Causes of Post-Fire Runoff and Erosion: Water Repellency, Cover, or Soil Sealing?

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Few studies have attempted to isolate the various factors that may cause the observed increases in peak flows and erosion after high-severity wildfires. This study evaluated the effects of burning by: (i) comparing soil water repellency, surface cover, and sediment yields from severely burned hillslopes, unburned hillslopes, and hillslopes where the surface cover was removed by raking; and (ii) conducting rainfall simulations to compare runoff, erosion, and surface sealing from two soils with varying ash cover. The fire-enhanced soil water repellency was only stronger on the burned hillslopes than the unburned hillslopes in the first summer after burning. For the first 5 yr after burning, the mean sediment yield from the burned hillslopes was 32 Mg ha^{-1}, whereas the unburned hillslopes generated almost no sediment. Sediment yields from the raked and burned hillslopes were indistinguishable when they had comparable surface cover, rainfall erosivity, and soil water repellency values. The rainfall simulations on ash-covered plots generated only 21 to 49% as much runoff and 42 to 67% as much sediment as the plots with no ash cover. Soil thin sections showed that the bare plots rapidly developed a structural soil seal. Successive simulations quickly eroded the ash cover and increased runoff and sediment yields to the levels observed from the bare plots. The results indicate that: (i) post-fire sediment yields were primarily due to the loss of surface cover rather than fire-enhanced soil water repellency; (ii) surface cover is important because it inhibits soil sealing; and (iii) ash temporarily prevents soil sealing and reduces post-fire runoff and sediment yields.

Abbreviations: CEC, cation exchange capacity; CST, critical surface tension.

Wildfires increase hillslope- and watershed-scale runoff and sediment yields by several orders of magnitude (e.g., Prosser and Williams, 1998; Robichaud and Brown, 1999; Moody and Martin, 2001; Benavides-Solorio and MacDonald, 2005; Malmon et al., 2007). Land use and climate change have increased, or are projected to increase, the size and frequency of fires in many wildland environments (e.g., Mouillot et al., 2002; Hennessy et al., 2005; Westerling et al., 2006). The increase in fire risk is generating considerable concern about the potential adverse effects on water quality, aquatic habitat, and water supply systems (Rinne, 1996; Robichaud et al., 2000; Moody and Martin, 2001; Burton, 2005).

The large increases in runoff and sediment yields after high-severity fires have been attributed to several factors, including: (i) soil water repellency (DeBano, 2000; Doerr et al., 2000); (ii) loss of surface cover (Johansen et al., 2001; Pannkuk and Robichaud, 2003); (iii) soil sealing by sediment particles (Lowdermilk, 1930; Neary et al., 1999); and (iv) soil sealing by ash particles (Mallik et al., 1984; Etiégni and Campbell, 1991). The problem is that the relative contribution of each factor to the observed increases in post-fire runoff and sediment yields is largely unknown (Shakesby et al., 2000; Letey, 2001). This lack of understanding hampers our ability to predict post-fire sediment yields and design effective post-fire rehabilitation treatments.

Burning has been shown to induce or enhance soil water repellency (hydrophobicity) in a variety of shrub and forest ecosystems (Doerr et al., 2009), and this increase in soil water repellency has been commonly cited as a primary cause of the observed post-fire increases in peak flows and sediment yields (e.g., Krammes and Osborn, 1969; DeBano, 1981, 2000; Robichaud, 2000; Shakesby et al., 2000). High-severity fires also alter the

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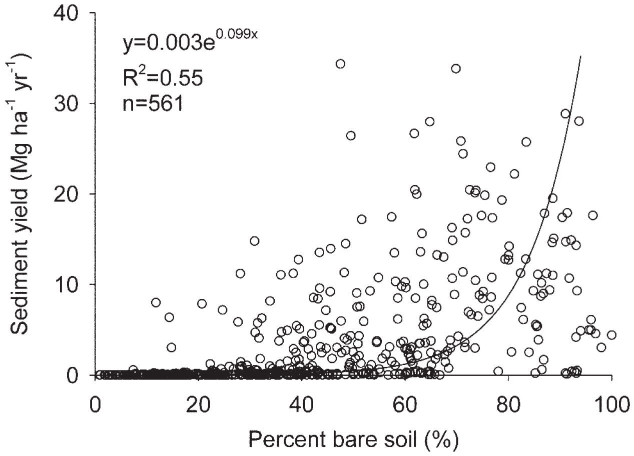

Fig. 1. Bare soil percentage vs. sediment yield for 10 wild and prescribed fires in the Colorado Front Range. Each data point is an annual sediment yield from a burned hillslope.

vegetative cover and characteristics of the mineral soil, making it very difficult to separate the effects of fire-induced soil water repellency from other changes in soil characteristics and surface cover. Doerr and Moody (2004) explicitly stated that the linkage between soil water repellency and post-fire runoff and erosion rates has rarely been demonstrated.

Our previous work in the Colorado Front Range has shown that post-fire soil water repellency breaks down within 1 to 2 yr after burning (MacDonald and Huffman, 2004). For the same fires, 3 to 5 yr may be required before hillslope-scale sediment yields return to background levels (Benavides-Solorio and MacDonald, 2005; Pietraszek, 2006; Wagenbrenner et al., 2006). This discrepancy in time implies that fire-induced soil water repellency is not the primary cause of the observed increases in runoff and erosion after high-severity wildfires. The role of soil water repellency is also called into question because although in nearby unburned areas the mineral soil surface is strongly water repellent (Huffman et al., 2001), overland flow and surface erosion (defined here as rain splash, sheetwash, and rilling) rarely occur on unburned hillslopes in the Colorado Front Range (Gary, 1975; Morris and Moses, 1987; Moody and Martin, 2001; Libohova, 2004).

The amount of surface cover is an important control on infiltration, runoff, and erosion in both burned and unburned areas (e.g., Walsh and Voigt, 1977; Brock and DeBano, 1982; Renard et al., 1997; Gyssels et al., 2005). In unburned areas, surface cover increases infiltration, decreases runoff, and decreases erosion by several mechanisms. These include rainfall interception (Kittredge, 1948; Clary and Ffolliott, 1969), maintaining high porosity by increasing soil organic matter and facilitating biological activity (DeBano et al., 2005), preventing soil sealing (Morin et al., 1989; Moss and Watson, 1991), and increasing surface roughness (Lavee et al., 1995; Pannkuk and Robichaud, 2003). In general, a decrease in surface cover causes a nonlinear increase in sediment yields at scales ranging from 1 m^2 to 100 km^2 (Brock and DeBano, 1982; Johansen et al., 2001; Gyssels et al., 2005; Vanacker et al., 2007).

Our empirical data from the Colorado Front Range show that the surface cover percentage (or conversely the bare soil percentage) is the dominant control on post-fire sediment yields (Fig. 1), followed by rainfall erosivity (Benavides-Solorio and MacDonald, 2005; Pietraszek, 2006). The problem is that these empirical relationships do not explain the underlying causal process(es). In severely burned areas, the loss of surface cover is greatly confounded by the increase in soil water repellency noted above, a decrease in soil organic matter and an associated reduction in aggregate stability (Giovannini and Lucchesi, 1983; Soto et al., 1991; Badí and Martí, 2003), and the presence of ash, which has been reported to induce soil sealing and inhibit the infiltration of runoff (e.g., Mallik et al., 1984; Durgin, 1985; Gabet and Sternberg, 2008).

Soil sealing refers to the development of a thin (0.1–1.0 mm), dense soil layer at the mineral soil surface. The hydraulic conductivity of this seal or crust can be several orders of magnitude lower than that of the underlying soil (McIntyre, 1958; Terry and Shakesby, 1993; Shainberg and Levy, 1996; Assouline and Mualem, 2000; Assouline, 2004). Soil seals are categorized as either structural seals, which form due to raindrop impact and rapid wetting, or depositional seals, which form due to the settling of fine particles carried by runoff (Assouline, 2004). The processes that contribute to the formation of structural soil seals include: (i) the destruction of soil aggregates by raindrop impact and slaking; (ii) soil compaction and realignment of surface particles by raindrops; and (iii) pore clogging by the physical movement of fine particles or the chemical dispersion of clays (Assouline, 2004).

Structural soil seals are an important control on runoff rates in agricultural areas (e.g., Radcliffe et al., 1991; Bajracharya and Lal, 1998; Assouline, 2004), and it has been suggested that high-severity fires can increase the likelihood of soil sealing by several processes. These include the afore-mentioned reduction in soil organic matter and aggregate stability (Giovannini and Lucchesi, 1983; Soto et al., 1991; Badí and Martí, 2003) and the dispersion of clays (Durgin and Vogelsang, 1984; Durgin, 1985; Mills and Fey, 2004). Several studies also have proposed that ash can induce soil sealing by clogging pores (Mallik et al., 1984; Onda et al., 2008) and by swelling within pores when wetted (Etiégni and Campbell, 1991). Other studies have shown, however, that an ash layer adsorbs rainfall and inhibits both runoff and sediment production (Cerdà, 1998a; Martin and Moody, 2001; Cerdà and Doerr, 2008; Woods and Balfour, 2008).

The overall goal of this study is to better understand the relative effects of soil water repellency, loss of surface cover, and soil sealing on post-fire sediment yields by a combination of field studies and rainfall simulation experiments. The first set of field studies (the burning experiment) compared soil water repellency, surface cover, and sediment yields from hillslopes burned by a high-severity wildfire to nearby unburned hillslopes that had no recent signs of logging or other disturbance. Given the observed strong relationship between surface cover and sediment yields from different fires (Fig. 1), a second field study repeatedly removed 80% of the surface cover from three unburned hillslopes by raking (the raking experiment). The sediment yields from the raked hillslopes and their paired controls were compared with the values from the burned and unburned hillslopes, respectively.

The results of these field experiments led to the third, laboratory-based study, which consisted of two sets of rainfall simulation experiments. The first set of rainfall simulations compared runoff and sediment yields from two different soils and the same soils with both a thin and a thick ash cover (single simulations).

Thin sections made from soil cores collected after these simulations were used to determine whether any differences between these treatments could be attributed to soil sealing. The second set of rainfall simulation experiments compared runoff and sediment yields from three successive simulated rainstorms on each bare soil and the same soils with an initial ash cover (successive simulations). The goal of these simulations was to determine how runoff and erosion rates would change as a result of successive storms with and without an ash cover.

Taken together, these experiments provide unique empirical and process-based insights into the causes of the observed large increases in surface runoff and sediment yields after forest fires. The results have important implications for predicting runoff and erosion after land management activities and wildfires, and for developing effective post-fire rehabilitation techniques.

METHODS

Field Experiments

The field experiments were conducted in the Colorado Front Range, approximately 60 km southwest of Denver (Fig. 2). The dominant vegetation is ponderosa pine (*Pinus ponderosa* P. Lawson & C. Lawson) with some Douglas-fir [*Pseudotsuga menziesii* (Mirb.) Franco] at higher elevations and on north-facing slopes. The annual precipitation is about 400 mm, and winter precipitation is primarily snow, whereas summer (1 June–31 October) precipitation is dominated by high-intensity thunderstorms (Gary, 1975) that generate nearly all the post-fire sediment yield (Pietraszek, 2006; Rough, 2007). The dominant bedrock is Pikes Peak granite, and the resultant soils are very coarse textured and highly erodible. The dominant soil in the burned, raked, and unburned areas is the Sphinx series, which is a sandy-skeletal, mixed, frigid, shallow Typic Ustorthent (U.S. Forest Service, 1992).

