Research Article

Post-Fire Soil Water Repellency, Hydrologic Response, and Sediment Yield Compared between Grass-Converted and Chaparral Watersheds

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ABSTRACT

In 2002, the Williams Fire burned >90% of the San Dimas Experimental Forest, providing an opportunity to investigate differences in soil water repellency, peak discharge, and sediment yield between grass-converted and chaparral watersheds. Post-fire water repellency and moisture content were measured in the winter and summer for four years. Peak discharge was determined using trapezoidal flumes with automated stage-height recorders. Sediment yields were measured by making repeated sag-tape surveys of small debris basins. Other than the high summer 2005 increase in repellency on the grass watersheds, only small differences in repellency were observed between the grass and chaparral sites. In general, soil water repellency increased with depth, decreased with time following the fire, and was inversely related to soil moisture content (i.e., least repellent during the winter and most repellent during the summer). Reduction in repellency occurred at moisture contents ranging between 8% to 16%. Approximately 85% of the sediment delivered to the debris basins occurred during the first year, with first year sediment yields being greatest in the chaparral watersheds. Peak discharge was similar for both the grass and chaparral watersheds and was highest following the record rainfall of the 2005 hydrologic year. However, only minor sedimentation followed the record rain events and was similar in both watershed types, suggesting that percent plant cover was sufficient and that the supply of easily mobilized sediment and ravel was depleted after the first post-fire winter.

Keywords: dry ravel, hydrologic response, post-wildfire, sediment yield, soil water repellency

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INTRODUCTION

Wildfires commonly occur on shrubland ecosystems linked with Mediterranean climates throughout the world. Chaparral is a shrubland plant community found primarily in California and in the northern portion of the Baja California peninsula of Mexico. In steep chaparral shrublands, fire coupled with winter rain events can increase flooding and erosion, threatening life, property, and infrastructure. Post-fire recovery, fire effects, hydrologic response, and magnitude of erosion events, however, can be highly variable depending on fire severity and extent; intensity, duration, and amount of rainfall; geology, topography, and soils; and aspect and location (Kutiel and Inbar 1993, Cerda and Lasanta 2005, Moody *et al*. 2008, Robichaud *et al*. 2008). Factors controlling post-wildfire erosion differ from site to site, and therefore a one-size-fits-all erosion model does not explain distinctive attributes of a fire-prone area (Shakesby 2011). In southern California as well as other similar regions, there is usually a first year peak of sediment following wildfire, which can remain elevated for 3 yr to 10 yr (Rowe *et al*. 1954, Florsheim *et al*. 1991, Wohlgemuth *et al*. 1999), with sediment yield amounts decreasing as vegetation becomes reestablished (Barro and Conard 1991). Important factors driving erosion in shrubland ecosystems include: loss of cover through foliage and litter consumption (Rice 1974, Wohlgemuth *et al*. 1999); loss of soil structure resulting in loose, easily detachable soil particles (Giovannini and Lucchesi 1983, Kutiel and Inbar 1993); reduced interception and exposure of bare soil to rainsplash (Farres 1987); surface sealing and pore clogging (Lavee *et al.* 1995, Neary *et al*. 1999), and soil water repellency (DeBano 1981).

In the 1960s, it was believed that type-conversion from shrublands to grass would improve fire control, provide cover, and enhance water yield (Rice *et al*. 1965). A dense cover of grass vegetation with interlocking root networks can substantially contribute to mechanical reinforcement of soils, providing protection against surficial rainfall and wind erosion (Gyssels and Poessen 2003). Following wildfire, however, Rice (1982) reported that landslides occurred on 18 % of the grass-converted land for a 32-year storm, while landslides occurred on only 0.7 % in 50-year-old chaparral. Most grass roots are shallow and do not penetrate into the underlying bedrock fractures. The deep roots of woody chaparral remain viable after fire, providing hillslope stabilization by establishing deep roots that penetrate into the fractures of the weathered bedrock, thus binding together the highly unstable rock and soil at the bedrock interface (Sampson 1944, Hubbert and Oriol 2005).

Post-fire hydrophobicity contributes to reductions in soil infiltration resulting in potential increased overland flow during rain events (Doerr *et al*. 2000). Under natural conditions, however, it is believed that water repellent soils typically alternate seasonally or over shorter intervals between repellent and non-repellent states in response to seasonal weather conditions, specifically rainfall and temperature patterns (Dekker *et al*. 1998, Doerr and Thomas 2000, Shakesby *et al*. 2000). Therefore, soil water repellency tends to increase in dry soils, limiting infiltration, while it decreases or vanishes following precipitation or extended periods of soil moisture, thus increasing infiltration capacity (Crockford *et al*. 1991, Ritsema and Dekker 1994). Repellency properties are greatly reduced or disappear at soil moisture thresholds ranging from 10 % to 13 % (Dekker *et al*. 2001, MacDonald and Huffman 2004, Hubbert and Oriol 2005). Additionally, much of the reduction in soil infiltration due to soil water repellency is minimized because of the high variability in the spatial distribution of repellency on the landscape (Hubbert *et al*. 2006, Spigel and Robichaud 2007).

Both dry erosion and water erosion can occur in wildland systems following wildfire. On steep slopes of 50 % to 60 %, even relatively small disturbances can initiate downslope movement of unconsolidated soil material as dry ravel (Anderson *et al*. 1959, Rice 1974). Colluvial material trapped in litter and behind living and dead biomass is liberated as the litter and biomass are consumed by fire and redistributed downslope by gravity. It is eventually delivered to the drainage channel (Krammes 1960, Hubbert and Oriol 2005). As a result, intermittent and ephemeral stream channels become loaded with sediment, which becomes mobilized during the first major storm event that generates sufficient channel flow (Rice 1982). In parts of southern California, dry ravel movement accounts for over half of all hillslope erosion, independent of fire (Anderson *et al*. 1959, Krammes 1969, Rice 1982). Infiltration excess occurs when rainfall intensity exceeds the soil infiltration rate, generally under high rainfall intensities and where soil infiltration has been reduced (Horton 1945). Saturation overland flow occurs when the water storage capacity of the regolith is exceeded (Anderson and Burt 1990). Storage capacity is often low in southern California shrublands that are noted for their shallow soils (Bailey and Rice 1969). When fire removes the aboveground biomass, soil moisture increases since transpiration and interception are negligible, thus allowing near-saturation of soil to be reached much sooner during rain events (Rowe and Colman 1951).

