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Effectiveness of prescribed fire to re-establish sagebrush steppe vegetation and ecohydrologic function on woodland-encroached sagebrush rangelands, Great Basin, USA: Part II: Runoff and sediment transport at the patch scale

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ABSTRACT

Woody species encroachment into herbaceous and shrub-dominated vegetations is a concern in many rangeland ecosystems of the world. Arrival of woody species into affected rangelands leads to changes in the spatial structure of vegetation and alterations of biophysical processes. In the western USA, encroachment of pinyon (*Pinus* spp.) and juniper (*Juniperus* spp.) tree species into sagebrush steppes poses a threat to the proper eco-hydrological functioning of these ecosystems. Prescribed fire has been proposed and used as one rangeland improvement practice to restore sagebrush steppe from pinyon-juniper encroachment. Short-term effects of burning on the ecohydrologic response of these systems have been well documented and often include a period of increased hydrologic and erosion vulnerability immediately after burning. Long-term ecohydrologic response of sagebrush steppe ecosystems to fire is poorly understood due to lack of cross-scale studies on treated sites. The aim of this study is to evaluate long-term vegetation, hydrologic, and erosion responses at two pinyon-juniper-encroached sagebrush sites 9 years after prescribed fire was applied as a restoration treatment. Thirty-six rainfall simulation experiments on 6 m × 2 m plots were conducted for 45 min under two conditions: a dry run (70 mm h^{-1}; dry antecedent soils) and a wet run (111 mm h^{-1}; wet antecedent soils). Runoff and erosion responses were compared between burned and unburned plots. Overall, increases in herbaceous cover in the shrub-interspace areas (intercanopy area between trees) at both sites 9 years post-burn resulted in runoff- and erosion-reduction benefits, especially under the wet runs. While the initially more degraded site characterized by 80% bare ground pre-burn, registered a higher overall increase (40% increase) in canopy cover, greater post-fire reductions in runoff and erosion were observed at the less degraded site (57% bare ground pre-burn). Runoff and erosion for the wet runs decreased respectively by 6.5-fold and 76-fold at the latter site on the burned plots relative to control plots, whereas these decreases were more muted at the more degraded site (2.5 and 3-fold respectively). Significant fragmentation of flow paths observed at the more-degraded site 9 years post-fire, suggests a decreased hydrologic connectivity as a mechanism of runoff and erosion reduction during post-fire recovery.

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1. Introduction

Encroachment of woody plant species into grasslands, shrublands and savannas is a pressing issue that affects arid and semi-arid ecosystems worldwide (Eldridge et al., 2011; House et al., 2003; Stephens et al., 2016). Woody species encroachment into spatially uniform arid and semi-arid vegetation systems has often been associated with an increase in vegetation patchiness (e.g., Cammeraat and Imeson, 1999; Kakembo, 2009) which affects surface processes (e.g., Bergkamp et al., 1996; Ludwig et al., 2005; Puigdefabregas et al., 1999; Valentin et al., 1999). The effects of woody plant encroachment on this pattern and process relationship have been documented for grassland-to-shrubland community transitions in the southwestern US (Schlesinger et al., 1990; Turnbull et al., 2008, 2012; Wainwright et al., 2000), woody plant encroachment in Africa (Manjoro et al., 2012) and South America (Chartier and Rostagno, 2006), and in association with coarsening of woodland community structure in Australia (Ludwig et al., 2007) and dry forests in the western US (Davenport et al., 1998; Wilcox et al., 1996a, 1996b). Encroachment by pinyon (*Pinus* spp.) and juniper (*Juniperus* spp.) tree species into native sagebrush-steppe communities in the western US, the focus of this study, results in fragmentation of these shrub-dominated ecosystems (Bates et al., 2000, 2005; Miller et al., 2000, 2005; Roberts and Jones, 2000) and a decline in delivery of ecosystem services (Davies et al., 2011). Pinyon-juniper encroachment is associated with an increase in vegetation patchiness through loss of herbaceous and shrub vegetation cover, creating vast areas of interconnected bare ground where runoff and sediment accumulate and move rapidly off-site (Pierson et al., 2007, 2010, 2013; Roundy et al., 2017; Williams et al., 2014a).

