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Quantifying long-term post-fire sediment delivery and erosion mitigation effectiveness

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ABSTRACT: Large wildfires can have profound and lasting impacts not only from direct consumption of vegetation but also longer-term effects such as persistent soil erosion. The 2002 Hayman Fire burned in one of the watersheds supplying water to the Denver metropolitan area; thus there was concern regarding hillslope erosion and sedimentation in the reservoirs. The efficacy of various treatments for reducing erosion was tested, including hand scarification on contour, agricultural straw mulch, wood mulch, burned controls and unburned reference plots. Simulated rill erosion experiments were used both immediately after the fire and again 10 years post fire. To better understand untreated recovery, the same experiments were applied to control plots in post-fire years 1, 2, 3 and 4, and in unburned reference plots in years 4 and 10. Results indicate that control and scarified plots produced significantly greater sediment flux rates – 1.9 and 2.8 g s^{-1} respectively – than the straw and wood mulch treatments – 0.9 and 1.1 g s^{-1} – immediately after the fire. Mulch treatments reduced runoff rate, runoff velocity, and sediment concentration and flux rate. The straw mulch cover was no longer present, whereas the wood mulch was still there in year 10. Vegetation regrowth was slow and mulch treatments provided effective cover to reduce sediment right after the fire. In post-fire year 10, there were no significant differences in sediment flux rates across treatments; it is notable, however, that the wood mulch treatment (0.09 g s^{-1}) most closely approached the unburned condition (0.07 g s^{-1}). The burned control plots had high sediment flux rates until post-fire year 3, when flux rates significantly decreased and were statistically no longer higher than the unburned levels from year 4 and 10. These results will inform managers of the longer-term post-fire sediment delivery rates and of the ability of post-fire emergency hillslope treatments to mitigate erosion rates. Published 2019. This article is a U.S. Government work and is in the public domain in the USA.

KEYWORDS: rill erosion; wood mulch; wood strands; scarification; Hayman Fire; straw mulch; post-wildfire; recovery

Introduction

Severe wildfires often effect landscape-level change, which creates highly disturbed conditions in watersheds. The consequences of an extensive wildfire disturbance can include increased hillslope erosion and sedimentation downstream of the burned area. The correlations between severe wildfire and subsequent increases in flooding, debris flows and sedimentation are well documented (Kunze and Stednick, 2006; Lane *et al.*, 2006; Moody *et al.*, 2008; Moody and Martin, 2009; Shakesby and Doerr, 2006; Schmeer *et al.*, 2018; Wilson *et al.*, 2018). Many historically wildfire-prone landscapes are in a repeating loop of earlier spring snowmelt, drought and other effects of climate change that are conducive to fire (Abatzoglou and Kolden, 2011); therefore, the number, size and severity of wildfires are likely to increase (Brown *et al.*, 2004; Flannigan *et al.*, 2000; Miller *et al.*, 2009; Westerling *et al.*, 2006). Additionally, the wildland–urban interface (WUI) is growing, with increases in the number of people living in and around forested areas (Theobald and Romme, 2007). In addition to the direct hazards from wildfire, secondary effects such as increased erosion and sedimentation can impact human life and safety, infrastructure, buildings, roads, and natural (aquatic habitat) and cultural resources (historically significant sites) (Emelko *et al.*, 2011; Murphy *et al.*, 2018; Rust *et al.*, 2018; Stewart *et al.*, 2003; Theobald and Romme, 2007). Consequently, when post-fire erosion risk is high, management efforts often include the prescription of mitigation treatments to minimize increases in runoff and erosion. These treatments are designed to protect public safety and reduce the potential for damage to natural and cultural resources (Robichaud *et al.*, 2010a).

Studies conducted over the past few decades have identified the most important factors that influence the likelihood and

rates of post-fire runoff and erosion, including: changes in soil hydraulic properties (Fox *et al.*, 2007; Ebel *et al.*, 2012); the degree of soil burn severity (Doerr *et al.*, 2006; Moody *et al.*, 2008) and subsequently the amount of bare soil (Benavides-Solorio and MacDonald, 2005); the rainfall intensity (Benavides-Solorio and MacDonald, 2005; Robichaud *et al.*, 2010a); the time since the fire (Gimeno-García *et al.*, 2007); and to a lesser extent the degree of post-fire soil water repellency (DeBano, 2000; Shakesby and Doerr, 2006).

Mulch treatments applied to burned soils reduce post-fire erosion by providing immediate ground cover for exposed soil, helping to protect the soil from raindrop impact and overland flow (Wagenbrenner *et al.*, 2006; Robichaud *et al.*, 2013d). Mulching is one of the most direct and effective emergency stabilization techniques used postfire (Robichaud *et al.*, 2010a; Bautista *et al.*, 2009; Lucas-Borja *et al.*, 2019). Mulches stabilize soil, reduce sediment movement, prevent loss of soil productivity and reduce the risk of flooding (Bautista *et al.*, 1996; Robichaud *et al.*, 2010a; Robichaud and Ashmun, 2013). Several researchers have suggested that at least 60% ground cover is needed to reduce post-fire hillslope erosion rates (Benavides-Solorio and MacDonald, 2005; Pannkuk and Robichaud, 2003; Robichaud *et al.*, 2010a). Other short-term studies of wheat straw mulch treatment effectiveness have reported reductions in erosion rates of 48–99% in the first two post-fire years, with the greatest reductions obtained when the wheat straw mulch provided 70% or more ground cover (Badia and Marti, 2000; Bautista *et al.*, 1996; Groen and Woods, 2008; Rough, 2007; Robichaud *et al.*, 2013a; Wagenbrenner *et al.*, 2006). In the ponderosa pine forests burned in the 2000 Bobcat Fire in Colorado, Wagenbrenner *et al.* (2006) found reduced sediment movement with increased mulch cover, and more vegetation cover on mulched areas compared to unmulched areas.

There are also potential negative effects from mulch. Some studies indicate that wheat straw mulch is susceptible to dislocation by wind (Robichaud *et al.*, 2017), which can leave exposed slopes in some areas and deep piles of straw in other areas. Thick mulch layers may prevent sunlight from reaching the soil surface and physically obstruct emerging natural and seeded vegetation (Bautista *et al.*, 2009; Beyers, 2004). In addition, agricultural straw has been found to contain seeds and can be the source of non-native vegetation introduction (Bautista *et al.*, 2009; Beyers, 2004; Kruse *et al.*, 2004; Robichaud *et al.*, 2003).