In 2002, the Williams Fire burned >90 % of the San Dimas Experimental Forest (SDEF), providing an opportunity to investigate differences in soil water repellency, peak discharge, and sediment yield between grass-converted and chaparral watersheds. The primary objectives of this study were to compare post-fire changes in soil water repellency, vegetation cover, hydrologic response, and sediment yield between mixed chaparral and type-converted grass watersheds over a four year period. Specifically, the objectives were to: 1) assess whether type conversion of chaparral to grass would change the persistence of repellency on the landscape; 2) assess how vegetation recovery would affect hydrologic response between the grass and chaparral sites; and 3) compare temporal changes in sediment yield between the grass and chaparral watersheds. Accelerated post-fire runoff and erosion following wildfire can overwhelm the ephemeral and intermittent stream channels in the headwater tributaries, scouring channels and generating floods and debris flows, resulting in the loss of life, property, and structures located in and around natural debris basins and drainages (Munns 1920, Kraebel and Sinclair 1940, Wells 1987). In this regard, results from this study will aid federal, state, county, and municipal land and watershed managers worldwide who must be able to quantitatively predict the effects of post-fire management actions on the hydrologic response and subsequent sediment yield for shrubland watersheds.

METHODS

Environmental Setting

The SDEF (part of the Angeles National Forest) comprises 6947 ha in the foothills of the San Gabriel Mountains, located north of Glendora, California, USA, approximately 58 km northeast of Los Angeles, California (Figure 1). The forest is characterized by rough topography of deeply cut channels, steep slopes up to 76 %, and elevations ranging from 396 m to 1737 m. The climate is Mediterranean with cool, wet winters and hot, dry summers. Mean temperatures (monthly) range from 8 ºC to 40 ºC during the year (Crawford 1962). Annual precipitation values for the period 1987 to 2011 have ranged from 211 mm to 1443 mm with a mean of 664 mm. Following a hot, dry summer, the Williams Fire burned >90 % of the SDEF (~6880 ha) from 22 September to 2 October, 2002, consuming almost 100 % of the vegetation (Napper 2002). Absence of wind, indicated by a smoke plume that rose straight up, allowed the fire to burn

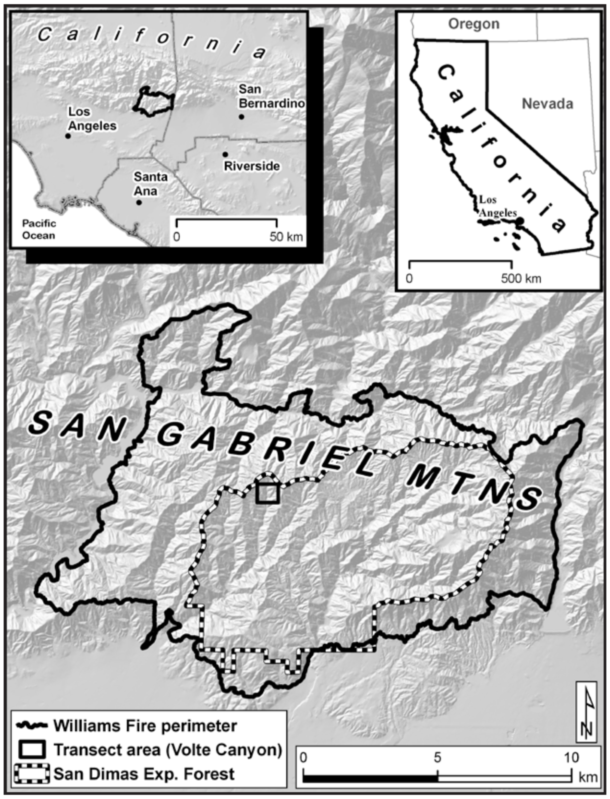

Figure 1. Site map showing study area, the Williams Fire, and San Dimas Experimental Forest boundaries, and their general location relative to California, USA.

relatively slowly. This allowed the fire to spread faster upslope, and slower when backing downslope (Weise and Biging 1997). The BAER (burned area emergency response) team reported that the majority of the Williams Fire burned at moderate-to-high severity, with the portion containing the SDEF burning at high severity. We derived our own assessment by calculating the Relative differenced Normalized Burn Ratio (RdNBR; Miller and Thode 2007) from an immediate pre-fire Landsat 5 pass from 20 September 2002, two days before the fire started, and the next pass of the same instrument on 6 October, by which time the fire was out. Both scenes are cloud-free, ready for use, and free of charge from the USGS GLOVIS (http://glovis.usgs.gov/) program. To get a sense of reported RdNBR values in the literature, note that Miller and Thode (2007) regard a value of 650 as the threshold for high severity (for fourteen forest fires in the Sierra Nevada), and above 750 resulted in 100% tree mortality in the work of Miller *et al.* (2009). In comparison, the average RdNBR value for the Williams Fire is 1111. Within our transect area, 99% of the pixels are above 1000 in RdNBR value. Our RdNBR analysis supports the on-the-ground observations of a high-severity fire event.

The wildfire consumed most of the standing biomass and litter, leaving only shrub skeletons and a mixture of surface ash and disturbed soil (Figure 2; Napper 2002, Hubbert and Oriol 2005).

Figure 2. Dry ravel deposited directly in channel immediately following wildfire.

Over the last million years, tectonic activity has intensely faulted and fractured the San Gabriel Mountains, resulting in steep, rugged, and unstable slope topography. Metamorphic and plutonic basement rocks, consisting of banded gneisses and schists, metamorphosed quartz monzonite, and intermixed igneous dike rocks, form the bulk of the mountains (Storey 1948, Lave and Burbank 2004). Extensive fracturing has allowed deep weathering of the rock, providing substantial storage capacity for water. Exposed rock weathers rapidly, contributing erodible sand, gravel, cobble, and stonesize material to the surface (Sinclair 1953). Soils that form on the steep slopes are shallow, coarse loamy sands, which are excessively drained, and exhibit little profile development (Williamson *et al*. 2004, Hubbert *et al*. 2006). Soil type is dominated by Typic Xerorthents (Ryan 1991, Hubbert *et al*. 2006), but Typic Haploxeralfs are also common (Williamson *et al*. 2004). Because of the high degree of weathering, boundaries between the soil and underlying parent material are often indistinguishable (Crawford 1962).