Various factors have been proposed as key drivers of pinyon-juniper woodland expansion into sagebrush ecosystems. These include fire suppression, livestock grazing (Miller and Rose, 1999), natural recovery of pinyon-juniper previously cleared by European settlers (Romme et al., 2009), climate variability (Miller and Wigand, 1994; Romme et al., 2009) and increased atmospheric CO_2 concentration (Knapp and Soulé, 1996). The degree of tree encroachment has been described by various authors to occur through time in three phases (Johnson and Miller, 2006; Miller et al., 2000; Roundy et al., 2014). Phase I corresponds to an incipient tree encroachment marked by a dominant shrub and herbaceous vegetation cover interspersed by pinyon-juniper trees. In Phase II, trees are co-dominant with shrub and herbaceous vegetation and each vegetation type significantly contributes to ecosystem processes. Continued tree recruitment and growth in Phase II leads to Phase III, a state characterized by a considerable reduction in shrub and herbaceous cover and a dominance of trees. The transition from Phase II to Phase III is associated with heightened intraspecies competition between trees, shrubs, and grasses followed by profound changes to understory vegetation (Miller et al., 2000) with significant ecohydrologic consequences (Petersen et al., 2009; Williams et al., 2016b). Experimental research has found that interspaces between trees and shrubs in these systems form patches of hydrologically connected bare ground, leading to reduced rainfall interception and infiltration and increased runoff and amplified soil loss (Petersen and Stringham, 2008; Pierson et al., 2007, 2010, 2013; Roundy et al., 2017; Williams et al., 2014a).

Treatment options involving tree removal are commonly used to restore sagebrush steppe vegetation and ecohydrologic function on pinyon and juniper encroached sites (Bates and Svejcar, 2009; Miller et al., 2005; Pierson et al., 2014, 2015; Sheley and Bates, 2008; Stephens et al., 2016). The ecological trajectory toward recovery of a sagebrush ecosystem treated by tree removal is dependent on various factors including the initial degree of tree encroachment, local soil and environmental conditions, and the type of treatment applied (Bates et al., 1998; Chambers et al., 2014; Miller et al., 2005, 2014). In a more general sense, each ecological path taken in these rehabilitation efforts is an expression of complex interplays between biotic processes dominated by vegetation and abiotic processes controlled by hydrologic function (Turnbull et al., 2012; Wilcox et al., 2003; Williams et al., 2016a). Hydrologic function describes the capacity of a site to store water resources for safe release and the resilience of the site to changes in this capacity (Pyke et al., 2002).

Prescribed fire is commonly used to induce pinyon and juniper mortality on tree-encroached sagebrush steppe sites (McIver and Brunson, 2014; Miller et al., 2014), but can have both positive and negative short-term ecosystem impacts (Pierson et al., 2013, 2014, 2015; Williams et al., 2014a; 2016a; 2018). Fire has profound effects on soil nutritional status (Caon et al., 2014; Girona-García et al., 2018; Guinto et al., 2001; Kennard and Gholz, 2001; Mataix-Solera and Doerr, 2004; Rau et al., 2007) and physical properties (e.g., Chief et al., 2012; Debano et al., 1970; Granged et al., 2011b; Morris and Moses, 1987; Prosser and Williams, 1998; Stavi et al., 2017). Soil water repellency is one of the most commonly researched fire-induced alterations to soil physical properties (e.g., Alcañiz et al., 2018; Granged et al., 2011a; Kennard and Gholz, 2001; Pierson et al., 2008; Scharenbroch et al., 2012). These fire-induced changes to intrinsic soil properties have been linked to exacerbated levels of sediment yield and runoff in the immediate post-fire period (Inbar et al., 1998; Pierson et al., 2011; Robichaud, 2005; Williams et al., 2014b). Nevertheless, adverse effects of fire on ecosystem hydrology are often transient and can dissipate over time (Huffman et al., 2001; MacDonald and Huffman, 2004; Williams et al., 2014a). Studies of pinyon and juniper woodlands by Pierson et al. (2013, 2015) and Williams et al. (2014a) found prescribed burning increased hydrologic connectivity between sediment producing bare patches and thereby amplified soil erosion the first year post-fire. Williams et al. (2014a, 2016b) found that increases in herbaceous cover two years post-fire on a woodland-encroached sagebrush site reduced connectivity of hydrologic and erosion processes and improved hydrologic function. The short-term studies of Pierson et al. (2015) and Williams et al. (2014a, 2016b) demonstrate the temporal nature of fire impacts in the short-term, but literature and knowledge remain limited regarding the long-term hydrologic and erosion impacts of pinyon and juniper removal by fire across the vast ecological domain occupied by these species (Williams et al., 2018).