In the summer of 2001, 20 pairs of convergent, zero-order hillslopes (swales) were identified and began to be monitored as part of an experiment to determine the effects of mechanical forest thinning on surface cover and sediment production. These hillslopes had no evidence of recent timber harvest, surface erosion, or overland flow, and sediment fences were built to measure sediment yields (Fig. 3a) (Robichaud and Brown, 2002). In the summer of 2002, many of these hillslopes were burned at high severity in the Hayman wildfire (Fig. 3b) (Pietraszek, 2006). The sediment fences were reconstructed within a few weeks after burning, and several additional hillslopes were added at the adjacent 2002 Schoonover wildfire to yield a set of 21 hillslopes that had burned at high severity and were not subjected to any post-fire rehabilitation treatments. Additional paired hillslopes were subsequently established in adjacent unburned areas to replace the sites that

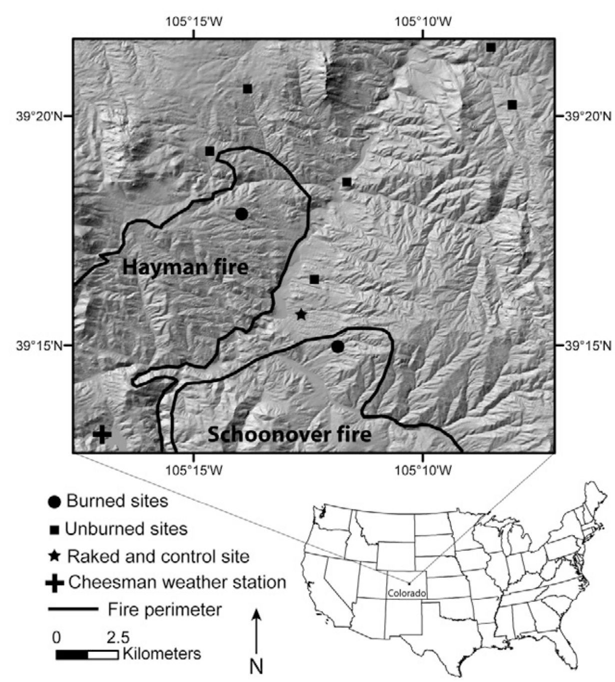

Fig. 2. Map of the study area, the northern perimeters of the 2002 Hayman and Schoonover wildfires, the unburned, burned, and raked study sites, and the long-term weather station at Cheesman Reservoir.

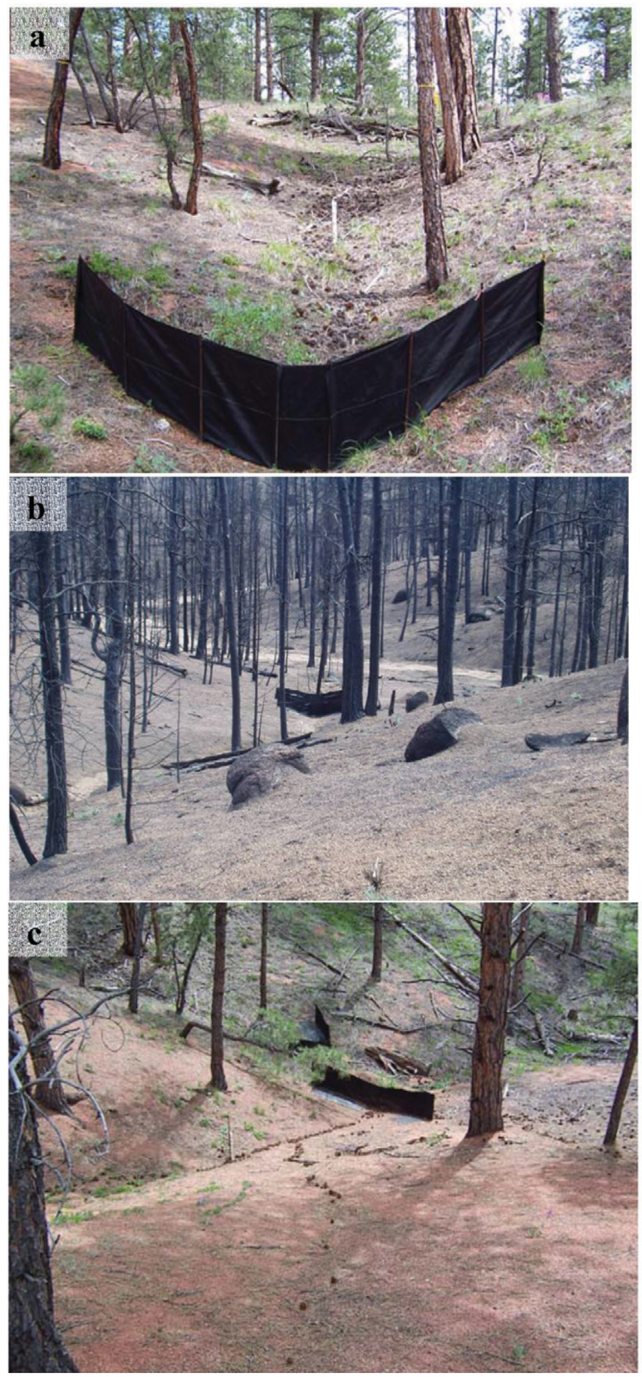

Fig. 3. Photographs of a typical (a) unburned, (b) burned, and (c) raked hillslope in the field study. The photo of the raked swale was taken several months after raking and hence some needlefall had occurred.

had burned, and in most of these pairs one hillslope was mechanically thinned between fall 2002 and fall 2005. Hence there were from 13 to 34 unburned and undisturbed hillslopes being monitored between 2002 and 2006 as part of the burning experiment (Table 1).

The raking experiment used three pairs of unburned hillslopes that were originally to be part of the thinning study but were never disturbed. One hillslope in each pair was randomly selected and left as an untreated control while the other hillslope was annually raked just before the summer thunderstorm season to remove the surface litter and any shallow-rooted vegetation (Table 1). The goal of the raking was to separate the single effect of reducing the surface cover from the multiple effects of burning on sediment yields. The three treated hillslopes were first raked in July 2004, and the raking was repeated at the beginning of the summers of 2005 and 2006 to remove the litter and any new vegetation that had accumulated since the last raking. After raking there was no surface cover except a small amount of fine humus, sparse but securely rooted understory vegetation, a few rocks, and the scattered ponderosa pine trees (Fig. 3c).

All of the sediment generated from the hillslopes was manually removed from the sediment fences and weighed on a storm-by-storm basis. The values from individual storms were corrected for moisture content, normalized by the contributing area, and summed to obtain an annual sediment yield. The total number of hillslope-years of sediment yield was 105 for the burned hillslopes, 105 for the unburned hillslopes, nine for the raked hillslopes, and nine for the paired, untreated controls for the raked hillslopes (Table 1).

Soil water repellency was measured using the critical surface tension (CST) method (Letey, 1969; Doerr, 1998). The CST for non-water-repellent soils at 20°C is 0.0728 N m^{-1}, and CST values decline as the strength of the soil water repellency increases. The mineral soil was exposed by sweeping away the litter in the unburned sites and the ash in the burned sites, and the CST was then measured at depths of 0, 3, and 6 cm at three randomly located sites in the upper, central, and lower portions of each hillslope. Soil water repellency was measured in 16 unburned hillslopes in 2002 (Libohova, 2004) and annually in each of the burned hillslopes from 2002 to 2004 (Rough, 2007). In 2006, soil water repellency was measured in the three raked hillslopes and their corresponding controls as well as in three of the burned hillslopes. All measurements were made during dry periods in the summer to minimize the effect of any differences in soil moisture.

Since the soil water repellency values were very consistent between the unburned hillslopes in 2002 and the three unburned control hillslopes for the raking study in 2006, we compared the soil water repellency values from the unburned hillslopes in 2002 to the values from the burned hillslopes in 2002 to 2004 and 2006 using ANOVA ($P \leq 0.05$). An ANOVA with a post-hoc Fisher's LSD test was used to compare soil water repellency among the burned hillslopes in the summer of 2003 and both the raked and the control hillslopes in the summer of 2006.

The surface cover in each swale was measured at a minimum of 100 points along multiple transects (Parker, 1951). At each point, the surface cover was classified as bare soil, live vegetation, litter, rock, ash, or wood. Measurements for the burned hillslopes were made at the beginning and end of each summer, and these two values were averaged to obtain an annual value. The surface cover was measured once each summer for the unburned hillslopes and immediately after each raking for the raked hillslopes.

Rainfall was measured from 1 June to 31 October adjacent to the hillslopes with tipping-bucket rain gauges, which had a resolution of 0.20 to 0.25 mm. The maximum 30-min rainfall intensity (I_{30}) was calculated for each summer storm, as the I_{30} is used to calculate both storm and total summer erosivity (Brown and Foster, 1987) and the I_{30} and maximum 10-min rainfall intensity (I_{10}) had nearly identical explanatory power for predicting storm-based sediment yields at our study sites. Storms were defined as periods of rainfall separated by at least 1 h with no rainfall. The erosivity of each storm with at least 5 mm of rainfall was calculated following Brown and Foster (1987), and these were summed to obtain a summer erosivity for each rain gauge. Mean summer rainfall and mean summer erosivity for the burned, unburned, and raked hillslopes were calculated by weighting the values from each rain gauge by the number of hillslopes represented by that rain gauge. The summer rainfall data from our study sites were compared with the long-term means from the Cheesman weather station, as this is the nearest weather station and is at a similar elevation only 7 to 9 km south and slightly west of our burned study sites (Fig. 2). The period of 1 June to 31 October accounts for >90% of the annual rainfall erosivity (Renard et al., 1997) and nearly 100% of the post-fire sediment yields in the Colorado Front Range (Benavides-Solorio and MacDonald, 2005; Pietraszek, 2006).

The effect of soil water repellency on sediment yields was assessed for the burned and unburned hillslopes by comparing the mean annual soil water repellency at each depth against the mean annual sediment yields. The 2004 sediment yield data from the raked swales were excluded because the raking did not occur until early July and there were no subsequent summer storms with >12 mm of rainfall. This meant that no sediment was generated from the raked swales, and the much higher rainfall on the burned hillslopes in the summer of 2004 precluded a meaningful comparison between the two groups.

In contrast, the mean annual rainfall erosivity for the raked hillslopes in 2005 and 2006 was nearly identical to the mean value measured for the burned hillslopes in 2003, and 2003 is also when the burned hillslopes had nearly the identical amount of surface cover as the raked hillslopes. Given the variations in annual erosivity and the progressive increase in surface cover on the burned hillslopes, the most valid comparison of the effects of burning vs. raking is between the 2003 sediment yield data from the burned hillslopes (n = 21) and the mean annual sediment yields from 2005 and 2006 for the raked (n = 6) and control hillslopes (n = 6). Significant differences in sediment yields among these three groups were tested using Fisher's LSD. The mean sediment

Table 1. Number of hillslopes per year for each treatment, mean contributing area, mean slope, and years with soil water repellency and sediment yield data for each treatment. Values after the ± symbol are standard deviations.

Treatment	Hillslopes	Mean contributing area	Mean slope	Years with soil water repellency data	Years with sediment yield data
	no. yr^{-1}	m^2	%		
Burned	21	1970 ± 1450	27 ± 7	2002–2004, 2006	2002–2006
Unburned	13–34	1700 ± 1500	26 ± 12	2002	2002–2006
Raked	3	790 ± 300	26 ± 3	2006	2004–2006
Controls for raked hillslopes	3	750 ± 390	32 ± 3	2002, 2006	2004–2006

yields from each storm for the burned, raked, and untreated hillslopes also were plotted against I_{30} to determine whether there were any differences in the threshold for sediment generation and the amount of sediment produced for a given I_{30}.

Rainfall Simulation Experiments

Each rainfall simulation experiment was conducted on both a granitic soil and a micaceous soil derived from schist and gneissic bedrock. These two soils are representative of the soils in the mid-elevation forests in the Colorado Front Range that are most prone to high-severity wildfires and most likely to be subjected to prescribed burns (Romme et al., 2003; Benavides-Solorio and MacDonald, 2005). The single simulation experiment used three different treatments on each soil with three replicates of each treatment for a total of 18 simulations. The three treatments were bare soil, a low-ash treatment of 2.9 kg m^{-2} or ~5-mm depth, and a high-ash treatment of 6.3 kg m^{-2} or ~12-mm depth. The second rainfall simulation experiment used one replicate of three successive rainfall simulations on each bare soil, and the same successive simulations on one replicate of the low-ash treatment for each soil type for a total of 12 simulations.

The rainfall simulations were conducted in the lab using a metal box that was 50 cm long, 30 cm wide, 20 cm deep, and filled with 15 cm of soil. The slope of the box was set to 25%, as this was comparable to the mean slope of our hillslopes (Table 1). A trough at the lower end of the box collected runoff and sediment but did not capture the ash and sediment lost over the sides by rain splash erosion. Rainfall was applied to the soil with a Purdue-type rainfall simulator (Neibling et al., 1981; Foster et al., 1982) at 40 mm h^{-1} for 45 min. This intensity and duration was selected to ensure near steady-state runoff by the end of the simulation and because this is roughly the same intensity and magnitude as the largest storms observed during the field experiment (Pietraszek, 2006). Such storms have an estimated recurrence interval of 1.5 to 2 yr based on CLIGEN (Nicks et al., 1995) data for the Cheesman weather station. The rainfall intensity was periodically checked by raining into the metal box without any soil, and three rain gauges were placed around the box during each simulation. The standard deviations of the rainfall measured in each of the three gauges were only 1.1 to 2.6 mm for all of the simulations. The water used in the simulations was obtained directly from Horsetooth Reservoir, primarily low-ionic-strength snowmelt runoff with a pH of 6.8 to 7.3 and an electrical conductivity of 4 to 8 mS m^{-1} (USGS, 2008).