Native vegetation in the SDEF is composed primarily of mixed chaparral characterized by sclerophyllous leaves, 1 m to 4 m plant height, and dense canopies. Common chaparral species include chamise (*Adenostoma fasciculatum* Hook & Arn.), hoary-leaf ceanothus (*Ceanothus crassifolius* Torr.), sugar bush (*Rhus ovata* S. Watson), Eastwood manzanita (*Arctostaphylos glandulosa* Eastw.), bigberry manzanita (*Arctostaphylos glauca* Lindley), scrub oak (*Quercus dumosa* Nutt.), black sage (*Salvia mellifera* Greene), wild buckwheat (*Eriogonum fasciculatum* Benth.), and yerba santa (*Eriodictyon trichocalyx* A.A. Heller). At the time of the Williams Fire, stand age of the chaparral was 42 years, with the watershed last burning during the 1960 Johnstone Fire that consumed 88 % of the forest. Following the Johnstone Fire of 1960, researchers selected replicate watersheds similar in size, shape, aspect, and potential erodibility, and subjected them to a variety of post-fire rehabilitation treatments (including seeding with perennial grasses and leaving controls consisting of native chaparral alone). Grasses included wheatgrass (*Agropyron* spp.), barley (*Hordeum vulgare* L.), perennial veldt grass (*Ehrharta calycna* Sm.), big bluegrass (*Poa ampla* Merr.), smilo grass (*Piptaherum miliaceum* [L.] Coss.), and blando brome (*Bromus hordaceous* L.). It was believed that type-conversion would improve fire control and enhance water yield (Rice *et al*. 1965). Perennial veldt grass made up only 15 % of the original seed mixture (Williamson *et al*. 2004). By 1964, veldt grass had become the predominant species in the watersheds that had been part of the perennial grass treatments. By 2002, buckwheat and sage had also established on the type-converted watersheds.

Sampling Design and Field Methods

Each watershed was instrumented with a trapezoidal flume to measure discharge, and a debris basin was constructed to capture sediment yield (Rice *et al*. 1965). Six representative watersheds were selected from the above group: three in previously type-converted grass vegetation and three in native mixed-chaparral (Figure 2). Existing field installations were refurbished and comparative data-sets of post-fire runoff and sediment yield from the 2002 wildfire were acquired over the four-year period from 2003 to 2006. The stilling wells of the trapezoidal flumes were instrumented with a float and pulley water level recorder. The rating curves of the flumes were then used to compute flow discharge from stage height water levels from 2002 to 2006. The sediment wedges in the debris basins were surveyed (sag tape surveys of permanent cross sections), excavated to increase reservoir capacity, and then the basins were resurveyed in November 2002 to establish new baselines (Ray and Megahan 1978). Sediment accumulations were resurveyed to calculate sediment yield during January 2003, May to June 2003, January 2004, June 2004, May 2005, June 2006, and June 2007. Therefore, sediment yields were cumulative following the fire to the first resurveying, and cumulative following each subsequent resurveying. A tipping bucket rain gauge with data logger was installed at the two study site and collected rain data from November 2002 to April 2007. Total rainfall duration, amount, and 15 minute rainfall intensities (I_{15}) were calculated for each rain event.

Vegetation sampling was conducted during the spring from 2003 through 2006. Each of the six watersheds was divided longitudinally into three equal sections (upper, mid, and lower drainages). Three horizontal transect lines were located randomly within each section. Ten 10 m line transects were randomly placed along the horizontal lines, yielding 30 line transects per watershed section. Shrub, subshrub, forb, and grass cover was measured along the 10 m line transects. Litter and bare ground were also measured along each transect. To augment March 2003 vegetative cover not measured using transects, satellite imagery was used, specifically NDVI (Normalized Difference Vegetation Index), which is a spectral band ratio that is a simple and widely used index of vegetation greenness and cover (Henry and Hope 1998, Nagler *et al*. 2001). Average NDVI (calculated for pixels) was determined within the watershed perimeters pre- and post-fire from 2002 through 2006 using all available cloud-free Landsat 5 satellite scenes.

Soil water repellency was measured at each of the 10 m vegetation line transects using the water drop penetration time method (WDPT) (Krammes and DeBano 1965). Measurements were conducted twice per year (late winter and midsummer) from 2003 through 2006. Because of personnel changes, the March 2004 measurements were not conducted. Twenty water drops were placed on the mineral soil surface and at the 2 cm depth within a 30 cm square area (900 cm^2). Another 10 water drops were placed at the 4 cm depth. Drop penetration time was measured with a stop watch and the times were aggregated to yield the following classification scheme: wettable, 0 s to 5 s; slightly water repellent, 5 s to 30 s; and moderate to highly repellent, >30 s (Hubbert and Oriol 2005). For every water repellency location, soil moisture samples were taken at 0 cm to 5 cm and 5 cm to 10 cm depths and ambient soil wetness was measured gravimetrically after oven drying (Gardner 1986). Repellency measurements immediately following the fire and for March 2003 were collected from adjacent watersheds highly similar to the study watersheds.

Terrain attributes were described at randomly selected vegetation sampling points. Attributes included: slope (local), slope (profile), slope length, slope shape, aspect, hillslope position, geomorphic component, soil depth, and surface cover (rock). Soil depth to weathered bedrock was sampled by auger (n = 99 for chaparral and n = 98 for grass). Depth of weathered bedrock was estimated by observing exposed roadcuts and contour trails located in or adjacent to subject watersheds. To determine soil and weathered bedrock water storage capacity, soil water characteristic curves (drying water release curve) were obtained for both the soil (n = 24) and the weathered bedrock (n =13) using a combination of pressure plate, suction table, and hanging water column apparatus. The samples used for the suction table and pressure plate methods were weighed after they reached equilibrium at each matric potential. Volumetric water content and bulk density of these samples were then determined by oven drying at 105 ºC for 24 h. For the purpose of this paper, soil and weathered bedrock storage capacity were calculated as the difference between field capacity and oven-dry moisture contents (Gregory *et al*. 1999). Previous research conducted by Williamson *et al*. (2004) and Hubbert *et al*. (2006) provided data for soil infiltration rate, soil temperature, saturated hydraulic conductivity (soil and weathered bedrock), soil bulk density, color, structure, and particle size.