This study and its companion study, Part I (Williams et al., 2020), fit within a broader study, the Sagebrush Steppe Treatment Evaluation Project (SageSTEP) aimed at investigating the ecological impacts of pinyon and juniper encroachment and tree removal practices in sagebrush steppe (McIver and Brunson, 2014). Several studies have been conducted within SageSTEP to understand the immediate and short-term effects of pinyon and juniper removal by fire on ecosystem hydrologic function and erosion (Pierson et al., 2013, 2014, 2015; Williams et al., 2014a, 2016a). These studies used a combination of small- and large-plot rainfall simulation and concentrated flow experiments to track and understand short-term changes to surface processes, such as runoff generation, splash-sheet erosion, and concentrated flow runoff and erosion. The Part I and Part II (this study) studies expand on the short-term findings from the earlier SageSTEP studies by quantifying longer-term ecohydrologic responses of the study sites to tree removal by prescribed fire across multiple spatial scales. Although this research is focused on the sagebrush steppe ecosystem in the Great Basin, USA, the ecohydrologic relationships assessed and fire impacts on runoff and erosion processes are likely broadly applicable to similar sparsely vegetated and water-limited rangeland and woodland ecosystems around the World.

The objective of this study is to evaluate the long-term (9 yr post-fire) effectiveness of prescribed fire to re-establish sagebrush steppe vegetation structure and thereby improve ecohydrologic function at two sagebrush sites within the Great Basin, USA, in the later stages (Phase II-III) of pinyon and juniper encroachment. Our companion study, Part I specifically quantified long-term impacts of tree removal by prescribed fire on: (1) vegetation and ground surface conditions at the small-plot scale (0.5 m^2), patch scale (~10 m^2), and hillslope scale (990 m^2 plots), (2) infiltration, runoff generation, and sediment

delivery by rainsplash and sheetflow (splash-sheet) processes during rainfall simulations at the small-plot scale, and (3) runoff and sediment delivery solely by concentrated overland flow processes at the patch scale. Part II (this study) expands on the inference space of the Part I study through use of large plot (12 m^2) rainfall simulation experiments in burned and unburned (control) treatments at the same study sites as in Part I to quantify long-term prescribed-fire treatment effects on canopy and ground surface conditions, vegetation structure, and runoff and erosion from combined splash-sheet and concentrated overland flow processes at the patch scale. Rainfall simulations at the large plot scale in this study were designed to quantify treatment effects for areas representative of the intercanopy and areas immediately underneath/adjacent to tree canopies.

2. Materials and methods

2.1. Study sites

This study was conducted at two experimental sites (Marking Corral and Onaqui) that were part of the SageSTEP network and have been extensively described in previous studies (e.g., Pierson et al., 2010, 2015; Williams et al., 2016a). General site characteristics are summarized in Table 1. The Marking Corral site is a single-leaf pinyon-Utah juniper (*P. monophylla* Torr. & Frém – *J. osteosperma* [Torr.] Little) community located 27 km northwest of Ely, Nevada. The Onaqui site is a Utah juniper community located 76 km southwest of Salt Lake City, Utah. These sites are public lands and were managed for grazing purposes by the Bureau of Land Management (BLM) until autumn 2005, when they were temporarily excluded from grazing during the SageSTEP study.

