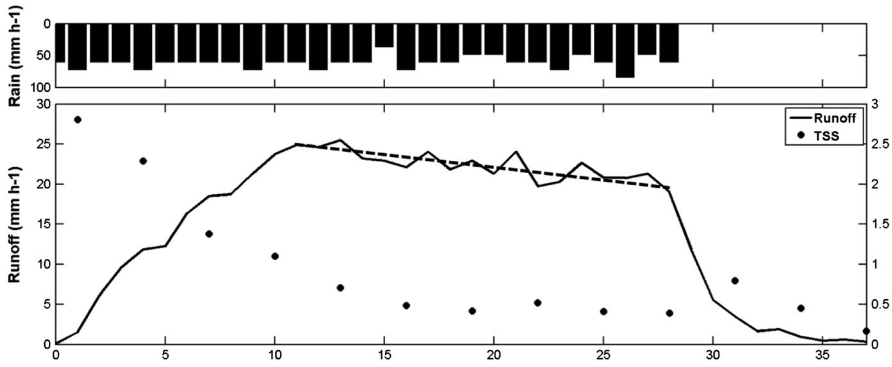

Figure 3. Event hydrograph for Site 3, depicting the first of three rainfall simulation experiments for the high gravel treatment. Field hydraulic conductivity, K_s (mm hr^{-1}), was estimated as the median value of rainfall rate minus run-off rate during conditions approximating steady-state run-off (i.e. from 11 to 28 min in Figure 3, as indicated by the dashed line). These time periods represented partially saturated soil conditions. Notice that run-off decreased because of decreasing rainfall intensity from the middle to the end of the experiment

software (SAS Institute Inc., 2012), which included all possible interactions among the following fixed effects: treatment (no gravel, low gravel, and high gravel), rainfall simulation group (1, 2, and 3), and slope group (<10% slope and >10% slope), as well as the random effect of site ID (road approaches 1–6) nested within the slope group. Treatment was included to account for the effect of gravel surface cover on road surface run-off and suspended-sediment delivery. Rainfall simulation group was included to account for decreases in sediment as a result of the chronological order in which experiments were completed at each stream crossing approach (i.e. three for the no gravel treatment, followed by three for low gravel, and lastly three for high gravel). The rainfall simulation group 1 included the first in a series of three rainfall simulations for each treatment, while the rainfall simulation groups 2 and 3 included the second and third rainfall simulations within a given treatment respectively. The six stream crossing approaches were categorized by slope, with Sites 1, 2, and 6 (slope = 9.4, 4.7, and 6.4% respectively) in the <10% slope group and Sites 3, 4, and 5 (slope = 13.7, 15.4, and 16.1% respectively) in the >10% slope group. Virginia's technical manual for forestry BMPs suggests that road grades be kept between 2 and 10%, whenever possible (VDOF, 2011).

The site ID variable takes into account that observations within one study plot will be correlated with each other as opposed to observations from different study plots. It is used as a random effect because the study plots represent a subset of all possible forest road–stream crossing approaches. Model fitting involved the retention in the final model of all significant fixed effects (alpha = 0.05), as well as the random effect of site ID nested within the slope group (denoted as 'site ID [slope group]'). Separation of least-squares means was performed with Tukey's honestly significant difference in JMP 10 software (SAS Institute Inc., 2012). Model residuals were analysed for normality and homogeneity of variance with quantile–quantile plots in JMP 10.

Table III. Overall ranges of running surface cover on the stream crossing approaches and median TSS concentrations of road surface run-off by treatment

Treatment	Surface cover (%)	Grand median TSS concentration ($g\,l^{-1}$)	Range of median TSS concentration per event ($g\,l^{-1}$)
No gravel	10–19	2.84	0.57–7.42
Low gravel	34–60	1.10	0.39–3.16
High gravel	50–99	0.82	0.15–1.24

$N = 18$ rainfall simulation experiments per treatment.