Unpaired student *t*-test P values were calculated comparing sediment yield, peak flow, total cover, shrubs and sub-shrubs, forbs, and grasses between grass-converted and mixed chaparral watersheds for 2003, 2004, 2005, and 2006. Values were considered significant at $P < 0.05$.

RESULTS

Yearly Precipitation and Intensity, Soil-Rock Water Storage Capacity, and Landscape Topography

Annual precipitation totals were 609 mm in water year (WY) 2003, 406 mm in WY 2004, 1863 mm in WY 2005, and 675 mm in WY 2006, compared to a long-term mean of 678 mm. During the 2003 hydrologic year, I_{15} values remained <5 mm hr^{-1}, except for the 8 November 2002 rain event with an I_{15} value of 5.1 mm hr^{-1} (Table 1). The highest I_{15} value reached in 2004 was 6.4 mm hr^{-1}. In the 2005 hydrologic year, I_{15} values were >12 mm hr^{-1} for two dates, and were probably >12 mm hr^{-1} for large rain events that occurred between 8 December 2004 and 10 January 2005, when the gauge malfunctioned (Table 1).

Soil depth was slightly greater for the grass-converted watersheds as compared to the chaparral watersheds; however, depth of soil plus weathered bedrock was similar between

Table 1. Rain events, duration, 15 minute rainfall intensity (I_{15}), and total event rainfall amount for 2003, 2004, and 2005 hydrologic years. Table represents rain events with total amounts >2.5 cm. No rainfall events <2.5 cm resulted in I_{15} >3 mm hr^{-1}.

Rain event start and end date	Duration (hr)	I_{15} (mm hr^{-1})	Total amount (mm)
2003 hydrologic year			
8 Nov to 9 Nov, 2002	47	5.1	110
16 Dec to 16 Dec, 2002	10	2.5	51
20 Dec to 20 Dec, 2002	11	2.0	32
10 Feb to 13 Feb, 2002	72	3.8	103
24 Feb to 28 Feb, 2002	79	2.0	42
14 Mar to 15 Mar, 2003	24	3.8	114
14 Apr to 15 Apr, 2003	26	2.0	57
2 May to 2 May, 2003	27	2.0	38
2004 hydrologic year			
24 Dec to 25 Dec, 2003	25	5.1	114
31 Jan to 1 Feb, 2004	10	6.4	38
21 Feb to 23 Feb, 2004	50	2.0	47
24 Feb to 25 Feb, 2004	12	2.5	92
2005 hydrologic year			
17 Oct to 21 Oct, 2004	90	7.6	280
25 Oct to 26 Oct, 2004	26	4.6	79
21 Nov to 22 Nov, 2004	6	3.8	35
8 Dec to 10 Dec, 2004	na[a]	na	178
16 Dec to 18 Dec, 2004	na	na	156
8 Jan to 10 Jan, 2005	na	na	533
28 Jan to 31 Jan, 2005	72	5.1	38
9 Feb to 11 Feb, 2005	41	4.6	75
17 Feb to 20 Feb, 2005	74	12.7	182
27 Feb to 1 Mar, 2005	65	6.4	146
23 Mar to 23 Mar, 2005	17	5.1	29
27 Apr to 28 Apr, 2005	6	2.5	30
6 May to 7 May, 2005	15	12.7	37

[a] Gauge malfunction data not available. Only total rainfall amounts collected.

the chaparral and grass-converted watersheds (Table 2). Total mean water storage capacity was similar in the two watersheds: 30.4 cm in the chaparral as compared to 33.5 cm in the grass-converted (Table 2). The presence of convex slope positions in both chaparral and grass-converted watersheds was almost four times greater than concave slope positions. Linear slope positions were similar between chaparral and grass-converted watersheds (Table 3).

Post-Fire Soil Water Repellency—Surface, 2 cm, and 4 cm Depths

Soil surface moderate-to-high repellency was measured at 47% immediately following the wildfire in a mixed chaparral watershed adjacent to the study watersheds. In the same watersheds, the proportion of surface moderate-to-high repellency declined sharply to ≤5% at soil wetness >12% for both grass-converted and chaparral watersheds following

Table 2. Soil and bedrock hydrologic properties calculated for chaparral and grass watersheds.

Substrate	PAW[a] (cm^3 cm^{-3})	WS[b] (cm^3 cm^{-3})	Depth[e] (cm)	SC[c] (cm)	Soil + Cr[d] (cm)
Chaparral watershed					
Soil	0.065	0.376	34	12.8	
Weathered bedrock (Cr)	0.076	0.160	110	17.6	30.4
Grass watershed					
Soil	0.123	0.427	39	16.7	
Weathered bedrock	0.076	0.160	105	16.8	33.5

[a] PAW = plant available water.
[b] WS = water storage.
[c] SC = storage capacity.
[d] Soil + Cr = Soil and weathered bedrock storage capacity (Depth × WS).
[e] Soil depth: n = 98 grass, n = 99 chaparral.

Table 3. A comparison of site attributes and peak discharge between grass-converted and native chaparral watersheds.