Soils for both sites are classified using the United States Department of Agriculture taxonomy. At Marking Corral, the soil is mapped as a complex Segura (Loamy, mixed, superactive, frigid Aridic Lithic Argixerolls) – Upatad (Loamy-skeletal, mixed, superactive, mesic Aridic Lithic Argixerolls) – Cropper (Loamy-skeletal, mixed, superactive, frigid Aridic Lithic Argixerolls) while the soil at Onaqui is mapped as a Borvant soil series (Loamy-skeletal, carbonatic, mesic, shallow Petrocalcic Palexerolls). Slopes measured at Marking Corral in this study ranged between 7% and 15% while at Onaqui they ranged from 12 to 21%. Surface soil texture at both sites was sandy loam. Soil bulk densities at 0–5 cm soil depth at Marking Corral averaged 1.26 g cm^{-3} in interspaces between shrubs and trees, 1.02 g m^{-3} in shrub canopy areas (shrub coppices), and 1.03 g m^{-3} in tree canopy areas (tree coppices). The same measure averaged 1.08 g cm^{-3} in interspaces, 1.05 g cm^{-3} in shrub canopy areas, and 0.90 in tree canopy areas at Onaqui.

Measured tree canopy cover in 2006 before prescribed burning was applied (Pierson et al., 2010; Williams et al., 2016a) was 27% (21% pinyon; 6% juniper) at Marking Corral and 28% (juniper) at Onaqui. Tree densities were 465 (pinyon) and 114 (juniper) trees ha^{-1} at Marking Corral and 532 trees ha^{-1} (juniper) at Onaqui. Average tree heights were 2.3 m and 1.9 m for pinyon and juniper trees at Marking Corral and 2.3 m for juniper trees at Onaqui. Shrubs occupied 20.6% and 0.5% of the open area between tree canopies (intercanopy area, shrub-interspace zones) at Marking Corral and Onaqui, respectively, while herbaceous species occupied 8.5% and 10.4% of these intercanopy areas. Bare soil and rock cover made up 68.3% of the intercanopy area at Marking Corral and 84.1% of the intercanopy at Onaqui. Prescribed fires at the sites were applied in late summer – early autumn of 2006 as described in Williams et al. (2020). Burn severity across the sites ranged from low to moderate as evident by presence of burned shrub skeletons, residual live and scorched tree needles, blackened litter, and downed-woody debris following the fires (Pierson et al., 2014, 2015; Williams et al., 2016b, 2020).

2.2. Experimental design

The rainfall simulation experiments were conducted in 4 separate field campaigns. The field campaigns occurred on June 1–6 and June 15–20, 2015 respectively for burned and control plots at Onaqui and August 27–September 3 and September 14–20, 2015 respectively for the burned and control plots at Marking Corral. Rainfall simulations were performed on a total of 20 plots at Marking Corral and 16 plots at Onaqui. All plots were 2 m wide × 6 m long and were oriented with the long axis perpendicular to the hillslope contour. Plots were bordered on the upslope end and both sides with sheet metal walls inserted approximately 5 cm into the ground and contained a runoff collection trough and plot outlet (supercritical flume) at the downslope end (Fig. 1). This experimental setup has also been used in other rainfall simulation studies on rangeland (e.g., Cadaret et al., 2016; Nouwakpo et al., 2016). At both sites, the total number of plots was evenly distributed across burned and control treatments. In each treatment area, plots were selected in two microsites: in the intercanopy open space between tree canopies (shrub-interspace zones) and on the tree coppices (tree zones). The microsite experimental design is consistent with that of pre-treatment and short-term post-treatment SageSTEP companion studies of the sites (Pierson et al., 2010, 2015; Williams et al., 2016b). Shrub-interspace zones encompassed 2–3 shrub coppices in addition to bare or grass-covered interspace areas. Tree zones were dominated by the tree coppices, but also overlapped with bare or grass-covered interspace areas with occasional shrub cover (especially in the burned area). Four plots were selected in the shrub-interspace zones and 6 plots in the tree zones within both the burned and control treatments at Marking Corral, resulting in 10 plots in the burned area and 10 plots in the control area. The choice of 6 plots in the tree zones at Marking Corral was to capture any potential difference between pinyon and juniper zones (3 pinyon tree zones and 3 juniper zones). Such species effect was not found to be statistically significant ($P > 0.05$) in preliminary analyses and therefore data across pinyon and juniper tree-zone plots were combined in the analysis presented in this paper. At Onaqui, an equal number of 4 plots was selected for each microsite within burned and control areas. In tree zones at both sites, burned and live trees were carefully removed by chainsaw (leaving a ~50 cm tall stump) to allow for placement of the rainfall simulator and to ensure an even distribution of rainfall across each plot.