Other materials, such as hydromulches and dry mulches made from forest materials (e.g. wood strands, wood chips or wood shreds) have been developed, tested and, in some cases, applied as post-fire hillslope treatments to avoid some of the disadvantages inherent in agricultural straw mulches (Robichaud *et al.*, 2013c; Prats *et al.*, 2012, 2013, 2016a, 2016b; Wohlgemuth *et al.*, 2011). Wood-based mulches have been produced from wood manufacturing waste (e.g. wood strands such as WoodStraw®; Forest Concepts, Inc., Auburn, WA, USA); wood shreds and wood chips have come from burned timber or forest thinning and harvest operations, and shredded forest floor material have come from nearby unburned areas (Bautista *et al.*, 2009; Robichaud *et al.*, 2010a, 2013c). A clear advantage of these materials is that they are derived from forest materials and are less likely to carry non-native seeds and/or agricultural chemical residues (Foltz and Dooley, 2003). In addition, laboratory studies have established that wood strands have greater resistance to wind displacement and longer persistence on-site as compared to agricultural straw (Copeland *et al.*, 2009; Robichaud *et al.*, 2013a, 2017). Furthermore, both wood strands and wood shreds provide equal or greater protection from erosion as compared to wheat straw mulch at equal areal coverage rates (Foltz and Dooley, 2003; Foltz and Wagenbrenner, 2010; Yanosek *et al.*, 2006).

Soil scarification is a less common practice but can be used post-fire to prepare the seedbed for seeding, and to theoretically break up fire-induced soil water repellency or sealing and increase infiltration (Napper, 2006). On gentle slopes less than 20%, an all-terrain vehicle can pull small harrows on the contour to scarify the soil and, on steeper slopes, hand tools such as rakes or McLeods can be used to scarify the soil surface following the contour.

Although many studies have been done in the past two decades on post-fire treatment effectiveness, there are still few quantitative measurements of runoff and erosion in burned areas that exceed 5 years (Robichaud *et al.*, 2010a). Few long-term post-fire erosion studies have been implemented because observations and early research suggested that erosion rates decline in the first few years post fire (Moody *et al.*, 2013; Robichaud *et al.*, 2000). One example is from a study by Sheridan *et al.* (2007), who found rill erosion was 540 times greater immediately after a wildfire compared to 2 years later. However, recent studies have shown that burned sites are not always stabilized after 3 years (Robichaud *et al.*, 2008b, 2013a). Treatments therefore may need to be designed to be effective for longer post-fire periods. Longer-term effects of post-fire treatments such as straw mulch and wood mulches have only recently been studied (Bontrager *et al.*, 2019; Jonas *et al.*, 2019; Robichaud *et al.*, 2013a, 2013b).

Overland flow, begins as inter-rill (sheet) flow when rainfall or snowmelt has exceeded the infiltration rate or, less commonly in burned areas, when soils are saturated. Sheet flow can converge to rill flow, and this can occur over a short distance in steep terrain. Because of its greater depth, concentrated flow in rills has greater hydraulic power and thereby more erosive energy than inter-rill flow, and about 80% of the sediment eroded from bare hillslopes is transported in rills (McCool *et al.*, 1989; Pietraszek, 2006). Consequently, sediment delivery from steep hillslopes in disturbed forests with exposed mineral soil is likely to be dominated by rill erosion (Lei *et al.*, 1998).

The 2002 Hayman Fire provided a setting where multiple hillslope treatments were prescribed for erosion mitigation and could be compared using a rill experiment. Our goals were to assess the effectiveness of various treatments immediately after the fire and longer term (10 years), and to compare the treatments to untreated controls and unburned reference conditions to gauge recovery dynamics. The specific objectives of this project were: (1) to determine whether wheat straw mulch, wood strand mulch or hand soil scarification reduced rill erosion rates compared to untreated hillslopes immediately after the fire; (2) to determine changes in rill erosion rates over 10 years post fire and compare them to rates in unburned reference areas; and (3) to compare rill erosion rates from mulch and scarification treatments 10 years after application and determine whether ancillary factors (ground cover, water repellency or vegetation regrowth) affect erosion rates.

Methods

Study sites

Within weeks of containment of the 2002 Hayman Fire on the Colorado Front Range, four study sites were selected on hillslopes burned at high soil burn severity (Lewis *et al.*, 2006; Parsons *et al.*, 2010; USDA Forest Service, 2002) (Figure 1) with an average elevation of 2420 m and an east or northeast aspect.

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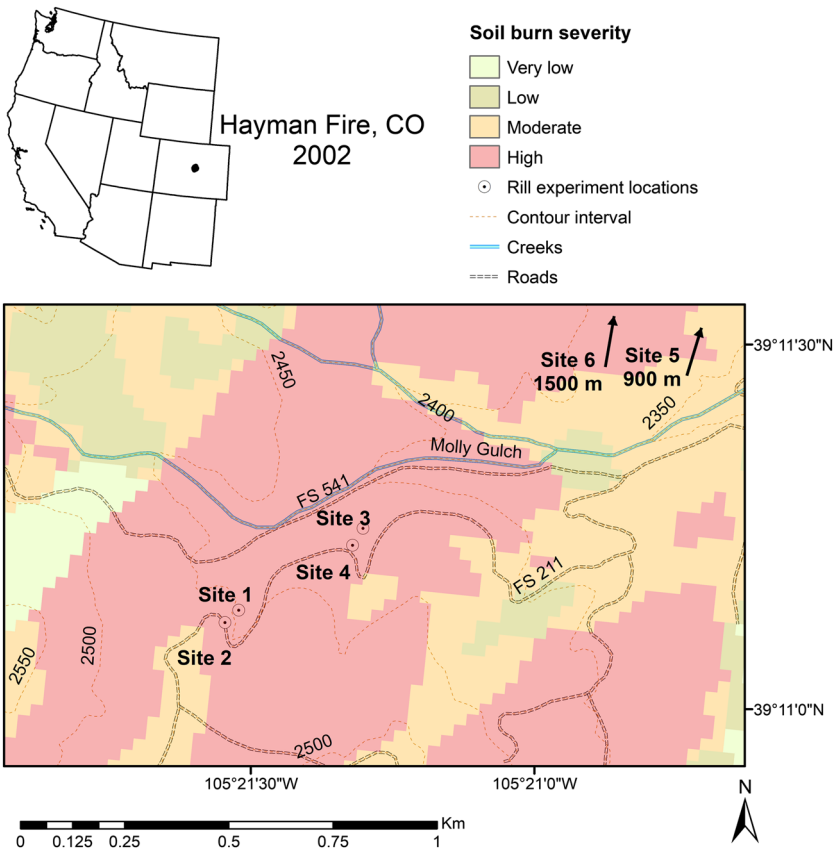

Figure 1. The Hayman Fire study sites. [Colour figure can be viewed at wileyonlinelibrary.com]

Sites were stratified by gradual (20%) and steep (40%) hillslope topography (Table I); otherwise the four sites had no apparent differences in soils, rainfall or pre-fire vegetation conditions. Each burned site had three to five replicates of randomly assigned treatments of wheat straw, wood strands, hand scarification or no treatment (controls), for a total of 16 replicates per treatment. The two unburned reference sites that were added in 2006 were also split between 20% and 40% slopes, and had the same approximate topographical and vegetation conditions as the burned sites. Short-duration high-intensity summer monsoonal rainfall is common in this region (Moody and Martin, 2009). The historic annual precipitation was derived from the Manitou Experimental Forest weather station (Asherin, 2016) (Table II).

