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Ecohydrologic response and recovery of a semi-arid shrubland over a five year period following burning

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ABSTRACT

Increasing trends in wildfire activity on semi-arid rangelands necessitate advancement in understanding of fire impacts on vegetation, soils, and runoff and erosion processes. This study used artificially applied rainfall and concentrated overland flow experiments to evaluate the ecohydrologic response and recovery of a semi-arid shrubland in the Great Basin Region, USA, following fire. Rainfall experiments were conducted at the 0.5 m^2 plot scale to assess fire impacts on rainsplash and sheetflow processes. Concentrated flow experiments were applied on 9 m^2 plots to evaluate fire impacts on concentrated overland flow processes. Vegetation, soil, hydrologic, and erosion variables were assessed at each scale pre-fire and 1, 2, and 5 yr post-fire. Infiltration and runoff on rainfall simulation plots were affected more by measured background soil water repellency than fire effects on vegetation and soils. Runoff from rainfall on shrub-dominated plots was unchanged 1 yr post-fire, but runoff from interspace plots between shrubs declined 1 yr post-fire. Runoff increased on shrub and interspace rainfall plots 2 yr post-fire and then declined in the 5 yr post-fire. Bare ground generally declined across study years, implicating the temporal variability in soil water repellency as the causal factor for infiltration and runoff trends. Erosion on rainfall plots increased by factors of 8 to more than 10 following fire removal of vegetation and ground cover and declined with vegetation recovery through five growing seasons. Concentrated overland flow plots generated slightly more total runoff and 26-fold more total sediment 1 yr following burning relative to pre-fire measures. Erosion from concentrated overland flow remained greater on burned than unburned plots after five growing seasons even though ground cover returned to approximately 85%. The relative recovery of vegetation and total ground cover were typical for the shrubland community assessed, but elevated erosion with 85% ground cover 5 yr post-fire was unexpected. The persistent high sediment delivery from concentrated plots is attributed to the fine textured soils and thin litter accumulation. The importance of considering erodibility in context with sediment supply and vegetative recovery is discussed. The results demonstrate the complexity of post-fire ecohydrologic interactions, advance process understanding of post-fire ecohydrologic responses for semi-arid rangelands, and underscore the need for additional studies on post-fire recovery over time.

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1. Introduction

Wildfire activity is increasing across arid and semi-arid rangelands and woodlands and dry forests around the globe (Flannigan et al., 2009; Littell et al., 2009; Miller et al., 2011; Shakesby, 2011; Litschert et al., 2012; Pausas and Keeley, 2014; Williams et al., 2014a). For arid to semi-arid regions, periods of high fire activity are commonly linked with invasive plants and climate extremes that promote oscillating periods of fuel accumulation, drying, and ignition and fire spread (Rogers and Vint, 1987; Knapp, 1996; Kitzberger et al., 1997; Swetnam and Betancourt, 1998; Brooks et al., 2004; Littell et al., 2009; O'Donnell et al., 2011). In the western United States (US), the invasion of cheatgrass (*Bromus tectorum* L.) on 4 to 7 million ha of sagebrush (*Artemisia* spp.) rangelands has dramatically increased fire frequency and annual area burned (Knapp, 1996; Miller et al., 2011; Balch et al., 2013). At higher elevations, infill of sagebrush rangelands by pinyon (*Pinus* spp.) and juniper (*Juniperus* spp.) conifers has increased woody fuel loading, the frequency of high-severity woodland fires, and annual area burned (Keane et al., 2008; Romme et al., 2009). Much of western US is now in an ecological condition where rangeland and woodland wildfires have a greater likelihood of progressing upslope into dry forests, where fire activity is also increasing (Westerling et al., 2006). In the European Mediterranean Region, recent increases in wildfire activity

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are attributed to drier summers, woody fuel build-up associated with scrub vegetation and tree encroachment of abandoned lands, extensive planting of highly flammable tree species, and increased urban population of fire-prone areas (Pausas et al., 2008; Shakesby, 2011). Recent periods of high wildfire activity in southeastern Australia are attributed to substantial fuel loading of highly flammable vegetation, drought conditions, and extreme fire weather (Cruz et al., 2012; Attiwill and Adams, 2013). The overall trend in increased fire activity for many arid- to semi-arid regions around the globe is expected to continue into the future based on climate projections (Running, 2006; Flannigan et al., 2009; Abatzoglou and Kolden, 2011; O'Donnell et al., 2011; Shakesby, 2011; Pausas and Keeley, 2014).

The increasing role of wildland fire necessitates increased understanding of fire impacts (Pierson et al., 2011; Shakesby, 2011; Moody et al., 2013; Williams et al., 2014a). Hydrologic responses to intense or prolonged rainfall on the immediate post-fire environment pose substantial threats to resources and surrounding communities (Cannon et al., 1998, 2001; Moody and Martin, 2001; Pierson et al., 2002; Nyman et al., 2011; Neary et al., 2012). Post-fire emergency response teams are commonly challenged with rapidly identifying and assessing post-fire threats and risks and implementing stabilization and mitigation treatments to limit impacts to values-at-risk (Robichaud et al., 2000; Calkin et al., 2007; Robichaud et al., 2014). Research over the past decade has greatly advanced knowledge of fire impacts on hydrology and erosion (Shakesby and Doerr, 2006; Cannon et al., 2011; Pierson et al., 2011; Nyman et al., 2011; Cawson et al., 2012), but key knowledge gaps remain (Shakesby, 2011; Moody et al., 2013; Williams et al., 2014a). In particular, knowledge is lacking regarding the spatially and temporally variable effects of ash cover and soil water repellency on post-fire hydrology and erosion and how to represent those effects in hydrologic modeling (Doerr et al., 2000, 2003; Cerdà and Doerr, 2008; Pierson et al., 2008a; Zavala et al., 2009; Jordán et al., 2010; Woods and Balfour, 2010; Bodí et al., 2011, 2012; Moody et al., 2013; Al-Hamdan et al., 2015; Pereira et al., 2015). The spatial scaling of runoff and erosion remains challenging given the current trend towards larger fires over diverse landscapes (Shakesby and Doerr, 2006; Pierson et al., 2011; Miller et al., 2011; Stoof et al., 2012; Cerdà et al., 2013; Moody et al., 2013; Williams et al., 2014a). Knowledge is scant regarding long-term soil health and the ecological impacts of high soil loss rates associated with frequent re-burning and community conversions to fire-prone vegetation (Pierson et al., 2011; Shakesby, 2011; Williams et al., 2014a). Lastly, data are commonly limited for specific plant community and soil groups, and these limitations can hinder development of model parameterizations, assessment and prediction of post-fire hydrologic response, and targeting of post-fire emergency stabilization treatments (Williams et al., 2014a).

Post-fire runoff and erosion rates from rangelands and woodlands can be difficult to predict and are strongly influenced by the pre-fire community structure and soil type and soil burn severity (Al-Hamdan et al., 2012a; Nyman et al., 2013; Al-Hamdan et al., 2015; Vieira et al., 2015). Fire and other disturbances increase the risk for hillslope runoff and erosion on these landscapes by altering the patchy structure of vegetation and ground cover (Puigdefábregas et al., 1999; Puigdefábregas, 2005; Ludwig et al., 2007; Pierson et al., 2009; Williams et al., 2014b, 2016a, 2016b). Vegetative patches effectively intercept and store precipitation input, limit soil detachment, reduce runoff and sediment transport, and promote infiltration and soil development (Cerdà, 1997; Wilcox et al., 2003; Ludwig et al., 2005; Pierson et al., 2009, 2010). Bare patches within the interspaces between shrub or tree canopies are sources for runoff and sediment, but delivery of these sources to the base of a hillslope is minimal except following disturbance (Pierson et al., 1994; Wilcox et al., 2003; Ludwig et al., 2007; Williams et al., 2016a). Following disturbance, sediment stored in vegetation patches becomes available for erosion processes, and vegetation and ground cover degradation enhances runoff generation and downslope sediment delivery (Cerdà, 1998; Pierson et al., 2009; Al-Hamdan et al., 2012a; Pierson et al., 2013, 2015; Williams et al., 2014b, 2016a). In the post-fire environment, ash may buffer raindrop impact and limit runoff through storage of water input, however, these attributes are commonly short-lived and highly variable spatially (Cerdà and Doerr, 2008; Woods and Balfour, 2008; Zavala et al., 2009; Woods and Balfour, 2010; Pereira et al., 2015). Ash can clog soil pores, form infiltration inhibiting crusts, promote runoff generation, and contribute substantially to sediment yield depending on the physical characteristics, depth, and packing of the ash material (Onda et al., 2008; Cerdà and Doerr, 2008; Gabet and Sternberg, 2008; Woods and Balfour, 2010; Balfour et al., 2014; Jordán et al., 2015). Overall, the magnitude of runoff and sediment delivery for a particular rainfall event following disturbance is dictated by the rate or amount of water input and the connectivity of runoff and erosion processes and sediment supply (Nyman et al., 2013; Williams et al., 2014b, 2016a). Sediment supply may be depleted on long-degraded sites prior to burning, limiting post-fire erosion potential (Al-Hamdan et al., 2015). However, these sites may produce substantial post-fire runoff, and the amplified runoff rates may still pose threats to downstream values-at-risk. In contrast, high sediment supply on well-vegetated landscapes may result in high rates of post-fire erosion during runoff generating storms (Nyman et al., 2015). Post-fire hydrologic recovery is also affected by the pre-fire vegetation and the time required to reproduce the pre-fire vegetation and ground cover structure (Cerdà, 1998; Cerdà and Doerr, 2005; Pierson et al., 2009; Miller et al., 2013). Knowledge regarding these complexities and the respective influences on post-fire responses is critical to assessing post-fire hydrologic risk and the period required for hydrologic recovery.