Watershed	Area[a] (ha)	Average slope (%)	Average slope length (m)	Total length channels[b] (m)	Drainage density[c] m m^{-2}	Basin length[d] (m)	Relative relief (m)
Chaparral	1.9	58	18	583	0.030	350	93
Grass	2.7	58	24	636	0.026	241	72

	Slope shape[e]			Peak discharge[f] (m^3 s^{-1} ha^{-1})			
	Convex	**Linear**	**Concave**	**2003**	**2004**	**2005**	**2006**
Chaparral	42	47	11	0.37	0.10	0.43	<0.01
Grass	46	43	11	0.39	0.09	0.43	<0.01

[a] Area from perimeter survey corrected to horizontal.
[b] Total length of channels is the average sum of lengths of main stem and all tributaries.
[c] Drainage density is the total length of channels divided by watershed area.
[d] Basin length calculated from check dam to watershed divide along main stem.
[e] Vertical and horizontal slope shapes. Convex includes: convex-convex, convex-linear, and linear-convex. Concave includes: concave-concave, concave-linear, and linear-concave. Linear includes: linear-linear.
[f] Average cubic meters per second per hectare.

rain events of the 2002-2003 winter. From July 2003 to July 2006, an increase in moderate-to-high repellency was observed during the dry summers at soil moistures <2%, and a decrease in repellency was observed for moist soils >8% measured during the late winter (Figure 2). Following above average rainfall of the 2004-2005 winter, moderate-to-high repellency recorded in July 2005 increased to 11% in the grass-converted watershed, whereas there was only a slight increase of 3% in the chaparral watershed (Figure 3).

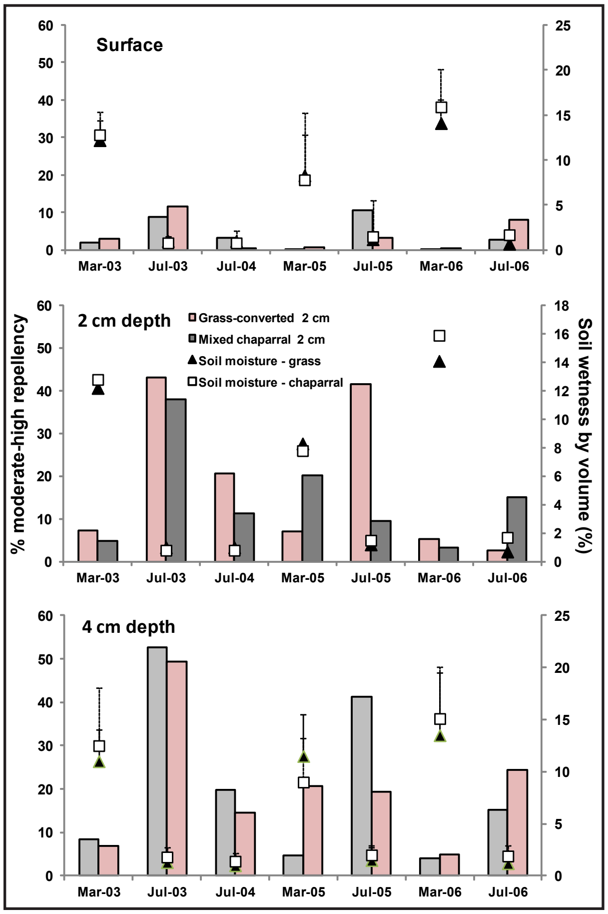

Figure 3. Seasonal fluctuations of soil water repellency and soil wetness measured at the soil surface, 2 cm, and 4 cm depths from November 2002 to July 2006 in grass-converted and mixed chaparral watersheds. Measurements for 2 November 2002 and 3 March 2003 were taken in similar watersheds adjacent to the study watersheds. Error bars for soil wetness represent one standard deviation of the mean. Error bars not shown for 2 cm depth as they are the same as the soil surface.

Wetting of the soil during 2002-2003 winter reduced moderate-to-high repellency at the 2 cm depth from 56% immediately following the fire to <7% for both grass and chaparral watersheds. As soil moisture declined to <2% during the summer of 2003, moderate-to-high repellency returned, increasing to 43% in the grass and 38% in the chaparral watersheds. From July 2003 to July 2006, seasonal patterns and percentages of moderate-to-high repellency remained similar between the 2 cm and 4 cm depths for both the grass and chaparral watersheds. Moderate-to-high repellency increased to 41% at both 2 cm and 4 cm depths in the grass-converted watersheds at soil moistures <2% in July 2005, but was only 9% at the 2 cm depth and 19% at the 4 cm depth in the chaparral watersheds (Figure 3).

Post-Fire Hillslope Erosion, Sediment Yield, and Hydrologic Response

First year 2003 mean sediment yields of 43 t ha^{-1} in the chaparral watersheds were not significantly greater than the 32 t ha^{-1} recorded for the grass-converted watersheds (P = 0.395; Figure 4; Table 4). Post-fire observations revealed that soil heating reduced the weak subangular blocky structure of the surface soil to structureless, single grain soil components. Wind events shortly following the wildfire removed and redistributed the ash layer and loose soil material (Figure 2). Additionally, cones of dry ravel were observed being deposited at the base of hillslopes, in channels, and on roads during and immediately following the wildfire (Figure 2). Plant cover >60% and below average precipitation of low intensity (Table 1) during the 2004 hydrologic year contributed to the large decrease in sediment collected during the second year from the grass-converted (0.5 t ha^{-1}) and chaparral watersheds (0.2 t ha^{-1}) (Figure 4). Even though record amounts of rainfall of higher intensity were recorded for the 2005 hydrologic year, sediment production remained relatively low during 2005, re-