The region is underlain by the granitic Pikes Peak batholith with frequent rocky outcroppings (USDA Forest Service, 2002). The soils are coarse textured (Robichaud *et al.*, 2003) and belong to the Legault soil series (sandy-skeletal micaceous, shallow *Typic Cryorthents*) with granitic parent material (NRCS 2010, 2011). The mean bulk density (0–5 cm depth) was 1.39 g cm^{-3}, and the clay, silt and sand fractions of surface composite soil samples (top 1 cm of soil) were 1%, 11% and 88%, respectively. Surface (~0–3 cm) soil samples were collected for gravimetric soil moisture measurement before each rill simulation (Gardner, 1986).

Table I. Site characteristics and plot counts

	Plots in 2002	Plots in 2012	Plot slope (%)	
Site	(#)	(#)	Mean	Range
1	20	4	41	37–44
2	16	4	23	22–24
3	12	4	39	37–40
4	16	4	18	17–19
5	0	4	38	37–39
6	0	4	19	17–21

The dominant tree species in this area are ponderosa pine (*Pinus ponderosa*) and Douglasfir (*Pseudotsuga menziesii*). Understory shrub and forb species include mountain mahogany (*Cercocarpus montanus*), juniper (*Juniperus* spp.), wax currant (*Ribes cereum*), Woods' rose (*Rosa woodsii*), kinnikinnik (*Arctostaphylos uva-ursi*), yucca (*Yucca glauca*), geranium (*Geranium caespitosum*) and asters (*Aster* spp.) (USDA Forest Service, 2002).

Experiment description

In 2002, rill simulations were run on 64 plots, which were randomly assigned control or treatment. The controls were left untreated and the straw plots were treated with wheat straw mulch at a rate of 2.2 Mg ha^{-1}. Wood plots were treated with wood strands (WoodStraw®) in a test mix of 3–4 mm thick wood strands in two lengths (120 and 60 mm) and two widths (4 and 16 mm) at a rate of 12.5 Mg ha^{-1}. The hand scarification treatment was completed with McCleod hand tools by raking (scarifying) the soil surface on the contour over the entire plot area. Pre-wetting the plots occurred in 2002 only by using a CSU-type rainfall simulator (Holland, 1969), where ~20 mm of rainfall was uniformly applied prior to the runoff experiment. In subsequent runoff experiments, pre-wetting was not done because of logistical constraints higher antecedent moisture conditions.

All plots were 9 m long and unbounded on the sides. Each simulated runoff experiment was conducted by releasing water through an energy dissipater at the top of each plot. The 60 min simulation included a sequence of five inflow rates (7, 22, 30, 15 and 48 L min^{-1}) that ran for 12 min each (Robichaud *et al.*,

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Table II. Mean annual precipitation at the Manitou Experimental Forest weather station (elevation 2387 m, 73 yr record, 24 km to sites) is 400 mm

Post-fire year (year)	Month sampled	Annual precipitation (mm)	Treatments sampled (number of plots)
0 (2002)	August	140	Control (16), scarified (16), straw (16), wood[a] (16)
1 (2003)	May	215	Control (16)
2 (2004)	May	343	Control (16)
3 (2005)	May	219	Control (16)
4 (2006)	May	368	Control (16), unburned (8)
10 (2012)	June	242	Control (8), scarified (8), straw (8), wood (8), unburned (8)

[a]WoodStraw® (Forest Concepts LLC, Auburn, WA, USA): test mix of 3–4 mm thick wood strands in two lengths (120 and 60 mm) and two widths (4 and 16 mm).

2010b). Overland flow velocity was measured using a dyed saline solution and two conductivity probes placed in the flow 2 and 7 m from the top of the plot during each inflow rate (King and Norton, 1992). The flow in each plot would often divide into separate rills, and the width and depth of flow in each rill were measured with a ruler to the nearest millimeter at 2 m and 7 m from the top of the plot during each inflow rate. The rill width and depth measurements were made after flow stabilized and a steady-state flow was reached. The within-flow measurements were not coordinated owing to the time required to collect each; rather, the summed total width and average depth of all rills at each location were averaged to produce a mean flow width and depth for each inflow rate by plot. The runoff and sediment from six timed samples were collected at the bottom of the plot during each flow rate when runoff flow was high enough to reach the bottom of the plot for a maximum of 30 samples total. If needed, sheet metal was used to funnel or redirect flow to the sampling point at the bottom of the plot (Robichaud *et al.*, 2010b). All samples were processed in the laboratory to measure runoff rates (L min^{-1}), sediment concentrations (g L^{-1}) and sediment flux rates (g s^{-1}).

Prior to each simulation, ground cover measurements were collected in 1 × 1 m quadrats in multiple (two or three) locations per plot and site. Ground cover was averaged across the quadrats for each plot, and combined into primary ground cover classes (live vegetation (vegetation hereafter), litter, mulch treatment) and mineral soil, which included ash and gravel < 25 mm. The measurements from the unburned plots in 2006 and 2012 were averaged into a single value for each ground cover variable to approximate the baseline unburned condition.

The presence and degree of soil water repellency at the soil surface was evaluated using the Mini Disk Infiltrometer (MDI) test (METER Group Inc, 2018 Pullman, WA, USA; Robichaud *et al.*, 2008a). The MDI tests were located in undisturbed soil adjacent to the study plots. The volume of water that infiltrated in 1 min at a suction head of 0.5 cm was recorded, and the mean of three replicates was calculated for each location. Data were classified as having: no trace of soil water repellency (MDI: >8 mL), low soil water repellency (MDI: 3–8 mL) or moderate/strong soil water repellency (MDI: <3 mL) (Robichaud *et al.*, 2008a). The control and scarified plots were tested in post-fire year 1 (2003); the control plots were also tested yearly in post-fire years 2–4 (2004–2006). For comparison, unburned plots were sampled in post-fire years 4 and 10 (2006 and 2012), and all control and treated plots were sampled in year 10 (2012).