The granitic soil was collected after the 2002 Hayman wildfire (Fig. 2) and the micacous soil was taken from the 2003 Dadd Bennett prescribed fire approximately 60 km west of Fort Collins (for details, see Pietraszek, 2006). In both cases, the soils were taken from areas that had burned at high severity. Particles >9.5 mm were removed by sieving. Both soils were coarse grained, as determined by sieving and the hydrometer method, and had 3 to 4% organic matter as determined by loss-on-ignition (Table 2). Both soils had a low cation exchange capacity (CEC) (Table 2). Ten bulk density samples from 0- to 10.5-cm depth were collected for each soil; the mean (± one standard deviation) bulk density was 1.25 ± 0.11 g cm^{-3} for the granitic soil and 1.14 ± 0.17 g cm^{-3} for the micaceous soil. Filling the box with soil resulted in a bulk density that was about 15% higher than the field values, and this was presumably due to the loss of roots and macropores.

The ash was a mixture of black and white ash collected from a ponderosa pine forest 3 wk after a high-severity wildfire and before any runoff-generating storms. The mean ash depth was 12 mm, which is a relatively typical value given the data reviewed by Cerdà and Doerr (2008). The mean mass of 6.3 ± 1.7 kg m^{-2} was determined by collecting and weighing 10 samples, where each sample was collected from an area that was 0.25 by 0.25 m. A mean mass of 6.3 kg m^{-2} was used for the high-ash treatment, and the mass of ash for the low-ash treatment was 2.9 kg m^{-2}, or two standard deviations below the mean value measured in the field. Scanning electron microscopy analysis indicated the ash was composed of approximately 32% C, 34% O, 13% Si, and 12.5% Ca. Since the predominant cation was Ca, the pH of the ash in deionized water was around 7.5 to 8. The CEC of the ash was 74 cmol kg^{-1} (Table 2), and the integrated Munsell color was N3 or dark gray.

The time to the beginning of runoff from the trough was recorded for each simulation, and the runoff was collected in 1-L bottles for 30 s of each minute during the simulations. The measured volumes were used to calculate the mean runoff rate during the last 5 min of rainfall application (the final runoff rate), and the cumulative mass of runoff was converted to a volume and used to calculate the percentage of the rainfall that was converted to runoff (the runoff coefficient). Filtration of the runoff through a 5-μm filter indicated that nearly all of the sediment was trapped in the trough, so the total sediment yield for each simulation was the dry mass of sediment and ash collected from the trough after each simulation. Fisher's LSD was used to test for differences among treatments in the time to runoff, runoff coefficient, final runoff rate, and sediment yield.

The formation of a surface seal was evaluated from one soil core for each treatment on each soil (n = 3 per soil). The cores were 5 cm in diameter and 3 cm long and were collected after the single simulations. Each core was impregnated with epoxy and one thin section was prepared perpendicular to the soil surface. The seal formation in each thin section was classified following Valentin and Bresson (1992) using optical microscopy in plane-polarized light.

The successive simulations on the granitic soil were conducted 5 d apart, and the surface cover was classified as ash or bare soil at 104 points after each simulation. There was 1 d between the first two simulations on the micaceous soil and 7 d between the second and third simulations. Surface cover and sediment yields were not measured for the consecutive simulations on the micaceous soil.

Table 2. Texture, organic matter content, and cation exchange capacity data for the two soils and the ash used in the rainfall simulation experiments. Texture and organic matter for the soils are averages based on three measurements ± one standard deviation. The ash data and cation exchange capacity are based on one measurement from a composite sample.

Soil property	Soil type		
	Granitic	Micaceous	Ash
Coarse fraction (>2 and <9.5 mm)	51 ± 2	42 ± 0.9	–
Organic matter, %	3.3 ± 0.3	3.2 ± 0.2	14.7
Fine fraction (<2 mm)			
Sand, %	64 ± 1	73 ± 2	76
Silt, %	26 ± 1	22 ± 3	13
Clay, %	9 ± 1	6 ± 1	11
Cation exchange capacity, cmol kg^{-1}	13.9	9.8	74.3
Exchangeable bases, cmol kg^{-1} in water extract			
Ca	1.5	1.9	5.8
Mg	0.4	0.6	5.2
Na	0.2	0.2	0.7
K	0.1	0.1	1.1

RESULTS
Field Component
Soil Water Repellency

Shortly after burning, the soils were strongly water repellent at 0- and 3-cm depths and moderately repellent at 6 cm (Fig. 4a). Soil water repellency declined very rapidly with time at the soil surface and at 6 cm. The CST values at 3 cm declined more slowly, but by the summer of 2004, the soil water repellency was weak or negligible at all depths (Fig. 4a).

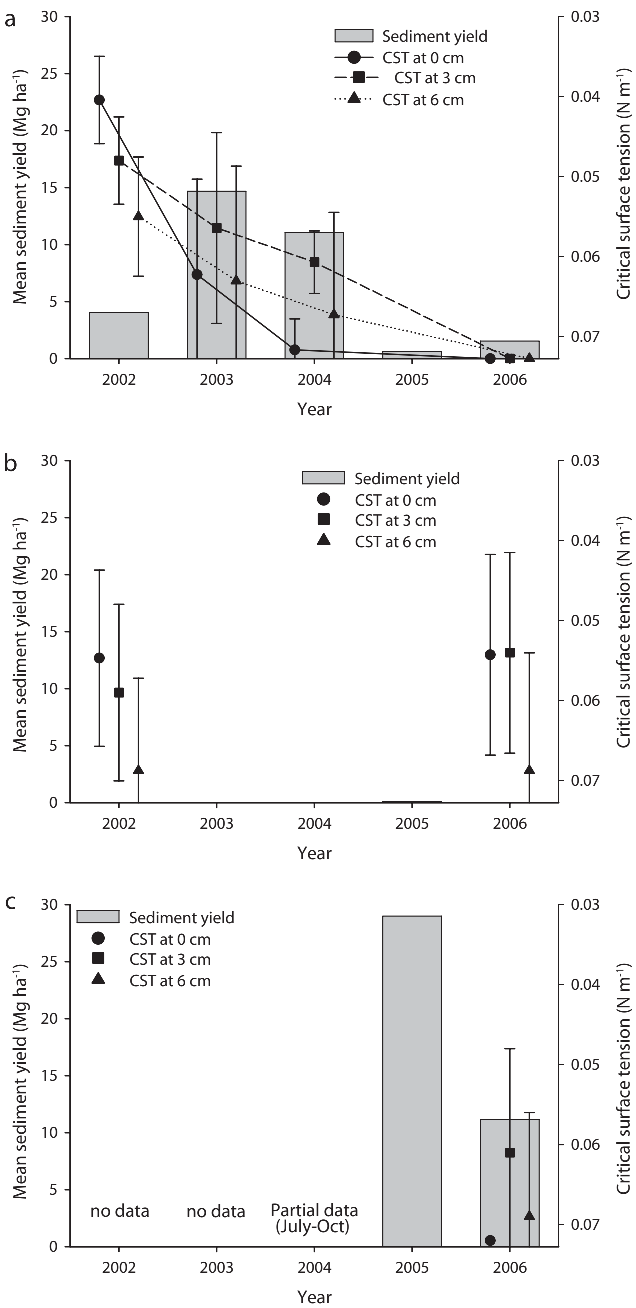

Fig. 4. Mean critical surface tension (CST) values at 0-, 3-, and 6-cm depths and mean sediment yields for (a) burned hillslopes, (b) unburned hillslopes, and (c) raked hillslopes from 2002 to 2006. Lower CST values indicate stronger soil water repellency, symbols represent the mean, and the bars indicate one standard deviation. No soil water repellency data were collected in 2005 for the burned hillslopes, or in 2003 to 2005 for the unburned hillslopes. The raking experiment was initiated in July 2004.

On the unburned hillslopes, the soils were water repellent at the surface and at 3-cm depth, but at 6-cm depth there was very little water repellency (Fig. 4b). The soil water repellency values from the unburned hillslopes were very consistent between 2002 and 2006 (Fig. 4b), supporting the assumption that variations in soil moisture were small and had little effect on the summer CST values.

In the summer of 2002, which was shortly after burning, the soil water repellency at all depths in the burned hillslopes was significantly stronger than at the same depths in the unburned hillslopes. In the summer of 2003, the burned hillslopes had significantly weaker soil water repellency at the soil surface than the unburned hillslopes. At a depth of 3 cm, there was no significant difference in soil water repellency between the burned and unburned hillslopes, while at 6 cm the soil water repellency was significantly stronger on the burned hillslopes. The continued decline in soil water repellency on the burned hillslopes meant that by 2004 the only significant difference between the burned and unburned hillslopes was the stronger soil water repellency at the soil surface of the unburned hillslopes.

In 2006, the raked hillslopes had no water repellency at the mineral soil surface, and this was probably due to the disturbance from raking plus some erosion of the finer, water-repellent soil particles. Like the other unburned hillslopes, the soil water repellency was relatively weak at 3 cm (0.061 ± 0.013 N m^{-1}) and largely absent at 6 cm (0.069 ± 0.012 N m^{-1}) (Fig. 4). The unraked control hillslopes had significantly stronger soil water repellency at the mineral soil surface (0.054 ± 0.013 N m^{-1}) than the raked hillslopes, and similar values of soil water repellency at depths of 3 and 6 cm.

Surface Cover

Shortly after burning, >90% of the surface on the burned hillslopes was characterized as either bare soil (47 ± 11%) or ash (54 ± 12%) (Fig. 3b and 5). Vegetative regrowth on the burned hillslopes caused the cover percentage to increase with time, and the mean total surface cover for each of the first 5 yr after burning was 5, 18, 40, 53, and 58%, respectively (Fig. 5). The ash cover was rapidly removed by overland flow and wind, as it dropped from 54% in the summer of 2002 to 34 ± 14% by the spring of 2003, and to just 5 ± 4% by the end of the summer in 2003 (Fig. 5). Since the ash was 85% mineral matter and the sediment fences trapped a large amount of ash following the first several post-fire storms, the observed loss of ash had to be due to erosion rather than dissolution.

The mean surface cover on the unburned hillslopes was 81 ± 11%, and this consisted primarily of litter (70 ± 8%), with much smaller amounts of live vegetation (11 ± 5%) (Libohova, 2004). As with soil water repellency, there was little variation in the amount and type of surface cover among years.

The raking treatments in 2005 and 2006 reduced the mean surface cover to just 17 ± 4%. A comparison of this value to Fig. 5 shows that only in the summer of 2003 did the burned hillslopes have nearly the identical amount of surface cover (18%) as the raked hillslopes.

Rainfall

Summer precipitation and erosivity were generally similar on the unburned and burned hillslopes (Table 3). In 2002, only one storm generated substantial amounts of sediment after the Hayman and Schoonover fires (one storm of 16 mm occurred on the Hayman Fire site before the sediment fences were reinstalled). The mean summer erosivity for the burned and unburned sites in 2002 was only 72 to 95 MJ mm ha^{-1} h^{-1} (Table 3), which is only about one-fourth of the long-term mean of 364 MJ mm ha^{-1} h^{-1} (Foster, 2004).

From 2003 to 2006, there were generally three to six sediment-generating storms each summer. The summers of 2003 and 2005 were quite dry, as the total summer rainfall at our field sites was 45 to 67% of the long-term mean at Cheesman, and the respective summer erosivity values were about 32 to 85% of the long-term average (Table 3). In contrast, summer rainfall in 2004 and 2006 was close to or above average, and the summer erosivity in these 2 yr was about 30% above the long-term mean (Table 3).