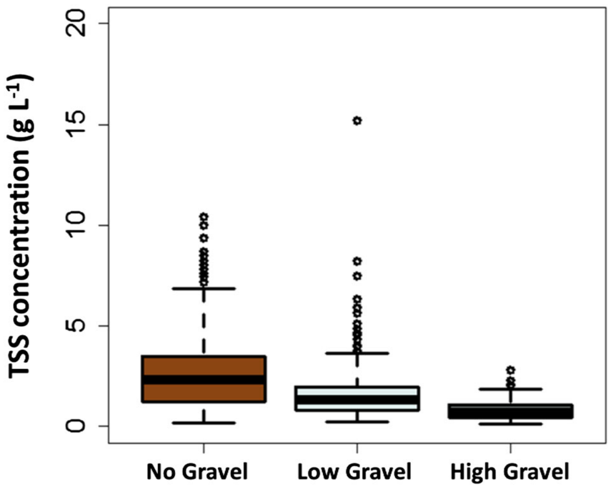

Figure 4. Box and whisker plots showing the 5th, 25th, 50th, 75th, and 95th percentiles of all samples of TSS concentrations of road surface run-off ($g\,l^{-1}$), aggregated by treatment type (no gravel, low gravel, and high gravel). $N = 228$, 222, and 231 for the no gravel, low gravel, and high gravel treatments respectively

RESULTS AND DISCUSSION

TSS concentration

Overall, gravel application reduced TSS concentrations of road surface run-off (Table III). TSS concentration for the no gravel treatment experiments was 2.6 and 3.5 times greater than that of the low gravel and high gravel treatments respectively (Figure 4). The effect of treatment on TSS concentration was significant ($p < 0.0001$; Table IV), and separation of the least-squares means indicated that all groups were significantly different (no gravel > low gravel > high gravel).

TSS concentration was greatest during the first rainfall simulation experiment within the no gravel treatment, while subsequent no gravel treatment experiments were similar (Figure 5). This finding indicates that the supply of loose sediment was approaching depletion after the first no gravel treatment rainfall simulation experiment. The treatment effect of gravel application (i.e. increased road surface cover) is evidenced by further declines in TSS concentrations of road surface run-off with successive gravel treatments (Figure 5). Renewed sediment sources associated with the application of the gravel treatments included truck traffic on the stream crossing approaches and dust associated with the unwashed gravel. Consequently, TSS concentrations in road surface run-off were also greatest for the first rainfall simulation experiments within the low gravel and high gravel treatments, while subsequent experiments within each treatment were similar (Figure 5). The effect of simulation group on TSS concentration was significant ($p < 0.0001$; Table IV), and separation of the least-squares means indicated that simulation group 1 > simulation groups 2 and 3 (2 and 3 are not different). Least-squares mean separation for site ID

Table IV. Fixed effect tests for the analysis of TSS concentration of road surface run-off, hydraulic conductivity (K_s), and sediment yield per unit rainfall

Dependent variable	Fixed effects	DF	DF denominator	F ratio	Probability > F	Model adjusted R-squared
TSS	Treatment	2	44.0	57.2	<0.0001	0.80
	Simulation group	2	44.0	14.0	<0.0001	
K_s	Slope group	1	3.7	30.0	0.01	0.34
	Treatment	2	45.9	4.5	0.02	
Sediment yield	Treatment	2	48.1	33.0	<0.0001	0.59
	Simulation group	2	48.1	3.9	0.03	

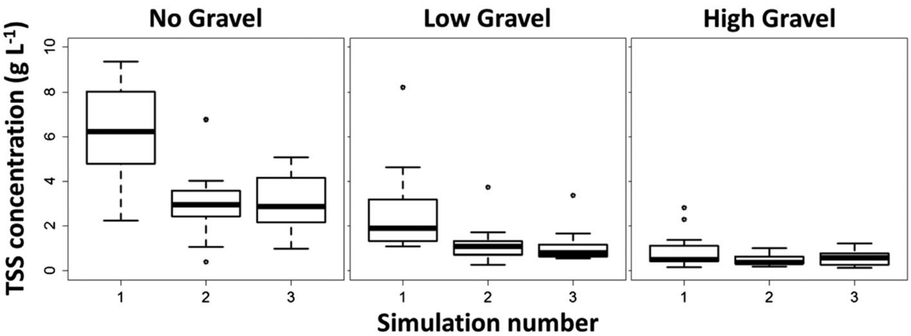

Figure 5. Box and whisker plots showing the 5th, 25th, 50th, 75th, and 95th percentiles of TSS concentrations of road surface run-off ($g\,l^{-1}$) for successive rainfall simulation experiments within a given treatment. Data are shown for Site 3, where the low gravel and high gravel treatments resulted in 60 and 90% cover respectively on the road approach running surface

nested within the slope group indicated that, in general, higher sediment concentrations were associated with steeper slopes. For example, the ranking of sites from greatest to least in terms of TSS concentration was as follows: Site 4 (15.4% slope), Site 1 (9.4% slope), Site 5 (16.1% slope), Site 3 (13.7% slope), Site 2 (4.7% slope), and Site 6 (6.4% slope). Overall, TSS concentration for Site 4 was greater than that of Sites 2 and 6 but not significantly different than Sites 1, 3, and 5.

