Catena 159 (2017) 93–105

Contents lists available at ScienceDirect

Catena

journal homepage: www.elsevier.com/locate/catena

Effects of closed roads, traffic, and road decommissioning on infiltration and sediment production: A comparative study using rainfall simulations

Gabriel Sosa-Pérez[a,*], Lee H. MacDonald[b]

[a] Instituto Nacional de Investigaciones Forestales, Agrícolas y Pecuarias, Km. 33.3 Carretera Chihuahua-Ojinaga, Aldama, Chihuahua C.P.32910, México
[b] Department of Ecosystem Science and Sustainability, Colorado State University, 1476 Campus Delivery, Fort Collins, CO 80523-1476, United States

A R T I C L E I N F O

Keywords:
Road decommissioning
Road erosion
Infiltration
Sediment production
Rainfall simulations
Traffic

A B S T R A C T

Road closures and road decommissioning are increasingly being used to reduce runoff and sediment production from unpaved roads, but few studies have quantitatively assessed the effectiveness of these treatments. This study used rainfall simulations to: 1) quantify the differences in infiltration and sediment production among five treatments: undisturbed forest, closed roads, closed roads exposed to all-terrain vehicle (ATV) traffic, and two decommissioning treatments (ripping only, and ripping plus wood-strand mulch); and 2) quantify the effects of key site variables on infiltration and sediment production. Four replicate rainfall simulations were conducted for each treatment in northcentral Colorado, with 44 mm h^{-1} of rainfall being applied to 1 m^2 bounded plots for 45 min. The mean infiltration rate for the last 5 min ("infiltration capacity") for the forest was 28 mm h^{-1} and highly variable, while the closed roads with and without traffic had nearly identical mean values of only 5 and 4 mm h^{-1}, respectively. Ripping only increased the mean infiltration capacity to 9 mm h^{-1}, while adding mulch more than doubled this to 20 mm h^{-1}. Mean sediment production from the forested plots was only 3 g m^{-2} as compared to 43 g m^{-2} from the closed roads with no traffic. Eighty passes of an ATV tripled the mean sediment production compared to the closed roads with no traffic. The mean sediment production for the ripping treatment was 72 g m^{-2} or 67% higher than the mean value from the closed roads, while adding mulch decreased the mean sediment production to just 16 g m^{-2}. These results first show the importance of roads and even small amounts of traffic for increasing plot-scale runoff and sediment production, and second that ripping plus mulching is a more effective road decommissioning treatment than just ripping. The results provide important guidance for future road decommissioning efforts.

1. Introduction

Sediment production and delivery from unpaved forest roads is a key environmental concern due to the potential effects on water resources infrastructure and the physical characteristics of water, particularly turbidity and total suspended solids (Goode et al., 2012, MacDonald and Stednick, 2003; Motha et al., 2003). Changes in these parameters can adversely affect the beneficial uses of domestic water supply, recreation, and aquatic ecosystems, particularly coldwater fisheries (Wood and Armitage, 1997). The documented impacts of roads are a direct result of the very large changes in runoff and erosion due to the highly compacted road surface, even though roads typically represent a small proportion of most forested and rural landscapes (Ramos-Scharrón and LaFevor, 2016; Ziegler and Giambelluca, 1997).

A common way to reduce the adverse environmental impacts of roads is to remove or decommission a road that is no longer needed or desirable (Switalski et al., 2004; Weaver et al., 2015). Decommissioning treatments can vary from relatively cheap and simple methods, such as closing the road by installing a gate or other barrier, to more expensive treatments such as full recontouring (Madej, 2001; Switalski et al., 2004; Weaver et al., 2015). While closing a road is the least expensive treatment, closing a road—even for several decades—may not restore infiltration rates to the values observed in an undisturbed forest. In Idaho the saturated hydraulic conductivity of an abandoned road after thirty years with no traffic was still only 7–28 mm h^{-1} (Foltz et al., 2009), which is much lower than the value of 40–80 mm h^{-1} for an undisturbed forest (Robichaud, 2000). In Peninsular Malaysia an abandoned logging road had > 80% vegetation cover after 40 years, but the saturated hydraulic conductivity of 62 mm h^{-1} was still an order of magnitude lower than the value of 675 mm h^{-1} for the adjacent hillslopes (Ziegler et al., 2007).

Although closing a road may not restore infiltration rates, the partial recovery in infiltration—when combined with the lack of traffic and the increase in surface cover by vegetation and litter—can greatly

* Corresponding author.
E-mail addresses: sosa.gabriel@inifap.gob.mx (G. Sosa-Pérez), lee.macdonald@colostate.edu (L.H. MacDonald).

http://dx.doi.org/10.1016/j.catena.2017.08.004
Received 5 September 2016; Received in revised form 13 July 2017; Accepted 7 August 2017

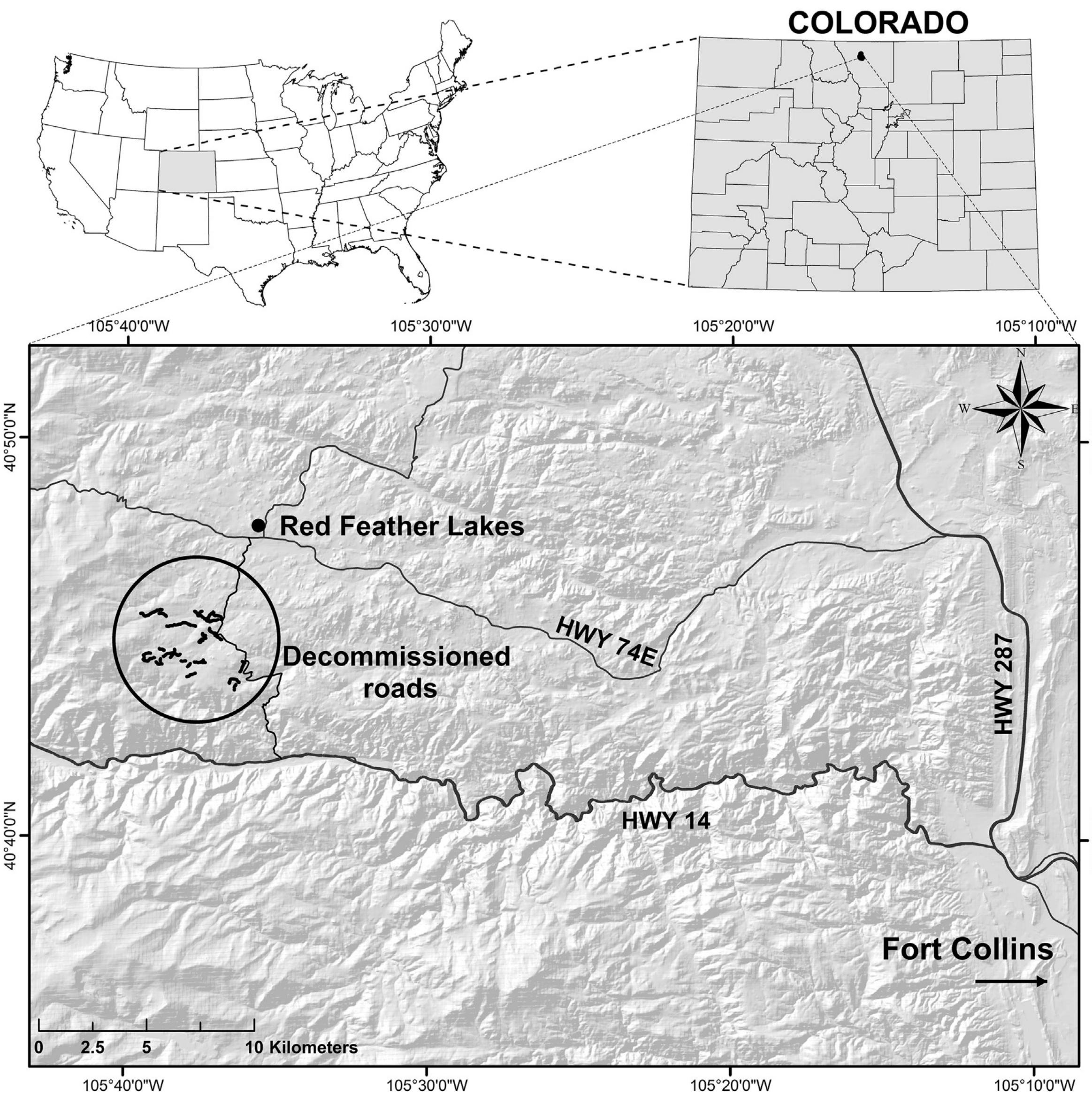

Fig. 1. Location of the road sections in the Arapaho-Roosevelt National Forest that were decommissioned in fall 2013 by ripping or ripping plus mulching. The closed roads and forested plots were immediately adjacent to the decommissioned roads.

reduce road sediment production. Rainfall simulations on an abandoned road in Idaho with 98% ground cover yielded a mean sediment concentration of 2.2 g L^{-1}, which was only 14% of the value from a similar road that had been subjected to logging traffic two years before the rainfall simulation (Foltz et al., 2009).

Numerous studies have shown that traffic greatly increases road sediment production by increasing the fine sediment supply through abrasion and crushing, as well as forcing fine sediment to the surface (Luce and Black, 1999; Reid and Dunne, 1984; Sheridan et al., 2006; Ziegler et al., 2001). High numbers of log trucks increased sediment production by 7.5 times compared to the same roads on days with no logging traffic (Reid and Dunne, 1984), and 2 to 25 times for roads heavily used by logging trucks as compared to roads with light traffic (Foltz, 1996). The type of traffic also may be important, as the erosion from unmanaged ATV and dirt bike trails can be similar to or greater than an active forest road with regular car and truck traffic (Meadows et al., 2008; Welsh, 2008).

A second common road decommissioning treatment is to rip the roadbed by metal tines being pulled behind a bulldozer to eliminate the compaction (Luce, 1997; Weaver et al., 2015). The ripping can be followed by the addition of mulch to reduce surface erosion, but the effectiveness of ripping, or ripping plus mulching, is still controversial. In Alberta, Canada ripping only decreased the bulk density from 1.60 to 1.40 Mg m^{-3}, or 13% (McNabb, 1994). In Idaho ripping initially decreased the bulk density to 1.50 Mg m^{-3} and increased the hydraulic conductivity from 8 to 30 mm h^{-1}, but after 90 mm of simulated rainfall the bulk density increased back up to 1.70 Mg m^{-3} and the hydraulic conductivity dropped by half to 15 mm h^{-1} (Luce, 1997). Similarly, the hydraulic conductivity two years after ripping was only 9 mm h^{-1} (Foltz et al., 2007). These results indicate that the initial increase in infiltration due to ripping is very transient, and the resultant infiltration rate is still substantially less than the typical infiltration rate of approximately 40–120 mm h^{-1} for undisturbed coniferous forests (Robichaud, 2000; Moody and Martin, 2001).