For the raked hillslopes, the rainfall and erosivity values in 2004 were much lower than for the burned and unburned hillslopes, and this is partly because the raking did not take place until early July. In 2005 and 2006, all three treatments—raked, burned, and unburned—had similar amounts of summer rainfall (Table 3). Summer erosivity values were much more variable between treatments, as in 2005 the raked hillslopes had more erosivity than the burned and unburned hillslopes, while in 2006 the erosivity on the raked hillslopes was less than half of the values from the other two treatments. Most importantly, the mean summer erosivity values for the raked hillslopes for 2005 and 2006 was 272 MJ mm ha^{-1} h^{-1}, which is nearly identical to the value for the burned hillslopes in the summer of 2003 (Table 3). This similarity allowed us to compare sediment yields from the burned and raked hillslopes for periods with similar erosivity values and nearly identical amounts of surface cover.

Sediment Yields: Burning Experiment

Mean sediment yields from the burned hillslopes were very low in 2002 due to the lack of large storms after the sediment fences had been reinstalled (Fig. 4a). In 2003, the mean sediment yield peaked at 14.8 Mg ha^{-1} despite the below-average rainfall and erosivity. In 2004, the mean sediment yield decreased to 11.0 Mg ha^{-1} (Fig. 4a) even though summer rainfall and erosivity nearly doubled relative to 2003 (Table 3). This decline in the mean sediment yield from 2003 to 2004 is attributed to the increase in mean surface cover from 18% in 2003 to 40% in 2004 (Pietraszek, 2006).

In 2005, the mean sediment yield from the burned hillslopes was only 0.6 Mg ha^{-1} due to the below-normal erosivity and the increase in mean surface cover to 53%. From 2005 to 2006, there was only a small increase in mean surface cover, but the fourfold increase in summer rainfall and erosivity caused sediment yields to nearly triple to 1.5 Mg ha^{-1} (Fig. 4a). Overall, the decline in soil water repellency on the burned hillslopes was much more rapid than the decline in annual sediment yields (Fig. 4a), and this discrepancy was most pronounced at the soil surface.

In contrast to the burned hillslopes, the unburned hillslopes generated a measurable amount of sediment for only two of the 105 plot-years of measurements. More specifically, in 2005 one of the untreated control hillslopes for the raking experiment generated 1.5 Mg ha^{-1} of sediment during a 19-mm storm with an I_{30} of 31 mm h^{-1}, 0.46 Mg ha^{-1} of sediment from a storm with an I_{30} of 19 mm h^{-1}, and 0.05 Mg ha^{-1} of sediment from a storm with an I_{30} of 17 mm h^{-1}. In 2006, the same hillslope produced 0.48 Mg ha^{-1} of sediment from five smaller storms with I_{30} values of 5 to 12 mm h^{-1}, and most of the sediment is believed to have originated from a steep, bare patch immediately adjacent to the sediment fence.

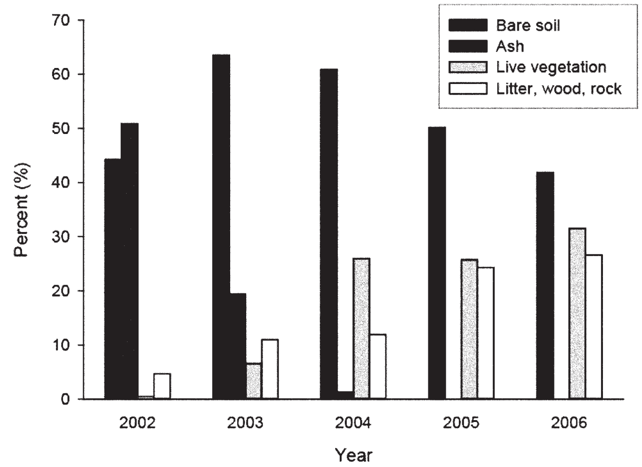

Fig. 5. Mean percentage of bare soil, ash, live vegetation, and other surface cover (litter, wood, and rock) for the burned hillslopes from 2002 to 2006.

Sediment Yields: Raking Experiment and Comparison of Sediment Yields vs. Thirty-Minute Rainfall Intensities

The lack of large storms in the latter part of the summer meant that none of the raked hillslopes produced any sediment in 2004, but in the summer of 2005 the mean sediment yield was 29 Mg ha^{-1}. In summer 2006 the total erosivity was 28% less than in the summer of 2005, and the mean sediment yield from the raked hillslopes was 62% lower at 11 Mg ha^{-1}. The mean sediment yield from the raked hillslopes in 2005 was nearly double the mean value from the burned hillslopes in 2003, despite having nearly identical amounts of surface cover and only 20% more rainfall erosivity. In 2006, the mean sediment yield from the raked hillslopes was 26% less than the mean sediment yield from the burned hillslopes in 2003, but the summer erosivity for the raked hillslopes was 20% lower than the corresponding value for the burned hillslopes. These variations in the summer erosivity for the raked hillslopes were largely negated by comparing the mean sediment yield of 20 Mg ha^{-1} yr^{-1} for the raked hillslopes

Table 3. Weighted mean summer rainfall and erosivity for the burned, unburned, and raked plus control hillslopes from 2002 to 2006. A dash indicates no data. For comparison, the mean summer rainfall from the long-term record for the nearby Cheesman weather station is 230 mm, and the estimated long-term mean summer erosivity for the study area is 364 MJ mm ha^{-1} h^{-1} (Foster, 2004).

Year	Summer rainfall			Summer erosivity		
	Burned	Unburned	Raked/control	Burned	Unburned	Raked/control
	mm			MJ mm ha^{-1} h^{-1}		
2002	84	99	–	95	72	–
2003	127	104	–	271	193	–
2004	260	218	114	467	456	59
2005	124	154	123	116	246	311
2006	289	265	231	470	556	223

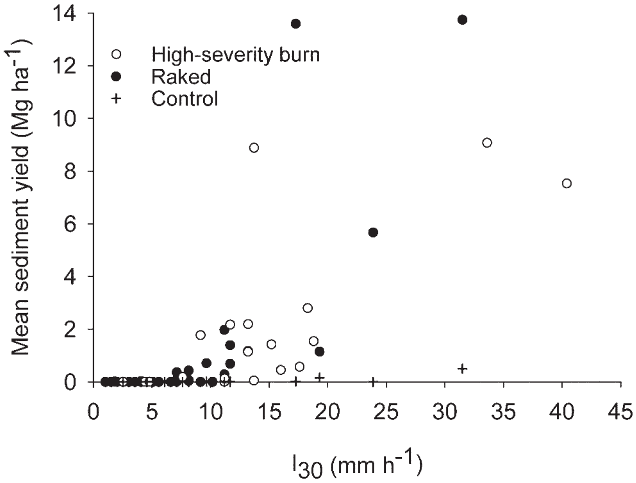

Fig. 6. Mean storm-based sediment yields vs. the maximum 30-min intensity (I_{30}) for the burned hillslopes in 2003, the raked hillslopes in 2005 to 2006, and the three control hillslopes for the raking experiment in 2005 to 2006. For clarity, the data from the other unburned hillslopes are not plotted as none of these hillslopes (n = 103 plot-years) produced any sediment despite I_{30} values of up to 65 mm h^{-1}.

in 2005 and 2006 to the mean sediment yield of 15 Mg ha^{-1} for the burned hillslopes in 2003 (Table 3). The resulting difference of 5 Mg ha^{-1} was not significant (P = 0.17).

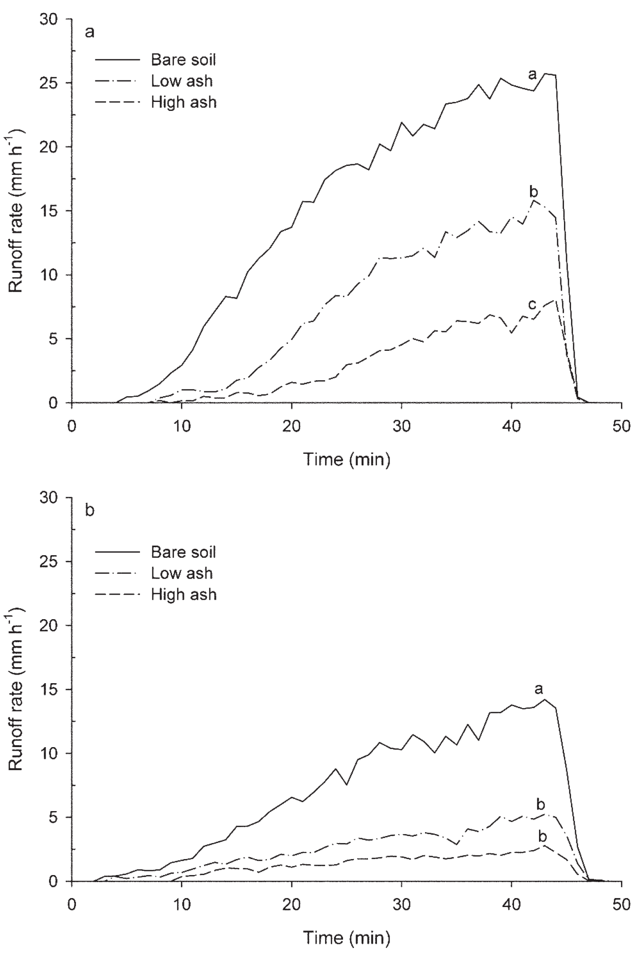

Fig. 7. Mean runoff rate with time for the bare-soil, low-ash, and high-ash treatments for (a) the granitic soil and (b) the micaceous soil. There were three replicates for each treatment on each soil; final runoff rates from treatments with different letters are significantly different at $P \leq 0.05$.

The similar behavior of the raked and burned swales can also be evaluated by comparing the storm-based sediment yields for those same years when each group averaged 17 to 18% surface cover. For both groups, the I_{30} threshold for sediment generation was about 7 mm h^{-1} (Fig. 6). The similarity in the threshold for sediment production indicates a similarity in runoff response that is independent of sediment availability, as infiltration-excess overland flow is the only mechanism for delivering substantial amounts of sediment to the sediment fence. The analogous threshold for the single control hillslope that generated sediment was 17 mm h^{-1}, while the other two control hillslopes did not generate any sediment from storms with I_{30} values of up to 32 mm h^{-1}.

There also is no clear distinction in the unit area sediment yields for a given I_{30} between the burned hillslopes in 2003 and the raked hillslopes in 2005 to 2006 (Fig. 6). In contrast, the mean storm-based sediment yields for the three control hillslopes were generally at least an order of magnitude lower than the corresponding mean sediment yields from the burned and raked hillslopes, respectively (Fig. 6). It should be noted that some of the other unburned hillslopes were subjected to rainfall intensities of up to 65 mm h^{-1} (approximately a 5-yr storm), and these unburned hillslopes still generated little or no evidence of overland flow and no measurable amounts of sediment (Libohova, 2004; Brown et al., 2005).

Rainfall Simulations

Single Simulations

The relative effects of the two ash treatments on runoff were similar for each of the two soils (Fig. 7). In the case of the granitic soil, the mean time to runoff increased from 6 min for the bare-soil treatment to 9 min for the low-ash treatment and 12 min for the high-ash treatment (Table 4). An increase in ash thickness also decreased the rate at which runoff increased with time (Fig. 7a).

The ash cover had a substantially greater effect on the mean runoff coefficient. For the granitic soil, the mean runoff coefficient declined from 35% for the bare-soil treatment to just 17% for the low-ash treatment and 7% for the high-ash treatment (Table 4). The final runoff rates followed the same pattern; the mean final runoff rate declined from 23 mm h^{-1} for the bare granitic soil to 14 mm h^{-1} for the low-ash treatment and 6.8 mm h^{-1} for the high-ash treatment (Fig. 7a). For the granitic soil, the mean runoff coefficients and final runoff rates were significantly different for each treatment (Table 4, Fig. 7a), but the time to runoff was significantly different only between the bare-soil and high-ash treatments (Table 4).

Less runoff was generated from each treatment on the micaceous soil than the granitic soil (Table 4). The mean runoff coefficient for the bare micaceous soil was only 7.8% and the mean final runoff rate was 13 mm h^{-1} (Table 4, Fig. 7b). The low-ash treatment reduced the runoff coefficient and final runoff rate by >50%, and these values were significantly lower than the bare-soil treatment. The high-ash treatment further reduced the runoff coefficient and final runoff rate and increased the time to runoff relative to the low-ash treatment, but the differences between the low- and high-ash treatments were not significant (Table 4, Fig. 7b).