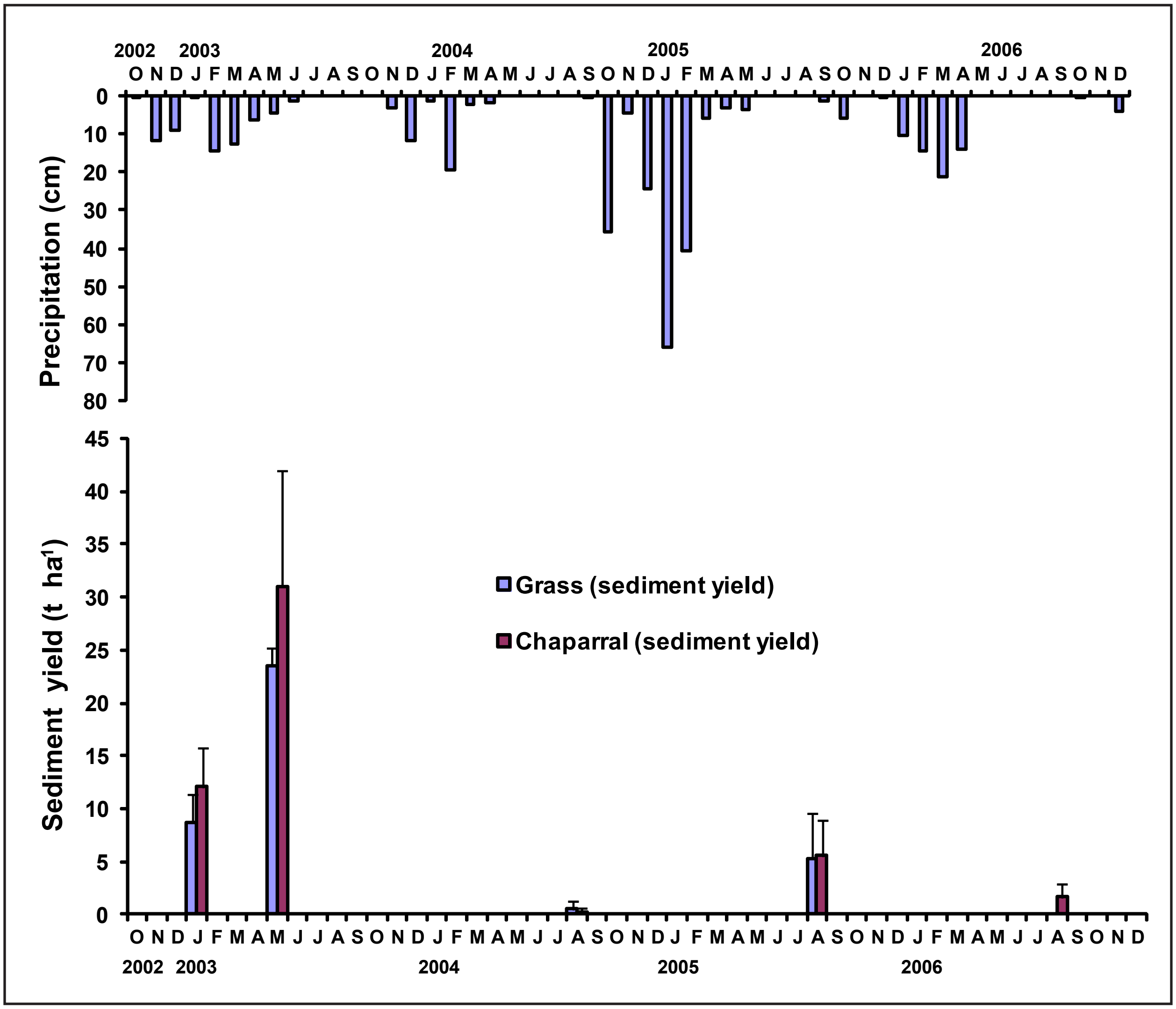

Figure 4. Mean sediment yield (m^3 ha^{-1}) compared between mixed chaparral and grass-converted watersheds from the winter of 2002-2003 through the year 2007. Precipitation (mm) measured from October 2002 through September 2007. Error bars for sediment yield represent one standard deviation of the mean.

Table 4. Rain events, duration, 15 minute rainfall intensity (I_{15}), and total event rainfall amount for 2003, 2004, and 2005 hydrologic years. Table represents rain events with total amounts >2.5 cm. No rainfall events <2.5 cm resulted in I_{15} >3 mm hr^{-1}.

Year	Sediment yield	Peak flow	Total cover	Shrubs and subshrubs	Forbs	Grass
2003	0.395	0.676	0.711	0.127	0.010*	0.010*
2004	0.644	0.770	0.714	0.880	0.041*	0.021*
2005	0.777	0.937	0.332	0.484	0.078	0.010*
2006	na	na	0.809	0.032*	0.601	0.033*

* Gauge malfunction data not available. Only total rainfall amounts collected.

sulting in the highest peak discharges (0.43 m^3 s^{-1} ha^{-1}) observed during the study (Figure 4; Table 1; Table 3). Low plant cover contributed to 2003 high peak discharges and were not significant between watershed types (P = 0.676) despite below average rainfall and low intensities. During 2004, peak discharges were low as precipitation dropped below normal. There were no significant differences in sediment production or peak discharge between watershed types for either 2004 or 2005 (Figure 4; Table 4).

Vegetation

Herbaceous plants (grasses and forbs) dominated the initial recovery in both type watersheds following the fire. March 2003 plant cover was >30 % and >50 % by mid April 2003 for both the chaparral and grass watersheds (Table 5). By August 2003, plant cover was 61 % in the chaparral and 57 % in the grass watersheds, with no significant differences between the two. From 2004 to 2006, total plant cover remained fairly constant, with no significant differences observed between watershed types (Table 4; Table 5). As shrub and subshrub cover increased into the third year, grasses and forbs declined, resulting in an increase in litter cover (total cover approaching 90 %). Forb cover differed significantly from shrubs and sub-shrubs between the chaparral and grass-converted watersheds in 2003 and 2004 (P = 0.010 and 0.041, respectively). Grass cover also differed significantly from shrubs and sub-shrubs for the two watershed types for the years 2003, 2004, and 2005 (P = 0.010, 0.021, and 0.010, respectively; Table 4). From 2005 on, all watersheds appeared to be reverting back to their pre-fire communities (Table 5). No veldt grass was observed on the chaparral sites, but it made up ~43 % of the total grass cover on the grass-converted sites from 2003 through 2006.

Table 5. Post-fire percent plant cover compared between chaparral and type-converted grass watersheds from 2003 to 2006.

	Mar 2003[a]	Apr 2003[a]	Aug 2003	Jul 2004	Jul 2005	Jul 2006
Chaparral						
Grass	na[c]	na	1	3	1	3
Forbs	na	na	48	46	5	12
Subshrubs[b]	na	na	3	8	26	19
Shrubs	na	na	10	11	18	27
Bare ground	na	na	38	15	12	12
Litter	na	na	0	17	39	27
Total plant cover (live)	38	54	61	68	50	61
Plant cover + litter	na	na	61	85	89	88
Type-converted grass						
Grass	na	na	18	31	13	13
Forbs	na	na	30	12	2	2
Subshrubs[b]	na	na	5	4	32	29
Shrubs	na	na	4	3	9	9
Bare ground	na	na	43	21	13	12
Litter	na	na	0	19	31	35
Total plant cover (live)	34	51	57	60	56	53
Plant cover + litter	na	na	57	79	87	88

[a] Values determined using mean values of Normalized Difference Vegetation Index.
[b] Subshrubs = small bushy plants that are woody except for the tips of the branches.
[c] Not available.