The problem is that relatively few studies have experimentally quantified the effects of different decommissioning treatments on infiltration and sediment production, even though road decommissioning has become an important restoration treatment on both public and private lands (Madej, 2001; Weaver et al., 2015). For instance, from 1998 to 2002 the USDA Forest Service (USFS) decommissioned 3200 km of road per year at an average cost of $2500 per kilometer (Schaffer, 2003), and more recently the USFS has been decommissioning over 2000 km yr^{-1} (USDA Forest Service, 2010–2014). More specific data are needed to quantify the benefits of different decommissioning treatments, and allow these benefits to be compared to the costs of each treatment.

The initial goal of our research was to evaluate the effectiveness of road decommissioning treatments in northcentral Colorado using a combination of sediment fences and road surveys at the road segment scale. While segment-scale measurements provide useful data for managers, they do not provide explicit data on infiltration rates over time or key surface erosion processes, particularly the combination of rainsplash and sheetwash. Rainfall simulations are very useful for quickly providing comparative runoff and sediment production data along with process-based insights (Arnáez et al., 2004; Butzen et al., 2014; Croke et al., 2006; Foltz et al., 2009; Jordan and Martinez-Zavala, 2008; Sheridan et al., 2008; Ziegler et al., 2000). The implication is that the best strategy is to conduct studies at different spatial scales using different techniques, with rainfall simulations providing detailed comparisons of infiltration and smaller-scale sediment production rates, and segment-scale measurements providing larger scale but less directly comparable data. This paper reports our plot-scale results using rainfall simulations, while a separate paper presents the results of a multi-year study of road erosion and decommissioning treatments at the road segment and landscape scales (Sosa-Pérez and MacDonald, 2017).

Hence the objectives of this paper are to: 1) quantify the differences in infiltration and sediment production between undisturbed forest, closed roads, closed roads exposed to all-terrain vehicle (ATV) traffic, and two decommissioning treatments (ripping only, and ripping plus wood-strand mulch); 2) quantify the effects of the measured site variables on infiltration and sediment production; and 3) understand how ATV traffic affects plot-scale sediment availability and sediment production. The results will help guide the design and quantify the benefits of future road closures and decommissioning projects.

2. Methods

2.1. Study area

The study area is at an elevation of 2630 to 2850 m in the Arapaho-Roosevelt National Forest in northcentral Colorado (Fig. 1). Average annual precipitation at the Red Feather Lakes weather station is 460 mm (WRCC, 2016), with about 36% of this falling as snow between October and April (NOAA, 2013). From May through September the precipitation falls primarily as rain, often in brief but occasionally intense thunderstorms (NOAA, 2013). Soils are predominantly Redfeather-Schofield-Rock outcrop association. The Redfeather and Schofield soils vary only in their depth to bedrock, and they are shallow to moderately deep (40–100 cm), well-drained sandy loams formed on granitic bedrock; the taxonomic description is loamy-skeletal, mixed, superactive Lithic Glossocryalfs (Moreland, 1980; USDA NRCS, 1998).

The vegetation is predominantly lodgepole pine (*Pinus contorta*) with some ponderosa pine (*Pinus ponderosa*), Douglas fir (*Pseudotsuga menziesii*), and quaking aspen (*Populus tremuloides*) according to aspect, soil wetness, and elevation. Much of the area was patch cut in the 1950s to 1970s (Veblen and Donnegan, 2005), and since then the roads have been primarily used for recreation, especially by all-terrain vehicles (ATV) and dirt bikes. Many of these roads are no longer needed given the reduction in logging and lack of property risk from fires.

2.2. Road decommissioning

In early summer 2013 the Arapaho-Roosevelt National Forest identified 14 km of roads for decommissioning distributed among 30 road sections. These road sections were selected because they were no longer needed for access, posed a disturbance to wildlife, and/or represented a risk to water resources due to their proximity to a stream. Most of the road sections had been closed to traffic for about 25 years, but there are no records of when the various road sections were closed. A few of these road sections were still open to recreational traffic, particularly ATVs. The roads were decommissioned in September–October 2013, and the primary treatment was ripping the road surface to a depth of approximately 0.4 m with a tracked bulldozer pulling three unwinged ripping teeth. After ripping wood-strand mulch and organic fertilizer were applied to about 40% of the total length, primarily to the steeper sections or sections close to a stream; target application rates were 6.2 Mg ha^{-1} of mulch and 0.3 Mg ha^{-1} of fertilizer. The initial depth of the mulch was not measured but was estimated at one to two centimeters. By the time of the rainfall simulations in summer 2014—nearly one year after decommissioning—much of the mulch had washed or fallen into the furrows created by ripping (Section 3.1).

2.3. Experimental design and plot measurements

The experimental design was five treatments with four replicates each, making a total of 20 rainfall simulations. The five treatments included undisturbed forest as an overall control, closed roads with little to no administrative traffic, closed roads subjected to traffic (80 passes of an ATV), and two decommissioning treatments (only ripping, and ripping plus mulching and fertilizer). The simulations on the closed roads are considered a treatment when compared to the undisturbed forest, and a control for evaluating the effects of traffic and decommissioning. The choice of five treatments was set according to the study design and desired information by the funding agency, while the number of replicates (n = 4) was a function of the available time and funding.

The four plots on closed roads were necessarily placed on two road sections because these were the only closed roads that were not subject to illegal ATV traffic. The effect of traffic was assessed by obtaining permission for an ATV to make 80 passes on the lower portion of each of the two closed roads, as this is a relatively typical amount of traffic for lower-traffic recreational roads on a summer weekend (Sosa-Pérez and MacDonald, 2017). The four plots for each decommissioning treatment were each on a different road section in order to capture as much of the between-road variability as possible. The four forested plots were randomly placed in mature forest with no evidence of recent disturbance, but the tree density was low due to low site quality and possibly some natural or human disturbances decades earlier.

Sediment availability before and after the 80 ATV passes was evaluated by sweeping and collecting the loose surface soil from three 30-cm wide swaths across the active width of each of the two road sections subjected to traffic (Fig. 2). This yielded six samples before and six samples after the 80 ATV passes. Each sample was dried for 24 h at 105 °C and sieved to determine the mass and particle-size distribution (Topp and Ferre, 2002), and the mean mass of available sediment by size class before and after the ATV passes was determined for each road.

The rainfall simulations were conducted on 1 m^2 plots bounded on the sides and top by sheet metal borders inserted 5–10 cm into the ground (Fig. 3). The edges were sealed by a mixture of native soil and bentonite. The plots on the closed roads were placed to include one wheel track and a portion of the center of the road. Similarly, the plots on the decommissioned roads were placed in the center of the road to include one of the ripped furrows (Fig. 3b). A thin plastic sheet was fastened to the ground with staples to collect the overland flow and direct it into a sheet metal collector for sampling (Fig. 3b). A plexiglass shield over the plastic sheet and sheet metal collector excluded the simulated rainfall.

Measurements before each rainfall simulation included slope, soil bulk density, soil moisture, surface roughness, and percent ground cover. Slope was measured with a digital level (Smart Tool®). Bulk density was measured at three locations around the perimeter of each plot by determining the volume of fine sand needed to fill an excavated volume that was approximately 10 cm × 10 cm × 10 cm. One of these

Fig. 2. Swath across the active width of a closed road after sweeping and collecting the loose surface soil.

samples was taken in a furrow or a wheel track if present, while the other two samples were taken between the furrows or wheel tracks. The excavated soil was dried for 24 h at 105 °C to determine the gravimetric soil moisture and dry mass (Topp and Ferre, 2002).

Topographic and fine-scale surface roughness was measured both longitudinally (downslope flow direction) and laterally along three transects in each plot by placing a fine-linked chain over the surface. Roughness was calculated as the length of the chain divided by the length or width of the plot, respectively (Butzen et al., 2014). In each of the decommissioned plots the width and depth of the furrow was measured at three locations along with the depth of any mulch. Ground cover was measured on a grid of 100 points in each plot, with each point being classified as bare soil, rock (> 1 cm), live vegetation, litter and wood, or wood-strand mulch. Cover was defined as 100 minus percent bare soil.

The amount and pattern of infiltration over time suggested that at least some of the plots in the undisturbed forest had water repellent soils. We therefore measured soil water repellency in the forested plots using the critical surface tension method (CST) (Watson and Letey, 1970). Drops of de-ionized water with increasing concentrations of ethanol were successively used to determine the surface tension at which four of five drops infiltrated within five seconds (King, 1981). The solutions used in this study were 0, 1, 3, 5, 9, 14, 19, 24, 34, and 48% ethanol, and these were converted to surface tension following Huffman (2001). Water repellency was measured at three points around the perimeter of the plots at depths of 0, 3, 6, and 9 cm. The water repellency at each depth was the surface tension associated with the concentration of the last solution that indicated soil water repellency, and this surface tension was averaged among the three sample locations to determine the water repellency for each depth for each plot. Lower CST values indicate stronger soil water repellency (Watson and Letey, 1970). Water repellency was not measured in any of the other treatments as soil water repellency is primarily in areas with permanent vegetation cover (Doerr et al., 2006), compacted areas are not water repellent (Wagenbrenner et al., 2015), and ripping breaks up any water repellent layer.

2.4. Rainfall simulations

Rainfall simulations were carried out between July and September 2014, which was 10–12 months after the roads had been decommissioned. Rainfall was applied for 45 min using a Purdue-type rainfall simulator (Fig. 3a), which has an oscillating nozzle centered 3 m above the plot (Foster et al., 1982). The rainfall simulator and the plot were shielded with a tarp to minimize wind effects, and the rainfall intensity was measured at the end of each simulation by raining for an additional 5 min onto a 1 m^2 plastic-lined box and measuring the steady-state runoff rate. Mean rainfall intensity was 44 mm h^{-1} with minimum and maximum values of 42 and 46 mm h^{-1}. The rainfall intensity was intentionally higher than the maximum 30-min rainfall intensity of 25 mm h^{-1} recorded by five tipping bucket rain gages in the summers of 2013 and 2014 in order to ensure that some Horton (infiltration-excess) overland flow was generated from the forested and ripped plots. This intensity corresponds to a 30-min storm with about a 20-year recurrence interval (NOAA, 2016), and is lower than the intensities used in other rainfall simulation studies (e.g., Foltz et al., 2011; Benavides-Solorio and MacDonald, 2001; Sheridan et al., 2008). A single

Fig. 3. a) The Purdue-type rainfall simulator set up above a plot on a decommissioned road that had been ripped and mulched with wood strands and fertilizer. b) Detailed view of the plot immediately prior to the rainfall simulation showing the plot construction, central furrow, uneven coverage of the wood-strand mulch, and limited vegetative growth in the 10 months since the road had been treated.

simulation was conducted on each plot because this most closely mimics the summer thunderstorms that typically occur under dry conditions and generate nearly all of the road surface runoff and sediment production (Sosa-Pérez and MacDonald, 2017).

The time from the start of rainfall to the beginning of runoff was recorded, and runoff samples were collected for the first 30 s of each minute in 1000 mL plastic bottles. Each sample from the first 20 min of each simulation and every other runoff sample from 20 to 45 min was taken back to the lab. These samples were weighed to determine the mass of runoff, while the volume of runoff for the other samples was measured in the field. The mass of sediment in each lab sample was determined by filtering it through a pre-weighed 5 μm paper filter, drying the filter, and calculating the dry mass of sediment. Sediment concentrations were calculated by dividing the mass of sediment by the volume of runoff.