Table 1
General description of the Marking Corral and Onaqui study sites.

	Marking Corral	Onaqui
Coordinates (latitude, longitude)	39°27′17″N, 115°06′51″W	40°12′42″N, 112°28′24″W
Altitude (m)	2250	1720
Soil texture	Sandy loam	Sandy loam
Slope range (%)	7–15	12–21
Annual precipitation (mm)	307	312
Tree species	Single leaf pinyon (*P. monophylla* Torr. & Frém) Utah juniper (*J. osteosperma* [Torr.] Little)	Utah juniper (*J. osteosperma* [Torr.] Little)
Tree density (trees/ha)	579	532

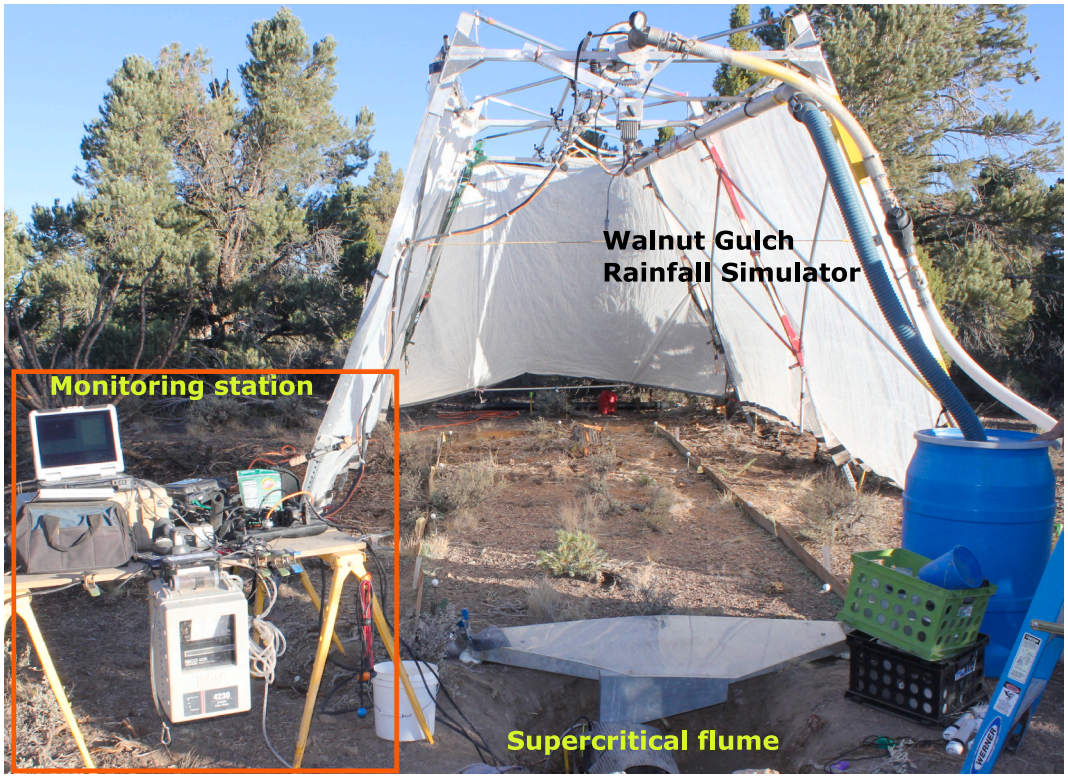

Fig. 1. Field experimental setup showing the Walnut Gulch Rainfall simulator, the runoff measurement flume and the instrument monitoring station.

2.3. Vegetation measurements and rainfall simulation experiments

Vegetation cover was measured on each plot prior to rainfall simulation using a laser point frame (VanAmburg et al., 2005). Canopy cover and ground cover on each plot were sampled at 5 transects running the width of the plot and along which 20 vertical laser lines were projected to record vegetation and ground cover features intercepted. At each point, the intersection between the vertical laser ray and a standing plant leaf or branch was classified as a canopy point with the species of the plant recorded while ground points include soil, litter, plant base, biological soil crust or rock. Plot level canopy cover was grouped by life form (shrubs, annual and perennial grasses and forbs) and derived as the number canopy points in each life form divided by 100 (the number of sample points). Canopy cover represents the proportion of aerial coverage occupied by plants leaves and stems and was measured for various vegetation life forms including, shrubs, annual grasses (regenerating annually from seeds) and perennial grasses (regenerating from roots) and forbs. The sum of annual and perennial grass cover and forb cover is the total herbaceous plant cover. Plot level ground cover in each ground cover class was calculated as the number of points in each class divided by 100. Plot slope was measured with a Nikon NPR 352 total station (Nikon Corporation, Tokyo, Japan) which was also used to survey ground control points for the 3D reconstruction procedure described below.