Silt fences were installed at the bottom of one half (32) of the original plots (64) in fall 2002, and the sediment delivered from rainfall was collected after each rainfall event through August of 2009 (Robichaud *et al.*, 2013a) when the fences were removed. Alternate sets of eight control plots in each slope class were identified for rill simulations adjacent to the original plots for the following 4 years (2003–2006). In 2006, eight unburned plots were added to the 16 control plots, and they were also split between slope classes. In 2012 – the last year of the simulations – four plots of each treatment, including unburned plots, were resampled on each slope class, for a total of 40 plots (Table II). No treatments were reapplied in 2012; rather, the simulations were run on the original plots in the original locations.

Statistics

Linear mixed-effects models (Littell *et al.*, 2006) were developed in SAS 9.4 (SAS Institute, Cary, NC, USA) using the post-fire year, treatment, slope class and the potential interactions between these factors as fixed effects, and the plot nested in site as a random effect. The dependent variables were runoff width, runoff depth, runoff velocity, sediment flux rate, sediment concentration and runoff rate. Flow width and sediment flux values were $\log_{10}$ transformed to improve the normality of the residuals; other variables met normality assumptions. Untransformed means are presented in the results tables for ease of interpretation. There were no significant differences found in any dependent variables between the slope classes; therefore, the slope classes were collapsed into one and slope was no longer considered in the model. Runoff and sediment flux rates approached a steady-state condition by the fourth sample in each experimental flow rate; thus only samples 4–6 were used to compare treatments (Robichaud *et al.*, 2010b). These 'steady-state' dependent variables were averaged by plot for modeling. The flow velocities were also averaged by plot for analysis.

Similarly, linear mixed-effects models were run on the ground cover data. On all plots, the percent vegetation cover was the dependent variable and post-fire year, treatment, and potential interactions were the fixed effects. On the control plots only, total ground cover (litter + vegetation + wind blown straw) was the dependent variable and post-fire year was the fixed effect. Least significant differences were used to compare differences in Tukey-adjusted least squares means of total ground cover among post-fire years on the control plots, and of vegetation cover among the interactions between post-fire year and treatment on all plots. Results were considered significant at $p < 0.05$.

Results

Ground cover

While nearly all plots showed an increase in ground cover over the 10-year study, the most meaningful temporal trends are from the control plots, where we have six annual measurements over 10 years (Figure 2). Litter and vegetation cover doubled from 26% to 54% over the 10-year study period on the

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control plots (Figure 2), and cover on the scarified plots increased similarly from 19% to 57%. Total ground cover (which included treatment) also increased on the wood mulch plots, from 65% to 81%. More than 20% of the initial 62% wood mulch cover remained on the plots 10 years after it was applied. The agricultural straw mulch was not as persistent as the wood mulch, and was completely gone by spring of post-fire year 3 (Robichaud *et al.*, 2013a). Consequently, total ground cover in the straw mulch plots decreased from 78% to 56% over 10 years. For comparison, ground cover on the unburned plots was 85%.

There was no measurable vegetation cover immediately after the fire in 2002 on any of our plots, and at the end of the 10-year study vegetation cover ranged from 23% on the straw mulch plots to 33% on the control plots (Figure 2). Vegetation appeared to respond linearly from 2003 to 2006 on the control plots, increasing by 7% a year (Figure 2). In post-fire year 10, the wheat straw plots had significantly less vegetation than the control plots and the wood mulch plots. The results of the linear mixed-effects models indicate that time since fire was a significant control on vegetation and total ground cover on the control plots, and the corresponding exposed mineral soil ($p < 0.0001$). The effect of slope on ground cover was not significant ($p = 0.09$).

Soil water repellency

In the first post-fire year, low soil water repellency was found at the soil surface on the control and scarified plots (respective MDI means 7 and 4 mL min^{-1}); there was no difference in the means between these treatments. The most consistent measure of soil water repellency was on the control plots in post-fire years 1–4 (2003–2006) and year 10; over this period soil water repellency was low or undetectable (MDI range 4–12 mL min^{-1}). We found no significant differences among control, unburned or treated plots (MDI mean 9 mL min^{-1}, range 7–10 mL min^{-1}).

Flow variables

The mean depth of flow on the control plots increased from 5 mm in 2002 to 8 mm in 2012 (Table III). Other treatments responded similarly, with the depth of flow in the first post-fire year averaging 4–5 mm across all treatments, and increasing significantly to 8–12 mm in year 10, with the deepest flows measured on the straw plots. Conversely, the width of the rill flow generally decreased over time in the control plots, although there were few significant differences among measurements. Flow was wider on the treated plots than the controls. The wood mulch flow width decreased from 931 mm in 2002 to 367 mm in 2012, and no other width changes between these periods were significant.

Runoff velocity on the control plots was 0.26 m s^{-1} in the first post-fire year, and the mean of the scarified plots was not significantly different. The velocities in the straw and wood mulch plots were both significanly less (0.14 and 0.10 m s^{-1}, respectively; Table III) than the control plots. The runoff velocity on the control plots decreased significantly to 0.18 m s^{-1} in post-fire year 10, and there was still no difference between the scarified and control values, but there was also no longer a difference between the controls and either of the mulch treatments. Runoff velocity on the control plots was constant until 2005, when it became more variable among years (Table III). The runoff velocities on the control and scarified plots in 2002 were much greater than the unburned velocity. In contrast, compared to the unburned plots, runoff velocities on the straw and wood mulch plots were similar in the first post-fire year. The velocities in the control, scarified and straw plots were approaching the unburned value by the 10th post-fire year (2012), while the velocity on the wood plots was identical at this point in time.

The runoff flow rate was higher on the control and scarified plots (17.5 and 15.7 L min^{-1}, respectively) than on the straw and wood mulch plots (13.6 and 14.1 L min^{-1}, respectively) in post-fire year 0 (Figure 3). Runoff remained high on the control plots through the first and second post-fire years, after which it decreased significantly to 2.8 L min^{-1} in the third post-fire year (2005). By year 10, runoff on the control, scarified and straw mulch plots was significantly lower than immediately after the fire (range 2.4–2.8 L min^{-1}); however, the only treatment that was approaching the unburned runoff value of 0.5 L min^{-1} was the wood mulch (Figure 3). Overall runoff was much lower in year 10 on all plots.

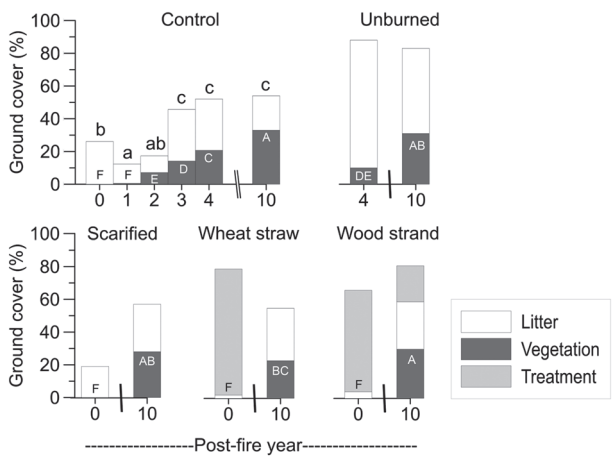

Figure 2. Mean ground cover by year and treatment. Different lower-case letters on the control graph indicate a significant difference in the total ground cover (litter + vegetation) between years. On all graphs, different upper-case letters indicate significant differences in vegetation cover across all years and treatments.