The infiltration rate for each minute was determined by subtracting the runoff rate from the measured rainfall intensity, and infiltration capacity (mm h^{-1}) was defined as the average infiltration rate for the last 5 min of each simulation. This implicitly assumes a constant depth of ponding, which was generally true except for the first 5–7 min of the simulations. For each simulation the sediment production in g m^{-2} was calculated by multiplying the runoff volume by the corresponding sediment concentration in g L^{-1}, and summing these. After each simulation trenches were cut through the plot to observe the depth and spatial variation of the wetting front.

2.5. Statistical analysis

Differences between treatments were first analyzed with the nonparametric Kruskal-Wallis test (SAS Institute, Inc., 2002–2010) given the small number of plots per treatment and that at least some of the independent and the two dependent variables did not appear to be normally distributed. If there was a significant difference at $p < 0.05$, the data were transformed to ranks to satisfy the assumptions for an analysis of variance (ANOVA), and the LSMeans test was used to determine which means were significantly different. Tukey's method was used for all pairwise comparisons (SAS Institute, Inc., 2002–2010). Spearman correlation coefficients and simple linear regressions were used to evaluate the interrelationships between plot characteristics, log-transformed infiltration capacities, and log-transformed sediment production. We also used Random Forest (Mohr et al., 2013), another nonparametric statistical technique, to further assess the relative importance of each measured variable on log-transformed infiltration capacity and sediment production.

3. Results

3.1. Plot characteristics

The mean slope of all plots was 6% with a range of 4 to 10%, but there were no significant differences in mean slope by treatment (Table 1). Mean bulk density for the forested plots was 1.28 g cm^{-3} (s.d. = 0.21 g cm^{-3}), and this was significantly lower than the mean value of 1.75 g cm^{-3} (s.d. = 0.08 g cm^{-3}) for the closed roads. Mean bulk density for the road plots subjected to traffic was only slightly higher than the mean value for closed roads (Table 1). Ripping only reduced the bulk density to 1.54 g cm^{-3} (s.d. = 0.21 g cm^{-3}) or just 14% less than the closed roads, and the mean bulk density for the ripping and mulching treatment was nearly identical at 1.52 g cm^{-3} (s.d. = 0.13 g cm^{-3}). Mean bulk densities for both decommissioning treatments were significantly higher than the forested plots, and significantly lower than the closed roads (Table 1). For the decommissioned roads there was no significant difference in mean bulk density in the furrows versus outside of the furrows ($p = 0.63$), and we attribute this to the lateral disturbance caused by the ripping, especially as the tines hit and moved rocks. This lateral disturbance was greatest towards the surface where we measured bulk density.

Mean soil moisture prior to the rainfall simulations varied from 4% to 18%, with higher and more variable values for the forested plots and the ripping plus mulching treatment than the closed roads or the ripping treatment. In contrast, mean soil moisture for the closed roads and the closed roads with traffic was significantly lower at < 5% and less variable (Table 1). The mean soil moisture of 4.1% for the decommissioning treatment of only ripping was very similar to the closed roads and roads with traffic, but the mean soil moisture value of 8.4% for ripping plus mulching plots was significantly higher and more variable (Table 1). We attribute much of the higher soil moisture for the forested plots and the ripping plus mulching treatment to the higher litter and mulch cover, which would reduce evaporation (Jalota et al., 2001).

Both the lateral and longitudinal roughness values were relatively high for the forested plots due to the high litter and vegetative cover, while the closed roads had virtually no surface roughness. Both decommissioning treatments had significantly more roughness than the closed roads, particularly in the lateral direction as the furrows had a mean width of 0.33 m (s.d. = 0.08 m) and a mean depth of 0.08 m (s.d. = 0.01 m). The mulch increased both the lateral and longitudinal roughness, in part because it was so unevenly distributed (Fig. 3b). Much of the mulch had washed or fallen into the furrows so these had a mean mulch depth of 0.05 m (s.d. = 0.02 m). Mulch coverage outside of the furrows varied from none to several times the approximately 5-mm thickness of the wood strands.

Ground cover and the amount of bare soil varied significantly among the different treatments (Fig. 4). The forested plots had 93–100% ground cover, and this was primarily litter and wood rather than live vegetation. In contrast, the closed roads and closed roads subjected to ATV traffic both had an average surface cover of just 13% (s.d. = 3%), and this cover was mostly rock (Fig. 4). The mean ground cover of 33% (s.d. = 5%) in the ripped plots was more than twice the value of the closed roads, and this was a roughly equal mixture of rocks and live vegetation plus litter. Mean ground cover for the ripped and mulched plots was 67% (s.d. = 4%), or twice the value of the plots that had only been ripped, and this difference was due to the 48% (s.d. = 4%) cover provided by the wood-strand mulch (Fig. 4).

3.2. Infiltration rates over time by treatment

The overall mean time to the beginning of runoff was 4.5 min, and this varied from 3.4 to 6.2 min among the different treatments. While there were no significant differences in the time to runoff among

Table 1
Mean (standard deviation) of the plot characteristics by treatment. Different letters indicate significant differences at $p < 0.05$.

Plot characteristic	Forest	Closed roads	Closed roads with traffic	Ripping	Ripping and mulching
Slope (%)	8^a (1)	6^a (1)	5^a (1)	6^a (3)	8^a (2)
Bulk density (g cm^{-3})	1.28^a (0.21)	1.75^b (0.08)	1.79^b (0.08)	1.54^c (0.21)	1.52^c (0.13)
Soil moisture (%)	8.4^a (4.2)	4.7^b (1.7)	4.1^b (0.2)	4.1^b (2.1)	8.4^a (5.1)
Roughness ratio (lateral)	1.15^a (0.10)	1.01^b (0.01)	1.02^b (0.01)	1.13^a (0.02)	1.20^a (0.03)
Roughness ratio (longitudinal)	1.13^a (0.08)	1.01^b (0.01)	1.01^b (0.01)	1.06^a (0.04)	1.14^a (0.05)
Bare soil (%)	1^a (3)	86^b (5)	88^b (2)	68^c (5)	34^d (4)

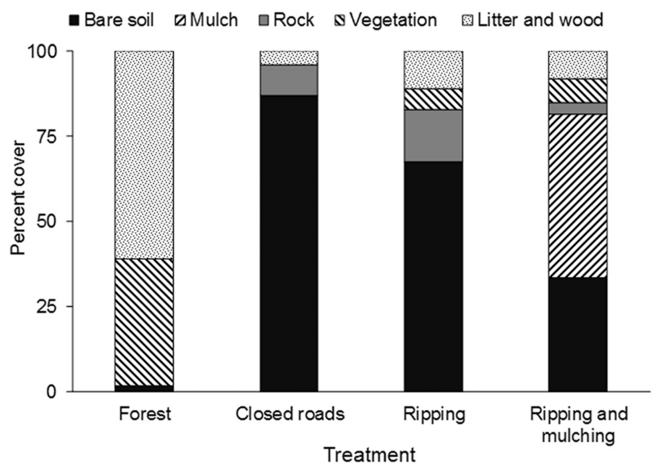

Fig. 4. Mean percent ground cover for the forested plots, closed roads, and two decommissioning treatments. Cover data for the closed roads with traffic are combined with closed roads because the values were nearly identical.

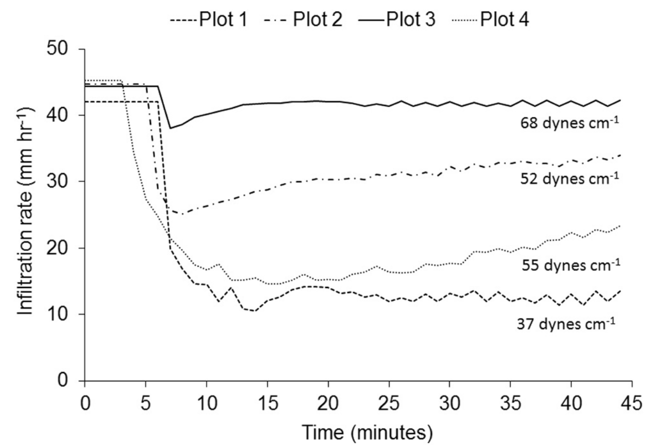

Fig. 6. Infiltration rates over time for each rainfall simulation in the forest. The values underneath each line are the mean critical surface tension at the soil surface in dynes cm^{-1}, where 72 dynes cm^{-1} is the value for pure water and lower values indicate stronger soil water repellency.

treatments (p = 0.07), the shortest mean time of 3.4 min was for the closed roads subjected to traffic, while the longest mean time of 6.2 min was for the ripped and mulched plots followed by 5.5 min for the forested plots. The longer time to runoff for these two treatments can be attributed to the greater moisture storage capacity due to the litter, mulch, and surface roughness (Kittredge, 1948).

Infiltration declined sharply once runoff began except for the ripping and mulching treatment, where infiltration declined more gradually (Fig. 5). In the forested plots infiltration rates tended to increase after about 6–7 min, and this increase is in marked contrast to the declines in infiltration for the other treatments (Fig. 5). The forested plots had both the highest mean infiltration capacity at 28 mm h^{-1} and the highest variability as values ranged from 13 to 42 mm h^{-1} (Fig. 6).

Both the increase in infiltration over time and the high variability in the forested plots can be largely explained by the variations in soil water repellency at the mineral soil surface (Fig. 6). The plot with the highest infiltration capacity had the weakest soil water repellency (68 dynes cm^{-1}), while the plot with the lowest infiltration capacity had the strongest soil water repellency (37 dynes cm^{-1}). The other two plots had intermediate surface tension values of 55 dynes cm^{-1} (plot 4) and 52 dynes cm^{-1} (plot 2), and both of these plots showed an increase in infiltration over time as the soil wetted up. In contrast, the plots with no and very strong water repellency (plots 3 and 1, respectively) showed very little change in infiltration over time (Fig. 6). No soil water repellency was observed below the mineral soil surface.

The closed roads and the closed roads with traffic had the sharpest decline in infiltration, the lowest mean infiltration capacities (4 and 5 mm h^{-1} respectively), and very low variability (s.d. of 1 and 3 mm h^{-1}, respectively) (Figs. 5, 7). The ripped plots also had a sharp but slightly later decline in infiltration, with infiltration dropping to 15 mm h^{-1} at 15 min and then slowly declining to the final value of 9 mm h^{-1} (s.d. = 3 mm h^{-1}). The ripped and mulched plots initially had a much higher mean infiltration rate than the ripped plots, but had a greater decline from 30 mm h^{-1} at 15 min to just under 20 mm h^{-1} (s.d. = 6 mm h^{-1}) at the end of the simulation (Fig. 5). Mean infiltration capacities for the forested plots and the ripping plus mulching treatment were significantly higher than each of the other treatments (Fig. 7).