Rainfall simulation experiments were performed on the 12 m^2 plots using a Walnut Gulch Rainfall Simulator (WGRS) (Paige et al., 2004) (Fig. 1). The WGRS consists of an oscillating central boom fitted with four Veejet 80–100 nozzles (Spraying systems, Inc., Wheaton, Ill.). This simulator has an effective spray area of 6.1 m × 2 m which dictated the 6 m × 2 m, plot size used in this study. As recommended in Paige et al. (2004), a nozzle height of 2.44 m was used in this study to approach raindrop energy within the range encountered during natural rainfall events. Simulation water used in this study was obtained from a municipal fire hydrant at Marking Corral and from a local reservoir at Onaqui. All water was pumped through a filtration system prior to application by the rainfall simulator. Two rainfall events were applied on each plot. A first rainfall event of intensity 70 mm h^{-1} was applied for 45 min on the soil surface in its initial moisture condition (dry run). Initial soil moisture content was measured before the dry run. A second rainfall event occurred approximately 45 min after the dry run. The 45-min wait time was needed to collect various data including final soil moisture content of the soil after the dry run simulation and the amount of rainfall applied. The second rainfall event (wet-run) was applied at an intensity of 111 mm h^{-1} for 45-min. Post-event soil moisture content and rainfall amount were also collected at the end of the wet-run. Initial and final volumetric soil moisture contents were measured for each event using a HydroSense II soil moisture sensor with a 12-cm-long probe (Campbell Scientific, Logan, UT) inserted at an oblique angle. Soil moisture measurements were performed near the 4 corners of each plot and an average soil moisture calculated. The 70 mm h^{-1} and 111 mm h^{-1} rainfall intensities were respectively chosen to correspond to the 5-min thunderstorms of 10- and 50-year return periods for the area (Bonnin et al., 2006). It is important to note that the simulated rainstorms were applied for 45 min to facilitate the generation of steady-state runoff, but this resulted in actual event return periods greater than 100 years. Runoff during rainfall simulations was conveyed into a supercritical flume at the downslope end of each plot where a Teledyne 4230 flow meter (Isco, Inc., Lincoln, NE) measured discharge at a rate of four samples per minute. Manual timed runoff samples were also collected approximately every 2 min to validate runoff discharge measures from the flow meter. Runoff discharge measurements from the flow meter were displayed in real-time on a computer screen via a serial communication.

Runoff sediment samples were collected in 1 L bottles at 3 min intervals during all rainfall simulation events. Sediment concentration for each sample interval was determined in the laboratory by decanting sediment samples, oven-drying and weighing the sediments and dividing the mass of sediments by the volume of runoff sample. From these runoff and sediment data, a suite of variables was extracted. For each rainfall simulation event, the steady-state condition was defined as

the greater of runoff discharge during the trendless period that occurred after the rapid rising limb of the hydrograph or during the last 10 min. A steady-state runoff discharge (mm h^{-1}), and sediment concentration (g L^{-1}) were calculated as the average runoff discharge and the sediment concentration measured during the steady-state condition. The steady-state discharge was subtracted from the rainfall intensity to obtain the steady state infiltration rate. Cumulative or total runoff (mm) and sediment loss (g) were calculated for each event by integrating discharge rates and sediment concentrations throughout the simulation duration and were divided by runoff duration to get an event average runoff rate (mm h^{-1}) and average sediment discharge (g s^{-1}). Sediment flux (g s^{-1} m^{-2}) was calculated by dividing the average sediment discharge by the plot area (12 m^2). A rainfall to runoff ratio was calculated for each simulation event by dividing cumulative runoff by cumulative rainfall depth. The average sediment yield per unit runoff (g m^{-2} mm^{-1}) for each event was calculated by dividing the cumulative sediment loss (g) by the plot area and further dividing the result by the cumulative runoff depth (mm). The organic matter content of eroded material for each simulation event was determined using the loss-on-ignition method (Nelson and Sommers, 1996).