The purpose of this study was to quantify fire impacts on vegetation, soils, runoff and erosion processes on a steeply-sloped sagebrush rangeland underlain by fine-textured soils, and to assess the temporal persistence of these impacts over a five year period. Specific objectives were: 1) quantify the relative contributions of runoff and erosion by rainsplash, sheetflow, and concentrated flow across point to patch scales (<1 m^2 to 10 m^2) before and after fire, and 2) evaluate vegetation and surface-soil factors that influence persistence of changes in post-fire hydrologic and erosion processes. A suite of small plot rainfall simulations (0.5 m^2) and concentrated overland flow experiments (9 m^2) were used to measure runoff and erosion for unburned and burned conditions. Vegetation and surface soil characteristics were quantified on rainfall simulation and overland flow plots to assess interactions of vegetation, soils, and hydrology and erosion responses to burning. The study results will contribute to process understanding on the hydrologic impacts of fire and post-fire hydrologic recovery and provide unique quantitative post-fire hydrologic response data for a commonly occurring ecological community along the rangeland-dry forest ecotone in the western US.

2. Materials and methods

2.1. Study area

Experiments were conducted in the Upper Sheep Creek (USC) study area (Fig. 1; 43°7′18.6″N latitude, 116°43′44.4″W longitude, 1900 m MSL) located within the Reynolds Creek Experimental Watershed in the Owyhee Mountains of southwestern Idaho, USA. The lands in and adjacent to the site are managed by US Department of Interior, Bureau of Land Management. The US Department of Agriculture, Agricultural Research Service maintains an extensive instrumentation network at the USC site for long-term research on climate, vegetation, and hydrologic processes (Flerchinger et al., 1998; Flerchinger and Cooley, 2000; Chauvin et al., 2011; Flerchinger and Seyfried, 2014). Average annual precipitation at the site is 527 mm, with 60% occurring as snow (Fig. 2A). Average annual air temperature is 5.9 °C (Fig. 2B). Streamflow at the site is intermittent, with about 44 mm of flow exiting the 0.26 km^2 catchment annually (USDA-ARS-NWRC, 2015). Summers are

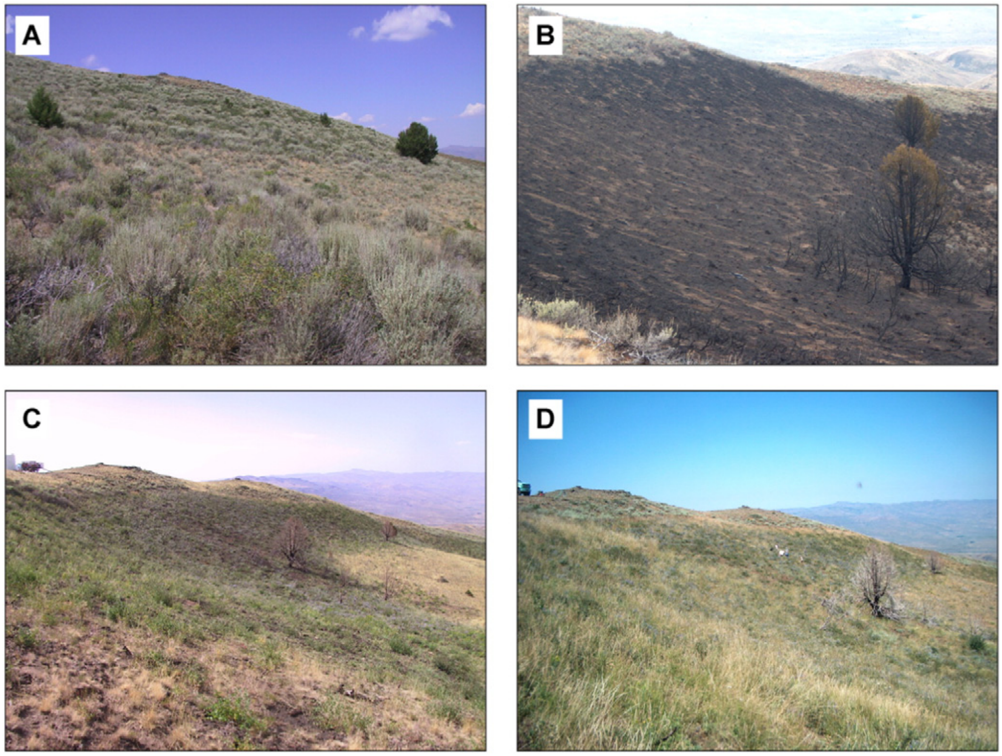

Fig. 1. Study hillslope within the Upper Sheep Creek study area pre-fire (A), immediately post-fire (B), 1 yr post-fire (C), and 2 yr post-fire (D).

hot and dry. Soils wet-up with autumn rainfall (Fig. 2C), but soil moisture stabilizes as the precipitation regime shifts to snow in winter months. The highest soil moisture contents and peak streamflow occur during the spring snowmelt period.

Vegetation at USC varies with aspect and elevation. Groves of aspen (*Populus tremuloides* Michx.) and willow (*Salix* spp.) occur at higher elevations in snowdrift zones. Middle elevations on northeast aspects contain dense pockets of mountain big sagebrush (*Artemisia tridentata* Nutt. subsp. *vaseyana* [Rydb.] Beetle) and snowberry (*Symphoricarpos oreophilus* A. Gray) shrubs, and various grasses and forbs (Fig. 1A). Southerly aspects are sparsely covered with low sagebrush (*Artemisia arbuscula* Nutt.) and very limited grass cover. The vegetation distribution and productivity are largely driven by soil water recharge from melting of snowdrifts (Flerchinger and Cooley, 2000). Experimental plots in this study were restricted to steeply-sloped (20°–25°) northeast facing aspects that commonly support dense cover of mountain big sagebrush, snowberry, and grasses (Fig. 1A). Soils on the study hillslope are classified as loam (USDA-NRCS, 2003). The soils are derived from basalt parent rock and wind-blown silt, have low rock content (0–25%), and are 0.5 m to more than 1 m deep.

2.2. Experimental design

Experimental plots were established to characterize vegetation, the ground surface, and hydrologic and erosion function over small plot (0.5 m^2) and patch (9 m^2) scales before (1 yr pre-fire, July 2007) and one (1 yr post-fire, August 2008), two (2 yr post-fire, September 2009), and five (5 yr post-fire, September 2012) years after prescribed fire (Fig. 1). The study site was burned by prescribed fire in September of 2007. Soil burn severity was not quantified following the fire, but the observed nearly 100% canopy and ground cover consumption and extensive coverage of ash post-fire were indicative of a moderate to high soil burn severity (Parsons et al., 2010).

Eight small rainfall-simulation plots (0.7 m × 0.7 m) were installed in July 2007 on randomly-selected unburned shrub coppice microsites (areas underneath shrub canopies) and on randomly-selected unburned interspace microsites (areas between shrub canopies). Small rainfall plots were selected such that all plots occurred on the same hillslope (same aspect, hillslope gradient, plant community, and soils) and small plots were interspersed between patch-scale concentrated flow plots. Each small-plot frame was pounded into place with use of a steel bar and was left in place for sampling in subsequent years after the prescribed fire (Fig. 3A and B). Vegetation, soil, runoff, and erosion data measured from small rainfall plots and experiments were used to quantify fine-scale vegetation and soil responses to burning and the impact of those responses on runoff and erosion from splash-sheet processes (rainsplash and sheetflow, interrill). Stratification of plots by microsite (shrub coppice vs. interspace) was used to partition microsite cover/soil differences and respective runoff and erosion contributions to the patch scale (Pierson et al., 2009, 2010; Williams et al., 2014b).

Patch-scale plots (concentrated flow plots, 2 m wide × 4.5 m long, Fig. 3C and D) were used to quantify vegetation and soil responses to burning and the impacts of those responses on runoff and erosion from concentrated overland flow processes (Pierson et al., 2009, 2010; Williams et al., 2014b). Twelve 9-m^2 concentrated overland flow plots were installed and removed each year of the study on the same hillslope with the small plots. The plot area of each concentrated flow plot encapsulated both shrub coppice and interspace microsites. Concentrated flow plots were installed with collection troughs arranged in a down-slope pointing "V" pattern at the base of each plot (Fig. 3C and D) as described by Pierson et al. (2010). The collection troughs were designed to route plot runoff directly to the plot outlet. Concentrated flow plots were un-bounded on the sides and upslope end. Plot length was dictated by the 4.5 m distance between the plot outlet and the overland flow release point.