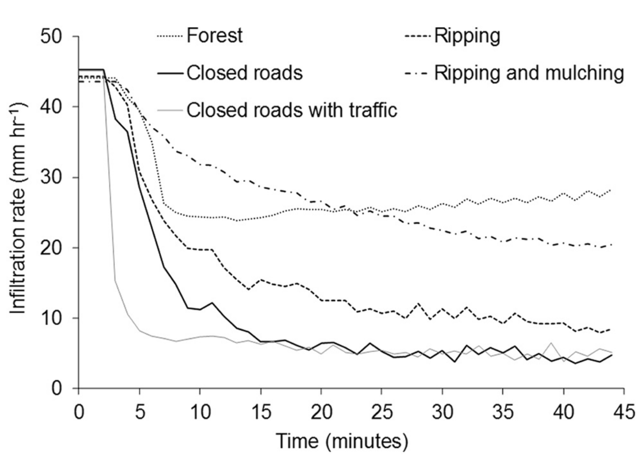

Fig. 5. Mean infiltration rates over time for each of the five treatments.

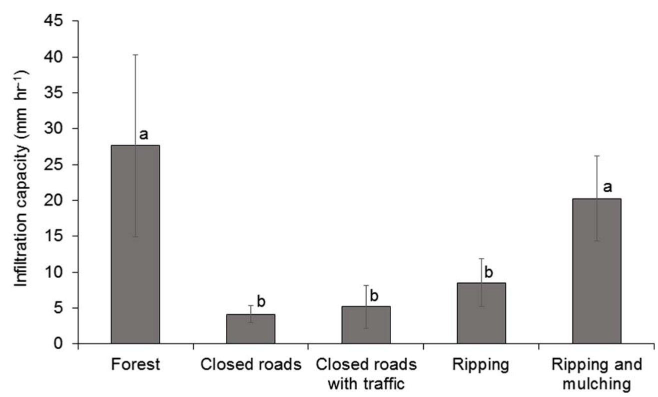

Fig. 7. Mean infiltration capacity for each of the five treatments. Error bars represent the standard deviation, and different letters indicate significant differences.

The excavations after the simulations showed that the forested plots had considerable variability in the depth and spatial extent of infiltration, with completely dry soils where there was stronger soil water repellency and wet soils in areas with preferential flow. Infiltration was much more uniform for the plots on the closed roads, but there usually was not a clear line between the wetting front from the simulated rainfall and pre-existing soil moisture. Both decommissioning treatments tended to have nearly saturated soil under the furrows, while the soils outside of the furrows were not nearly as wet. This high spatial variability meant that it was not possible to clearly determine and measure the depth of the wetting front for the different treatments. We did note that the saturated zone underneath the furrow was > 20 cm deep for some of the ripped and mulched plots as compared to about 10 cm for the plots that had only been ripped.

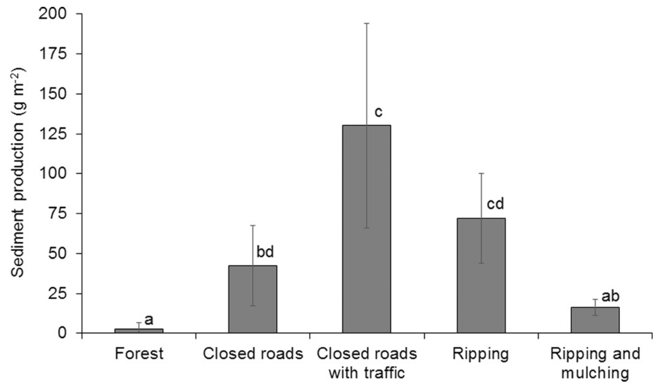

Fig. 8. Mean sediment production by treatment. Error bars represent the standard deviation, and different letters indicate significant differences.

3.3. Sediment production

Mean sediment production from the forested plots was only 2.8 g m^{-2} (s.d. = 3.7 g m^{-2}) (Fig. 8), but this appeared to be primarily organic matter rather than mineral soil. Sediment production was slightly higher in the first 15 min but there wasn't a sharp peak (Fig. 9), indicating a lack of readily available sediment.

Mean sediment production for the closed roads was 43 g m^{-2} (s.d. = 25 g m^{-2}) or 15 times the mean value from the forested plots, and this difference was significant (Fig. 8). Sediment production was highest for the first 15 min of runoff, with relatively little change over the last 25 min (Fig. 9) despite a slow increase in runoff. The initial flush of sediment in the first 15–20 min is commonly observed in sediment studies and is generally attributed to the presence of readily available sediment (e.g., Sheridan et al., 2006; Walling and Webb, 1982; Ziegler et al., 2001).

Mean sediment production from the closed roads subjected to 80 ATV passes was 130 g m^{-2} (s.d. = 64 g m^{-2}) or three times the mean sediment production from the closed roads with no ATV traffic, and this difference was significant (Fig. 8). The pattern of sediment production over time was remarkable for the very high initial peak as soon as runoff began (Fig. 9), indicating a relatively large supply of readily available sediment. Sediment production then declined over time, suggesting a decreasing supply of sediment, but mean sediment production over the course of the simulation was always higher than any other treatment (Fig. 9).

Sediment production from the ripped plots was relatively similar to the closed roads over the first eight minutes of the simulation, but the mean sediment production stayed relatively high for the entire

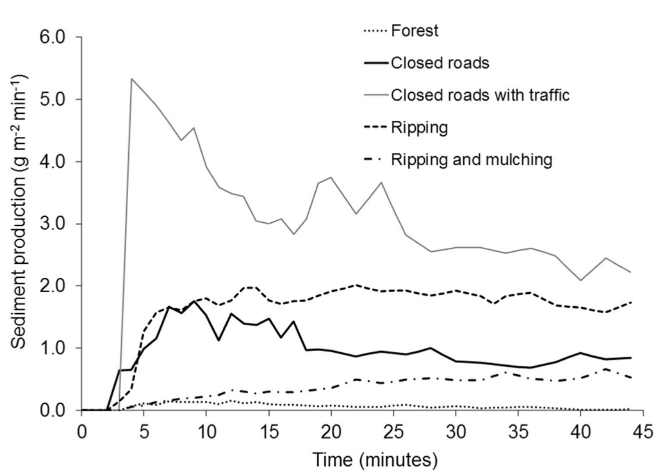

Fig. 9. Mean sediment production in grams per square meter per minute for each of the five treatments.

simulation (Fig. 9). This pattern suggests that sediment production for the ripped plots was not as supply limited, and this is supported by the strong correlation between runoff and sediment production (R^2 = 0.67; p < 0.0001). Mean sediment production for the ripped plots of 72 g m^{-2} (s.d. = 28 g m^{-2}) was 40% higher than the closed roads but 45% lower than the closed roads with traffic; neither of these differences were significant due to the variability within treatments (c.v. = 0.38–0.58) (Fig. 8).

Adding mulch reduced the mean sediment production to only 16 g m^{-2} (s.d. = 5 g m^{-2}) or 22% of the mean value from the ripped plots, and this difference was significant (Fig. 8). In contrast to the other treatments, the ripped and mulched plots had no initial flush of sediment and sediment production slowly increased over time (Fig. 9). Like the ripping treatment, sediment production for these four plots was strongly and linearly related to runoff (R^2 = 0.90, p < 0.0001), but neither infiltration capacity nor sediment production was significantly related to percent mulch cover or the depth of mulch in the furrows. By the end of the simulation the mean sediment production rate from the ripped and mulched plots was approaching the value from the closed roads, indicating a continuing decrease in the effectiveness of the mulch for reducing sediment production.

3.4. Controls on infiltration capacity and sediment production

Infiltration rates for the four forested plots were strongly correlated with soil water repellency (R^2 = 0.74, p = 0.004), but were not significantly related to bulk density, roughness, soil moisture, or ground cover. When the data from all 20 plots were pooled, the log-transformed infiltration capacity was positively and significantly correlated with ground cover (r = 0.89), longitudinal roughness (r = 0.88) and slope (r = 0.54), and negatively correlated with bulk density (r = − 0.83; Table 2). These correlations are not independent as Table 2 shows that ground cover was very strongly correlated with bulk density and roughness, while the significant correlation between plot slope and infiltration capacity was due to the fact that the forested plots and the plots that had been ripped and mulched had the highest infiltration capacities and slightly higher slopes (Table 1, Fig. 7).

The analysis using Random Forest yielded very similar results, with ground cover and roughness being the two strongest controls on the log-transformed infiltration capacity, followed by bulk density and soil moisture (Table 3). The data from Table 2 and a scatterplot of infiltration capacity versus ground cover (Fig. 10a) provide a more explicit view of these relationships, as closed roads and roads with traffic fall near the origin with their low infiltration capacity, high bulk density, low ground cover, and low surface roughness, while the forested plots lie at the upper end with their high infiltration capacity, high ground cover, generally low bulk density, and high surface roughness. Similarly, the scatterplot of infiltration capacity versus bulk density shows that the road plots were relatively consistent in having a high bulk density and low infiltration capacity, while both the forested and ripped plots had substantially more variation in bulk density but still tended to have high and low infiltration capacities, respectively (Fig. 10b).

The key point is that infiltration rates for the plots that had been ripped and mulched were most similar to the forested plots, while infiltration rates for the plots that had only been ripped were always lower and more similar to the closed roads in absolute terms. These results indicate a clear benefit to mulching after ripping (Fig. 7).

Log-transformed sediment production significantly increased with decreasing ground cover (r = − 0.74), infiltration capacity (r = − 0.65), slope (r = − 0.60), and longitudinal roughness (r = − 0.58), and with increasing bulk density (r = 0.66; Table 2). These relationships with the plot characteristics are almost exactly the inverse of the correlations with infiltration capacity. The analysis using Random Forest confirmed that ground cover was the most important control on sediment production, followed by slope and infiltration

Table 2
Correlation matrix of Spearman correlation coefficients for the 20 rainfall simulations. Both infiltration capacity and sediment production values were log transformed.

	Slope (%)	Bulk density (g cm^{-3})	Soil moisture (%)	Longitudinal roughness ratio	Ground cover (%)	Time to runoff (min)	Infiltration capacity (mm h^{-1})
Bulk density (g cm^{-3})	−0.61**						
Soil moisture (%)	0.38	−0.24					
Longitudinal roughness ratio	0.50*	−0.76**	0.25				
Ground cover (%)	0.59*	−0.91**	0.28	0.86**			
Time to runoff (min)	0.37	−0.49*	0.00	0.59**	0.64**		
Infiltration capacity (mm h^{-1})	0.54*	−0.83**	0.41	0.88**	0.89**	0.58**	
Sediment production (g m^{-2})	−0.60*	0.66**	−0.16	−0.58*	−0.74**	−0.60**	−0.65**

* Indicates that the correlation is significant at p ≤ 0.05.
** Indicates that the correlation is significant at p ≤ 0.01.

Table 3
Percent increase in mean standard error for each of the independent variables evaluated in Random Forest for the log-transformed infiltration capacity and log-transformed sediment production, respectively. Higher values indicate greater importance.

Variable	Infiltration capacity	Variable	Sediment production
Ground cover (%)	9.17	Ground cover (%)	7.05
Roughness ratio	9.10	Slope (%)	4.94
Bulk density (g cm^{-3})	6.31	Infiltration capacity (mm h^{-1})	4.83
Soil moisture (%)	5.84	Water repellency	3.98
Water repellency	0.66	Roughness ratio	3.34
Slope (%)	0.28	Soil moisture (%)	2.76
		Bulk density (g cm^{-3})	1.71

capacity (Table 3).