Table III. The mean and standard deviation of the flow depth, width and velocity for the unburned reference plots, control plots across different post-fire years, and control and treated plots in 2002 and 2010

Post-fire year (year)	Treatment	Flow depth (mm)	Flow width (mm)	Velocity (m s^{-1})
0 (2002)	Control	5 (1.3)d	447 (178)bcd[a]	0.26 (0.11)ab
0 (2002)	Scarified	5 (1.6)d	375 (196)def	0.24 (0.08)abc
0 (2002)	Straw	4 (2.0)d	584 (395)bc	0.14 (0.07)de
0 (2002)	Wood	4 (2.1)d	931 (412)a	0.10 (0.05)e
10 (2012)	Control	8 (2.8)bc	313 (166)cdef	0.18 (0.07)de
10 (2012)	Scarified	8 (2.0)b	299 (116)ef	0.16 (0.09)de
10 (2012)	Straw	12 (3.0)a	293 (143)cdef	0.20 (0.07)cd
10 (2012)	Wood	9 (2.2)bc	367 (201)bcde	0.12 (0.05)de
0 (2002)	Control	5 (1.3)d	447 (178)bcd	0.26 (0.11)ab
1 (2003)	Control	7 (1.9)c	287 (138)f	0.26 (0.10)ab
2 (2004)	Control	5 (1.7)d	311 (152)ef	0.26 (0.18)ab
3 (2005)	Control	5 (1.5)d	430 (240)bcde	0.14 (0.06)de
4 (2006)	Control	7 (2.4)bc	597 (250)ab	0.30 (0.11)a
10 (2012)	Control	8 (2.8)bc	313 (166)cdef	0.18 (0.07)de
Reference	Unburned	6 (4.3)	772 (185)	0.12 (0.07)

[a]Different letters within a column indicate a significant difference between year or treatment.

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Sediment variables

The sediment concentration in the control plots in 2002 was 7.8 g L^{-1}, and the sediment concentration in the scarified plots was slightly but non-significantly greater than the controls (9.6 g L^{-1}) (Figure 4). The straw and wood mulch plot sediment concentrations were both significantly lower than the control values in 2002 (2.9 and 3.3 g L^{-1}, respectively). The sediment concentration from the control plots nearly doubled to 14.6 g L^{-1} in 2003–2004, before significantly decreasing to a mean of 3.5 g L^{-1} in 2012 (Figure 4), which was the same as the scarified plots. The sediment concentrations from the straw and wood mulch plots decreased over 10 years, which resulted in none of the treatments having different values in 2012. However, only the sediment concentration from the wood mulch plots in 2012 (0.3 g L^{-1}) approached the value from the unburned plots (0.1 g L^{-1}). Interestingly, the sediment concentration on the straw mulch plots was nearly unchanged over time, remaining at about 3 g L^{-1}, which was significantly less than the control in 2002 and no different from the control and scarified plots in 2012.

The greatest mean sediment flux rate right after the fire was in the scarified plots (2.8 g s^{-1}), followed by the control plots (1.9 g s^{-1}), which was not significantly different (Figure 4).