Suspended-sediment concentrations of road surface run-off for unsurfaced roads that have an abundant supply of loose sediment can reach $100\,g\,l^{-1}$ at the onset of high-intensity rainfall (Ziegler *et al.*, 2000a). In our study, maximum TSS concentrations were commonly greater than $8\,g\,l^{-1}$ and were observed at the onset of applied rainfall for the experiments that were conducted immediately following bulldozer trafficking over the stream crossing approaches (i.e. the first no gravel treatment rainfall simulation in a series of three). Such high concentrations of TSS in road surface run-off at the road–stream interface pose important issues for water quality protection (McCaffery *et al.*, 2007; Pepino *et al.*, 2012).

However, suspended sediment has been shown to decline rapidly over the course of a run-off-producing event because of a diminishing supply of loose sediment available for downslope transport (Croke *et al.*, 1999a; Ziegler *et al.*, 2000a,2000b; Ziegler *et al.*, 2001; Foltz *et al.*, 2009). Sheridan *et al.* (2008) reviewed the literature regarding forest road rainfall simulation experiments and found that net sediment concentration of surface run-off ranged from 1.0 to $18.9\,g\,l^{-1}$ for gravelled roads (from Selkirk and Riley, 1996; Costantini *et al.*, 1999; Riley *et al.*, 1999; Croke *et al.*, 1999b) and from 7.5 to $18.0\,g\,l^{-1}$ for unsurfaced roads (from Croke *et al.*, 1999a; Ziegler *et al.*, 2001). In our study, median TSS concentration of road surface run-off ranged from 0.57 to $7.42\,g\,l^{-1}$ and from 0.15 to $1.24\,g\,l^{-1}$ respectively, for the no gravel and high gravel treatment rainfall simulations (Table III). The higher values are associated with the first rainfall simulation experiment following a disturbance event (i.e. bulldozer trafficking or tailgate spreading of gravel from a dump truck), while the lower values are associated with the final (third) rainfall simulation experiment within a given treatment.

Field hydraulic conductivity, K_s

Run-off coefficients by treatment, calculated as event run-off (mm)/ event rainfall (mm), ranged from 0.11 to 0.84 (no gravel), 0.02 to 0.58 (low gravel), and 0.07 to 0.46 (high gravel). The median run-off coefficients of the no gravel and low gravel treatment rainfall simulation experiments (0.36 and 0.35 respectively) were greater than that of the high gravel treatment rainfall simulation experiments (0.24) by a factor of 1.5. Increases in hydraulic conductivity ($mm\,h^{-1}$) were associated with increasing gravel surface cover and ranged from 7.2 to 41.6 (no gravel), 11.9 to 46.3 (low gravel), and 16.0 to 58.6 mm h^{-1} (high gravel).

Luce (1994) estimated hydraulic conductivities from plots on freshly bladed, native-surfaced forest roads comprising six different soils in Colorado, Idaho, and Montana. Estimated hydraulic conductivities ranged from 5×10^{-5} to 8.82 mm h^{-1}. Very low (near zero) rates of hydraulic conductivity are more representative of active haul roads with frequent traffic. Much greater road surface hydraulic conductivities have been estimated for roads that were abandoned for approximately 30 years (Foltz *et al*., 2009) and also for larger road segment-scale (>10 m long) rainfall simulation experiments (e.g. Ziegler *et al*., 2000a). Foltz *et al*. (2009) measured run-off and suspended-sediment concentrations during rainfall simulation experiments on the tire track and non-tire track components of abandoned and reopened forest roads in northern Idaho. Saturated hydraulic conductivity ranged from 7 to 28 mm h^{-1} for the abandoned ('brushed-in') road and from 13 to 21 mm h^{-1} for the recently reopened road. Overall, infiltration was greatest in the non-track portions of the abandoned road that had been left to regrow.