2.4. Three-dimensional soil surface characterization

Before and after each rainfall event, 200–300 overlapping photographs of the respective plot were taken and used in Structure-from-Motion (SfM) software Agisoft PhotoScan v. 1.3 to reconstruct soil surface microtopography. In this study, three-dimensional (3D) data was used to quantify soil surface roughness and a flow path Connectance Index as described below. To calculate soil roughness, a bare earth model extraction described by Zhang et al. (2003) was first applied to the original point clouds to remove aerial vegetation features (Fig. 2A and B). A plane was then fitted to the bare earth model and the surface roughness calculated as the average of distances from each point of the bare earth model to the plane. The Connectance Index was calculated by sampling the bare earth point cloud into a 0.01 m gridded Digital Elevation Model (DEM) in ESRI ArcGIS. A bottom-hat operator was applied to the DEM to detect areas of lower elevation in the DEM where runoff is likely to converge (Fig. 2C). These local flow convergence areas have been successfully used as a representation of flow path network in a DEM (Nouwakpo et al., 2016; Rodriguez et al., 2002; Schwanghart et al., 2013).

The flow path maps were binary rasters of value 1 for grid points classified as flow paths and 0 otherwise. These maps were then imported in the landscape spatial pattern analysis program FRAGSTATS v. 4.2 (McGarigal et al., 2002) to compute the Connectance Index (CONNECT) parameter. FRAGSTATS was originally developed to quantify the spatial organization of the landscape using binary maps of vegetation as inputs. In this study, the inputs were binary maps of the flow pathways of each plot which provided a characterization of the spatial structure of water flow pathways on each plot. The Connectance Index is calculated in FRAGSTATS as the proportion of functional joining among flow path patches. Two patches of flow paths are considered connected when the distance between them is less than 0.2 m.

2.5. Statistical analyses

In this paper, statistical analyses were conducted using R (R Development Core Team, 2015). A two-way mixed model Analysis of Variance followed by Tukey's post-hoc test was used to compare vegetation, hydrology, and erosion variable mean differences across microsites and treatments. All hydrology and erosion comparisons were performed at each site separately (i.e. data at Marking Corral were not compared to Onaqui) with variance in the dependent variables (e.g., runoff duration, cumulative runoff, etc.) evaluated against independent factors microsites (Tree coppice vs. Shrub-Interspace) and treatment (Burned vs. Control). For the dependent hydrology and erosion response variables, a separate analysis was conducted for the dry and wet runs. Comparisons for vegetation and site characteristics were carried out across sites (i.e. Marking Corral compared to Onaqui data), microsites and treatments.

Factors controlling hydrology and erosion responses were further investigated using a multiple linear regression. Explained variables were: the time needed for runoff to initiate (time to runoff), steady state discharge, infiltration rate, cumulative runoff, average runoff, total