The trends in sediment yields between treatments were similar to the trends in runoff, but the magnitude of the differences between treatments was somewhat smaller and there was more variability between replicates (Fig. 8). The simulations on the bare granitic soil yielded 24 g of sediment compared with mean values of 16 and 11 g for the low- and high-ash treatments, respectively. The differences in sediment yields between the bare soil and the two ash treatments were significant, but there was no significant difference in the sediment yields between the ash treatments (Fig. 8).

The mean sediment yield for the bare micaceous soil was 24 g, which was the same as the granitic soil despite the much lower runoff (Fig. 8). The mean sediment yields for the two ash treatments on the micaceous soil also were very similar to the treatments on the granitic soil. Again, the bare soil generated significantly more sediment than the two ash treatments, but there was no significant difference in sediment yields between the two ash treatments (Fig. 8). The similarity in sediment yields from the two soils, despite the large differences in runoff, indicates that most of the sediment yield was due to splash erosion, which is consistent with observations made during the simulations.

Successive Simulations

The three successive rainfall simulations on the bare granitic soil caused only a small increase in the runoff coefficients (Fig. 9a) and final runoff rates (Fig. 9b). In contrast, there was a fivefold increase in the runoff coefficient from the first to the second simulation on the low-ash treatment (Fig. 9a), and nearly a threefold increase in the final runoff rate (Fig. 9b). The increases in runoff were smaller from the second to the third simulation, but the runoff coefficient still increased from 43 to 55% and the final runoff rate increased from 26 to 29 mm h^{-1} (Fig. 9). These increases mean that the runoff values from the low-ash treatment progressively approached the values from the bare-soil treatment (Fig. 9). The percentage of the surface covered by ash dropped from 100 to 77% at the end of the first simulation, to 45% at the end of the second simulation, and to just 25% by the end of the third simulation. Sediment yields were 28 to 29 g for the first two simulations on the bare granitic soil, but decreased to 22 g for the third simulation.

The first simulation on the low-ash treatment yielded 17 g of sediment or about 60% of the sediment yield from the first simulation on the bare soil. The sediment yield from the second simulation increased to 28 g, which was identical to the second simulation on the bare soil. The sediment yield from the third simulation was 19 g, which again was very similar to the corresponding sediment yield from the bare-soil treatment.

Runoff rates for the micaceous soil greatly increased from the first to the second rainfall simulation. For the bare soil, there was a threefold increase in the runoff coefficient between the first and second simulations (Fig. 9a) and a slightly greater than twofold increase in the final runoff rate (Fig. 9b). The greater magnitude of these increases relative to the bare granitic soil is at least partially due to the shorter time period between the simulations (1 d) and hence the wetter antecedent conditions for the second simulation. There was a 7-d period between the second and the third simulations on the bare micaceous soil, and the drier conditions relative to the second simulations can explain the observed small decline in both the runoff coefficient and final runoff rate from the second to the third simulation. Runoff rates from the second and third simulations on the bare micaceous soil were comparable to the values measured from the bare granitic soil (Fig. 9).

Table 4. Mean time to runoff and runoff coefficients for each soil by treatment. Different letters in the same column indicate significant differences. Mean values ± one standard deviation.

Treatment	Time to runoff		Runoff coefficient	
	Granitic soil	Micaceous soil	Granitic soil	Micaceous soil
	min		%	
Bare soil	6 ± 1 a	6 ± 3 a	35 ± 5.8 a	7.8 ± 1.3 a
Low ash	9 ± 1 ab	8 ± 4 a	17 ± 3.1 b	3.7 ± 0.7 b
High ash	12 ± 4 b	12 ± 2 a	7.2 ± 0.8 c	3.2 ± 1.4 b

The successive rainfall simulations on the low-ash treatment on the micaceous soil resulted in an increase in runoff that was similar to the results observed for the low-ash treatment on the granitic soil. From the first to the second simulation there was a fivefold increase in the runoff coefficient (Fig. 9a) and a nearly fourfold increase in the final runoff rate (Fig. 9b); these increases were nearly double the increases observed for the bare soil treatment. From the second to the third simulation there was a much smaller increase in the final runoff rate and little change in the runoff coefficient, partly due to the wetter antecedent conditions for the second simulation compared with the third simulation. As with the granitic soil, by the third simulation the amount of runoff was very similar between the bare soil and the low-ash treatment (Fig. 9). This means that the ash cover substantially reduced runoff only for the first simulation, and each successive simulation progressively reduced the differences in runoff between treatments. Differences in runoff between the two soils were largely eliminated by the second simulation, although this comparison is confounded by the wetter antecedent conditions for the second simulation on the micaceous soil.

Thin-Section Descriptions

The thin section taken after a single simulation (40 mm of rainfall) on the bare granitic soil showed that the soil surface was dominated by larger grains up to 2 mm in diameter (Fig. 10a). Underneath this coarse surface layer there was a dense layer of fine particles that averaged <1 mm in thickness (Fig. 10a). This sequence of loose, skeletal grains overlying a dense, low-porosity layer indicates a sieving structural crust or seal (Valentin and Bresson, 1992).

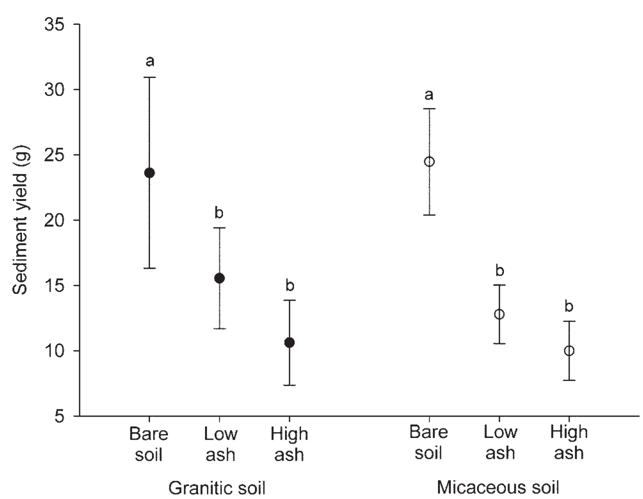

Fig. 8. Mean sediment yields for the granitic and micaceous soils by treatment. The bars indicate one standard deviation. Sediment yields from treatments with different letters are significantly different at $P \leq 0.05$.

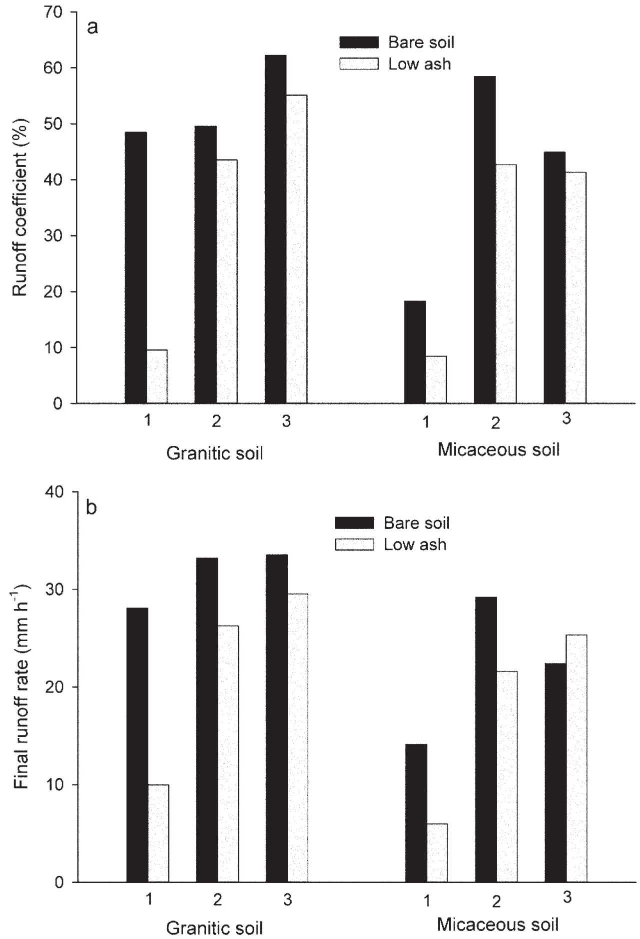

Fig. 9. Runoff (a) coefficients and (b) final rates for three successive rainfall simulations on the bare-soil and low-ash treatments for the granitic and micaceous soils.

The corresponding thin section after 40 mm of simulated rainfall on the low-ash treatment showed a 3-mm-thick ash layer overlying the mineral soil (Fig. 10b). The sharp boundary between the two indicates that the ash particles were generally too large to move into and clog the pores in the underlying mineral soil. This is supported by the fact that 85% of the mass of the ash is mineral matter, and 76% of the mineral mass is sand-sized particles. The profile of the thin section taken from the simulation on the high-ash treatment was similar except that the overlying ash layer was still about 8 mm thick. In both ash treatments, there was a concentration of smaller ash particles in the upper 0.1 to 0.2 mm of the ash layer (Fig. 10b).

The thin section of the bare micaceous soil showed the development of a similar layer of predominantly coarse particles over a dense, structural seal (Fig. 10c). The thin sections from the low- and high-ash treatments had a distinct boundary between the ash and mineral soil, and this again indicates that the ash particles generally did not move down into the pores of the mineral soil (Fig. 10d).

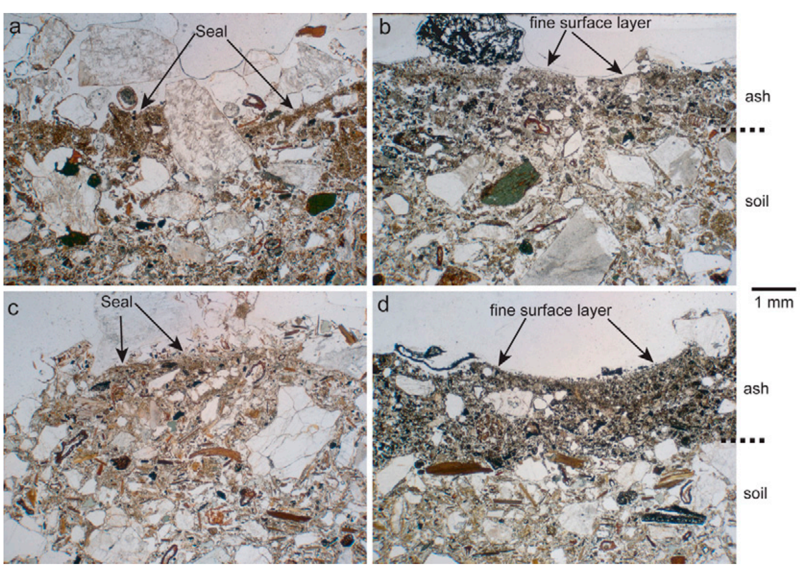

Fig. 10. Photographs of thin sections after 40 mm of simulated rainfall on (a) bare granitic soil, (b) granitic soil with an initial ~5-mm-thick ash layer, (c) bare micaceous soil, and (d) micaceous soil with an initial ~5-mm thick ash layer. The 1-mm scale bar applies to all photos.

DISCUSSION

Contribution of Soil Water Repellency to Post-Fire Runoff and Erosion

The rapid decay of soil water repellency after burning suggests that soil water repellency cannot be the primary cause of the observed increases in post-fire runoff and surface erosion. The severely burned areas at the Hayman and Schoonover fires generally had stronger soil water repellency than the unburned areas only through the first summer after burning (Fig. 4). An even more rapid decline in post-fire soil water repellency was observed after the June 2000 Bobcat fire in the northern Colorado Front Range (MacDonald and Huffman, 2004), yet sediment yields were high in each of the first 2 yr after burning (Benavides-Solorio and MacDonald, 2005; Wagenbrenner et al., 2006). Rapid declines in post-fire soil water repellency also have been documented for other conifer and chaparral ecosystems in western North America (Henderson and Golding, 1983; McNabb et al., 1989; Hubbert et al., 2006). The longer duration of high post-fire sediment yields relative to the fire-induced soil water repellency, as shown in Fig. 4, is strong evidence that fire-enhanced soil water repellency is not the dominant control on post-fire sediment yields.