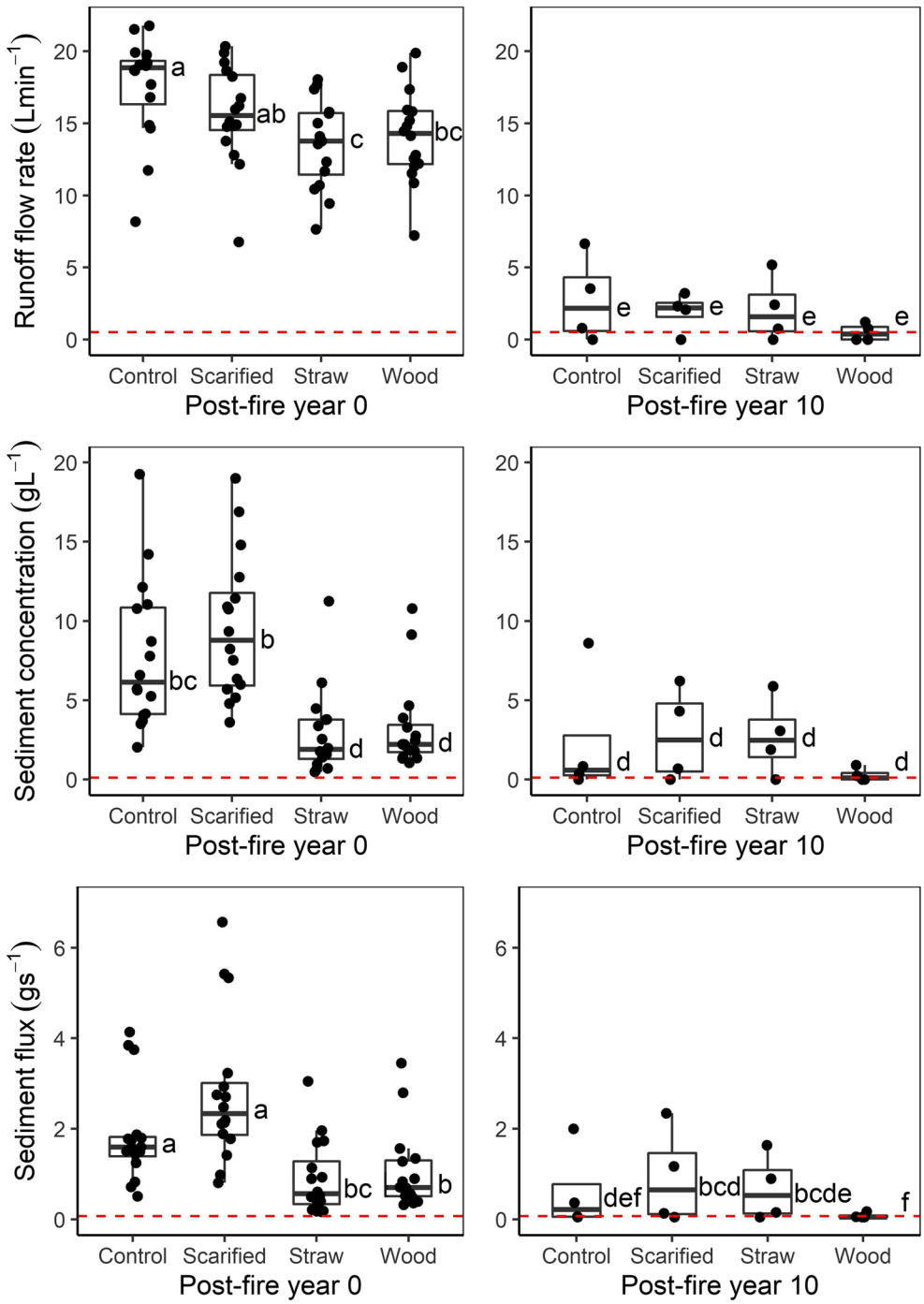

Figure 3. Runoff flow rate, sediment concentration and sediment flux by treatment. Left-hand column is post-fire year 0 (2002) and right-hand column is post-fire year 10 (2012). Dashed lines represent the unburned values. The box plots indicate 25th and 75th quantiles and the median. Within a row (e.g. runoff flow rate), significant differences between mean values across treatment and year are indicated by different letters. [Colour figure can be viewed at wileyonlinelibrary.com]

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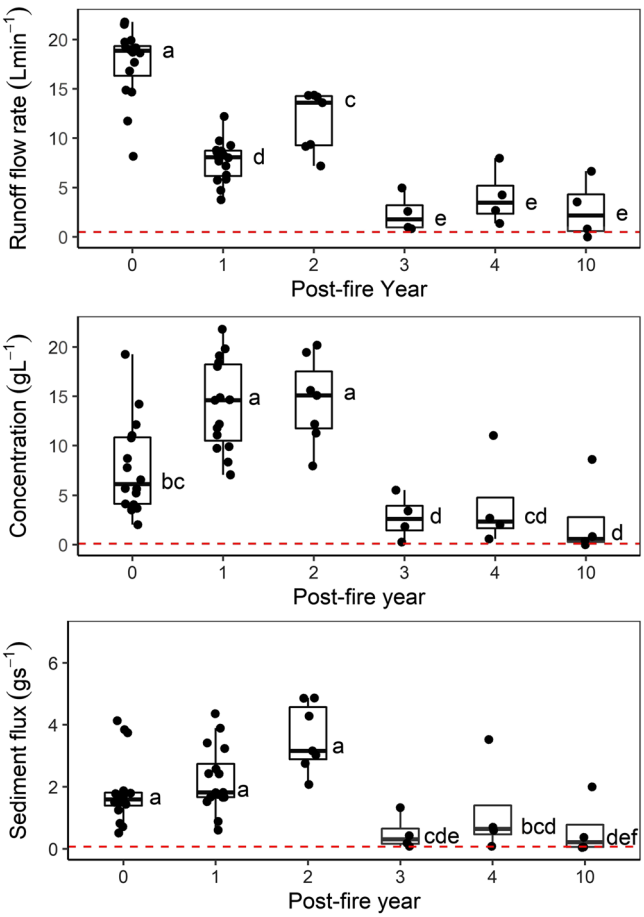

Figure 4. Runoff flow rate, concentration and sediment flux in the control plots. The dashed lines represent the unburned values. Boxplots indicate 25th and 75th quantiles and the median. Significant differences amongst dependent variable means are shown by different letters within a plot. [Colour figure can be viewed at wileyonlinelibrary.com]

The straw and wood mulch plots had initial sediment flux rates of 0.9 and 1.1 g s^{-1} respectively, both of which were significantly less than the control and scarified plots. By year 10 (2012), the sediment flux rate from the control and scarified plots decreased to about 1 g s^{-1}, while the sediment flux on the straw mulch remained at 0.9 g s^{-1}, which was not significantly different than the control and scarified plots in 2012. The sediment flux rate from the wood mulch plots decreased to 0.1 g s^{-1} by 2012; again, the wood mulch plots were most similar to the unburned plots (0.07 g s^{-1}).

Interestingly, there is an increase in both sediment concentration and sediment yield in post-fire years 1 and 2 (2003 and 2004) on the control plots. The decrease in sediment delivery is apparent in post-fire year 3 (2005), and by year 10 (2012) sediment concentration measures are still approaching the values on the unburned plots.

The shapes of the sediment flux–runoff relationships changed over time and among the treatments (Figure 5). For the control plots, in the year of the fire (2002) the apparent slope of a line fit to the data would be fairly flat due to high variability in the data, even as the runoff rate increased, suggesting the sediment detachment was limited (Figure 5). This was similarly observed on the straw mulch plots, and to some extent on the wood mulch plots. However, on the scarified plots, there was an abundance of sediment even at the highest runoff rates. The slope of the data on the scarified plots is steeper than the control and the straw mulch plots. In post-fire year 10, there were no significant differences between the treatments, including the unburned reference plots. The low sediment flux and runoff values measured on the wood mulch plots in year 10 were nearly the same as on the unburned plots, which is shown by the zoom detail in Figure 5.

Discussion

As documented in a related study using silt fences on these sites (Robichaud *et al.*, 2013a), the ground cover in the control plots did not attain the total cover values found in the unburned plots despite steady initial understory regrowth (Figure 2). Other studies focused on vegetation regeneration after the Hayman Fire found comparable understory regeneration rates in areas burned at high severity (Fornwalt and Kaufmann, 2014; Fornwalt *et al.*, 2018), but little change in the amount of bare soil or overstory cover between the fire year and post-fire year 10. Fornwalt *et al.* (2018) forecasts that surface cover will increase as sloughing bark and other vegetative material accumulate on the ground, but that pre-fire surface cover levels will take decades to achieve in this dry forest. We also theorize that a much longer period will be needed to allow overstory regeneration, which will lead to replenishment of litter and understory vegetation found in the unburned forest. The gravelly coarse-grained soils at this site also contribute to slow vegetation production and low organic accumulation in the soil (Fornwalt *et al.*, 2018; Moody and Martin, 2009).