Study findings from Luce (1994), Ziegler *et al*. (2000a), and Foltz *et al*. (2009) indicate that infiltration rates of unpaved forest roads can be highly variable and dependent on factors such as traffic (i.e. type, weight, and frequency), time since disturbance, soil type, and climate, as well as the scale of rainfall simulation experiments. Frequent traffic leads to highly compacted road surfaces and consequently lower infiltration rates. Following road abandonment and traffic cessation, cracks and other macropores can form over time as a result of weathering, vegetation re-establishment and burrowing by organisms, resulting in increased infiltration rates. Also, the greater surface areas of large-scale rainfall simulation experimental plots can result in a higher probability of overall infiltration rates being influenced by one or more macropores (i.e. areas of high infiltration). Therefore, it is arguable that large-scale rainfall simulation experiments offer a more representative estimate of hydraulic conductivity values at the road segment scale, while being well aligned with the spatial scale of forest road management and erosion modelling.

We found relatively high values for hydraulic conductivity, ranging from 7.2 to 41.6 mm h^{-1}, for abandoned forest road–stream crossing approaches that were recently reopened by bulldozer blading. Earlier work at RHFRRC that estimated road surface infiltration rates with double-ring infiltrometers found that rates ranged between 0.6 and 7.2 mm hr^{-1} (Kadak *et al*., 2000, unpublished data). It is hypothesized that the relatively large scale of our rainfall simulation experiments (61–116 m^2) had a greater likelihood of interaction with macropores at the road segment scale, which resulted in the greater conductivity rates. In addition, our estimates of hydraulic conductivity were representative of partially saturated conditions, as opposed to completely saturated conditions. Our estimates of K_s would likely be lower had each applied rainfall experiment reached steady-state run-off conditions and soil was sufficiently saturated. This is because infiltration rates decrease over time as wetting front potential in the soil decreases. It is also important to note that the stream crossing approaches were abandoned prior to this study and that following reopening by bulldozer blading, traffic was limited to light-vehicle use to perform the rainfall simulation experiments (one to two passes per week), as well as two passes by a dump truck to spread gravel on the approaches.

We found that median K_s for the high gravel treatment (36.7 mm h^{-1}) was greater than that of the low gravel (33.9 mm h^{-1}) and no gravel (33.1 mm h^{-1}) treatments. The slope group effect was significant ($p = 0.01$), with higher hydraulic conductivities occurring on steeper slopes, suggesting a potential interaction with soil texture (Table I). The treatment effect was significant ($p = 0.02$), and separation of the least-squares means indicated that hydraulic conductivity was greatest for the high gravel rainfall simulation experiments (Table IV). This finding suggests that infiltration rates tended to be higher with the highest gravel surface cover. Increased surface roughness likely acted to slow run-off velocities and allowed greater infiltration. Luce (2002) suggests that we may not be able to ameliorate road surface infiltration for active roads because they are impervious by design. However, we found that the most gravel surface cover was associated with increased infiltration. More importantly, gravel surface cover helps to reduce TSS concentration of road surface run-off at stream crossings and thus sediment delivery to streams.

Standardized sediment yield

Median TSS load per rainfall simulation experiment was 1.73, 0.38, and 0.17 kg respectively for the no gravel, low gravel, and high gravel treatment rainfall simulation experiments (Table V). Sediment yield per unit rainfall (mg m^{-2} mm^{-1}) ranged from 185 to 6470 (no gravel treatment), 30 to 1130 (low gravel treatment), and 58 to 268 (high gravel treatment). Sheridan and Noske (2007) quantified sediment yield per unit rainfall for different forest road types in the State Forest in the Central Highlands area of the Great Dividing Range, Victoria, Australia, and estimated a range of 216 to 5373 mg m^{-2} mm^{-1}. The lower value represented a minimum-traffic, high-quality gravel road, while the higher value represented an unsurfaced road, with erodible soil and moderate light-vehicle traffic. In our study, following road reopening by

Table V. Summaries of event TSS load (kg), run-off amount (mm), rainfall amount (mm), and run-off coefficient (run-off/rainfall) by treatment

	TSS load (kg)			Run-off (mm)			Rainfall (mm)			Run-off coefficient		
Treatment	Min	Median	Max	Min	Median	Max	Min	Median	Max	Min	Median	Max
No gravel	0.29	1.73	6.41	0.7	7.4	16.6	5.6	20.2	42.0	0.11	0.36	0.84
Low gravel	0.04	0.38	1.86	0.3	5.5	21.5	8.8	14.8	36.8	0.02	0.35	0.58
High gravel	0.04	0.17	0.57	0.9	3.5	9.6	4.2	15.0	29.6	0.07	0.24	0.46

bulldozer blading, traffic included light-vehicle use to complete the rainfall simulation experiments (i.e., one to two passes per week), as well as two passes by a dump truck to spread gravel on the approaches.