A scatterplot of sediment production versus infiltration capacity again provides useful insights, with the closed roads, roads with traffic, and ripping treatments at the high end of the regression, the ripped and mulched plots falling in the middle, and the two forested plots with the least water repellency, highest infiltration, and lowest sediment production anchoring the low end of the regression (Fig. 11). The greater scatter in the relationship between infiltration capacities and sediment production is primarily due to the high variability in infiltration rates in the forested plots as described above, and the high variability in sediment production from the closed roads with traffic as presented in the next section.

3.5. Effect of traffic on sediment availability and particle size distribution

The 80 passes of an ATV had no effect on infiltration capacity but tripled mean sediment production compared to the closed roads with no traffic (Figs. 7, 8). Much of this increase in sediment production can be attributed to the increase in available sediment. Prior to the 80 ATV passes the mean mass of loose sediment was 2.6 kg m^{-2} (s.d. = 0.6 kg m^{-2}) with a strong peak in sand-sized particles (0.063–2 mm). After 80 ATV passes the mean mass increased by 46% to 3.8 kg m^{-2} (s.d. = 0.9 kg m^{-2}) with the two roads having nearly identical amounts of readily available sediment.

Prior to any traffic the particle-size distribution of the loose sediment was generally similar between the plots on each road and between the two roads (Fig. 12). The main difference was that road 2 tended to have more particles larger than 2 mm. For road 1 the 80 ATV passes particularly increased the mass of particles larger than 0.25 mm, while for road 2 nearly all the increase in available sediment was in particles smaller than 1.0 mm (Fig. 12). Sediment production from the two rainfall simulations on road 1 was only 67 g m^{-2} and 86 g m^{-2}, respectively, while mean sediment production for the two plots on road 2

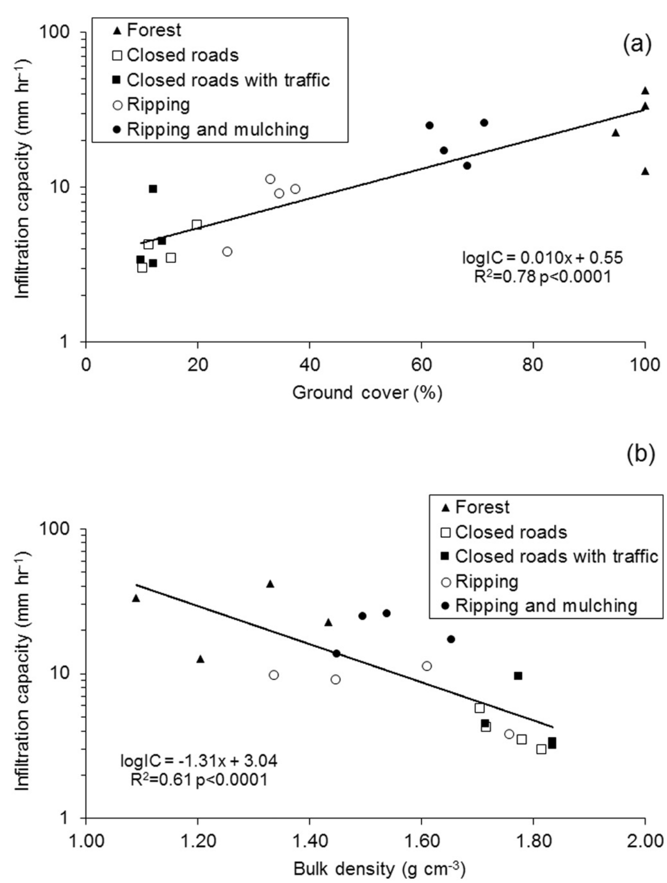

Fig. 10. Log-transformed infiltration capacity (logIC) for each rainfall simulation by treatment (n = 20) versus (a) ground cover and (b) bulk density.

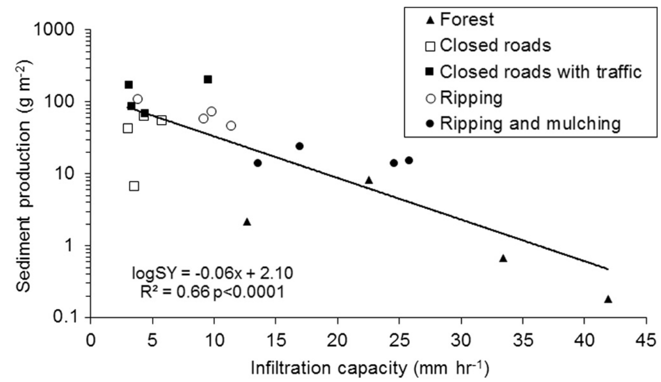

Fig. 11. Relationship between the log-transformed sediment production (logSY) and infiltration capacities for each simulation (n = 20).

was 2.4 times higher (169 g m^{-2} and 199 g m^{-2}). Since the two roads had very similar ground cover, bulk densities, roughness, slopes, soil moisture, and mass of readily available sediment after traffic, the much greater sediment production from road 2 must be attributed to the greater increase in readily available fine particles relative to road 1 (Fig. 12).

4. Discussion

4.1. Soil water repellency in the undisturbed forest

Soil water repellency at the soil surface was the most important factor influencing overland flow in the forested plots. Under dry conditions soil water repellency is typical for soils with permanent vegetation cover, such as grasslands and coniferous forests (Doerr et al., 2006; Shakesby et al., 2000). The lack of any soil water repellency below the surface also is consistent with most other studies in unburned coniferous forests (Doerr et al., 2009). The forested plots were the only treatment where mean infiltration increased over the course of the simulation, and this is consistent with how water repellent soils are initially resistant to wetting but become more hydrophilic as the critical soil moisture threshold is exceeded (Doerr et al., 2006; MacDonald and Huffman, 2004).

The variations in soil water repellency and infiltration capacities among the forested plots contributed to the lack of any significant difference in mean infiltration capacity between the forested plots and the plots that had been ripped and mulched. At larger scales and under wetter soil conditions the mean infiltration capacity in the forest could be substantially higher; reported infiltration rates range from 77 mm h^{-1} for undisturbed Douglas-fir/lodgepole pine forests in Idaho (Robichaud, 2000) to > 120 mm h^{-1} for ponderosa pine forests in Colorado (Martin and Moody, 2001). The infiltration rate also may be underestimated in the forested plots because some portions of the plots may still not have been generating any infiltration-excess overland flow (see Betson, 1964), or the plots would produce less runoff when the soils are wetter and hence less hydrophobic. A higher infiltration rate in the forest would result in a larger difference in infiltration between the forested plots and the two decommissioning treatments, and thereby reduce the apparent effectiveness of the two decommissioning treatments.

4.2. Infiltration and sediment production from closed roads and the effect of traffic

The low infiltration capacities of the closed roads match up well with other published studies. Saturated hydraulic conductivity values for unpaved roads from 18 studies around the world ranged from 1 to 10 mm h^{-1}, and steady-state infiltration rates were from 3 to 5 mm h^{-1} (Ramos-Scharrón and LaFevor, 2016). These low infiltration rates are due to compaction, the destruction of soil aggregates by traffic, and the associated sealing of the surface by fine particles (Ziegler et al., 2000). Our measured values confirm that low infiltration rates can persist for several decades after a road is closed (Foltz et al., 2009; Ziegler et al., 2007), and this means that road closures may provide relatively little benefit in terms of restoring the normal hydrologic regime.

Mean sediment production from the closed roads was 15 times higher than the mean value from the forested plots, but in absolute terms the mean sediment production from the closed roads was only 43 g m^{-2} (430 kg ha^{-1}) of active road surface area, and this is low compared to most published studies (Fu et al., 2010; MacDonald and Coe, 2008). This low sediment production rate is probably due in part to the relatively coarse particle-size distribution typical of granitic areas (Luce and Black, 1999). The implication is that closed roads can continue producing sediment long after a road is closed, but the absolute sediment production may still not be very large. In contrast, just 80 ATV passes increased mean sediment production by a factor of three, and higher sediment production rates would be expected with more traffic (Reid and Dunne, 1984; Sheridan et al., 2006). Hence road closures can provide large benefits in terms of reducing road sediment production, particularly in more productive areas with more litterfall and faster vegetative regrowth (Foltz et al., 2009), and in areas where bedrock weathering produces more of the silt- and clay-sized particles that are most easily eroded (Dunne and Leopold,1978).

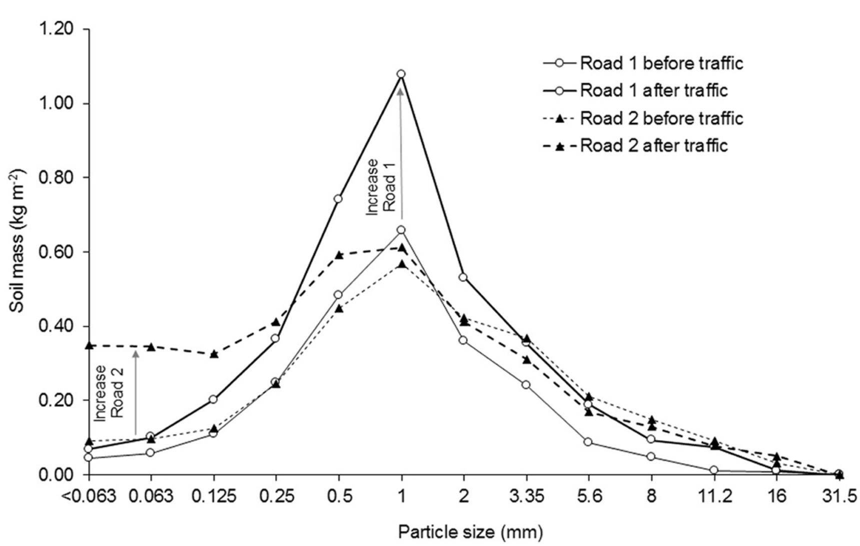

Fig. 12. Mean mass of loose sediment on the road surface by particle size for roads 1 and 2 before traffic (thin lines) and after 80 passes of an off-highway vehicle (heavy lines). Each line represents the mean of three samples. For clarity the particle sizes on the x axis are plotted on a phi ($\log_2$) scale.

The much higher sediment production rates for roads with traffic are generally due to the increased supply of readily erodible sediment by abrasion and crushing of the road surface materials, and the upward forcing of fine-grained sediment from the road bed (Reid and Dunne, 1984; Sheridan et al., 2006; van Meerveld et al., 2014; Ziegler et al., 2001). Both closed roads and the closed roads subjected to ATV traffic showed a sharp decline in sediment production after about 15–20 min, indicating a decrease in the supply of readily-available fine sediment. Other studies have shown a decrease in rainsplash detachment over time due to the compacted surface and less available sediment, as well as the development of a thin layer of overland flow that helps protect the road surface from raindrop impact (Arnaez et al., 2004; van Meerveld et al., 2014; Ziegler et al., 2000). The problem is that the mean measured sediment production for roads 1 and 2 was only about 2% and 5% of the total available sediment as measured by sweeping.