2.3. Vegetation and soil sampling

2.3.1. Small plot scale

Canopy cover, ground cover, and surface roughness on each small rainfall plot were measured using point-frame methodologies (Pierson

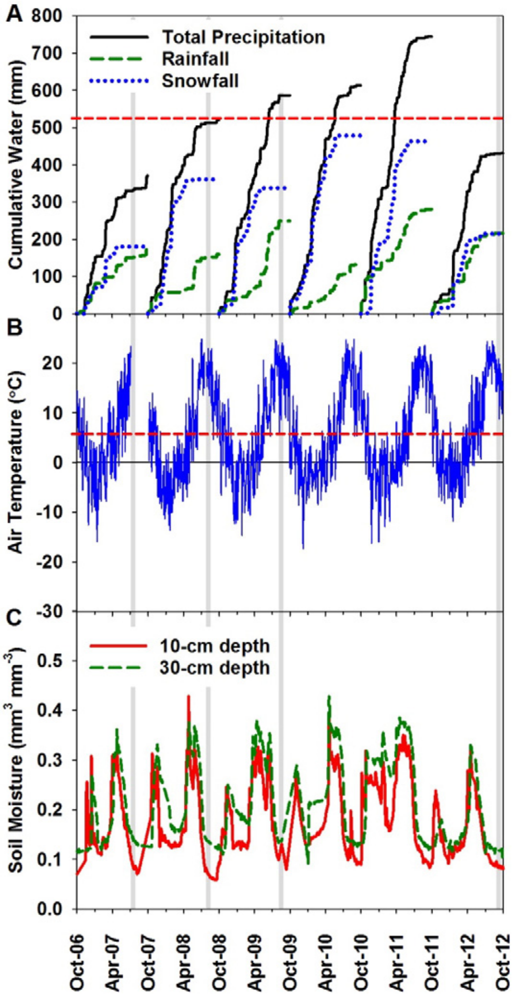

Fig. 2. Cumulative precipitation by water year (Oct–Sep; A), hourly air temperature (B), and soil moisture content (C) recorded in the Upper Sheep Creek study area from Oct 2006 to Oct 2012. Horizontal dashed red lines indicate site annual averages for total precipitation and air temperature. Vertical grey lines indicate approximate date of sample periods for this study. Data were obtained from a meteorological network (USDA-ARS-NRWC, 2015) located approximately 500 m from and at the same elevation (1900 m) as experimental plots in this study.

et al., 2010). Plot canopy and ground cover by cover type (bare soil, cryptogam, forb, grass, litter, rock, shrub, standing dead, and woody dead) and relative ground-surface height were recorded at 15 points (5-cm spacing) along each of seven point-frame transects spaced 10 cm apart and parallel to hillslope contour. Percent cover for each cover type was derived from the frequency of respective cover hits divided by the total number of points sampled within the plot. The relative ground-surface height at each sample point was calculated as the distance between the ground surface and the point-frame level line at the respective point. Plot ground surface roughness was estimated as the arithmetic average of the standard deviations of the ground surface heights for each of the seven transects sampled. Litter depth on the ground surface was measured adjacent to each small plot at four evenly spaced points along each of the two small plot borders oriented perpendicular to the hillslope contour (Pierson et al., 2010).

Surface soils for each plot were sampled for soil moisture and aggregate stability. Soil samples were obtained for 0–5 cm depth immediately adjacent to each plot each year and were analyzed gravimetrically for soil water content. Surface soil aggregate stability for each plot was determined using a modified sieve test described by Herrick et al. (2001, 2005). Six soil aggregates about 2–3 mm thick and 6–8 mm in diameter were excavated from the soil surface immediately adjacent to each plot and were subjected to the stability test. Each soil aggregate was assigned to a stability class defined by Herrick et al. (2001, 2005), as indicated in Table 1. For each plot, a mean aggregate stability class was derived as the average of the classes assigned to the six aggregate samples.

The soil water repellency was measured over 0–5 cm soil depth before rainfall simulation immediately adjacent to each plot using the water drop penetration time (WDPT) method (DeBano, 1981). The time required for water drop infiltration (up to a maximum of 300 s) was recorded for eight water drops (3-cm spacing) applied to the mineral soil surface (ash and litter removed). Following this procedure 1 cm of soil was excavated immediately underneath the previously sampled area and the WDPT procedure was repeated with eight more drops. WDPT sample iterations continued to a soil depth of 5 cm. Mean soil water repellency at each 1-cm depth (0-, 1-, 2-, 3-, 4-, and 5-cm depths) for each plot was recorded as the mean of the eight WDPT (s) samples for the respective depth. The mean soil water repellency across all sampled soil depths on each plot was derived as the average of the respective WDPT means for 0-, 1-, 2-, 3-, 4- and 5-cm soil depths. Soils were classified as wettable when WDPT was less than 5 s, slightly water repellent when WDPT ranged from 5 to 60 s, and strongly water repellent when WDPT exceeded 60 s (Bisdom et al., 1993).

2.3.2. Concentrated flow plots

Canopy and ground cover by cover type and the distance between plant bases (basal gaps) were measured on each concentrated flow plot using line-point intercept and gap-intercept procedures (Herrick et al., 2005). Canopy and ground cover were recorded at 21 points with 20-cm spacing, along each of nine line-point transects 4.0 m in length, spaced 20 cm apart, and oriented perpendicular to the hillslope contour (189 points/plot). Plant basal gaps exceeding 20 cm were recorded along each line-point transect of each plot. Average and maximum basal gap sizes for each plot were determined as the mean and the maximum of all basal gaps measured in excess of 20 cm. The relative ground-surface height along line-point intercept transects was calculated as the distance between the ground surface and a survey transit level line above the respective point. The ground surface roughness for concentrated flow plots was estimated as the arithmetic average of the standard deviations of the ground surface heights across the line-point transects sampled within each plot (Pierson et al., 2010).

2.4. Hydrologic and erosion responses

2.4.1. Small plot rainfall simulations

A Meyer and Harmon (1979) type portable oscillating-arm rainfall simulator with 80–100 Veejet nozzles was used to apply rainfall to each small plot (Pierson et al., 2008a, 2009, 2010, 2014). The simulator was fitted with nozzles positioned 3 m above plots and pressurized at 41 N m^{-2}. The nozzle configuration produces rainfall with similar drop size and kinetic energy to that of natural rainfall (Meyer and Harmon, 1979). Rainfall was applied to each small plot at rates of 64 mm h^{-1} under dry (dry run) and 102 mm h^{-1} under wet (wet run) antecedent soil moisture conditions. The rates were applied for 45 min each, separated by a 30-min interval between the dry and wet runs. The total amount of rainfall applied to each plot was determined by integrating a pan catch of a 5-min calibration run immediately prior to rainfall simulation. Calibration pans were designed to fit directly on plot frames without disturbing plot surfaces. Total water applied was estimated for plots where shrub cover prevented placement of calibration pans. The estimates were calculated as the average of all calibrations for the respective plot-frame slope. Rain rates were selected to simulate runoff and erosion generating storm events typical of the USC study area. The dry run intensity over 5-, 10-, and 15-min durations is approximately equivalent to storm return intervals of 4, 8, and 20 yr, and the wet run intensity over the same durations is approximately equivalent to storm return intervals of 15, 33, and 75 yr (Hanson and Pierson, 2001). Timed samples of plot runoff were collected at 1-min

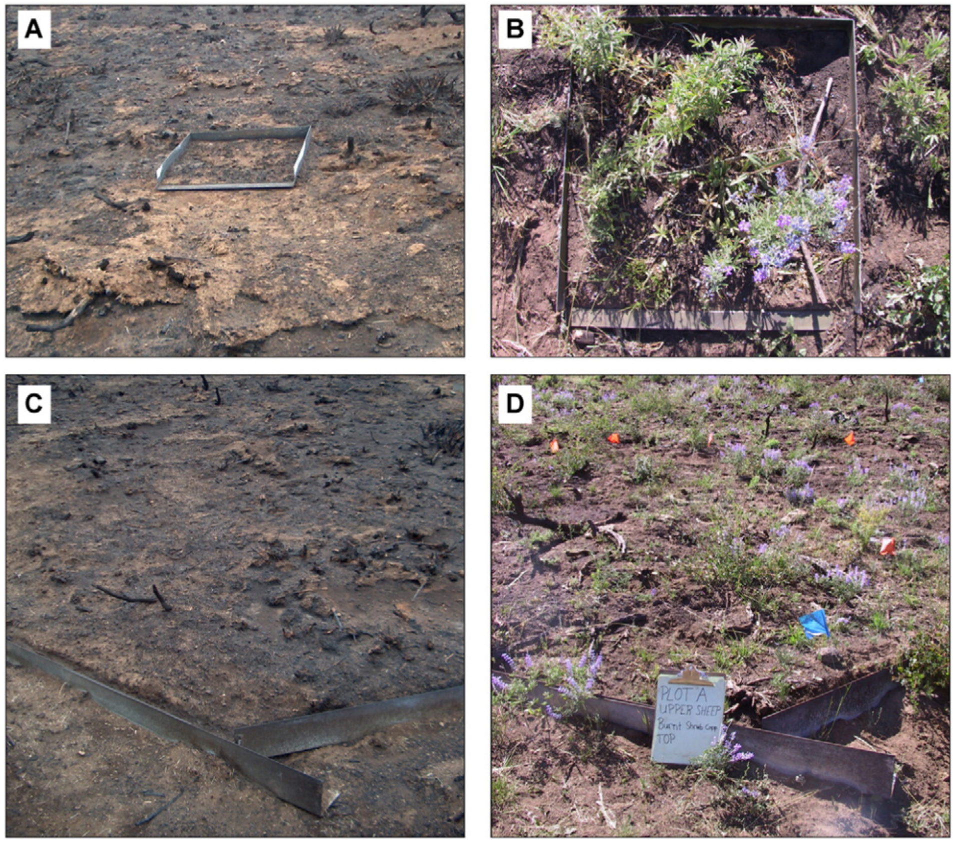

Fig. 3. Small rainfall plots (0.5 m^2) in burned area immediately post-fire (A) and 1 yr post-fire (B), and concentrated flow plots in burned area immediately post-fire (C) and 1 yr post-fire (D) at the Upper Sheep Creek study site. Photos of concentrated flow plots (C and D) show collection troughs at the downslope end of plots and unbounded upslope areas within the 9 m^2 plot area.

to 3-min intervals throughout each 45-min rainfall simulation and were analyzed in the laboratory for runoff volume and sediment concentration. Runoff volume and sediment concentration were determined for each runoff sample by weighing the sample before and after drying at 105 °C (Pierson et al., 2008b, 2009, 2010).