Vegetation on this dry forested site was seemingly highly sensitive to phenological timing between years, as evidenced by the change in vegetation cover (10–30%) on the unburned

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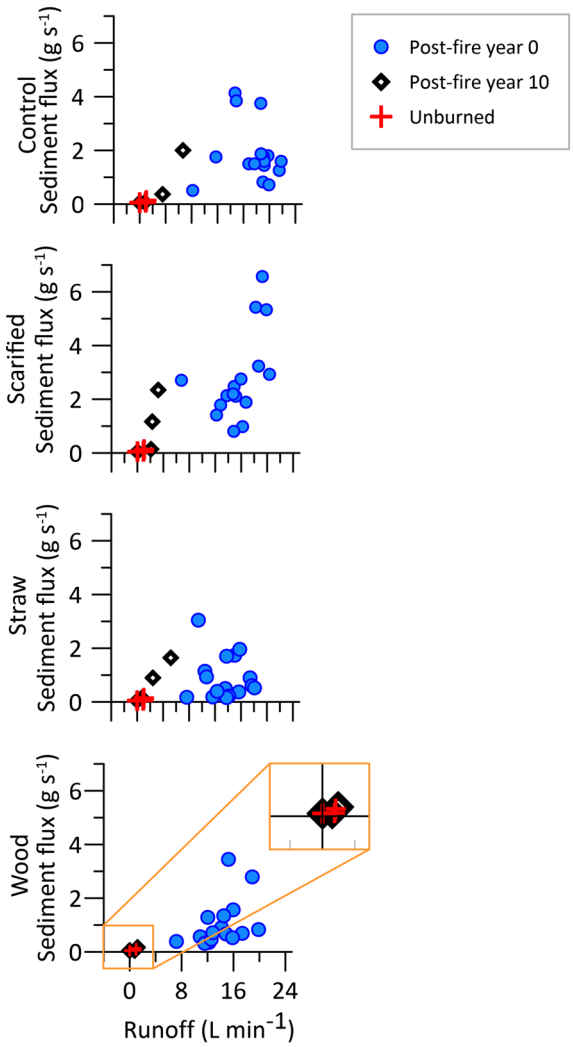

Figure 5. Sediment flux versus runoff rates in post-fire year 0 (2002) and post-fire year 10 (2012) by treatment. [Colour figure can be viewed at wileyonlinelibrary.com]

plots between the two field campaigns. The first measure of the unburned plots was in post-fire year 4 (mid-May 2006), which coincided with greater than the mean annual precipitation (Table II), and the second measure was later in the season in post-fire year 10 (mid-June 2012). Cover is often cited as the primary control on runoff and sediment yields (Larsen *et al.*, 2009), and our results support this. Sediment flux rates in the control plots were about the same until 2004, 3 years after the fire; then rilling decreased but the sediment fluxes were still higher than in the unburned plots (Figure 4). This is strongly related to the slow vegetation response after the Hayman Fire (>10 years) (Robichaud *et al.*, 2013a; Fornwalt *et al.*, 2018). These results are in contrast with other studies where decreasing trends in sediment delivery rates were observed in the first three post-fire years (Wagenbrenner *et al.*, 2016), and where longer-term sediment delivery rates in silt fence hillslope plots approached the presumed unburned condition (Robichaud *et al.*, 2013a; Wagenbrenner *et al.*, 2015).

The applied mulch cover reduced sediment delivery rates in the mulched plots as compared to the controls in the year of the fire. The straw mulch was completely gone from the plots by post-fire year 10, and Robichaud *et al.* (2013a) found most straw mulch was gone by the spring of post-fire year 3 on the silt fence plots established on these study sites. The wood strand mulch, with its larger pieces and higher density, persisted on the plots (Robichaud *et al.*, 2013a) and was still present in post-fire year 10, when we measured about 20% wood mulch cover. The amount of wood mulch remaining on the plots was likely higher than this, but some of the wood pieces were occluded by vegetation and thus not reflected in our cover assessment. There are no studies that assess the decomposition rates of wood strand mulches over relatively long time periods. One study of the decay of Douglasfir branches on the soil surface in western Oregon found an annual decay rate of 6 – 9% of branch weight per year (Fogel and Cromack, 1977). A study on the decomposition of pine wood stakes (2.5 × 2.5 × 20 cm) placed on the soil surface in the Hayman burned area showed that about 1.7% of mass loss occurs per year in the high-severity burn locations (C. Miller, Michigan Technological University, unpublished data). If we assume the more conservative rate, which better represents the conditions at our dry forest study sites, up to 84% of the original wood strand mass may still remain in our plots in the tenth post-fire year, when the observed contribution to total cover was only 20%. The moderately high coverage and longevity of the wood mulch dampened the regeneration of understory grasses and forbs in an eastern Washington burned area (Morgan *et al.*, 2014) and the nearby High Park Fire in the Colorado Front Range (Jonas *et al.*, 2019), yet the latter study measured an increase in tree seedling establishment with wood strand mulch. Our results are comparable to previous research showing that wood strand mulches of various sources and strand dimensions reduced post-fire sediment delivery (Fernández and Vega, 2014; Prats *et al.*, 2012, 2016a, b, 2019; Foltz and Wagenbrenner, 2010; Robichaud *et al.*, 2013a).

Another consequence of mulching was increased flow widths, which were attributed to the increased surface roughness of the mulched plots. This led to lower flow velocities, despite similar mean flow depths across treatments. The precision of our depth measurement was similar to the actual flow depths, which may have reduced our ability to detect any differences in flow depth among treatments in the fire year and post-fire year 10. In contrast, in the fire year, the flow widths were much greater in the mulched plots, particularly in the wood mulch plots, where the difference was significant. The lower velocities in the mulched plots also resulted in longer transit times of the runoff reaching the bottom of the plot as compared to the controls and scarified plots (data not shown). In both mulch treatments some of the energy available for soil detachment and transport was partitioned to the mulch, and presumably this also led to reduced incision and increased deposition. In the year of the fire, the net results were significantly lower sediment delivery rates as compared to the burned controls.

Several studies have demonstrated the ability of straw mulch to reduce post-fire sediment delivery rates for relatively short periods after application at multiple spatial scales (e.g. Bautista *et al.*, 1996; Fernández and Vega, 2016; Wagenbrenner *et al.*, 2006; Robichaud *et al.*, 2013a, 2013b, 2013c). The mechanisms for straw mulch reducing soil detachment and sediment delivery are similar to wood mulch outlined above, with the main difference of smaller strand sizes and reduced longevity of the straw as compared to wood mulch. The straw mulch also had a similar amount of cover to the unburned plots in the fire year, but the sediment delivery was greater in the straw plots than in the unburned areas. The elevated sediment delivery, despite 80% ground cover, was probably due to the observed lack of soil structure and the increased availability of fine sediment and ash immediately after the fire and would be similar on all of the soils affected by high severity fire. The added components of surface grain size demonstrate the complexity of predicting post-fire sediment delivery rates for a given amount of ground cover and erosive condition, and merit additional research.