Other field studies indicate that the effect of soil water repellency on runoff and erosion decreases with increasing spatial scale. Soil water repellency is most easily measured at the point scale, or at multiple points in small plots, and at this scale some studies have found a strong relationship between fire-induced soil water repellency and higher post-fire runoff and erosion rates (e.g., Robichaud, 2000). In the Colorado Front Range, soil water repellency explained 70 to 80% of the increase in runoff and sediment yields from rainfall simulations on severely burned 1-m^2 plots; however, the correlations with the surface cover percentage were nearly as strong (Benavides-Solorio and MacDonald, 2001, 2002). More compelling evidence comes from two studies showing that surfactants, which can increase soil wettability, significantly reduced plot-scale (0.4–40-m^2) runoff and sediment yields following fires in southern California chaparral (Osborn et al., 1964) and eucalyptus plantations in Portugal (Leighton-Boyce et al., 2007).

The contribution of soil water repellency to post-fire runoff and erosion is much more difficult to justify and document at larger spatial scales, and there are several reasons for this. First, there are no techniques to directly measure soil water repellency at larger scales. Second, post-fire soil water repellency varies with time due to both fluctuations in soil moisture and the breakdown of the water-repellent compounds (e.g., Doerr et al., 2009), and this makes it very difficult to collect repeated soil water repellency measurements at larger scales and relate these to runoff rates and storm-based sediment yields with time. Third, soil water repellency is highly heterogeneous across space (Woods et al., 2007). This patchiness, when combined with the high infiltration rates in the less repellent areas, provides a conceptual justification for soil water repellency to play a much more limited role in post-fire runoff and erosion than is commonly assumed.

Similar arguments can be made for unburned areas, as our point measurements at the Hayman and Schoonover fires indicated strong water repellency at 0 and 3 cm. There was no evidence of overland flow, however, and sediment was generated for only two of the 105 hillslope-years of monitoring. The implication is that, if any runoff is generated by point-scale soil water repellency, it quickly infiltrates farther downslope. Similarly, unburned soils in Portugal were water repellent but generally did not generate runoff or sediment until burning removed the surface cover (Shakesby et al., 2000).

In the Colorado Front Range, soil water repellency did explain about 40% of the variability in annual hillslope-scale sediment yields at some of the same fires where the rainfall simulations were conducted (Benavides-Solorio and MacDonald, 2005). A closer analysis then showed that this relationship depended entirely on the data from one 6-yr-old fire that had very low soil water repellency values and very low sediment yields. After removing these data, there was no significant relationship between soil water repellency and sediment yields for the other 82 hillslope-years of data from five fires (Benavides-Solorio and MacDonald, 2005).

The application of a surfactant to a 1.1-km^2 watershed after a chaparral fire in southern California also did not reduce sediment yields relative to an untreated control watershed (Rice and Osborn, 1970). The limited effect of soil water repellency on runoff and sediment yields at larger scales is also supported by the high sediment yields from the raked hillslopes compared with their controls, even though the controls had much stronger soil water repellency at the mineral soil surface. These combined results lead to the conclusion that soil water repellency is a much less significant control on sediment yields at the hillslope or small-catchment scale than at the small-plot scale.

Role of Surface Cover in Post-Fire Runoff and Erosion Processes

The burning and raking experiments demonstrated that surface cover was an important control on post-fire sediment yields, which is consistent with the results from other studies (e.g., Morris and Moses, 1987; Cerdà, 1998a; Prosser and Williams, 1998; Robichaud and Brown, 1999; Benavides-Solorio and MacDonald, 2001, 2005; Johansen et al., 2001). The problem is that the process(es) that drives the observed empirical relationship between bare soil percentage and post-fire sediment yields has heretofore been unclear, as high-severity fires reduce both soil organic matter and aggregate stability. These changes increase soil erodibility and are conducive to soil sealing. The Hayman fire did not significantly reduce soil organic matter (Libohova, 2004), and hence the loss of organic matter cannot explain the observed increase in sediment yields. Although we have not independently measured the effects of fire-induced changes on aggregate stability and soil erodibility, our various field and laboratory studies provide insights as to whether the post-fire increase in soil erodibility (Moody et al., 2005) or soil sealing were more important.

The effects of fire-induced changes in soil erodibility were evaluated by comparing the burned and raked hillslopes. These two sets of hillslopes had similar characteristics and generated similar amounts of sediment in years when the surface cover percentage and rainfall erosivity were similar, so the main difference is how the surface cover was removed. The absence of fire on the raked hillslopes means that the increased surface runoff and erosion cannot be attributed to a fire-induced increase in soil erodibility or reduction in aggregate stability. It can be argued that the surface disturbance due to raking increased the soil erodibility, but it is difficult to envisage how the litter can be removed in larger scale experiments without at least some associated soil disturbance.

More process-based studies provide strong evidence that a fire-induced increase in soil erodibility was not the primary cause of the increased sediment production after the Hayman wildfire. Our field data showed no evidence of overland flow or surface erosion on convergent hillslopes, while the first post-fire storm caused extensive rilling in the swale axes (Libohova, 2004). During successive storms, these rills incised up to several decimeters into the underlying soil, and this soil was not altered by the 2002 fires. Detailed measurements indicate that these rills and small gullies contributed 60 to 80% of the hillslope sediment yields measured after the Hayman and Schoonover fires (Pietraszek, 2006). Since this unaltered soil was the primary source of sediment, we infer that the fire-induced changes in soil erodibility and aggregate stability were not the primary causes of the observed increase in sediment yields. The same argument can be made for the raked hillslopes where similar rilling was observed.

The burned and raked hillslopes had nearly identical rainfall intensity thresholds for generating sediment, and these thresholds were much lower than the I_{30} threshold for the unburned control hillslopes. The increase in runoff was primarily responsible for the increase in sediment yields because most of the eroded sediment was derived from rill erosion. The key question is what caused the observed increase in runoff, and both the single and successive rainfall simulation experiments demonstrated that the development of a structural soil seal on the bare soil treatment led to the measured increases in runoff. The simulation experiments also demonstrated that an ash cover initially prevented soil sealing, but runoff rates sharply increased as the ash cover was removed by successive rain events. Numerous other studies also have shown that a wide variety of surface cover types can protect against soil sealing; these include straw mulch, crop residues, leaves, grasses, cryptogams, and stones (Morin and Benyamini, 1977; Poesen, 1986; Kinnell et al., 1990; Moss and Watson, 1991; Ruan et al., 2001). Hence, the relationship between sediment yields and surface cover shown in Fig. 1 can be best explained by the surface cover percentage controlling the extent of soil sealing, which in turn controls runoff production and surface erosion by overland flow. More detailed studies are needed to further separate the relative roles of soil sealing and changes in soil erodibility under different conditions, but our combined field and lab results indicate that soil sealing is the more critical process.

Role of Ash in Post-Fire Runoff and Erosion Processes

The runoff data from the rainfall simulations and the thin sections show that surface ash particles did not clog the soil pores and precluded the development of a structural soil seal in both the granitic and micaceous soils. While this result is contrary to the conclusions of two older studies that were based on measurements and observations of soil and ash properties (Mallik et al., 1984; Etiégni and Campbell, 1991), it is consistent with several more recent field studies that measured runoff from ash-covered soils. In Spain, for example, rainfall simulations on recently burned plots with 1 to 10 cm of ash failed to produce runoff despite a rainfall application rate of 60 mm h^{-1} (Cerdà, 1998a). Six months later the ash layer had been largely eroded; in a second round of rainfall simulations, 40 to 80% of the applied rainfall was converted to runoff. This change in runoff was attributed to soil sealing (Cerdà, 1998a).

A more recent rainfall simulation study in Spain showed that 18 times more runoff was produced from bare plots than plots covered with a 36-mm-thick ash layer, and that nearly all of the ash in a control area was eroded by the first natural post-fire storm (Cerdà and Doerr, 2008). Similarly, rainfall simulations on bare plots and plots with a 19-mm-thick ash layer conducted about 1 mo after a Montana wildfire showed that the ash cover reduced infiltration and protected the underlying mineral soil from sealing (Woods and Balfour, 2008). A second set of rainfall simulations conducted 9 mo after the fire, when much of the ash had been eroded, showed no significant difference in runoff between the bare and ash-covered plots (Woods and Balfour, 2008). In northern California, runoff measurements from a small plot indicated that 20 to 30 mm of rainfall was needed to generate runoff immediately after a fire in a pine forest (Onda et al., 2008). By the third post-fire storm, the threshold for runoff generation had decreased to <6 mm, and this was attributed to the erosion and compaction of a 20-mm-thick ash layer (Onda et al., 2008).

The results from these field studies support our findings and conclusions with respect to the dominant role of soil sealing, but our rainfall simulation experiments provide additional insights into the process(es) by which an ash layer reduces runoff and surface erosion. The ash layer after a high-severity burn in forests and shrub lands is typically a few millimeters to a few decimeters thick, and this generally is hydrophilic (Cerdà, 1998b; Grogan et al., 2000; Henig-Sever et al., 2001; Johansen et al., 2003) with the exception of the ash studied by Gabet and Sternberg (2008). This means that some of the initial rainfall is intercepted as the ash layer wets up (e.g., Woods and Balfour, 2008).

In our single simulation experiments, the delay in the onset of runoff for the ash treatments indicates that, on average, the 5-mm-thick, low-ash treatment intercepted 2 mm or 5% of the applied rainfall and the 12-mm-thick, high-ash treatment intercepted about 5 mm of water. These values are only about half of the theoretical storage as calculated from the depth of the ash, an ash bulk density of 0.5 g cm^{-3}, and an assumed particle density of 2.5 g cm^{-3} (Bookter, 2006). The twofold difference between the theoretical and actual storage of water in the ash layer can be attributed to the rapid compaction and erosion of the ash layer by raindrop impact as observed during the simulations and shown by the thin sections. Even if all of the water that was stored in the ash layer had become runoff, each of the ash treatments on the granitic soil still would have generated only about 8 to 9 mm of runoff vs. 14 mm for the bare soil. The only explanation for the additional reduction in runoff from the ash-covered plots was that there was an inherent difference in infiltration as a result of the ash layer protecting the mineral soil and inhibiting soil sealing. The ability of ash to reduce soil sealing is supported by the observation that soil columns with bare soils and identical soils covered with chaparral ash had identical infiltration rates (Burgy and Scott, 1952). Rainfall simulation experiments concurrent with our studies and after a high-severity burn also found that water storage in the ash layer could only account for only 75% of the reduction in runoff relative to bare plots. The additional 25% reduction in runoff was attributed to the reduction in soil sealing as a result of an ash layer (Woods and Balfour, 2008).

In our study, the storage of water in the ash layer was proportionally more important for the micaceous soil because there was so little runoff from the bare-soil treatment. As with the granitic soil, a rainfall rate of 45 mm h^{-1} will satisfy the moisture storage capacity of the ash layer in <5 or 10 min, so the much higher final runoff rate for the bare-soil treatment must be attributed to a lower infiltration rate into the underlying mineral soil. Again, the formation of a structural soil seal was the most plausible explanation for the significantly higher runoff rate from the bare-soil treatment. From our experimental results and the recent work by Woods and Balfour (2008), we conclude that the infiltration rate into an ash-covered soil is controlled by the characteristics of the underlying mineral soil, and that raindrop impacts on the exposed mineral soil is necessary for inducing post-fire soil sealing.

It is less clear why the single simulations on the granitic soil consistently generated more runoff than the comparable treatments on the micaceous soil. The granitic soil did have a slightly higher bulk density, which tends to reduce infiltration, and a higher clay content, which tends to increase soil sealing (Shainberg and Levy, 1996). Both soils had similar CEC and sodicity values (Table 2), and the chemical analyses indicate that the sealing is a physical rather than chemical process. The key point, however, is that runoff from the two soils was very similar for the second and third successive simulations (Fig. 9), indicating that a similar sealing process was acting to reduce infiltration in both soils.

Both our laboratory and field data indicate that a relatively high proportion of ash cover is needed to protect against soil sealing and substantially reduce runoff rates after a high-severity fire. In the case of the granitic soil, the amount of ash cover dropped from 77 to 45% during the second rainfall simulation, and the final runoff rate increased to nearly 80% of the corresponding value from the bare soil (Fig. 9a). The 561 hillslope-years of annual sediment production data from 10 fires in the Colorado Front Range show that 60 to 65% surface cover is needed to greatly reduce post-fire sediment yields (Fig. 1). Conversely, sediment yields are almost always high when the surface cover is <40% (Fig. 1) (Benavides-Solorio and MacDonald, 2005; Pietraszek, 2006). These results suggest that at least a 50 to 75% cover of ash, litter, or live vegetation is needed to prevent soil sealing and reduce post-fire runoff and erosion rates.