Hydrologic and erosion response variables were derived for each plot based on the timed runoff samples. A mean runoff rate (mm h^{-1}) was calculated for each sample interval as the cumulative runoff divided by the interval time. Cumulative runoff (mm) was calculated as the integration of runoff rates over the total time of runoff. The percentage of

Table 1
Ground surface, soil, and vegetation characteristics measured on rainfall simulation plots (0.5 m^2) pre-fire and one, two, and five years after burning. Means within a row followed by a different lowercase letter are significantly different ($P < 0.05$).

Plot characteristic	1 Yr pre-fire		1 Yr post-fire		2 Yr post-fire		5 Yr post-fire	
	Shrub coppice	Interspace	Shrub coppice	Interspace	Shrub coppice	Interspace	Shrub coppice	Interspace
Aggregate stability class (1–6)[a]	6 a	6 a	6 a	5 a	5 a	5 a	4 a	5 a
Soil water repellency (s)[b]	97 d	95 d	83 d	34 bc	33 bc	49 cd	4 a	11 ab
Surface roughness (mm)	25 b	16 a	18 a	15 a	20 ab	16 a	24 b	17 a
Total canopy cover (%)[c]	103 cd	81 bc	41 a	65 ab	96 cd	117 d	88 bc	84 bc
Shrub canopy cover (%)	64 b	2 a	6 a	2 a	13 a	0 a	13 a	10 a
Grass canopy cover (%)	19 b	42 cd	3 a	19 b	22 bc	47 d	47 d	44 cd
Forb canopy cover (%)	5 a	13 ab	23 bc	35 c	37 c	53 d	27 bc	30 c
Total ground cover (%)[d]	96 d	95 d	64 ab	57 a	75 abc	86 cd	79 bc	83 bc
Basal plant cover (%)[e]	4 ab	2 a	2 a	4 ab	7 bc	9 bc	11 c	8 bc
Litter cover (%)	89 c	91 c	56 a	51 a	64 ab	77 bc	63 ab	75 bc
Rock cover (%)	0 a	0 a	2 a	1 a	0 a	1 a	1 a	0 a
Bare soil (%)	4 a	5 a	36 cd	43 d	25 bcd	14 ab	21 bc	17 bc
Litter depth (mm)	34 b	32 b	9 a	6 a	4 a	4 a	8 a	5 a
No. of plots	8	8	8	8	8	7	8	8

[a] Stability classes: (1) less than 10% stable aggregates, 50% structural integrity lost within 5 s; (2) less than 10% stable aggregates, 50% structural integrity lost within 5–30 s; (3) less than 10% stable aggregates, 50% structural integrity lost within 30–300 s; (4) 10–25% stable aggregates; (5) 25–75% stable aggregates; (6) 75–100% stable aggregates (Herrick et al., 2001, 2005).
[b] Mean soil water repellency for 0–5 cm soil depth assessed as water drop penetration time (WDPT, 300 s maximum). Soils were classified slightly water repellent if WDPT ranged 5 to 60 s and strongly water repellent if WDPT exceeded 60 s (Bisdom et al., 1993).
[c] Includes shrub, grass, forb, standing dead, and woody dead canopy cover.
[d] Includes cryptogam, litter, live and dead basal plant, rock, and woody dead ground cover.
[e] Includes live and dead cryptogam, shrub, grass, and forb basal cover.

rainfall converted to runoff on each plot was derived as a runoff-to-rainfall ratio (mm mm^{-1}), cumulative runoff divided by cumulative rainfall applied and multiplied by 100%. Infiltration and sediment variables were calculated solely for plots that generated runoff. A mean infiltration rate (mm h^{-1}) for each sample interval was derived as the difference between applied rainfall and measured runoff divided by the sample interval duration. Cumulative sediment yield (g m^{-2}) was calculated as the integrated sum of sediment collected during runoff and was extrapolated to a unit area by dividing cumulative sediment by plot area. A sediment-to-runoff ratio (g m^{-2} mm^{-1}) was obtained by dividing cumulative sediment yield per unit area by cumulative runoff.

Soil wetting patterns were investigated over 0–20 cm depths immediately following dry-run rainfall simulations on each small plot. Wetting patterns were measured by excavating 50-cm long trenches to a depth of 20 cm. Trenches were excavated immediately adjacent to each plot so as to not affect wet run rainfall simulations. The percent wetted area of each exposed soil profile was measured using a 4 cm^2 grid. Each grid area was determined to be dry or wet based on the dominant condition in the grid area and a percent wetted area was calculated for soil depths 0–6 cm, 0–10 cm, and 0–20 cm (Pierson et al., 2010, 2014).

2.4.2. Concentrated flow experiments

Concentrated overland flow was applied to each plot using computer-controlled flow regulators (Pierson et al., 2010). Release rates of 15, 30, and 45 L min^{-1} were applied to each concentrated plot for 12 min from a single location, 4.5 m upslope of the plot outlet. Release rate progression was consecutive from 15 L min^{-1} to 45 L min^{-1}. Concentrated flow was routed through a metal box filled with Styrofoam pellets and was released through a 10-cm wide mesh-screened opening at the base of the box (Pierson et al., 2008b, 2009, 2010). Plot runoff samples were collected at about 2-min intervals for each 12-min flow rate simulation. For each plot, the number of flowpaths and width of each flowpath were recorded at a cross-section located 3 m downslope of the flow release point. Flow velocity on each plot was determined by releasing a concentrated salt solution ($CaCl_2$) into the fastest (determined by visual tracer) flowpath and calculating the mean travel time of the salt solution between flowpath cross-sections 1 m and 3 m downslope of the flow release point. Salt solution concentration and simulation time were monitored instantaneously by conductivity probes at the 1-m and 3-m cross-sections. Mean travel time was calculated as the time difference between the maximum conductivity readings on each conductivity probe. Flow velocity was calculated by dividing the measured flow path distance between the 1-m and 3-m transects by the associated mean travel time.

Runoff samples were processed in the laboratory for runoff and sediment concentration as described for small rainfall plots. Plot mean runoff and erosion variables for each flow release rate were derived for an 8-min time period beginning at runoff initiation. A mean runoff rate (L min^{-1}) was calculated for each sample interval as the cumulative runoff divided by the interval time. Cumulative runoff (L) by release rate for each plot was calculated as the integration of runoff rates over the 8-min time of runoff. Cumulative sediment (g) by release rate for each plot was calculated as the integrated sum of sediment collected during the 8-min runoff period. Total runoff (L) for each plot was calculated as the sum of cumulative runoff from all release rates. Total sediment (g) for each plot was calculated as the sum of cumulative sediment from all release rates. Initial and final sediment concentrations were determined for each release rate on each plot as the sediment concentration of the first and final runoff sample collected.

2.5. Statistical analyses

All statistical analyses were conducted using SAS software, version 9.2 (SAS Institute Inc., 2006). Small plot data were analyzed using a repeated measures split-plot mixed model with four whole-plot or treatment factors: 1 yr pre-fire, 1 yr post-fire, 2 yr post-fire, and 5 yr post-fire. Microsite was the small plot sub-plot factor and had two levels: shrub coppice and interspace. Sample year was the repeated measure. Covariance structure for all variables was evaluated using fit statistics suggested by Littell et al. (2006) and the best fit model was applied.

Cover variables, flow velocity, flowpath widths, and the number of flowpaths from concentrated flow experiments were analyzed using a mixed model with four treatments levels: 1 yr pre-fire, 1 yr post-fire, 2 yr post-fire, and 5 yr post-fire. Concentrated flow runoff and erosion variables were analyzed with a repeated measures mixed-model using four treatment levels: 1 yr pre-fire, 1 yr post-fire, 2 yr post-fire, and 5 yr post-fire. Flow release rate was the repeated measure for concentrated-flow runoff and erosion analyses, with three levels: 15, 30, and 45 L min^{-1}. Carryover effects of concentrated flow releases were modeled with an autoregressive order 1 covariance structure (Littell et al., 2006). Plot location was designated a random effect and treatment and microsite were considered fixed effects in all respective analyses. Normality and homogeneity of variances were tested prior to ANOVA using the Shapiro-Wilk test and Levene's test (SAS Institute Inc., 2006) and deviance from normality was addressed by data transformation. Where necessary, arcsine-square root transformations were used to normalize proportion data (e.g., canopy cover, percent wet). Logarithmic transformations were used to normalize WDPT, runoff, cumulative sediment, and sediment-to-runoff ratio data. Backtransformed results are reported. Mean separation was determined using the LSMEANS procedure with Tukey's adjustment. Significant effects for all analyses are reported at the $P < 0.05$ level.

3. Results

3.1. Vegetation and soils

Ample woody and fine fuels at the time of the prescribed fire facilitated a uniform burn, resulting in nearly 100% consumption of above ground vegetation, surface litter, and woody fuels across the study hillslope (Figs. 1 and 3). Dense shrub and grass cover provided approximately 90% total canopy cover across the site pre-fire (Table 1). About 90% of the pre-fire ground surface on shrub coppice and interspace microsites was covered with a more than 30 mm thick litter layer (Table 1). As expected, shrubs were the dominate canopy cover type on unburned shrub coppices. Pre-fire, grass canopy cover ranged from 19% underneath shrubs to more than 40% in interspaces. Visual observations immediately post-fire found that all sagebrush and above-ground cover of grasses were removed by the fire (Fig. 3A and C). Surface litter was also nearly 100% consumed, but a spatially heterogeneous charred mat of decomposed or embedded litter remained (Fig. 3A). The only woody fuels that remained were charred stumps from burned shrubs and several burned juniper trees (*J. occidentalis* Hook.) (Figs. 1B and 3C).