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In post-fire year 10 there was no difference in velocity among the treatments, but the wood mulch plots still produced significantly lower sediment fluxes than the other treatments. In fact, the mean wood mulch sediment flux in post-fire year 10 was the only value in any year that was comparable to the unburned sediment flux and was an order of magnitude less than the control plot sediment flux rate. We attribute this to the surface protection provided by the combination of vegetation, litter and residual wood mulch, which was similar to the value in the unburned plots (Figure 2). The wood strands, given their relatively greater size and density as compared to the straw or litter produced by understory regrowth, provided additional and more persistent protection to the soil surface (Robichaud *et al.*, 2013c). We suggest the protection is derived from a combination of (1) the runoff flowing over the wood rather than the soil surface, reducing the shear stress applied to the soil surface, and (2) the wood strands increasing the roughness and flow path length, thereby reducing the velocity and total shear stress of the flow that did pass over the soil surface (Gilley *et al.*, 1991). Given the relatively low decomposition rates in the Hayman burned area, it is likely that the wood strands will continue to provide ground cover for at least the next decade, thereby contributing to reduced erosion and sediment delivery for the near future as the long recovery period for the Hayman Fire continues.

The scarification treatment had no effect on ground cover. Other research on the broader-scale scarification and seeding treatment in the Hayman burned area also found no effect of scarification on understory vegetation regrowth (Fornwalt, 2009); and, in contrast to the mulching treatments, the scarification treatment actually increased sediment flux relative to the untreated controls in the year of the fire, although not statistically. In post-fire year 10, the mean sediment flux in the scarified plots was comparable to the controls and straw mulch plots, and there appeared to be more sediment per runoff on the scarified plots (Figure 5). We suggest that the light mechanical disturbance of the scarification treatment made the soil more detachable than the control plots, and this led to slightly greater sediment flux rates. The sediment flux versus runoff relationships (Figure 5) also suggest that the control plots may have been source or detachment limited in the year of the fire, whereas the scarified plots were transport limited. In our companion silt fence study on these plots, the scarified plots had slightly lower sediment yields than the controls for the first post-fire year, but the slight difference was gone by the second post-fire year (P. Robichaud, USDA Forest Service, unpublished data), suggesting that the increase in sediment availability related to the scarification treatment was short lived. No other studies that we are aware of assessed the impact of scarification as an erosion mitigation treatment on sediment delivery rates, but another study on the Hayman Fire assessed the impact of different cover levels on sediment delivery, and the low-cover level of their study was achieved by raking (Larsen *et al.*, 2009). In that study, plots were raked (scarified) in the summers of post-fire years 2–4, and sediment delivery rates in the second post-fire year were low because of a lack of high-intensity rainfall at that site. However, in the third post-fire year, sediment delivery rates in the recently re-raked plots were much greater than the yields from the control plots in the first post-fire year when ground cover was similar. The authors also suggested that raking increased the erodibility of the soil (Larsen *et al.*, 2009).

Another conclusion from the Larsen *et al.* (2009) study was that the high sediment delivery rates after fire can be primarily attributed to a lack of ground cover rather than an increase in soil water repellency. The raking treatment was intended to 'break up the hydrophobic soil surface and will also increase infiltration' (USDA Forest Service, 2002). Disrupting the water-repellent layer has also been at least a secondary goal of other post-fire management activities, including contour trenching (Robichaud *et al.*, 2008b), contour-felled log erosion barriers (Wagenbrenner *et al.*, 2006; Robichaud *et al.*, 2008c) and post-fire salvage logging (McIver and Starr, 2001). However, evidence is building that disrupting the water-repellent soils does not necessarily lead to reduced runoff or erosion rates (Wagenbrenner *et al.*, 2015, 2016) and that other factors, particularly the amount of ground cover, can have a greater impact on post-fire hydrologic responses (Robichaud *et al.*, 2016). In this study, a wetter post-fire year 2 (2004) (Table II) may have contributed to a significant increase in vegetation growth in post-fire year 3 (2005) (Figure 2) and, subsequently, significant decreases in sediment concentration and flow rate (Figure 4).

Given the controlled flow conditions, the differences in runoff rates and sediment delivery can be attributed to changes in either ground cover or soil properties. The result that the sediment flux from the straw mulch plots did not change between post-fire year 0 (2002) and post-fire year 10 (2012), despite a lower ground cover of mulch, vegetation and litter in the later year, suggests that there was some reduction in erodibility of the bare soil during this period as well as a likely increase in infiltration. The sediment flux results from the control plots support this theory, and there was a distinct shift in the controls between post-fire year 2 (2004) and post-fire year 3 (2005). In post-fire year 1 and particularly post-fire year 2, there were a large number of sediment delivery events in the silt fence plots at this location (Robichaud *et al.*, 2013a), suggesting that the initial, unstructured surface material was eroded during this period, leaving a less erodible soil subject to the rill flow in post-fire years 3–10 (2005–2012).

Some past research has shown the dynamic nature of burned soil erodibility over a period of years (Wagenbrenner *et al.*, 2010). However, changes in soil erodibility and hydrologic responses are aspects of post-fire recovery that are not well understood, and the specific mechanism(s) for these changes are uncertain.

Flow depths, widths, and velocities and sediment fluxes at our Hayman Fire sites were similar to those measured in 4 m rill erosion plots in the first post-fire year in a burned pine forest with granitic soils (Robichaud *et al.*, 2010b). High sediment flux coinciding with slow recovery during the first few years postfire has also been observed in various sites in the western USA (Wagenbrenner and Robichaud, 2011). The effects of the agricultural straw and wood mulch treatments from other studies also show sediment flux reduction during the first few years after fire especially with faster vegetation recovery rates (Robichaud *et al.*, 2013d).

Conclusions

A rill experiment was used to compare post-fire erosion mitigation treatments immediately and 10 years after the 2002 Hayman Fire. We measured runoff and sediment delivery responses to controlled inflows for straw mulch, wood mulch and scarification treatments relative to untreated controls. Additional measurements were made in post-fire years 1–4 in burned control plots and in post-fire years 6 and 10 in unburned reference plots. Responses from untreated controls decreased in the 10 year study, but were still greater than the unburned conditions. Our results showed that straw and wood mulches reduced both runoff and sediment delivery compared to the controls in the year of the fire. The additional disturbance from the scarification treatment may have increased soil detachment, which resulted in non-significantly higher sediment flux rates as compared to the

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controls. Responses in the control plots were relatively stable until the third post-fire year, when the runoff and sediment delivery decreased. Wood mulch persisted on site, and it was still present in substantial quantities in post-fire year 10. In post-fire year 10, while there was no statistical difference in sediment flux among the treated and control plots, only the plots treated with wood mulch behaved similarly to the unburned plots. Wood mulch was the most stable of the treatments, and it persisted longer on site; runoff and sediment flux rates from the wood mulch plots nearly reached unburned levels after 10 years, and the sediment flux rate in year 10 was an order of magnitude less compared to the control plots. These results support earlier conclusions that the Hayman Fire burned area is recovering slowly compared to other locations in the western USA. Land managers may find these results useful as they plan for potentially elevated post-fire runoff and sediment delivery rates, and select treatments to reduce hillslope erosion.

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