Sheridan *et al*. (2008) used rainfall simulation to quantify interrill erodibility index values of unsealed forest roads for comparison with observed forest road erodibility index values that were quantified from 1 year of detailed *in situ* erosion monitoring. Sediment yield per unit rainfall (mg m^{-2} mm^{-1}) estimated from 1 year of monitoring (natural rainfall) ranged from 839 to 8671 for the bare roads and from 312 to 2687 for the gravel roads. Sediment yield per unit rainfall (mg m^{-2} mm^{-1}) estimated from the rainfall simulation experiments ranged from 3650 to 7024 for the gravel roads and from 3462 to 28,426 for the bare roads (Sheridan *et al*., 2008). Our estimates of road surface erodibility index values are within the range of values estimated by Sheridan and Noske (2007) and Sheridan *et al*. (2008) for natural rainfall conditions but considerably less than those estimated from applied rainfall experiments by Sheridan *et al*. (2008). Our rainfall experiments were more representative of natural rainfall conditions in that rainfall intensity was variable, and on average, rainfall intensity was half that of experiments by Sheridan *et al*. (2008), where applied rainfall intensity was constant (100 mm h^{-1}) for 30 min. Dynamic erodibility of road surface soil (Ziegler *et al*., 2000b; Foltz *et al*., 2008a) is a function of rainfall intensity and soil type, as well as the cumulative rainfall amount since a disturbance event, such as road blading. Therefore, it may be difficult or impossible to assign a single soil erodibility parameter value with confidence to a specific road segment, which is common practice in parameterizing soil erosion models, even if the parameter value is based on well-controlled field experiments (Brazier *et al*., 2000).

We found that median sediment yield per unit rainfall (mg m^{-2} mm^{-1}) by treatment was 788 (no gravel), 389 (low gravel), and 161 (high gravel). Median sediment yield per unit rainfall for the no gravel treatment rainfall simulation experiments was greater than that of the low gravel and high gravel treatments by factors of 2.0 and 4.9 respectively.

Results for standardized sediment yield were similar to TSS concentration in that treatment ($p < 0.0001$) and simulation group ($p = 0.03$) were both significant model effects (Table IV). Separation of the least-squares means by treatment indicated that no gravel > low gravel > high gravel. Simulation group 1 was greater than simulation group 3 but not different than simulation group 2. There were no differences in standardized sediment yield among slope groups.

Cost-effectiveness of gravel treatments

The mean cost of the low gravel treatment was \$91.77 or \$3.34 m^{-2}. When applied to stream approaches with different lengths (19.2 to 41.3 m), the low gravel treatment (i.e. gravel application beginning near the stream and continuing uphill for a length of 9.8 m) resulted in 34–60% cover on the road approach running surfaces. Increasing surface cover led to decreases in sediment yield (Figure 6). Overall, the low gravel treatment reduced sediment yield by a factor of 2.0 over the no gravel treatment. The high gravel treatment doubled costs (\$183.54) and resulted in 50–99% coverage of the road approach running surfaces. Overall, the high gravel treatment reduced sediment yield by a factor of 4.9 over the no gravel treatment. However,

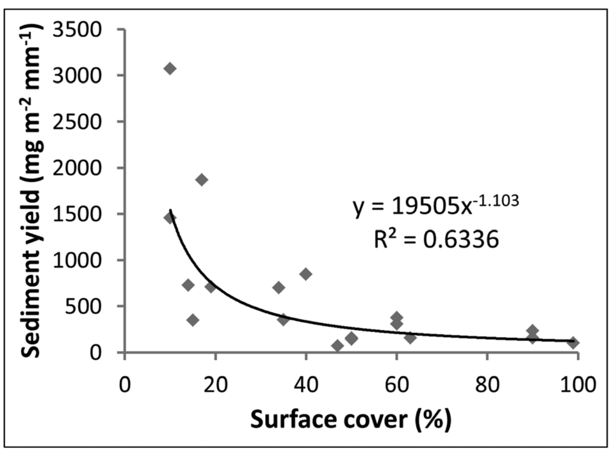

Figure 6. Mean sediment yield per unit rainfall (mg m^{-2} mm^{-1}) for each site and treatment combination (6 sites × 3 treatments = 18 observations) *versus* running surface cover