Natural re-vegetation of the site occurred within several years, but the community shifted from shrub- to herbaceous-dominated (Fig. 1, Table 1). Shrub canopy cover generally increased throughout the study, but remained five-fold lower on shrub coppice plots 5 yr post-fire than pre-fire (Table 1). Grass canopy cover was lower on shrub and interspace plots 1 yr post-fire than pre-fire, but returned to pre-fire levels across all plots within two growing seasons (Table 1). Grass canopy cover more than doubled on shrub plots by the fifth year. The fire enhanced forb cover, increasing forb canopy cover by three- to seven-fold across interspace and shrub coppices the first few years after burning (Table 1). By the fifth year post-fire, total canopy cover returned to pre-fire levels, but canopy cover on shrub plots had shifted from shrub-dominated to grass- and forb-dominated. Total canopy cover on interspace plots was grass-dominated pre-fire and grass- and forb-dominated 5 yr post-fire. Ground cover responses to burning were similar with trends in canopy cover (Table 1; Fig. 4A). Total ground

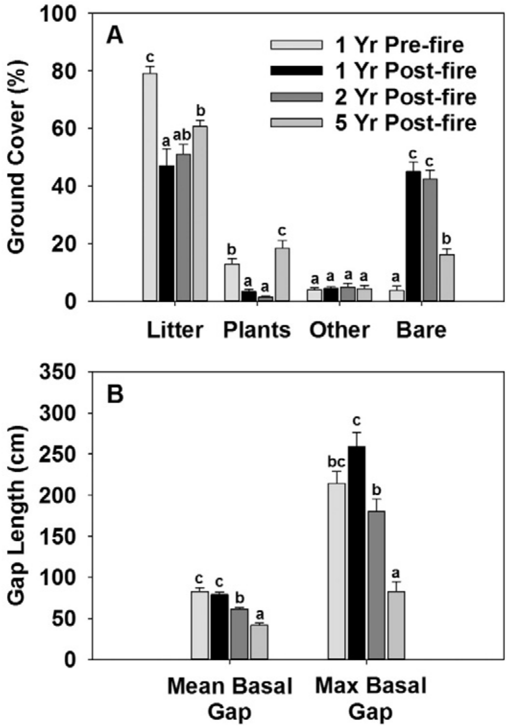

Fig. 4. Percent ground representation by litter, basal plants, other cover types (woody dead and rock cover), and bare soil (A), and the mean and maximum distances (B; 20 cm minimum) between plant bases on concentrated flow plots (9 m^2) at the Upper Sheep Creek study site. Error bars depict standard error. Means within a cover type (e.g., litter) or gap type followed by different lower case letters are significantly different ($P < 0.05$).

cover by 1 yr post-fire was nearly two-fold lower than pre-fire, but returned to or near pre-fire levels after the second year. The recovery of total ground cover to near pre-fire conditions by the 3–5 yr post-fire was mainly due to increases in litter in the interspace and herbaceous basal plant cover (Table 1). Post-fire basal cover recruitment reduced the mean and maximum distances between plant bases by 41 cm and 132 cm, respectively (Fig. 4B). Although the site rapidly revegetated, litter cover remained lower than pre-fire levels the fifth year after burning (Fig. 4A), and the depth of litter was 5-fold less for burned plots the 5 yr post-fire relative to pre-fire conditions (Table 1).

Dense vegetation pre-fire created stable surface conditions that were altered to varying degrees by burning. Soils underneath the unburned 30-mm thick litter layer were highly stable and strongly water repellent (Table 1; Fig. 5). The prescribed fire had no significant effect on aggregate stability as assessed in this study. The upper 2 to 3 cm of the soil profile underneath shrubs and in interspaces was strongly water repellent under unburned conditions and soil water repellency exhibited significant variability over the course of the study (Fig. 5). Soils at 0- to 2-cm depth on shrub plots were strongly water repellent pre-fire and 1 yr post-fire, but, only slightly water repellent or wettable 2 yr and 5 yr post-fire. Soil water repellency at 0- to 2-cm depth on interspace plots was strong pre-fire, but was reduced (for 0- to 1-cm depth only) by 70–90% 1 yr post-fire. In contrast to strongly water repellent surface conditions pre-fire, soil water repellency on interspaces 2 yr post-fire tended to be greater at depths 1 to 2 cm below the mineral soil surface. Soils were wettable at all depths across all plots the fifth year post-fire with few exceptions. The ground surface roughness on interspace plots was unaffected by burning and was generally lower than that of shrub coppices (Table 1). Burning slightly reduced surface roughness on shrub coppice plots, but roughness on shrub plots retuned to pre-fire levels 2 yr and 5 yr post-fire (Table 1). Surface roughness measured at the patch scale (concentrated flow plots) averaged 41 mm across all years and was not significantly affected by burning or changes in vegetation in the years post-fire ($P > 0.05$). Gravimetric soil moisture content at the time of sampling was uniformly low (<10%) across all microsites and treatments each year.

3.2. Rainfall simulations

Small plots exhibited significant temporal and fire-induced runoff responses to applied rainfall. As expected, runoff from dry- and wet-run rainfall simulations was at least two-fold higher for interspace than shrub microsites pre-fire, but microsite differences in hydrologic responses to rainfall were inconsistent following burning (Table 2). Runoff and infiltration of dry and wet runs on shrub coppices 1 yr post-fire were similar to those pre-fire and to the same measures on the burned interspace plots. For interspaces 1 yr post-fire, dry-run runoff and infiltration were similar to pre-fire measures, but wet-run runoff was 20% less than pre-fire and comparable to unburned shrub plots (Table 2, Fig. 6A and B). In the 2 yr post-fire, microsite differences were detected for runoff and infiltration, but only for the dry run. Wet-run runoff from shrub coppices in 2 yr post-fire was the highest measured during the study and was about 60% greater than pre-fire and 1 yr post-fire (Table 2, Fig. 6A). Dry- and wet-run runoff from interspaces the 2 yr post-fire were 50–100% higher than in 1 yr post-fire (Table 2). Runoff decreased and infiltration increased from 2 yr to 5 yr post-fire across shrub and interspace plots (Table 2). Wetting trench data indicate 90–100% of the soil profile from 0- to 10-cm soil depth was wet following the dry run simulations 5 yr post-fire (Table 2). Dry- and wet-run runoff responses for shrub plots 5 yr post-fire were consistent with the same measures pre-fire and 1 yr post-fire. Interspace runoff and infiltration in the 5 yr post-fire were lower and higher, respectively, than pre-fire and 2 yr post-fire measures for both the dry and wet runs. Overall, burning had minimal impact on infiltration and runoff from shrub coppices, and the fire-induced enhanced infiltration on interspace plots was superseded by generally high rates of wet-run runoff across all plots in the 2 yr post-fire (Fig. 6A and B).

Erosion rates for small plot rainfall simulations exhibited an initial post-fire pulse and were not consistent with the temporal trends of runoff responses (Fig. 6). Pre-fire erosion and sediment-to-runoff ratios were low for dry- and wet-run simulations and were similar for shrub and interspace microsites (Table 2; Fig. 6C and D). Sediment-to-runoff ratios and cumulative sediment increased by factors of 8 to 13 across all rainfall simulations 1 yr following the burn relative to pre-fire. In the 2 yr post-fire, erosion from dry-run simulations returned to pre-fire levels for shrub and interspace microsites, but wet-run erosion and the sediment-to-runoff ratio remained above pre-fire levels for shrub coppices (Table 2). Wet-run sediment discharge on shrub coppices was high in the 5 yr post-fire relative to pre-fire (Fig. 6C), but high variability across shrub plot responses resulted in no significant fire effect on erosion relative to the pre-fire response (Table 2). Dry- and wet-run erosion on interspaces in the 5 yr post-fire were generally consistent with responses pre-fire (Table 2). Overall, erosion from rainfall simulations was low pre-fire, increased sharply 1 yr post-fire, and returned to pre-fire levels in the 5 yr following burning (Table 2).