DISCUSSION

Soil Water Repellency

Soil water repellency was inversely related to soil moisture content, being lower during the winter wet seasons and the highest during the summer dry season. Water repellency followed this trend in both watershed types over the four-year sampling period at the surface and at the 2 cm and 4 cm depths. At the soil surface, fire effects on soil water repellency were very short-lived and did not return to the high levels observed immediately following the wildfire, or to levels observed pre-fire. Moderate-to-high repellency measured prior to the William Fire was 37% on mixed chaparral sites and 47% immediately following the wildfire in the SDEF (Hubbert *et al*. 2006). Both chaparral and grass watersheds exhibited rapid reduction in repellency at winter soil moisture contents ranging from 8% to 16%, suggesting a critical soil moisture threshold demarcating water-repellent and non-repellent conditions (Figure 3). A number of soil moisture thresholds have been reported (Doerr and Thomas 2000, MacDonald and Huffman 2004, Hubbert and Oriol 2005), and appear to be dependent on differences in soil properties, vegetation type, and the organic molecular structure of the hydrophobic compounds (Doerr *et al*. 2000). Because of the reduced repellency during the winter months, we assumed that water repellency played only a minor role in water erosion.

Moderate-to-high repellency returned to >38% at the 2 cm and 4 cm depths during the 2003 summer dry season, but was <12% at the surface in both the chaparral and grass watersheds (Figure 3). Many studies have reported soil water repellency returning as soils dry during the summer (Crockford *et al*. 1991, Dekker *et al*. 1998, Shakesby *et al*. 2000). However, Doerr and Thomas (2000) have suggested that repellency is not always re-established when soils become dry after wetting. Some factors explaining the lower values of dry season repellency at the soil surface include: (1) transport to and recondensation of hydrophobic compounds at the 2 cm and 4 cm depths during the fire (DeBano *et al*. 1979), (2) biologic productivity—lack of chaparral vegetation providing new hydrophobic compounds (Teramura 1980, Doerr and Thomas 2000), (3) soil erosion caused by wind and gravity, (4) soil bioturbation (Bond 1964, Hubbert *et al*. 2006, Jackson and Roering 2009), (5) downward leaching of hydrophobic compounds (Doerr *et al*. 2000), and (6) spatial distribution of soil moisture (Hubbert and Oriol 2005). Summer repellency levels >40% at the 2 cm and 4 cm depths in both the chaparral and grass watersheds could help generate overland flow during the first fall and winter rain events. Woods *et al.* (2007) suggested a threshold ranging from 35% to 75%, at which water repellency can be spatially contiguous and able to instigate overland flow. However, they noted that below these percentages, patches of repellency would not connect laterally and any generated infiltration excess flows would infiltrate near their point of origin.

The largest difference in moderate-to-high repellency between the grass and chaparral watersheds occurred at the 2 cm and 4 cm depths at the July 2005 sampling, which suggested that a source of repellency occurred in the grass watersheds that was not present in the chaparral. Much of the increase was due to an increase in root production by perennial veldt grass due to the above average 2005 water year (Figure 3). Veldt grass contributes hydrophobic compounds to the soil, either through exudates or decay (Smith *et al*. 1999). Following fire, veldt grass recovers rapidly, resprouting from the root crown, and produces an abundance of new roots near the soil surface after rain events, forming a dense, fibrous root system (Tothill 1962).

Post-Fire Hillslope Erosion, Sediment Yield, and Hydrologic Response

Approximately 85% (43 t ha^{-1} chaparral and 32 t ha^{-1} grass-converted) of the sediment that was transported out of the small watersheds over the duration of the study occurred during the first post-fire winter, even though the 2003 hydrologic water year was below average and individual storm events were of low intensity (Figure 4). Two major factors contributing to first year erosion and peak discharge were loss of plant cover and steep slopes, while soil water repellency played only a minor role as increases in soil moisture decreased the proportion of moderate-to-high repellency. The high severity fire consumed most of the plant cover, leaving a pattern of relatively contiguous smooth surfaces. These areas can promote runoff and erosion, whereas burned hillslopes interspersed with a mosaic pattern of partially and unburned rough patches produce little runoff (Kutiel *et al.* 1995, Lavee *et al.* 1995). In addition, time to runoff initiation is significantly decreased by loss of plant cover (Giordanengo *et al.* 2003), resulting in a reduction in transpiration and loss of interception. Aspect also plays a role, with south-facing slopes exhibiting higher hydrological connectivity and more runoff (Arnau-Rosalen *et al.* 2008). Comparing first year sediment amounts, Spigel and Robichaud (2007) measured post-fire sediment totaling 24.8 t ha^{-1} on a steep mixed conifer site in the Bitterroot National Forest of Montana, USA. Greater soil losses of 50 t ha^{-1} to 100 t ha^{-1} were reported by Shakesby and Doerr (2006) for a five-month period following fire in a southeastern Australian dry sclerophyll forest, whereas Menendez-Duarte *et al*. (2009) reported much lower sediment production of only 6.8 t ha^{-1} in post-fire shrub vegetation of northwest Spain.

Rainfall intensity did not appear to be a factor during the below average 2003 hydrologic year as I_{15} values were low (Table 1; Figure 4); nevertheless, even small storms generated moderately high peak discharges because of the lack of interception and low ground cover (Table 3). Spigel and Robichaud (2007) stated that short duration, high intensity storms produce greater sediment loads than rain events of low intensity and long duration, and were the driving factor for first year post-fire erosion. However, in the case of the Williams Fire, high erosion rates occurred during low intensity rain events. In southern California, where low intensity, long duration orographic events are more common, high intensity convective rain events of short duration occur infrequently (Tubbs 1972). Additionally, only a few small rill networks were observed in the loose surface soil-ash mixture during this time period, with only a few reaching drainage positions that provided sediment loading to channels (Wohlgemuth 2006).