Table VI. Costs associated with the gravel treatments and differences in sediment yield in comparison to the no gravel treatment

	Sediment yield (mg m^{-2} mm^{-1})			Gravel cost ($)		Sediment yield difference from no gravel (%)	
Site	No gravel	Low gravel	High gravel	Low gravel	High gravel	Low gravel	High gravel
1	1458	351	233	100.96	201.92	−76	−84
2	350	850	158	81.42	162.84	143	−55
3	1867	374	158	104.22	208.44	−80	−92
4	3072	699	144	81.42	162.84	−77	−95
5	728	71	156	94.45	188.90	−90	−79
6	709	309	100	84.68	169.36	−56	−86

increasing gravel surface cover did not always result in sediment yield reductions, indicating that BMP effectiveness is site specific. For example, rutting from the low gravel treatment application at Site 2 resulted in greater sediment yields than the no gravel treatment (Table VI). In four of the six study sites, however, the succession of gravel treatments led to successive decreases in sediment yield (Table VI).

The high gravel treatment covered 90–99% of the running surface area for the shortest approaches (Sites 1, 3, and 6; Table II), and sediment yields were reduced to less than 250 mg m^{-2} mm^{-1} (Figure 6). TSS concentrations were reduced to less than 1 g l^{-1} for sites with more than 90% cover. One option for reducing the costs associated with graveling stream crossing approaches is to use water-control structures (water bars/turnouts, cross drains, broad-based, or rolling dips) to reduce the length of the road segment draining directly to the stream. Shorter approaches (e.g. 15.2 m) require less gravel and less money to surface. In addition, shorter approaches manage surface run-off in smaller quantities. In our case, the main cost associated with graveling the stream crossing approaches was that of the gravel itself. Gravel was procured and tailgate spread with a dump truck that is maintained and operated by RHFRRC staff. The costs associated with graveling as presented in this research would be increased by additional factors, such as whether or not a dump truck is owned or must be rented, operator costs, and the distance to the nearest gravel retailer.

In this study, stream crossing approach lengths ranged from 19.2 to 41.3 m. The lengthy approaches are representative of legacy forest roads, indicating that upon reopening, additional measures to reduce the drainage length and maximize road approach surface cover may be necessary to protect water quality. The Virginia Department of Forestry BMP manual (VDOF, 2011) recommends that the entire length and width of active stream crossing approaches be covered with gravel, mulch, or other suitable material or for a minimum of 15.2 m on either side of the crossing. In many cases, low-cost water-control structures such as water bars, turnouts, or rolling dips could be installed on reopened stream crossing approaches to limit the drainage length to 15.2 m. The total cost of gravel to cover this section of road would then be 15.2 m x $9.41 per metre of road length = $143. Based on our study findings, completely gravelled stream crossing approaches with minimal traffic had TSS concentrations well below 1 g l^{-1} during rainfall events. This demonstrates that judicious BMP implementation can be used to reduce sediment delivery to streams in situations where original road conditions at the road–stream interface were less than ideal.

CONCLUSIONS

The direct contribution of surface run-off and sediment delivery to streams at forest road–stream crossings necessitates the careful management of storm water run-off to reduce negative impacts to water quality and aquatic habitat. In the case of reopening legacy road–stream crossing approaches, perhaps even greater efforts are necessary to protect water quality, especially if the approaches have minimal surface cover, inadequate water control, or steep grade. Quantification of road segment-scale surface hydrologic processes and sediment delivery during storm events, as influenced by factors such as road type and traffic frequency, are crucial to facilitate the development of cost-effective BMPs to control sediment transport where the probability of water quality impacts are highest.

This research used rainfall simulation experiments to quantify road segment-scale surface run-off and suspended-sediment delivery for reopened stream crossing approaches with different levels of gravel surface cover. The cost-effectiveness of gravel application was also quantified in terms of reductions in suspended-sediment delivery. Study findings suggest that newly reopened legacy road–stream crossing approaches can have much higher rates of hydraulic conductivity than highly trafficked crossings but that reopened (and unsurfaced) stream crossing approaches can represent significant sources of direct storm water run-off and sediment delivery to streams. This research demonstrates that judicious and low-cost BMP implementation can be used to ameliorate common water quality problems associated with reopening legacy road–stream crossing approaches.

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