3.3. Concentrated flow simulations

Runoff and sediment delivery from concentrated flow experiments were amplified following burning and elevated erosion persisted through the 5 yr post-fire. Runoff generally increased with increasing bare ground during the study (Fig. 7A). Runoff was lowest pre-fire and in the 5 yr after burning (Table 3). Overland flow released on burned plots in the 1 yr post-fire tended to form more flowpaths with narrow flow widths and faster velocity as compared with unburned conditions (Table 3). Cumulative runoff from release rates 15 and 30 L min^{-1} and total runoff (runoff from all rates) the first year post-fire were 25% to more than 100% higher than from pre-fire conditions, and flow

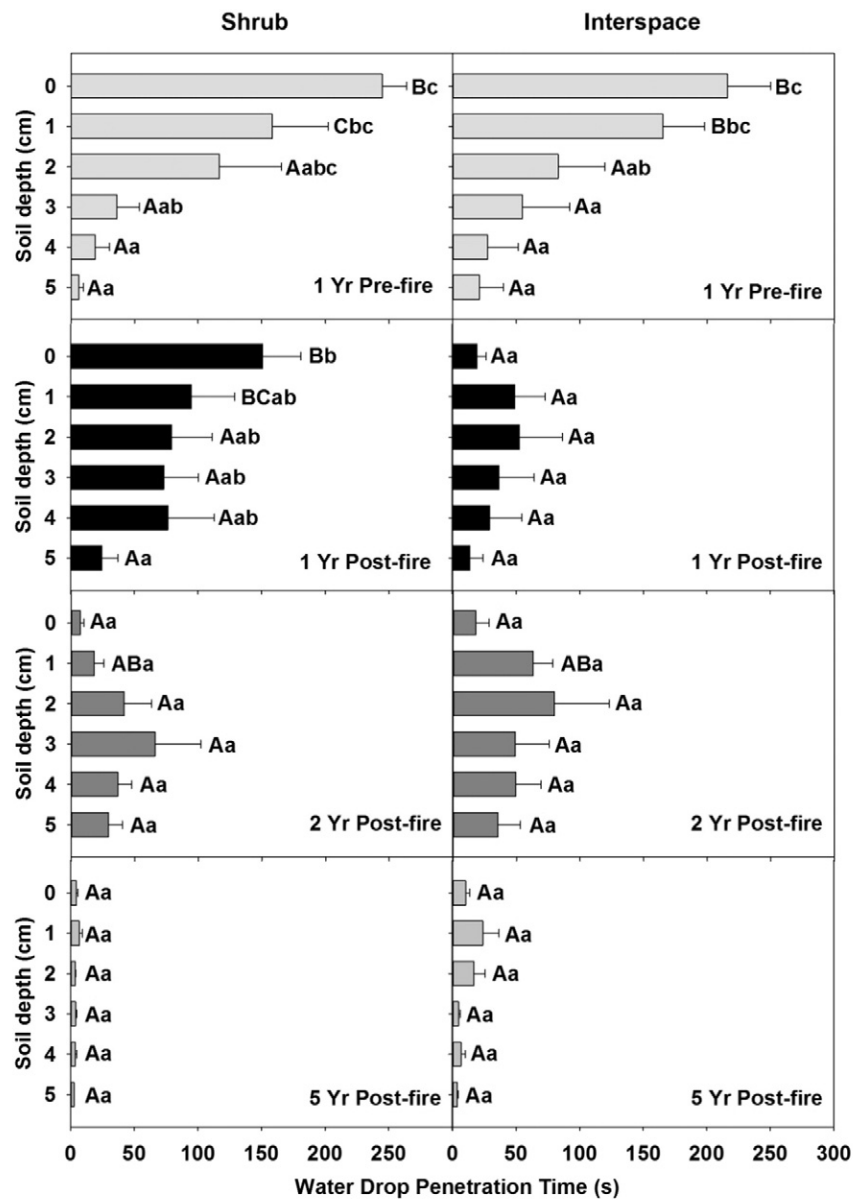

Fig. 5. Soil water repellency assessed as water drop penetration times (WDPT, 300 s maximum) measured at 0–5 cm soil depths on shrub and interspace rainfall simulation plots (0.5 m^2) at the Upper Sheep Creek study site. Soils were classified as slightly water repellent if WDPT was 5 to 60 s and strongly water repellent if WDPT exceeded 60 s (Bisdom et al., 1993). Error bars depict standard error. Means for a specific soil depth across years in a panel column (shrub or interspace microsite) followed by a different upper case letter are significantly different ($P < 0.05$). Means across depths within a treatment and year combination for a given microsite followed by a different lower case letter are significantly different ($P < 0.05$).

velocities 1 yr post-fire were three- to four-fold higher than pre-fire (Table 3). Cumulative runoff was stable from 1 yr to 2 yr post-fire for most release rates, but flow in the second year post-fire tended to form wider flowpaths with slower velocity than measured in the first year after the burn. Cumulative runoff from all but the 15 L min^{-1} rate returned to pre-fire levels by the 5 yr post-fire. Released overland flow in the 5 yr post-fire typically formed one to two flowpaths as wide or wider than those measured before the burn. Erosion responses were similar to those for runoff and, like runoff, increased with bare ground (Fig. 7B). However, total runoff and total sediment (sediment from all rates) were poorly correlated (Fig. 7C). Narrower and higher velocity flows in the 1 yr post-fire generated the highest cumulative erosion (Table 3) and sediment concentrations (Fig. 8). Total sediment decreased by a factor of 4 from the 1 yr to 2 yr post-fire and cumulative sediment by release rate was similar for the 2 yr and 5 yr after the burn. Cumulative sediment across all flow rates in the 5 yr post-fire remained five- to eight-fold greater than pre-fire (Table 3).

4. Discussion

The temporal variability in runoff and infiltration rates demonstrates the complexity of disentangling drivers of post-fire runoff responses (Cerdà and Doerr, 2005; Sheridan et al., 2007; Moody et al., 2013; Williams et al., 2014a). Runoff rates commonly increase immediately post-fire and then decline as vegetation and ground cover return (Cerdà, 1998; Robichaud et al., 2000; Cerdà and Doerr, 2005; Shakesby and Doerr, 2006; Pierson et al., 2011; Robichaud et al., 2016). In this study, runoff of applied rainfall on shrub plots was unchanged 1 yr post-fire and was highest the second year following the burn (Fig. 6A). Runoff declined on interspace plots in the first year post-fire and was highest pre-fire and the 2 yr after burning (Fig. 6B). Such trends are typically attributed to annual variability in soil moisture conditions and soil water repellency at the time of sampling (Shakesby and Doerr, 2006; Williams et al., 2014a; Robichaud et al., 2016). Pierson et al. (2008a, 2008b, 2009) reported similar temporal variability in post-

Table 2
Average rainfall, runoff, infiltration, sediment, and wetting depth response variables measured on burned and unburned rainfall simulation plots (0.5 m^2) pre-fire and one, two, and five years post-fire. Means within a row followed by a different lowercase letter are significantly different ($P < 0.05$).

Rainfall simulation variable	1 Yr pre-fire		1 Yr post-fire		2 Yr post-fire		5 Yr post-fire	
	Shrub coppice	Interspace	Shrub coppice	Interspace	Shrub coppice	Interspace	Shrub coppice	Interspace
Dry-run simulation (64 mm h^{-1}, 45 min)								
Applied rainfall (mm)	45 a	45 a	45 a	46 a	45 a	45 a	44 a	44 a
Cumulative runoff (mm)	6 a	16 bc	4 a	10 ab	9 ab	21 c	3 a	8 a
Runoff-to-rainfall (mm h^{-1}) × 100%	13 a	36 bc	9 a	21 ab	19 ab	47 c	6 a	18 a
Mean infiltration rate (mm h^{-1})[a]	48 c	36 ab	46 bc	44 bc	47 bc	33 a	55 c	48 c
Cumulative sediment (g m^{-2})[a]	6 ab	11 bc	76 d	83 d	15 bc	25 c	2 a	5 a
Sediment/runoff (g m^{-2} mm^{-1})[a]	0.61 a	0.60 a	7.88 c	4.58 b	1.54 ab	1.08 ab	0.69 a	0.53 a
Percent wet at 0–6 cm depth	78 a	79 a	85 a	91 ab	91 ab	87 ab	100 b	100 b
Percent wet at 0–10 cm depth	75 a	73 a	81 a	85 a	83 a	74 a	94 a	91 a
Percent wet at 0–20 cm depth	47 a	52 a	60 a	61 a	49 a	55 a	54 a	60 a
Percent of plots with runoff	63	88	38	75	83	100	67	100
Wet-run simulation (102 mm h^{-1}, 45 min)								
Applied rainfall (mm)	72 a	72 a	73 a	73 a	72 a	72 a	75 a	74 a
Cumulative runoff (mm)	21 a	42 c	22 a	32 ab	34 bc	47 c	21 a	28 ab
Runoff-to-rainfall (mm h^{-1}) × 100%	30 a	58 c	30 a	44 ab	47 bc	65 c	28 a	37 ab
Mean infiltration rate (mm h^{-1})[a]	68 bc	41 a	68 bc	55 bc	52 ab	35 a	72 c	62 bc
Cumulative sediment (g m^{-2})[a]	13 a	21 a	135 b	223 b	144 b	67 ab	62 ab	22 a
Sediment/runoff (g m^{-2} mm^{-1})[a]	0.59 a	0.52 a	4.81 b	6.72 b	3.72 b	1.31 a	2.42 ab	0.67 a
Percent of plots with runoff	100	100	100	100	100	100	100	100
No. of plots	8	8	8	8	6	6	6	6

[a] Means based solely on plots that generated runoff.

fire hydrologic responses from simulated rainfall on sagebrush rangelands with coarse-textured soils. They found the temporal variability in infiltration and runoff responses was driven by annual fluctuations in soil water repellency, even though experiments were conducted under similar soil moisture conditions (1% to 8% gravimetric) each year of the studies (Pierson et al., 2008a, 2008b, 2009). In this study, soil moisture conditions were relatively dry each year, and soil water repellency exhibited significant temporal variability (Fig. 5, Table 1). We attribute the decline in runoff on interspaces 1 yr post-fire to the associated decline in soil water repellency (Fig. 5; Table 1) and moderate recovery of vegetation and ground cover. The high amount of ground cover on shrub coppices 1 yr post-fire buffered persistence of strongly water repellent surface soils and repellency effects on runoff from those plots (Meeuwig, 1971; Imeson et al., 1992; Leighton-Boyce et al., 2007; Pierson et al., 2008a, 2009, 2010; Robichaud et al., 2016). The high levels of wet-run runoff in the second year post-fire were most likely associated with rainfall-shedding senescent vegetation (Pierson et al., 2002, 2008b) and soil water repellency undetected by the WDPT methods or the variability common to the WDPT procedure (Huffman et al., 2001; Doerr et al., 2009; Madsen et al., 2011). Pierson et al. (2002, 2008b) observed substantial shedding of rainfall applied on interspace plots in sagebrush with dense cover of senescent grasses and litter, similar to cover conditions in the 2 yr post-fire of this study (Table 1). Madsen et al. (2011) investigated effects of repellency