This low proportion of eroded sediment can be at least partly explained by the fact that the primary erosion processes on our 1 m^2 plots were the interrill processes of rainsplash and sheetwash (Ries et al., 2013), and these processes are very size selective (Costantini et al., 1999; Luce and Black, 1999; Sheridan et al., 2008). A study in Australia showed that particles < 0.02 mm were < 6% of the total available sediment, but accounted for 50–90% of the total sediment production from rainfall simulations on 6 m^2 plots (Costantini et al., 1999). Graveled roads on a silty clay loam soil in western Oregon yielded about nine times more sediment than comparable roads on a gravelly loam soil (Luce and Black, 1999), indicating that the mass of fine particles is more important than the total mass of loose soil. For roads 1 and 2 measured sediment production was respectively 45% and 26% of the readily available sediment < 0.125 mm in diameter. These percentages, plus the observed coarsening of the plot surface during the simulations, suggests that the applied rainfall was relatively effective in removing the finer particles that are most readily detached and transported. It should be noted that the force applied by sweeping also is probably very different and not size selective compared to the forces applied by rainsplash and sheetwash, and this further complicates direct comparisons between the available sediment and measured sediment production.

It is not clear why the same number of ATV passes had such a different effect on the amount and particle-size distribution of the available sediment on the two roads (Fig. 12). The two roads had similar lithology and particle-size distributions before traffic, but we could not control exactly how or where the ATV drove on each road. We hypothesize that repeated passes following the same wheel tracks might affect the amount and particle-size distribution of fine sediment, but we know of no studies that have directly evaluated this issue.

4.3. *Effectiveness of the ripping and ripping plus mulching treatments on infiltration and sediment production*

The bulk density data indicate that ripping the road surface caused only a small reduction in bulk density, and we attribute this to soil settling over the 10–11 months between when the roads were ripped and the rainfall simulations. Soil settling is the re-compaction of soil due to the rearranging of soil grains over time (SSSA, 2001). The roads to be decommissioned were ripped in early September 2013, and before any mulch could be applied much of the Colorado Front Range was subjected to a very unusual, widespread rainstorm. The mean 6-day rainfall in our study area was 206 mm, with 90 mm falling in just 18 h. The estimated return periods for the two- to six-day rainfalls was 200–500 years (NOAA-NWS, 2013), but the total erosivity and erosion was not that large given the generally low rainfall intensities (Schmeer, 2014; Sosa-Pérez and MacDonald, 2017). Since soil recompaction is primarily a function of cumulative rainfall and soil physical properties rather than the kinetic energy of the falling raindrops (van Wesemael et al., 1995), the large September storm was probably the primary cause of soil settlement. Soil settling and recompaction were probably further enhanced by the subsequent winter snowfall, spring snowmelt, and smaller rainstorms before we began our simulations in July 2013.

This hypothesis is supported by other work, as a study in Idaho showed that ripping increased the hydraulic conductivity from 8 to 30 mm h^{-1}, but after 90 mm of simulated rainfall the bulk density increased from 1.50 to 1.70 Mg m^{-3} and the hydraulic conductivity dropped by half to 15 mm h^{-1} (Luce, 1997). The same study also showed that straw mulch provided minimal protection against soil settling (Luce, 1997). We also found that mulching had no effect on bulk density. These results mean that the two-fold decrease in infiltration and the four-fold increase in sediment production from the ripped plots as compared to the ripped and mulched plots are due to processes other than soil settling. We found that mulching nearly doubled the mean ground cover and filled in more than half the furrow depth compared to the plots that were only ripped. As documented in other studies (Grismer and Hogan, 2005; Larsen et al., 2009; Moore and Singer, 1990; Thompson and James, 1985), a mulch cover absorbs the kinetic energy of raindrops and protects soil aggregates from being broken apart, thereby reducing both soil sealing and soil detachment by rainsplash. These beneficial effects can help explain the increased infiltration and decreased erosion due to mulching.

Another likely cause of the greater infiltration in the mulched plots is the greater infiltration in the furrow. The furrows in all of the ripped plots occupied about one-third of the plot area, and the furrow played an important role in collecting and directing the runoff from the plots to the plot outlet. In the plots that were only ripped, needles and small depressions created miniature dams, but these were broken or overtopped by the concentrated overland flow, allowing a relatively efficient delivery of runoff to the plot outlet. In contrast, the wood-strand mulch was concentrated in the furrows of the mulched plots, and this trapped more of the runoff, reduced the flow velocity, and increased the opportunity for infiltration. The vegetative regrowth also tended to be concentrated in the furrow (Fig. 3b), and root channels facilitate more and deeper infiltration by preferential flow (Beven and Germann, 2013). The greater infiltration in the furrow was clearly shown by the greater depth of the saturated zone when the plots were trenched after the simulations. Other studies confirm that wood strands increase depression storage and reduce overland flow velocities (Foltz and Dooley, 2003; Govers et al., 2000).

The greater effect of the mulch on sediment production than infiltration capacity can be attributed to both the increased ground cover and the accumulated mulch in the furrows. Particle detachment by rainsplash was observed in both decommissioning treatments, but there was visually more detachment and pedestal development in the ripped plots due to the greater amount of bare soil. In contrast to the closed roads, the greater small-scale surface roughness in the ripped soil prevented the development of a consistent layer of overland flow that would help protect the soil against rainsplash. The 33% ground cover in the ripped plots also is too low to greatly reduce erosion (Larsen et al., 2009), so the continuing exposure of the ripped soil to rainsplash provided a continuing supply of sediment as shown in Fig. 9. The mulch also greatly slowed overland flow in the furrow, which would reduce sediment detachment and allow more trapping of the sediment being transported.

The surprising result is that the mulched plots showed a consistent decrease in infiltration and increase in sediment production over the 45-min simulation. These trends indicate a relatively rapid decrease in the effectiveness of mulching after its application, and suggest that this decrease will be faster in areas with more rainfall. Over longer time periods the effectiveness of both decommissioning treatments will primarily depend on the rate of vegetative regrowth, as percent ground cover is a primary control on both infiltration and surface erosion (Larsen et al., 2009).

4.4. Scaling to hillslopes and road segments

A key issue is the extent to which these plot-scale results can be extrapolated to the road segment or hillslope scale, which is the smallest scale that is generally considered by resource managers. Extrapolation of the infiltration capacities from the forested plots is difficult given the high variability observed in our data (s.d. = 13 mm h^{-1}) and other studies on the spatial variability in infiltration and runoff generation (e.g., Betson, 1964; Loague and Gander, 1990; Beven, 2011). Our data indicate that soil water repellency had an important effect on infiltration, and other studies have confirmed the high spatial heterogeneity of soil water repellency (Doerr et al., 2006; Doerr et al., 2009; Huffman et al., 2001). At the hillslope scale we would expect less overland flow than we observed from our plots due to the commonly observed log-normal distribution of infiltration measurements and high spatial variability (e.g., Doerr et al., 2009; Loague and Gander, 1990; Martin and Moody, 2001). These considerations mean that surface runoff from a plot or portion of a hillslope can often infiltrate further downslope (Butzen et al., 2014; Larsen et al., 2009), resulting in less infiltration-excess overland flow than would be predicted from the mean of our plot-scale measurements.

Our measured infiltration rates from closed roads and closed roads with traffic can be more readily extrapolated to the road segment scale because compacted road surfaces are spatially much more uniform than forested hillslopes. This lower spatial variability is demonstrated by the low standard deviations of the infiltration capacities from the plots on the closed roads (1–3 mm h^{-1}). Our segment-scale sediment production data confirm that rainfall rates of only 5–11 mm h^{-1} were sufficient to generate overland flow and deliver sediment into our sediment fences (Sosa-Pérez and MacDonald, 2017). For closed roads in wetter areas or under different site conditions there may be more variability in infiltration as a result of more vegetative regrowth, burrowing by organisms, or other factors that increase preferential flow paths (Beven and Germann, 2013; Foltz et al., 2009; Ziegler et al., 2000), and hence potentially less segment-scale runoff (Brown et al., 2015).

Extrapolating our plot-scale infiltration capacities to the road segment scale is easier for the ripping treatment because this treatment—like the closed roads—had a standard deviation of only 3 mm h^{-1}. In contrast, infiltration capacities for the ripping and mulching treatment were more variable (s.d. = 6 mm h^{-1}), but our field observations and segment-scale results show that surface runoff and segment-scale sediment production depend to a large extent on the infiltration and trapping capacity of the furrows created by the ripping (Sosa-Pérez and MacDonald, 2017).

Extrapolation of the sediment production values from our 1 m^2 plots to the road segment scale is more difficult because the rainfall simulations do not incorporate the larger-scale process of rill erosion, which requires the accumulation and concentration of surface runoff from larger areas (Luce and Black, 1999; MacDonald et al., 1997). Nevertheless, the simulation results may be extrapolated to larger scales under certain conditions, such as outsloped roads where the flow distance is too short to induce much rilling, or relatively flat road segments where there is sheetflow rather than rilling. Extrapolation of our plot-scale sediment production data to longer and steeper road segments will probably underestimate unit area sediment production as there is a greater propensity for significant rilling (Ramos-Scharrón and MacDonald, 2005).

Additional scaling issues come into play when extrapolating the simulation results from the ripped or ripped plus mulched plots to the road segment scale. For our 1 m^2 plots ripping plus mulching was far more effective than only ripping in terms of increasing infiltration and reducing erosion. However, our segment-scale data indicate that both decommissioning treatments were relatively effective in reducing sediment production because most of the runoff and nearly all of the sediment was trapped in the furrows (Sosa-Pérez and MacDonald, 2017). This difference between our plot and segment scale results shows that rainfall simulations are useful for providing specific parameter values and process-based insights, but larger-scale observations and a process-based understanding are needed to reliably extrapolate from the plot to road segment or hillslope scales.

4.5. Management implications

Land managers have a choice of techniques to reduce road surface runoff, erosion, and the delivery of this material to streams. Road closure is the simplest and cheapest treatment, and our data indicate that closed roads will continue to generate large amounts of surface runoff but relatively little sediment compared to roads subjected to recreational ATV traffic. The decision of whether to close a road, or to undertake more expensive decommissioning treatments, will therefore depend on the management objectives. For example, closing a road may be a viable treatment if the primary objective is to reduce sediment delivery to a stream, regardless of the amount of surface runoff being generated. However, if the management objective is to restore the natural hillslope hydrology, or to eliminate road-stream connectivity, a more intensive road decommissioning treatment is necessary.

We found that ripping plus mulching was significantly more effective for increasing infiltration and reducing erosion than only ripping, but the ripping plus mulching treatment still did not restore the hydrologic functioning of the hillslope. The mean infiltration capacity of 20 mm h^{-1} for the ripping and mulching treatment at the end of the simulation was nearly 30% below the mean infiltration capacity of the forested plots, and less than the maximum summer rainfall intensities of 25 mm h^{-1} recorded in 2013 and 2014 (Sosa-Pérez and MacDonald, 2017). Over more or longer storms this difference in infiltration is likely to become even larger as the water repellent forest soils wet up and infiltration increases, while infiltration rates from the two ripping treatments should continue to decline (Fig. 5). Similarly, mean sediment production from the ripping and mulching treatment was nearly six times the mean value from the forested plots, and this difference kept increasing over the course of the simulations (Fig. 9). Managers have to decide when mulching is worth the additional cost compared to only ripping, but both our plot- and segment-scale data suggest that mulching may be justified—particularly for longer and steeper segments near a stream—when water quality protection is a high management priority.