Both the field studies and the rainfall simulations indicate that post-fire ash layers are short lived and therefore provide only brief protection against soil sealing. At the Hayman and Schoonover fires, the mean ash cover dropped from 54% almost immediately after burning to just 5% by the end of the second summer. Other studies have shown that a surface ash layer is easily eroded by wind and water (Johansen et al., 2003; de Luis et al., 2003; Whicker et al., 2006; Onda et al., 2008; Cerdà and Doerr, 2008; Woods and Balfour, 2008). Our visual observations and cover data from the successive simulations on the granitic soil confirm that a 5-mm-thick ash layer is easily eroded by rain splash and overland flow. A thicker ash layer is unlikely to persist much longer, given the ease with which it can be eroded by wind and surface runoff, and recent field studies confirm that a post-fire ash layer reduces runoff for only the first few storms after burning (Onda et al., 2008; Cerdà and Doerr, 2008).

Implications for Post-Fire Erosion Mitigation Treatments

The critical role of soil sealing for post-fire runoff and sediment yields has important implications for the design and application of post-fire emergency rehabilitation treatments. Several studies have shown that application of straw mulch reduces post-fire sediment yields by at least 90% in the first 1 to 3 yr after burning (Bautista et al., 1996; Wagenbrenner et al., 2006; Rough, 2007), and this further confirms the importance of surface cover in controlling post-fire sediment yields. Our present study provides the mechanistic explanation as to why post-fire mulching is effective, as it protects the mineral soil from raindrop impact and soil sealing. Experimental and plot-scale rainfall simulations conducted under unburned conditions also have shown that mulch reduces soil sealing (Morin and Benyamini, 1977; Morin et al., 1989; Moss and Watson, 1991).

In contrast to mulching, seeding generally does not reduce post-fire sediment yields (Robichaud et al., 2000). The lack of effectiveness has to be attributed to the fact that seeding generally does not significantly increase revegetation rates and the amount of surface cover (Robichaud et al., 2000; Wagenbrenner et al., 2006; Rough, 2007). These empirical observations also can be mechanistically explained as a lack of protection against soil sealing. Taken together, these results show that the most effective treatments for reducing post-fire sediment yields are those that immediately increase the amount of surface cover and thereby inhibit soil sealing.

CONCLUSIONS

A series of field and rainfall simulation experiments were conducted to evaluate the relative effects of soil water repellency, surface cover, and the presence of an ash layer on post-fire runoff and sediment yields in the Colorado Front Range. There was strong soil water repellency at and near the soil surface of the unburned hillslopes, but only one of the 34 unburned hillslopes generated any sediment during the 5-yr study. The hillslopes burned at high severity had stronger soil water repellency than unburned hillslopes only for the first summer after burning, but sediment yields were greatly elevated above background levels for the first three summers after burning and were still above background levels five summers after burning. Removing the surface litter by raking caused the unburned and burned hillslopes to have a nearly identical relationship between rainfall intensity and sediment yields.

Rainfall simulations on two soils with 0-, 5-, or 12-mm-thick layers of ash showed that runoff coefficients, final runoff rates, and sediment yields decreased as ash thickness increased. The additional storage of water in the ash layer could not account for the observed differences in runoff, and soil thin sections indicate that the much higher runoff and erosion rates from the bare soil were due to the formation of a thin structural soil seal. These field and laboratory results show that the observed empirical relationship between surface cover percentage and post-fire sediment yields is due primarily to soil sealing rather than soil water repellency or fire-induced changes in soil erodibility. The predominant role of soil sealing means that the most effective post-fire rehabilitation treatments will be those that immediately increase the amount of surface cover.

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REFERENCES

Assouline, S. 2004. Rainfall-induced soil surface sealing: A critical review of observations, conceptual models, and solutions. Vadose Zone J. 3:570–591.

Assouline, S., and Y. Mualem. 2000. Modeling the dynamics of soil seal formation: Analysis of the effect of soil and rainfall properties. Water Resour. Res. 36:2341–2349.

Badí, D., and C. Martí. 2003. Plant ash and heat intensity effects on chemical and physical properties of two contrasting soils. Arid Land Res. Manage. 17:23–41.

Bajracharya, R.M., and R. Lal. 1998. Crusting effects on erosion processes under simulated rainfall on a tropical Alfisol. Hydrol. Processes 12:1927–1938.

Bautista, S., J. Bellot, and V.R. Vallejo. 1996. Mulching treatment for postfire soil conservation in a semiarid ecosystem. Arid Soil Res. Rehabil. 10:235–242.

Benavides-Solorio, J., and L.H. MacDonald. 2001. Post-fire runoff and erosion from simulated rainfall on small plots. Hydrol. Processes 15:2931–2952.

Benavides-Solorio, J., and L.H. MacDonald. 2002. Errata: Post-fire runoff and erosion from simulated rainfall on small plots. Hydrol. Processes 16:1131–1133.

Benavides-Solorio, J., and L.H. MacDonald. 2005. Measurement and prediction of post-fire erosion at the hillslope scale, Colorado Front Range. Int. J. Wildland Fire 14:457–474.

Bookter, A. 2006. Erosional processes after wildfires: The impact of vegetative ash and the morphology of debris flows. M.S. thesis. Univ. of Montana, Missoula.

Brock, J.H., and L.F. DeBano. 1982. Runoff and sedimentation potentials influenced by litter and slope on a chaparral community in central Arizona. p. 372–377. *In* C.E.C. Conrad and W.C. Oechel (ed.) Proc. Symp. Dynamics and Manage. of Mediterranean-type Ecosystems. Gen. Tech. Rep. PSW-58. U.S. For. Serv., Pac. Southw. For. Range Exp. Stn., Berkeley, CA.

Brown, E., L.H. MacDonald, Z. Libohova, D. Rough, and K. Schaffrath. 2005. Sediment production rates from forest thinning, wildfires, and roads: What is important? Abstract H51E–0418. *In* AGU Fall Meet., San Francisco. 5–9 Dec. 2005. Am. Geophys. Union, Washington, DC.

Brown, L.C., and G.R. Foster. 1987. Storm erosivity using idealized intensity

distributions. Trans. ASAE 30:379–386.
Burgy, R.H., and V.H. Scott. 1952. Some effects of fire and ash on the infiltration capacity of soils. Trans. Am. Geophys. Union 33:405–416.
Burton, T.A. 2005. Fish and stream habitat risks from uncharacteristic wildfire: Observations from 17 years of fire-related disturbances on the Boise National Forest, Idaho. For. Ecol. Manage. 211:140–149.
Clary, W.P., and P.F. Ffolliott. 1969. Water holding capacity of ponderosa pine forest floor layers. J. Soil Water Conserv. 24:22–23.
Cerdà, A. 1998a. Post-fire dynamics of erosional processes under Mediterranean climatic conditions. Z. Geomorphol. 42:373–398.
Cerdà, A. 1998b. Changes in overland flow and infiltration after a rangeland fire in a Mediterranean shrubland. Hydrol. Processes 12:1031–1042.
Cerdà, A., and S.H. Doerr. 2008. The effect of ash and needle cover on surface runoff and erosion in the immediate post-fire period. Catena 74:256–263.
DeBano, L.F. 1981. Water repellent soils: A state-of-the-art. Gen. Tech. Rep. PSW-46. U.S. For. Serv., Pac. Southw. For. Range Exp. Stn., Berkeley, CA.
DeBano, L.F. 2000. The role of fire and soil heating on water repellency in wildland environments: A review. J. Hydrol. 231–232:195–206.
DeBano, L.F., D.G. Neary, and P.F. Ffolliott. 2005. Soil physical properties. p. 29–51. *In* D.G. Neary et al. (ed.) Wildland fire in ecosystems: Effects of fire on soil and water. Gen. Tech. Rep. RMRS-GTR-42. Vol. 4. U.S. For. Serv., Rocky Mountain Res. Stn., Fort Collins, CO.
de Luis, M., J.C. González-Hidalgo, and J. Raventós. 2003. Effects of fire and torrential rainfall on erosion in a Mediterranean gorse community. Land Degrad. Dev. 14:203–213.
Doerr, S.H. 1998. On standardizing the 'water drop penetration time' and the 'molarity of an ethanol droplet' techniques to classify soil hydrophobicity: A case study using medium textured soils. Earth Surf. Processes Landforms 23:663–668.
Doerr, S.H., and J.A. Moody. 2004. Hydrological impacts of soil water repellency: On spatial and temporal uncertainties. Hydrol. Processes 18:829–832.
Doerr, S.H., R.A. Shakesby, and L.H. MacDonald. 2009. Soil water repellency: A key factor in post-fire erosion? *In* A. Cerdà and P.R. Robichaud (ed.) Fire effects on soils and restoration strategies. Science Publ., Enfield, NH (in press).
Doerr, S.H., R.A. Shakesby, and R.P.D. Walsh. 2000. Soil water repellency: Its causes, characteristics and hydro-geomorphological significance. Earth Sci. Rev. 51:33–65.
Durgin, P.B. 1985. Burning changes the erodibility of forest soils. J. Soil Water Conserv. 40:299–301.
Durgin, P.B., and P.J. Vogelsang. 1984. Dispersion of kaolinite by water extractions of Douglas-fir ash. Can. J. Soil Sci. 64:439–443.
Etiégni, L., and A.G. Campbell. 1991. Physical and chemical characteristics of wood ash. Bioresour. Technol. 37:173–178.
Foster, G.R. 2004. User's reference guide for Revised Universal Soil Loss Equation, Version 2 (draft). USDA-ARS, Washington, DC.
Foster, G.R., W.H. Neibling, and R.A. Nattermann. 1982. A programmable rainfall simulator. Pap. 82-2570. Am. Soc. Agric. Eng., St. Joseph, MI.
Gabet, E.J., and P. Sternberg. 2008. The effects of vegetative ash on infiltration capacity, sediment transport, and the generation of progressively bulked debris flows. Geomorphology 101:666–673.
Gary, H.L. 1975. Watershed management problems and opportunities for the Colorado Front Range ponderosa pine zone: The status of our knowledge. Res. Pap. RM-139. U.S. For. Serv., Rocky Mountain For. Range Exp. Stn., Fort Collins, CO.
Giovannini, G., and S. Lucchesi. 1983. Effect of fire on hydrophobic and cementing substances of soil aggregates. Soil Sci. 136:231–236.
Grogan, P., T.D. Bruns, and F.S. Chapin III. 2000. Fire effects on ecosystem nitrogen cycling in a California bishop pine forest. Oecologia 122:537–544.
Gyssels, G., J. Poesen, E. Bochet, and Y. Li. 2005. Impact of plant roots on the resistance of soils to erosion by water: A review. Prog. Phys. Geogr. 29:189–217.
Henderson, G.S., and D.L. Golding. 1983. The effects of slash burning on the water repellency of forest soils at Vancouver, British Columbia. Can. J. For. Res. 13:353–355.
Henig-Sever, N., D. Poliakov, and M. Broza. 2001. A novel method for estimation of wildfire intensity based on pH and soil microarthropod community. Pedobiologia 45:98–106.
Hennessy, K., C. Lucas, N. Nicholls, J. Bathols, R. Suppiah, and J. Ricketts. 2005. Climate change impacts on fire-weather in south-east Australia. CSIRO Marine and Atmos. Res., Aspendale, VIC, Australia.
Hubbert, K.R., H.K. Preisler, P.M. Wohlgemuth, R.C. Graham, and M.G. Narog. 2006. Prescribed burning effects on soil physical properties and soil water repellency in a steep chaparral watershed, southern California, USA. Geoderma 130:284–298.
Huffman, E.L., L.H. MacDonald, and J.D. Stednick. 2001. Strength and persistence of fire-induced soil hydrophobicity under ponderosa and lodgepole pine, Colorado Front Range. Hydrol. Processes 15:2877–2892.
Johansen, M.P., T.E. Hakonson, and D.D. Breshears. 2001. Post-fire runoff and erosion from rainfall simulation: Contrasting forests with shrublands and grasslands. Hydrol. Processes 15:2953–2965.
Johansen, M.P., T.E. Hakonson, F.W. Whicker, and D.D. Breshears. 2003. Pulsed redistribution of a contaminant following forest fire: Cesium-137 in runoff. J. Environ. Qual. 32:2150–2157.
Kinnell, P.I.A., C.J. Chartres, and C.L. Watson. 1990. The effects of fire on the soil in a degraded semi-arid woodland: II. Susceptibility of the soil to erosion by shallow rain-impacted flow. Aust. J. Soil Res. 28:779–794.
Kittredge, J. 1948. Forest influences. McGraw-Hill Book Co., New York.
Krammes, J.S., and J. Osborn. 1969. Water-repellent soils and wetting agents as factors influencing erosion. p. 177–187. *In* L.F. DeBano and J. Letey (ed.) Proc. Symp. on Water-Repellent Soils, Riverside, CA. March 1968. Univ. of California, Riverside.
Lavee, H., P. Kutiel, M. Segev, and Y. Benyamini. 1995. Effect of surface roughness on runoff and erosion in a Mediterranean ecosystem: The role of fire. Geomorphology 11:227–234.
Leighton-Boyce, G., S.H. Doerr, R.A. Shakesby, and R.P.D. Walsh. 2007. Quantifying the impact of soil water repellency on overland flow generation and erosion: A new approach using rainfall simulation and wetting agent on *in situ* soil. Hydrol. Processes 21:2337–2345.
Letey, J. 1969. Measurement of contact angle, water drop penetration time, and critical surface tension. p. 43–47. *In* L.F. DeBano and J. Letey (ed.) Proc. Symp. on Water-Repellent Soils, Riverside, CA. March 1968. Univ. of California, Riverside.
Letey, J. 2001. Causes and consequences of fire-induced soil water repellency. Hydrol. Processes 15:2867–2875.
Libohova, Z. 2004. Effects of thinning and a wildfire on sediment production rates, channel morphology, and water quality in the upper South Platte River watershed. M.S. thesis. Colorado State Univ., Fort Collins.
Lowdermilk, W.C. 1930. Influence of forest litter on run-off, percolation, and erosion. J. For. 28:474–491.
MacDonald, L.H., and E.L. Huffman. 2004. Post-fire soil water repellency: Persistence and soil moisture thresholds. Soil Sci. Soc. Am. J. 68:1729–1734.
Mallik, A.U., C.H. Gimingham, and A.A. Rahman. 1984. Ecological effects of heather burning: I. Water infiltration, moisture retention, and porosity of surface soil. J. Ecol. 72:767–776.
Malmon, D.V., S.L. Reneau, D. Katzman, A. Lavine, and J. Lyman. 2007. Suspended sediment transport in an ephemeral stream following wildfire. J. Geophys. Res. Earth Surf. 112:F02006, doi:10.1029/2005JF000459.
Martin, D.A., and J.A. Moody. 2001. Comparison of soil infiltration rates in burned and unburned mountainous watersheds. Hydrol. Processes 15:2893–2903.
McIntyre, D.S. 1958. Soil splash and the formation of surface crusts by raindrop impact. Soil Sci. 85:261–266.
McNabb, D.H., F. Gaweda, and H.A. Froehlich. 1989. Infiltration, water repellency, and soil moisture content after broadcast burning a forest site in southwest Oregon. J. Soil Water Conserv. 44:87–90.
Mills, A.J., and M.V. Fey. 2004. Frequent fires intensify soil crusting: Physiochemical feedback in the pedoderm of long-term burn experiments in South Africa. Geoderma 121:45–64.
Mouillot, F., S. Rambal, and R. Joffre. 2002. Simulating climate change impacts on fire frequency and vegetation dynamics in a Mediterranean-type ecosystem. Global Climate Change Biol. 8:423–437.
Moody, J.A., and D.A. Martin. 2001. Initial hydrologic and geomorphic response following a wildfire in the Colorado Front Range. Earth Surf. Processes Landforms 26:1049–1070.
Moody, J.A., J.D. Smith, and B.W. Ragan. 2005. Critical shear stress for erosion of cohesive soils subjected to temperatures typical of wildfires. J. Geophys. Res. 110:F01004, doi:10.1029/2004JF000141.
Morin, J., and Y. Benyamini. 1977. Rainfall infiltration into bare soils. Water Resour. Res. 13:813–817.
Morin, J., R. Keren, Y. Benjamini, M. Ben-Hur, and I. Shainberg. 1989. Water