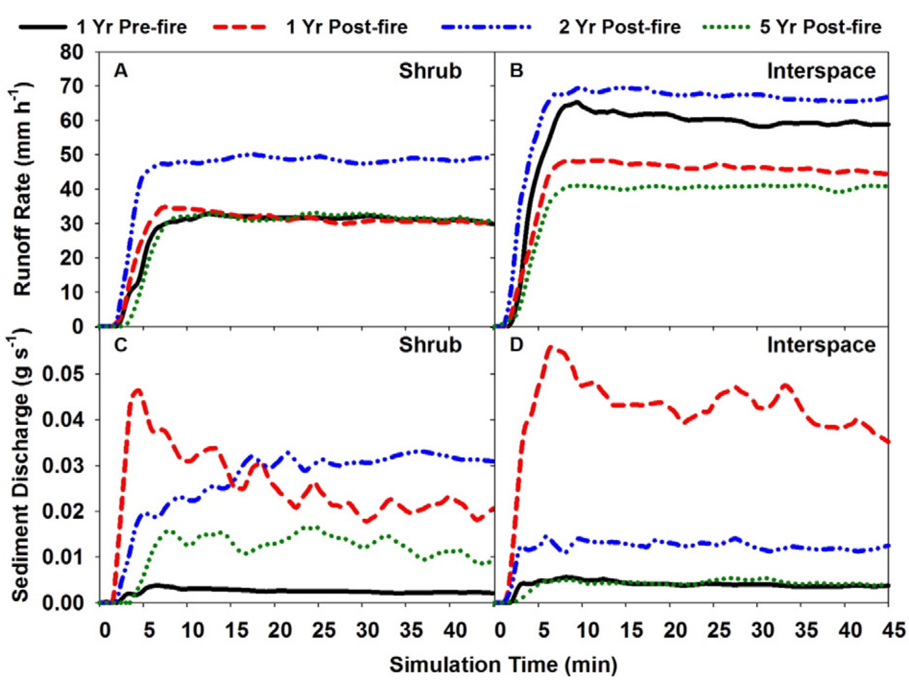

Fig. 6. Runoff rates (A and B) and sediment discharge (C and D) for wet-run (102 mm h^{-1}, 45 min) small-plot rainfall simulations (0.5 m^2) that generated runoff on shrub coppice (Shrub) and interspace (Interspace) microsites pre-fire, 1 yr post-fire, 2 yr post-fire, and 5 yr post-fire at the Upper Sheep Creek study site.

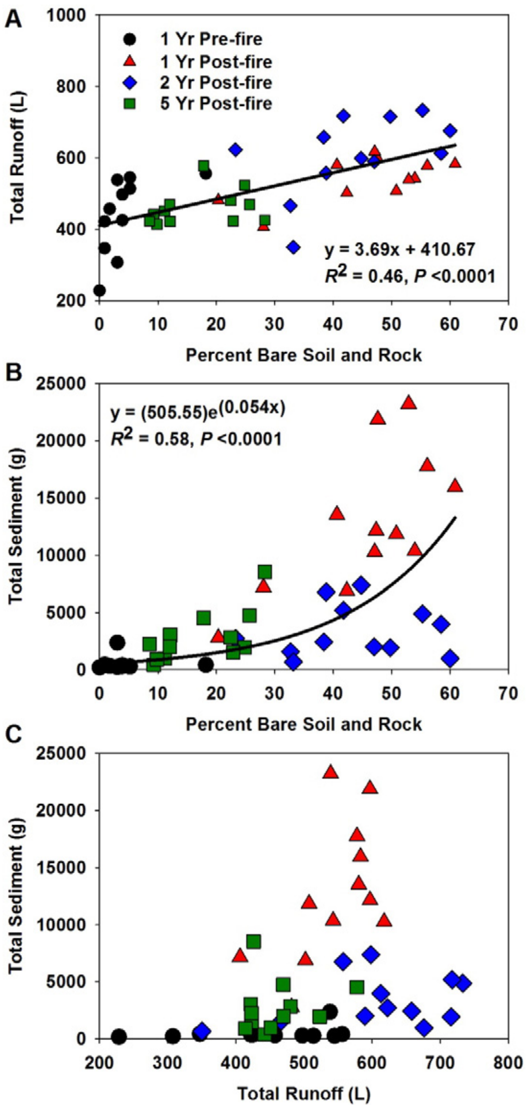

Fig. 7. Total runoff versus percent bare ground (bare soil and rock; A), total sediment versus percent bare ground (B), and total sediment versus total runoff (C) measured on concentrated flow plots (9 m^2) pre-fire, 1 yr post-fire, 2 yr post-fire, and 5 yr post-fire at the Upper Sheep Creek study site. Total runoff and sediment values are the sum of respective cumulative measures for 15, 30, and 45 L min^{-1} flow releases.

Table 3
Runoff, sediment, and flow variables by flow release rate for concentrated flow experiments (9 m^2) pre-fire, and one, two, and five years post-fire. Means within a row followed by a different lowercase letter are significantly different ($P < 0.05$).

Concentrated flow variable	Release rate (L min^{-1}) or total	1 Yr pre-fire	1 Yr post-fire	2 Yr post-fire	5 Yr post-fire
Cumulative runoff (L)	15	24 a	57 b	82 c	44 b
	30	147 a	187 b	205 b	165 ab
	45	263 ab	301 bc	322 c	250 a
	Total	435 a	544 bc	608 c	460 ab
Cumulative sediment (g)[a]	15	35 a	2929 c	355 b	279 b
	30	269 a	6362 c	1447 b	1526 b
	45	182 a	3526 c	1570 b	1001 b
	Total	486 a	12,816 c	3372 b	2805 b
Flow velocity (m s^{-1})[a]	15	0.03 a	0.12 b	0.04 a	–
	30	0.04 a	0.16 b	0.06 a	–
	45	0.06 a	0.18 b	0.07 a	–
Flow path width (cm)[a]	15	42 b	13 a	41 b	78 c
	30	68 bc	16 a	35 b	89 c
	45	81 bc	20 a	39 ab	93 c
Number of flow paths[a]	15	2 ab	3 b	3 b	1 a
	30	2 a	3 b	4 b	2 a
	45	2 a	4 b	4 b	2 a

- No data.
[a] Means based solely on plots that generated runoff.

on infiltration and post-fire vegetation establishment on burned pinyon and juniper woodlands in the Great Basin, USA. They found the WDPT test was able to detect relative differences in severely hydrophobic soil conditions, but was less effective at detecting subcritical soil water repellency (Tillman et al., 1989; Hallett et al., 2001) that also reduced infiltration and vegetation establishment. Madsen et al. (2011) and other studies (Lewis et al., 2006; Robichaud et al., 2008) have found mini-disk infiltrometers to be good tools for the assessment of repellency and its effect on infiltration. Our results and the literature suggest assessment of hydrologic response on potentially water repellent soils should consider multiple methods in assessing infiltration and the strength and persistence of soil water repellency after fire.

The poor correlation in total runoff and total sediment in this study (Fig. 7C) illustrates the influences of sediment supply, vegetation, and ground cover establishment on post-fire hydrologic and erosion responses. Sediment delivery showed an initial fire-induced increase across all plots (1 yr post-fire) as has been shown in previous studies (Robichaud et al., 2000; Cerdà and Doerr, 2005; Pierson et al., 2008b, 2009; Williams et al., 2014b; Shakesby et al., 2015; Robichaud et al., 2016); but elevated levels of runoff on small plots in the 2 yr post-fire generally yielded similar cumulative sediment to pre-fire conditions (Table 2). Erosion increased across all rainfall plots in the 1 yr post-fire without a concurrent increase in runoff relative to pre-fire levels (Table 2). On concentrated flow plots, both runoff and sediment delivery increased 1 yr post-fire (Table 3). Results from the small plots indicate sediment was readily available for detachment and transport immediately following burning. The higher bare soil conditions 1 yr post-fire resulted in amplified runoff, narrow flowpaths with high flow velocities, and elevated sediment detachment and delivery from concentrated flow plots (Table 3; Pierson et al., 2008b, 2009; Al-Hamdan et al., 2012a, 2012b, 2013). Changes in ground cover in interspaces (Table 1) between 1 and 2 yr post-fire reduced basal gaps (Fig. 4B) and forced overland flow to spread out with reduced flow velocity, erosive energy, and sediment delivery (Table 3; Pierson et al., 2009; Al-Hamdan et al., 2012b, 2013). Sediment variables from rainfall simulations and sediment concentration data from overland flow plots further reflect reduced sediment availability in the 2 yr post-fire

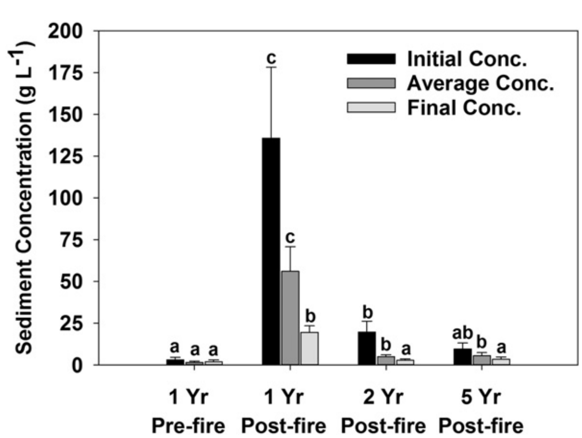

Fig. 8. Initial, average, and final sediment concentrations measured for 15 L min^{-1} concentrated flow simulations pre-fire, 1 yr post-fire, 2 yr post-fire, and 5 yr post-fire at the Upper Sheep Creek study site. Error bars depict standard error. Means for a sediment concentration type (i.e., initial, average, or final) followed by different lower case letters are significantly different ($P < 0.05$).