The time scale of the restoration objectives also is important, as the more intensive decommissioning treatments, such as ripping plus mulching or full recontouring, will immediately increase infiltration and reduce road-stream connectivity. Both our plot- and segment-scale data results show that closing a road will lead to a relatively rapid decrease in sediment production due to the lack of traffic, but our work and other studies indicate that many decades may be needed before the infiltration rate of a closed road begins to approach the value of a natural forest.

5. Conclusions

Rainfall simulations were used to quantify infiltration and sediment production rates from a lodgepole pine forest, closed roads, closed roads subjected to limited ATV traffic, and two road decommissioning techniques. The four forested plots had the highest but highly variable infiltration rates, with the variability being attributed to the spatial variations in surface soil water repellency. The forested plots also had the lowest mean sediment production because of the high surface cover, high surface roughness, and high mean infiltration capacity. The closed roads had a final mean infiltration capacity that was nearly an order of magnitude lower than the forested plots, and a mean sediment production that was more than an order magnitude higher. Eighty passes of an off-highway vehicle had no effect on the infiltration capacity but increased the amount of loose sediment by nearly 50% and tripled the mean sediment production compared to the same closed roads with no traffic. Sediment production was much greater for the road with a larger increase in the mass of readily-available fine sediment, indicating that plot-scale sediment production is highly sensitive to the size of the loose particles on the road surface.

Ripping doubled the mean infiltration capacity of closed roads to 9 mm h^{-1}, but this was still only 32% of the mean infiltration capacity from the forested plots. Ripping caused a 67% increase in mean sediment production relative to the closed roads. Adding wood-strand mulch after ripping more than doubled the final infiltration rate and reduced the mean sediment production by more than a factor of four compared to just ripping. The positive effect of mulching can be attributed to the protection from rainsplash and surface sealing, enhanced infiltration in the furrows where the mulch had accumulated, and reduced sediment transport due to greater roughness and slower flow velocities. Over longer periods and under wetter conditions both decommissioning treatments may be progressively less effective as infiltration declines and the sediment storage capacity in the furrows is exceeded, but this loss of effectiveness will depend on site-specific factors such as the amount and intensity of rainfall, soil type, slope, and vegetative regrowth rate.

Acknowledgments

The authors are very grateful to the Arapaho-Roosevelt National Forest and the USDA Forest Service National Stream and Aquatic Ecology Center for their financial support. Carl Chambers and Deb Entwistle provided encouragement, useful insights, and logistical support. Jim Dobrowolski kindly provided the rainfall simulator, and Junior Garza and Bob Brown were essential for refurbishing and modernizing it. Field assistance was provided by Hunter Gleason and Eric Boileau. Modeling simulations by Bill Elliot helped us better understand our results and their longer-term context, and John Turk provided key guidance on Random Forest and other statistical questions. We also want to thank Consejo Nacional de Ciencia y Tecnología (CONACYT) and Instituto Nacional de Investigaciones Forestales Agrícolas y Pecuarias (INIFAP) for their support of Gabriel Sosa-Pérez during his stay at Colorado State University. Comments from two reviewers and Bill Elliot substantially improved the clarity and focus of the paper.

References

Arnáez, J., Larrea, V., Ortigosa, L., 2004. Surface runoff and soil erosion on unpaved forest roads from rainfall simulation tests in northeastern Spain. Catena 57 (1), 1–14. http://dx.doi.org/10.1016/j.catena.2003.09.002.

Benavides-Solorio, J.D.D., MacDonald, L.H., 2001. Post-fire runoff and erosion from simulated rainfall on small plots, Colorado front range. Hydrol. Process. 15 (15), 2931–2952. http://dx.doi.org/10.1002/hyp.383.

Betson, R.P., 1964. What is watershed runoff? J. Geophys. Res. 69, 1541–1552. http://dx.doi.org/10.1029/JZ069i008p01541.

Beven, K.J., 2011. Rainfall-Runoff Modelling: The Primer. John Wiley & Sons (488 pp).

Beven, K., Germann, P., 2013. Macropore flow in soils revisited. Water Resour. Res. 49, 3071–3092. http://dx.doi.org/10.1002/wrcr.20156.

Brown, K.R., McGuire, K.J., Aust, W.M., Hession, W.C., Dolloff, C.A., 2015. The effect of increasing gravel cover on forest roads for reduced sediment delivery to stream crossings. Hydrol. Process. 29, 1129–1140. http://dx.doi.org/10.1002/hyp.10232.

Butzen, V., Seeger, M., Wirtz, S., Huemann, M., Mueller, C., Casper, M., Ries, J.B., 2014. Quantification of Hortonian overland flow generation and soil erosion in a central European low mountain range using rainfall experiments. Catena 113, 202–212. http://dx.doi.org/10.1016/j.catena.2013.07.008.

Costantini, A., Loch, R.J., Connolly, R.D., Garthe, R., 1999. Sediment generation from forest roads: bed and eroded sediment size distributions, and runoff management strategies. Aust. J. Soil Res. 37 (5), 947–964. http://dx.doi.org/10.1071/SR98088.

Croke, J., Mockler, S., Hairsine, P., Fogarty, P., 2006. Relative contributions of runoff and sediment from sources within a road prism and implications for total sediment delivery. Earth Surf. Process. Landf. 31 (4), 457–468. http://dx.doi.org/10.1002/esp.1279.

Doerr, S.H., Shakesby, R.A., Dekker, L.W., Ritsema, C.J., 2006. Occurrence, prediction and hydrological effects of water repellency amongst major soil and land-use types in a humid temperate climate. Eur. J. Soil Sci. 57 (5), 741–754. http://dx.doi.org/10.1111/j.1365-2389.2006.00818.x.

Doerr, S.H., Shakesby, R.A., MacDonald, L.H., 2009. Soil water repellency: a key factor in post-fire erosion. In: Fire Effects on Soils and Restoration Strategies. CRC Press, pp. 197–224.

Doerr, S.H., Woods, S.W., Martin, D.A., Casimiro, M., 2009. 'Natural background' soil water repellency in conifer forests of the north-western USA: its prediction and relationship to wildfire occurrence. J. Hydrol. 371, 12–21.

Dunne, T., Leopold, L.B., 1978. Water in Environmental Planning. W.H. Freeman and Company, San Francisco, CA (818 p).

Foltz, R.B., 1996. Traffic and no-traffic on an aggregate surface road: sediment production differences. In: Proceedings of the Seminar on Environmentally Sound Forest Roads and Wood Transport. U.N. Food and Agriculture Organization (FAO), Sinaia, Romania, pp. 195–204.

Foltz, R.B., Copeland, N.S., Elliot, W.J., 2009. Reopening abandoned forest roads in northern Idaho, USA: quantification of runoff, sediment concentration, infiltration, and interrill erosion parameters. J. Environ. Manag. 90, 2542–2550. http://dx.doi.org/10.1016/j.jenvman.2009.01.014.

Foltz, R.B., Dooley, J.H., 2003. Comparison of erosion reduction between wood strands and agricultural straw. Trans. ASAE 46 (5), 1389–1396.

Foltz, R.B., Elliot, W.J., Wagenbrenner, N.S., 2011. Soil erosion model predictions using parent material/soil texture-based parameters compared to using site-specific parameters. Trans. ASABE 54 (4), 1347–1356.

Foltz, R.B., Rhee, H., Yanosek, K.A., 2007. Infiltration, erosion, and vegetation recovery following road obliteration. American Society of Agricultural and Biological Engineers 50 (6), 1937–1943.

Foster, G.R., Neibling, W.H., Nattermann, R.A., 1982. A Programmable Rainfall Simulator. Paper 82–2570. American Society of Agricultural and Biological Engineers, St. Joseph, Michigan.

Fu, B., Newham, L.T.H., Ramos-Scharrón, C.E., 2010. A review of surface erosion and sediment delivery models for unsealed roads. Environmental Modeling & Software 25, 1–14. http://dx.doi.org/10.1016/j.envsoft.2009.07.013.

Goode, J.R., Luce, C.H., Buffington, J.M., 2012. Enhanced sediment delivery in a changing climate in semi-arid mountain basins: implications for water resource management and aquatic habitat in the northern Rocky Mountains. Geomorphology 139-140, 1–15. http://dx.doi.org/10.1016/j.geomorph.2011.06.021.

Govers, G., Takken, I., Helming, K., 2000. Soil roughness and overland flow. Agronomie 20, 131–146.

Grismer, M.E., Hogan, M.P., 2005. Simulated rainfall evaluation of revegetation/mulch erosion control in the Lake Tahoe basin 3: soil treatment effects. Land Degrad. Dev. 16 (5), 489–501. http://dx.doi.org/10.1002/ldr.679.

Huffman, E.L., 2001. Fire-induced Soil Hydrophobicity under Ponderosa and Lodgepole Pine. Colorado Front Range. M.S. Thesis. Colorado State University, Fort Collins, CO (186 pp).

Huffman, E.L., MacDonald, L.H., Stednick, J.D., 2001. Strength and persistence of fire-induced soil hydrophobicity under ponderosa and lodgepole pine, Colorado Front Range. Hydrol. Process. 15 (15), 2877–2892. http://dx.doi.org/10.1002/hyp.379.

Jalota, S.K., Khera, R., Chahal, S.S., 2001. Straw management and tillage effects on soil water storage under field conditions. Soil Use Manag. 17 (4), 282–287.

Jordán, A., Martínez-Zavala, L., 2008. Soil loss and runoff rates on unpaved forest roads in southern Spain after simulated rainfall. For. Ecol. Manag. 255 (3), 913–919. http://dx.doi.org/10.1016/j.foreco.2007.10.002.

King, P.M., 1981. Comparison of methods for measuring severity of water repellence of sandy soils and assessment of some factors that affect its measurement. Aust. J. Soil Res. 19 (3), 275–285. http://dx.doi.org/10.1071/SR9810275.

Kittredge, J., 1948. Forest Influences: The Effects of Woody Vegetation on Climate, Water, and Soil, With Applications to the Conservation of Water and the Control of Floods and Erosion. McGraw-Hill Book Co., Inc, New York, USA (394 pp).

Larsen, I.J., MacDonald, L.H., Brown, E., Rough, D., Welsh, M.J., Pietraszek, J.H., Libohova, Z., Benavides-Solorio, J.D.D., Schaffrath, K., 2009. Causes of post-fire runoff and erosion: water repellency, cover, or soil sealing? Soil Sci. Soc. Am. J. 73 (4), 1393–1407. http://dx.doi.org/10.2136/sssaj2007.0432.

Loague, K., Gander, G.A., 1990. R-5 revisited: 1. Spatial variability of infiltration on a small rangeland catchment. Water Resour. Res. 26 (5), 957–971.

Luce, C.H., 1997. Effectiveness of road ripping in restoring infiltration capacity of forest roads. Restor. Ecol. 5 (3), 265–270.

Luce, C.H., Black, T.A., 1999. Sediment production from forest roads in western Oregon. Water Resour. Res. 35 (8), 2561–2570.