The geomorphic process of dry ravel contributed to the first major hillslope erosion events during and immediately following the fire as there was evidence of newly deposited cones of eroded material at the bottom of hillslopes in the absence of rain (Figure 2). Ravel activity continued for months and was especially noticeable when soils were dry and during high wind events that allowed material to move downward at slope angles far less than the angle of repose (~50% to 60%). Gabet (2003) noted that seasonal soil drying can reduce interparticle cohesion, thus contributing to the dry ravel process. Dry ravel may have contributed to the larger 2003 sediment yields observed in the chaparral as compared to the grass-converted sites. Shrub boles, downed woody debris, low lying branches, and sporadic litter accumulation allowed ravel to accumulate over the years; whereas there is little space available for ravel entrapment in grasses because they grow close to the ground in a tight, continuous manner. Hubbert and Oriol (2005) estimated potential dry ravel ≥2 mm (pre-fire trapped material), under unburned conditions, at 43 t ha^{-1} in chaparral watersheds

>40 yr. A number of studies have highlighted the importance of dry ravel as a primary source of first year erosion. Florsheim *et al*. (1991) reported that at least 90 % of the sediment of the first winter flow following the Wheeler Fire in southern California was derived from colluvium delivered by dry ravel processes from hillslopes near the channel. Following the 2003 Sulphur Fire in the Oregon Coast Range, Jackson and Roering (2009) measured 47.4 t of colluvium accumulation over the first three months in a 1.9 ha watershed.

Both sediment yield and peak discharge decreased dramatically in the 2004 hydrologic year due to plant cover >60 %, very low rainfall amounts, and low rainfall intensities. Rapid recovery of herbaceous species was due to evenly distributed rain events that occurred from 13 February through 21 April, 2003. Robichaud *et al*. (2000) noted that erosion is effectively controlled at 60 % plant cover, even during high intensity rain events. A number of papers have suggested a 30 % threshold of cover at which soil erosion is reduced considerably (Quinton *et al.* 1997, Ludwig *et al.* 2002). In southern Spain, Cerdà (1998) measured plant cover of 74 % along with a large decrease in runoff during the second winter following wildfire. In 2005, SDEF experienced the wettest hydrologic year in 75 years of record keeping, with both watershed types recording peak discharges higher than what occurred in 2003 (Figure 3; Table 1; Table 5). Although the high peak discharges during this time were the result of unusually high intensity storms, they resulted in relatively minor sediment yields (Figure 4). Factors behind the low yields included: (1) plant and litter cover combined was >75 % on all sites; and (2) much of the loose soil and ash material had already eroded down the steep slopes, been deposited in the channels, and then flushed out during the first winter storm events (Wohlgemuth 2006). Therefore, watershed recovery is not solely a function of vegetation regrowth, but also involves the supply of easily mobilized sediment. Although not determined in this study, some portion of the peak discharge may be attributed to subsurface flow draining water from the hillslopes. Mosley (1979) noted that water in a saturated soil moved through macropores (mainly root channels) at rates two orders of magnitude greater than the soil matrix and generated channel storm flow.

The moderately high peak discharges observed in 2003 were possibly generated by saturation excess overland flow events (when soil and weathered bedrock storage capacity was surpassed). Wildfire lowers the potential storage capacity of both the soil and rock by eliminating plant interception and transpiration of water. Therefore, saturation can occur sooner during post-fire rain events. In this case, watershed storage capacity of ~300 mm may have been reached as rainfall totaled 486 mm from 8 November 2002 through 15 March 2003 (Figure 3; Table 3). Because the majority of rain events during this time were of low intensity (Table 1), there was less chance of infiltration excess events to occur. In 2005, although rainfall amounts were much higher, plant cover promoted transpiration and water loss by interception, thus increasing the potential storage capacity. In this case, water storage capacity was probably not reached, and there was a greater chance of infiltration excess events occurring due to more rain events of higher intensity (Table 1). Rice (1974) thought that overland flow events were rare in the SDEF and had little effect on runoff, noting that only 2.5 % of the precipitation exceeded the infiltration rate of the soil in a 24 yr time span. Williamson *et al*. (2004) also noted that ponded infiltration rates under both grass and chaparral exceeded historical rainfall rates, therefore decreasing the potential for infiltration excess. However, even rain events of low intensity can exceed soil infiltration rates when infiltration rates are reduced due to increased water repellency and pore clogging (Martin and Moody 2001, Valeron and Meixner 2010).

CONCLUSION

Denuded hillslopes, steep unstable terrain, soil disturbance, and reduced infiltration due to repellency produced first year watershed conditions that promoted heavy runoff and erosion events. Approximately 85 % of the total sediment was delivered to the debris basins in the first year, even though the 2003 hydrologic water year was below average and of low intensity. First year sediment yields were greatest in the chaparral watersheds, possibly due to larger accumulations of ravel behind shrubs and downed wood. Dry ravel, during and immediately following the fire, was deposited directly into the channels and accounted for the first major hillslope erosion events. Soil water repellency was inversely related to soil moisture content, being least repellent during the winter wet seasons and most repellent during the summer dry season. There was little difference in persistence of repellency between the grass and chaparral watersheds, except during the 2005 summer when there was a large increase in repellency on the grass watersheds, mainly due to veldt grass. Reduction in repellency occurred at moisture contents ranging between 8% to 16 %. Although storm events of the 2005 water year were the wettest in 75 years of record keeping and produced tremendous runoff, they generated only minor sediment yields in both watershed types. The low sediment yields can largely be explained by rapid plant recovery, and the fact that the supply of easily mobilized sediment had already eroded from the hillslopes and had been flushed from the channels during the first year. Therefore, watershed recovery is not solely a function of vegetation regrowth, but also involves the supply of easily mobilized sediment.

The rapid onset of dry erosion events and the magnitude of sediment produced during the first post-fire winter have major implications for the planning and establishment of emergency rehabilitation treatments. It is understandable that hillslope or stream channel treatments and mitigation measures must be in place before the rainy season begins. However, it is obvious that one cannot prevent dry erosion events that occur during and immediately following the fire. Therefore, it is very important to weigh the values at risk in relation to economical costs of the treatment. Furthermore, the persistence or longevity of any treatment after the first year appears to be considerably less critical, as witnessed by the rapid recovery of both the chaparral and grass-converted watersheds.

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