(Table 2; Fig. 8). The availability of sediment commonly declines over the first few years post-fire as ash and fine sediments are winnowed from the surface (Woods and Balfour, 2008; Smith et al., 2011; Al-Hamdan et al., 2012a; Nyman et al., 2013; Bodí et al., 2014). High runoff rates in the 2 yr post-fire lacked the energy to detach and transport the more limited sediment supply. Our results clearly show that post-fire hydrology and erosion responses are strongly affected by the interacting or ecohydrologic effects of sediment availability and vegetation and ground cover influences on runoff generation and overland flow dynamics (Cerdà and Doerr, 2005; Al-Hamdan et al., 2015; Williams et al., 2016a). Although vegetation and ground cover are typically considered in assessment and prediction of post-fire runoff and erosion (Pierson et al., 2011; Robichaud et al., 2007; Al-Hamdan et al., 2015; Williams et al., 2014a), sediment delivery is commonly modeled as a static variable based solely on inherent erodibility for the parent soil type and burn severity. Our study suggests sediment availability or erodibility in the years post-fire should be considered as a dynamic variable, dictated by inherent soil properties, changes in canopy and ground cover, and the sequence of post-fire runoff and erosion events (Cannon et al., 2001; Cerdà and Doerr, 2005; Shakesby, 2011; Smith et al., 2011; Al-Hamdan et al., 2012a; Nyman et al., 2013).

Vegetation re-established quickly after this fire, which is fairly typical of Great Basin mountain big sagebrush communities in years of favorable precipitation (Pierson et al., 2008b, 2009; Davies et al., 2012; Miller et al., 2013). Total precipitation and snowfall were near or above average (527 mm, 60% as snow) in each of the first four years after the fire (Fig. 2A). Vegetation density at the USC site strongly depends on deep soil recharge from snowmelt and the dense pre-fire vegetation and ground cover was indicative of a productive sagebrush community (Flerchinger et al., 1998; Flerchinger and Cooley, 2000; Chauvin et al., 2011). Pierson et al. (2009) investigated fire impacts on vegetation and hydrology at a nearby mountain big sagebrush site with coarse-textured soils. Precipitation at that site averages 600 mm annually and the majority of the precipitation occurs as snow. Total canopy and ground cover were near 60% and 80%, respectively, at the site pre-fire and were reduced to less than 1% and near 25% immediately post-fire. Within one growing season, canopy and ground cover reached near 100% and 35%, respectively, but, litter (54%) and total ground cover (62%) remained lower than pre-fire levels after two growing seasons. In this study, total canopy and ground cover were reduced from 90 to 95% pre-fire to near 50% the first year after burning and were 80–85% (only slightly below pre-fire) after five growing seasons. Litter cover in this study increased from 47% 1 yr post-fire to 61% after five growing seasons, slightly below pre-fire levels (Fig. 4A). In another mountain big sagebrush study, high severity burning removed nearly 100% of canopy and ground cover and both, along with litter cover, approached pre-fire levels after three growing seasons (Pierson et al., 2008b). Soils at that site are coarse-textured. Annual precipitation was 75%, 55%, and 48%, of the long-term average (350–400 mm) the year of the fire and each of two years thereafter. In a multi-site ($n = 6$) study, Davies et al. (2012) evaluated the recovery of mountain big sagebrush vegetation over a three year period following fire. Sites were located over elevation ranges from 2013 to 2166 m where annual precipitation ranged 400–510 mm. Bare ground and total canopy cover in control areas averaged about 20% and 65% during the study. Over the course of the study, bare ground decreased from about 35% 1 yr post-fire to about 25% the third year after burning, similar to this study, and total canopy cover increased from about 30% to 45% over the same period (Davies et al., 2012). Although vegetation establishment was rapid in this study, sagebrush recovery at the site will likely require 20 to 50 years (Wambolt et al., 2001; Miller and Heyerdahl, 2008; Miller et al., 2011), and the time period required to re-accumulate litter to pre-fire depth is unknown (Miller et al., 2013). Overall, vegetation recovery at the USC site was consistent with other studies from productive mountain big sagebrush communities with annual precipitation near or above 500 mm.

The experimental results from this study expand current knowledge on post-fire ecohydrologic responses and recovery for sagebrush steppe ecosystems in the Great Basin. Pierson et al. (2008b, 2009) conducted similar rainfall simulation experiments (85 mm h^{-1}, 60 min, 0.5 m^2) on burned and unburned areas of steeply-sloped mountain big sagebrush rangelands. In the Pierson et al. (2008b, 2009) studies, infiltration was near 70 mm h^{-1} for unburned and burned conditions 1 yr post-fire. In this study, dry-run simulations (64 mm h^{-1}, 45 min) were conducted under similar antecedent soil moisture (<10%) and hydrophobic soil conditions (unburned WDPT $\approx$ 95 s; burned WDPT $\approx$ 55 s) as those in the Pierson et al. (2008b, 2009) studies, however bare ground was generally 40% to 80% less (5–40% bare). Dry run infiltration rates in this study were about 40% less than infiltration rates for unburned and 1 yr post-fire in the Pierson et al. (2008b, 2009) experiments, but the difference is likely influenced by the lower rainfall application rate in this study. The fine-textured soils at USC produced higher sediment-to-runoff ratios 1 yr post-fire than the coarse-textured sites in the Pierson et al. (2008b, 2009) studies. Temporal variability in soil water repellency and runoff rates in this and the Pierson et al. (2008b, 2009) studies confound comparisons of multiple year post-fire responses (i.e., beyond the 1 yr post-fire). However, this study is the first, to our knowledge, that provides similarly collected infiltration, runoff, and erosion data from fine-textured soils on a mountain big sagebrush community and over five years post-fire. Infiltration and runoff rates in this and the studies by Pierson et al. (2008b, 2009) indicate post-fire hydrologic recovery of mountain big sagebrush communities may occur within one growing season, but may also be delayed where strong soil water repellency persists post-fire. Further, the studies indicate amplified soil loss from overland flow where it occurs may persist for two or more growing seasons following burning. Sediment delivery from similar overland flow experiments in the Pierson et al. (2009) study retuned to near unburned levels within two years post-fire; sediment delivery from similar overland flow experiments in the Pierson et al. (2008b) study remained elevated above unburned levels three years post-fire. In this study, sediment delivery from concentrated overland flow experiments declined from l yr to 5 yr post-fire, but remained elevated above pre-fire conditions the 5 yr after burning. Total ground cover in the 5 yr post-fire in this study was above the 60% threshold for limiting soil loss as suggested by Pierson et al. (2008b, 2009), but the more erodible surface may have counteracted the extensive, but thin layer of litter ground cover in the fifth year.

5. Summary and conclusions

This study investigated the short-term ecohydrologic impacts of fire on a shrubland community common throughout much of the Great Basin Region in the western US, but the process based findings have implications for semi-arid shrublands around the world. Fire removal of nearly 100% of above ground vegetation and ground cover dramatically increased soil erosion, but had less impact on infiltration and runoff of artificially applied rainfall than naturally occurring soil water repellency. Runoff initially was unchanged one year following burning underneath shrub canopies and declined on burned interspaces between shrub canopies. Runoff increased two years post-fire with hydrophobic conditions and decreased five years post-fire when soils were wettable. The initial runoff response was directly attributed to changes in soil water repellency from pre-fire to the 1 yr post-fire. The increase in the second year post-fire was not definitively identified, but was most likely associated with undetected strong hydrophobicity at the mineral soil surface or duff layers based on responses in other years of the study. The fire impacts on vegetation had a more direct effect on erosion from rainfall plots. Erosion increased eight- to more than ten-fold on rainfall plots the first year after burning and declined on rainfall plots as canopy and ground cover approached 85% through the 5 yr post-fire. Fire reduction of canopy and ground cover slightly increased runoff and greatly increased erosion (26-fold) from released concentrated overland flow. Fire removal of vegetation and ground cover allowed overland flow to form narrow flowpaths with high flow velocity and sediment detachment and transport capacity. Runoff from concentrated flow returned to pre-fire levels by the 5 yr post-fire, but erosion remained greater on burned than unburned plots after five growing seasons.

The results clearly demonstrate the complexity of causal drivers of post-fire runoff and erosion. Although erosion followed a typical pulse and decline with vegetation recovery, runoff of rainfall was strongly governed by non-fire related soil water repellency. Erosion was affected by burning, but the temporal changes in soil erosion were dictated by vegetation and repellency effects on hydrologic response and overall sediment availability. The collective results suggest post-fire assessment and prediction of runoff and erosion should carefully evaluate the potential effects of repellency on runoff responses and consider soil erodibility as a dynamic parameter, temporally variable with changes in cover and the sequence of post-fire erosion events. Ecohydrologic recovery in this study was typical for a mountain big sagebrush community in the Great Basin at elevations where precipitation approaches or exceeds 500 mm, but the high levels of sediment delivery from overland flow 5 yr post-fire was unexpected with approximateley 85% ground cover. The persistent high levels of sediment delivery 5 yr post-fire are attributed to the fine-textured soils and limited litter depth accumulation.

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