MacDonald, L.H., Anderson, D.M., Dietrich, W.E., 1997. Paradise threatened: land use and erosion on St. John, U.S. Virgin Islands. Environ. Manag. 21 (6), 851–863. http://dx.doi.org/10.1007/s002679900072.

MacDonald, L.H., Coe, D.B.R., 2008. Road sediment production and delivery: processes and management. In: Proceedings of the First World Landslide Forum. International Consortium on Landslides, Japan, pp. 385–388.

MacDonald, L.H., Huffman, E.L., 2004. Post-fire soil water repellency: persistence and soil moisture thresholds. Soil Sci. Soc. Am. J. 68, 729–1734. http://dx.doi.org/10.2136/sssaj2004.1729.

MacDonald, L.H., Stednick, J.D., 2003. Forests and Water: A State of the Art Review for Colorado. Colorado Water Resources Research Institute. Colorado State University, Fort Collins, CO. (65 pp).

Madej, M.A., 2001. Erosion and sediment delivery following removal of forest roads. Earth Surf. Process. Landf. 26, 175–190 (DOI: 10.1002/1096-9837(200102)26:2 < 175:AID-ESP174 > 3.0.CO;2-N).

Martin, D.A., Moody, J.A., 2001. Comparison of soil infiltration rates in burned and unburned mountainous watersheds. Hydrol. Process. 15, 2893–2903. http://dx.doi.org/10.1002/hyp.380.

McNabb, D.H., 1994. Tillage of compacted haul roads and landings in the boreal forests of Alberta, Canada. For. Ecol. Manag. 66 (1), 179–194.

Meadows, D., Foltz, R.B., Geehan, N., 2008. Effects of All-terrain Vehicles on Forested Lands and Grasslands. San Dimas Technology Development Center, USDA Forest

Service (124 pp).
Mohr, C.H., Coppus, R., Iroumé, A., Huber, A., Bronstert, A., 2013. Runoff generation and soil erosion processes after clear cutting. J. Geophys. Res. Earth Surf. 118 (2), 814–831.
Moore, D.C., Singer, M.J., 1990. Crust formation effects on soil erosion processes. Soil Sci. Soc. Am. J. 54 (4), 1117–1123. http://dx.doi.org/10.2136/sssaj1990.03615995005400040033x.
Moreland, D.C., 1980. Soil Survey of Larimer County Area, Colorado. USDA, Soil Conservation Service and Forest Service, CO (181 pp).
Motha, J.A., Wallbrink, P.J., Hairsine, P.B., Grayson, R.B., 2003. Determining the sources of suspended sediment in a forested catchment in southeastern Australia. Water Resour. Res. 39 (3), 1056. http://dx.doi.org/10.1029/2001WR000794.
NOAA (National Oceanic and Atmospheric Administration), 2013. Climatological Data. Red Feather Lakes, Colorado, USA. U.S. Dept. of Commerce, Asheville, N.C (ISSN 0145-0506).
NOAA (National Oceanic and Atmospheric Administration), 2016. Precipitation frequency data server. http://hdsc.nws.noaa.gov/hdsc/pfds/pfds_map_cont.html?bkmrk=co (Accessed 30 December 2016).
NOAA-NWS (National Oceanic and Atmospheric Administration – National Weather Service), 2013. Exceedance Probability Analysis for the Colorado Flood Event 9–16 September 2013. Hydrometeorological Design Studies Center, Silver Springs, MD.
Ramos-Scharrón, C.E., LaFevor, M.C., 2016. The role of unpaved roads as active source areas of precipitation excess in small watersheds drained by ephemeral streams in the northeastern Caribbean. J. Hydrol. 53, 168–179.
Ramos-Scharrón, C.E., MacDonald, L.H., 2005. Measurement and prediction of sediment production from unpaved roads, St John, US Virgin Islands. Earth Surf. Process. Landf. 30 (10), 1283–1304. http://dx.doi.org/10.1002/esp.1201.
Reid, L.M., Dunne, T., 1984. Sediment production from forest road surfaces. Water Resour. Res. 20 (11), 1753–1761.
Ries, J.B., Iserloh, T., Seeger, M., Gabriels, D., 2013. Rainfall simulations - constraints, needs and challenges for a future use in soil erosion research. Z. Geomorphol. 57, 1–10. http://dx.doi.org/10.1127/0372-8854/2013/s-00130.
Robichaud, P.R., 2000. Fire effects on infiltration rates after prescribed fire in northern Rocky Mountain forests, USA. J. Hydrol. 231-232, 220–229. http://dx.doi.org/10.1016/s0022-1694(00)00196-7.
SAS Institute Inc, 2002–2010. SAS 9.3 Online Documentation. Samples and SAS Notes. SAS Institute Inc, Cary, NC.
Schaffer, R., 2003. National Forest Service Road Decommissioning: An Attempt to Read Through the Numbers. Wildlands CPR (20 p).
Schmeer, S.R., 2014. Post-fire Erosion Response and Recovery, High Park Fire, Colorado. M.S. Thesis. Colorado State University, Fort Collins, CO (209 pp).
Shakesby, R.A., Doerr, S.H., Walsh, R.P.D., 2000. The erosional impact of soil hydrophobicity: current problems and future research directions. J. Hydrol. 231–232, 178–191. http://dx.doi.org/10.1016/S0022-1694(00)00193-1.
Sheridan, G.J., Noske, P.J., Lane, P.N.J., Sherwin, C.B., 2008. Using rainfall simulation and site measurements to predict annual interrill erodibility and phosphorus generation rates from unsealed forest roads: validation against in-situ erosion measurements. Catena 73 (1), 49–62. http://dx.doi.org/10.1016/j.catena.2007.08.006.
Sheridan, G.J., Noske, P.J., Whipp, R.K., Wijesinghe, N., 2006. The effect of truck traffic and road water content on sediment delivery from unpaved forest roads. Hydrol. Process. 20 (8), 1683–1699. http://dx.doi.org/10.1002/hyp.5966.
Sosa-Pérez, G., MacDonald, L.H., 2017. Reductions in road sediment production and road-stream connectivity from two decommissioning treatments. For. Ecol. Manag. 398, 116–129.
SSSA, 2001. Glossary of Soil Science Terms. Soil Science Society of America, Madison, WI. https://www.soils.org/publications/soils-glossary# (web archive link, 06 October 2015) (Accessed 6 October 2015).
Switalski, T.A., Bissonette, J.A., DeLuca, T.H., Luce, C.H., Madej, M.A., 2004. Benefits and impacts of road removal. Front. Ecol. Environ. 2 (1), 21–28 (DOI: 10.1890/1540-9295(2004)002[0021:BAIORR]2.0.CO;2).
Thompson, A.L., James, L.G., 1985. Water droplet impact and its effect on infiltration. Trans. ASAE 28, 1506.
Topp, G.C., Ferre, P.A., 2002. The soil solution phase. In: Jacob, H.D., Topp, G.C. (Eds.), Methods of Soil Analysis: Part 4 - Physical Methods. Soil Science Society of America, Inc, Madison, Wisconsin, pp. 417–545.
USDA Forest Service, 2010–2014. National Forest System Statistics FY2010–2014. USDA Forest Service, Washington, D.C. Available at: http://www.fs.fed.us/publications/ (web archive link, 10 September 2015) (Accessed 10 September 2015).
USDA Natural Resources Conservation Service, 1998. Official soil series description Available at: https://soilseries.sc.egov.usda.gov/OSD_Docs/R/REDFEATHER.html (Accessed 30 December 2016).
van Meerveld, H.J., Baird, E.J., Floyd, W.C., 2014. Controls on sediment production from an unpaved resource road in a Pacific maritime watershed. Water Resour. Res. 50 (6), 4803–4820. http://dx.doi.org/10.1002/2013WR014605.
van Wesemael, B., Poesen, J., de Figueiredo, T., 1995. Effects of rock fragments on physical degradation of cultivated soils by rainfall. Soil Tillage Res. 33 (3–4), 229–250. http://dx.doi.org/10.1016/0167-1987(94)00439-L.
Veblen, T.T., Donnegan, J.A., 2005. Historical Range of Variability for Forest Vegetation of the National Forests of the Colorado Front Range. Final report, USDA Forest Service Agreement no. 1102-0001-99-033. USDA Forest Service, Golden, CO (153 pp).
Wagenbrenner, J.W., MacDonald, L.H., Coats, R.N., Robichaud, P.R., Brown, R.E., 2015. Effects of post-fire salvage logging and a skid trail treatment on ground cover, soils, and sediment production in the interior western United States. For. Ecol. Manag. 335, 176–193. http://dx.doi.org/10.1016/j.foreco.2014.09.016.
Walling, D.E., Webb, B.W., 1982. Sediment availability and the prediction of storm-period sediment yields. In: Recent Developments in the Explanation and Prediction of Erosion and Sediment Yields, IAHS Publ. No. 137, pp. 327–337.
Watson, C.L., Letey, J., 1970. Indices for characterizing soil-water repellency based upon contact angle-surface tension relationships. Soil Sci. Soc. Am. J. 34 (6), 841–844. http://dx.doi.org/10.2136/sssaj1970.03615995003400060011x.
Weaver, W.E., Weppner, E.M., Hagans, D.K., 2015. Handbook for Forest, Ranch and Rural Roads: A Guide for Planning, Designing, Constructing, Reconstructing, Upgrading, Maintaining and Closing Wildland Roads, Mendocino County Resource Conservation District, Ukiah, California.
Welsh, M.J., 2008. Sediment Production and Delivery From Forest Roads and Off-highway Vehicle Trails in the Upper South Platte River Watershed, Colorado. M.S. thesis. Colorado State University, Fort Collins, CO (227 pp).
Wood, P.J., Armitage, P.D., 1997. Biological effects of fine sediment in the lotic environment. Environ. Manag. 21 (2), 203–217.
WRCC (Western Regional Climate Center), 2016. Monthly climate summary for Red Feather Lakes, Colorado. Available at. http://www.raws.dri.edu/cgi-bin/rawMAIN.pl?coCREF (web archive link, 25 May 2016) (Accessed 25 May 2016).
Ziegler, A.D., Giambelluca, T.W., 1997. Importance of rural roads as source areas for runoff in mountainous areas of northern Thailand. J. Hydrol. 196, 204–229. http://dx.doi.org/10.1016/S0022-1694(96)03288-X.
Ziegler, A.D., Negishi, J.N., Sidle, R.C., Gomi, T., Noguchi, S., Nik, A.R., 2007. Persistence of road runoff generation in a logged catchment in Peninsular Malaysia. Earth Surf. Process. Landf. 32 (13), 1947–1970. http://dx.doi.org/10.1002/esp.1508.
Ziegler, A.D., Sutherland, R.A., Giambelluca, T.W., 2000. Runoff generation and sediment production on unpaved roads, footpaths and agricultural land surfaces in northern Thailand. Earth Surf. Process. Landf. 25 (5), 519–534.
Ziegler, A.D., Sutherland, R.A., Giambelluca, T.W., 2001. Interstorm surface preparation and sediment detachment by vehicle traffic on unpaved mountain roads. Earth Surf. Process. Landf. 26 (3), 235–250.