infiltration as affected by soil crust and moisture profile. Soil Sci. 148:53–59.
Morris, S.E., and T.A. Moses. 1987. Forest-fire and the natural soil erosion regime in the Colorado Front Range. Ann. Assoc. Am. Geograph. 77:245–254.
Moss, A.J., and C.L. Watson. 1991. Rain-impact soil crust: III. Effects of continuous and flawed crusts on infiltration, and the ability of plant covers to maintain crustal flaws. Aust. J. Soil Res. 29:311–330.
Neary, D.G., C.C. Klopatek, L.F. DeBano, and P.F. Ffolliott. 1999. Fire effects on belowground sustainability: A review and synthesis. For. Ecol. Manage. 122:51–71.
Neibling, W.H., G.R. Foster, R.A. Nattermann. J.D. Nowlin, and P.V. Holbert. 1981. Laboratory and field testing of a programmable plot-sized rainfall simulator. p. 405–414. *In* Erosion and sediment transport measurement, Proc. Florence Symp. IAHS Publ. 133. Int. Assoc. Hydrol. Sci., Florence, Italy.
Nicks, A.D., L.J. Lane, and G.A. Gander. 1995. Weather generator. *In* D.C. Flanagan and M.A. Nearing (ed.) USDA–Water Erosion Prediction Project hillslope profile and watershed model documentation. NSERL Rep. 10. Natl. Soil Erosion Res. Lab., West Lafayette, IN.
Onda, Y., W.E. Dietrich, and F. Booker. 2008. Evolution of overland flow after a severe forest fire, Point Reyes, California. Catena 72:13–20.
Osborn, J.F., R.E. Pelishek, J.S. Krammes, and J. Letey. 1964. Soil wettability as a factor in erodibility. Soil Sci. Soc. Am. Proc. 28:294–295.
Pannkuk, C.D., and P.R. Robichaud. 2003. Effectiveness of needle cast at reducing erosion after forest fires. Water Resour. Res. 39(12):1333, doi:10.1029/2003WR002318.
Parker, K.W. 1951. A method for measuring trend in range condition on National Forest ranges. U.S. For. Serv., Washington, DC.
Pietraszek, J.H. 2006. Controls on post-fire erosion at the hillslope scale, Colorado Front Range. M.S. thesis. Colorado State Univ., Fort Collins.
Poesen, J. 1986. Surface sealing as influenced by slope angle and position of simulated stones in the top layer of loose sediments. Earth Surf. Processes Landforms 11:1–10.
Prosser, I.P., and L. Williams. 1998. The effect of wildfire on runoff and erosion in native eucalyptus forest. Hydrol. Processes 12:251–265.
Radcliffe, D.E., L.T. West, R.K. Hubbard, and L.E. Asmussen. 1991. Surface sealing in Coastal Plains loamy sands. Soil Sci. Soc. Am. J. 55:223–227.
Renard, K.G., G.R. Foster, G.A. Weesies, D.K. McCool, and D.C. Yoder. 1997. Predicting soil erosion by water: A guide to conservation planning with the Revised Universal Soil Loss Equation (RUSLE). Agric. Handbk. 703. USDA, Washington, DC.
Rice, R.M., and J.F. Osborn. 1970. Wetting agent fails to curb erosion. Res. Note PSW-219. U.S. For. Serv., Pac. Southw. For. Range Exp. Stn., Berkeley, CA.
Rinne, J.N. 1996. Short-term effects of wildfire on fishes and aquatic macroinvertebrates in the southwestern United States. N. Am. J. Fish. Manage. 16:653–658.
Robichaud, P.R. 2000. Fire effects on infiltration rates after prescribed fire in northern Rocky Mountain forests, USA. J. Hydrol. 231–232:220–229.
Robichaud, P.R., J.L. Beyers, and D.G. Neary. 2000. Evaluating the effectiveness of postfire rehabilitation treatments. Gen. Tech. Rep. RMRS-GTR-63. U.S. For. Serv., Rocky Mountain Res. Stn., Fort Collins, CO.
Robichaud, P.R., and R.E. Brown. 1999. What happened after the smoke cleared: Onsite erosion rates after a wildfire in eastern Oregon. p. 419–426. *In* D. Olsen and J.P. Potyondy (ed.) Proc. AWRA Specialty Conf. on Wildland Hydrol., Bozeman, MT. 30 June–2 July 1999. Am. Water Resour. Assoc., Herndon, VA.
Robichaud, P.R., and R.E. Brown. 2002. Silt fences: An economical technique for measuring hillslope soil erosion. Gen. Tech. Rep. RMRS-GTR-94. U.S. For. Serv., Rocky Mountain Res. Stn., Fort Collins, CO.
Romme, W.H., T.T. Veblen, M.R. Kaufmann, R. Sherriff, and C.M. Regan. 2003. Historical (pre-1860) and current (1860–2002) fire regimes. p. 181–195. *In* R.T. Graham (ed.) Hayman Fire case study. Gen. Tech Rep. RMRS-GTR-114. U.S. For. Serv., Rocky Mountain Res. Stn., Fort Collins, CO.
Rough, D. 2007. Effectiveness of rehabilitation treatments in reducing post-fire erosion after the Hayman and Schoonover fires, Colorado Front Range. M.S. thesis. Colorado State Univ., Fort Collins.
Ruan, H., L.R. Ahuja, T.R. Green, and J.G. Benjamin. 2001. Residue cover and surface sealing effects on infiltration: Numerical simulations for field applications. Soil Sci. Soc. Am. J. 65:853–861.
Shainberg, I., and G.J. Levy. 1996. Infiltration and seal formation processes. p. 1–22. *In* M. Agassi (ed.) Soil erosion, conservation, and rehabilitation. Marcel Dekker, New York.
Shakesby, R.A., S.H. Doerr, and R.P.D. Walsh. 2000. The erosional impact of soil hydrophobicity: Current problems and future research directions. J. Hydrol. 231–232:178–191.
Soto, B., E. Benito, and F. Diaz-Fierros. 1991. Heat-induced degradation processes in forest soils. Int. J. Wildland Fire 1:147–152.
Terry, J.P., and R.A. Shakesby. 1993. Soil hydrophobicity effects on rainsplash: Simulated rainfall and photographic evidence. Earth Surf. Processes Landforms 18:519–525.
U.S. Forest Service. 1992. Soil survey of Pike National Forest, eastern part, Colorado, parts of Douglas, El Paso, Jefferson, and Teller counties. U.S. Gov. Print. Office, Washington, DC.
USGS. 2008. Water quality samples for Colorado: USGS 06737500 Horsetooth Reservoir near Fort Collins, CO. Available at nwis.waterdata.usgs.gov/co/nwis/qwdata?site_no=06737500 (accessed 15 Oct. 2008, verified 19 Apr. 2009) USGS, Lakewood, CO.
Valentin, C., and L.M. Bresson. 1992. Morphology, genesis and classification of surface crusts in loamy and sandy soils. Geoderma 55:225–245.
Vanacker, V., F. von Blanckenburg, G. Grovers, A. Molina, J. Poesen, and J. Deckers. 2007. Restoring dense vegetation can slow mountain erosion to near natural benchmark levels. Geology 35:303–306.
Wagenbrenner, J.W., L.H. MacDonald, and D. Rough. 2006. Effectiveness of three post-fire rehabilitation treatments in the Colorado Front Range. Hydrol. Processes 20:2989–3006.
Walsh, R.P.D., and P.J. Voigt. 1977. Vegetation litter: An underestimated variable in hydrology and geomorphology. J. Biogeogr. 4:253–274.
Westerling, A.L., H.G. Hidalgo, D.R. Cayan, and T.W. Swetnam. 2006. Warming and earlier spring increase western U.S. forest wildfire activity. Science 313:940–943.
Whicker, J.J., J.E. Pinder III, and D.D. Breshears. 2006. Increased wind erosion from forest wildfire: Implications for contaminant-related risks. J. Environ. Qual. 35:468–478.
Woods, S.W., and V.N. Balfour. 2008. The effect of ash on runoff and erosion after a severe forest wildfire, Montana, USA. Int. J. Wildland Fire 17:535–548.
Woods, S.W., A. Birkas, and R. Ahl. 2007. Spatial variability of soil hydrophobicity after wildfires in Montana and Colorado. Geomorphology 86:465–479.