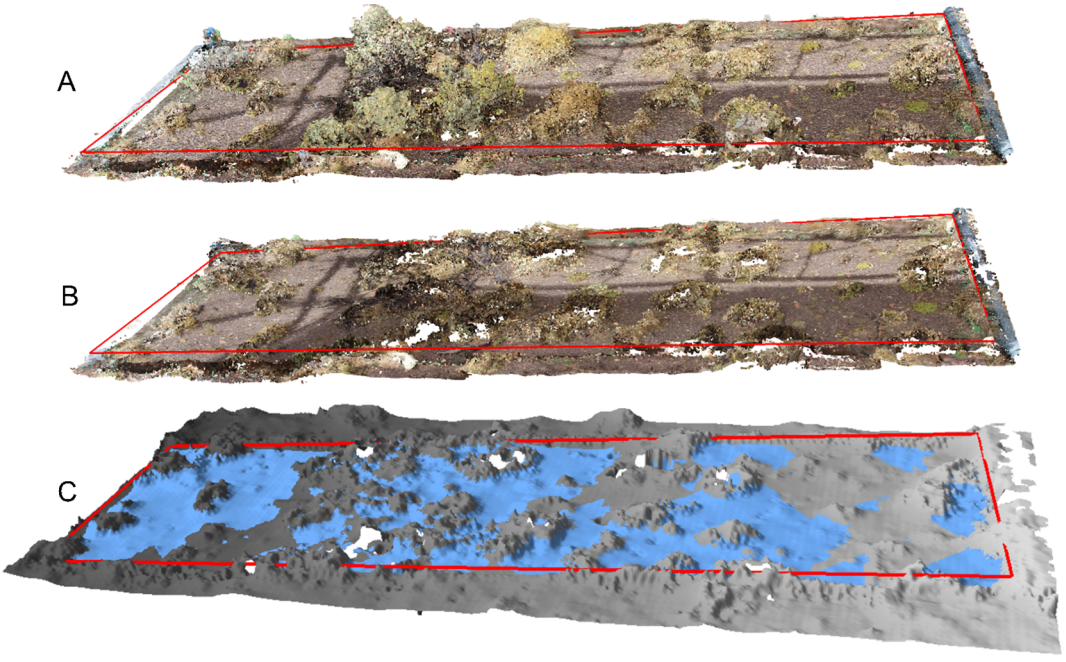

Fig. 2. Example of three-dimensional reconstruction of the 6 m × 2 m erosion plot showing the original point cloud (A), the same point cloud freed of aerial vegetative features (B), and the interpolated Digital Elevation Model (DEM) at 0.01 m resolution (C) with areas of flow concentration colored in blue. The plot outline is marked in red and holes in the DEM result from missing points after vegetation removal. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

sediment loss, sediment concentration, sediment flux and sediment organic matter content. Explanatory variables were: microsite (Tree coppice vs. Shrub-Interspace), treatment (Burned vs. Control), slope angle, total canopy cover, litter cover, and initial soil moisture content. This multiple linear regression analysis was performed on the dry and wet runs separately but initial soil moisture content was only used for the dry runs because soil moisture content after these events and before wet runs were considered to be homogenized and at or near saturation. A stepwise variable selection using the Akaike Information Criterion (Venables and Ripley, 2002) was applied to keep the best explanatory variables in each multiple linear regression. In addition, a dominance analysis (Nimon and Oswald, 2013) was performed on the retained multiple linear regression models to determine the contribution of each explanatory variable to the coefficient of determination (R^2) of the regression. The general dominance of each statistically significant variable is converted into a percentage of the R^2 and displayed in a graphical format as a dominance dot matrix in which dot sizes are proportional to the general dominance they represent. This dot matrix is presented in the result section while the detailed multiple regression analyses and dominance calculations are added as supplemental materials. The threshold for statistical significance for all analyses was set at an alpha p-value of 0.05.

Runoff and erosion data obtained in this study were compared to data collected before prescribed fire treatment and 1-year post-treatment at the same sites and presented in Williams et al. (2016a). In the Williams et al. (2016a) study, rainfall simulation data were collected on 13 m^2 (2 m × 6.5 m) plots similar to the 12 m^2 plots used in our study. In the Williams et al. (2016a) study, a Colorado State University-type rainfall simulator (Holland, 1969) was used. This simulator utilizes stationary sprinklers fixed on rigid pipes at 3.05 m above the soil surface to deliver rainfall at fixed rates. Rainfall intensities of 64 mm h^{-1} and 102 mm h^{-1} were applied in the Williams et al. (2016a) study, which were similar to the intensities applied in the current study. Like the current study, the dry and wet runs in the Williams et al. (2016a) study were consecutively applied and rainfall duration was also 45 min. Vegetation data (namely herbaceous canopy cover and litter) pre- and 1-year post-burn were presented in Williams et al. (2016a) and compared to data collected in this study. To compare hydrologic responses across studies, rainfall to runoff ratios for the wet runs were used. Soil erosion response was normalized across studies using sediment yield per unit runoff.

3. Results

3.1. Biophysical and site characteristics

Biophysical characteristics of the rainfall simulation plots at Marking Corral and Onaqui are summarized in Table 2. As illustrated in the control areas, initial canopy cover was greater at Marking Corral compared to Onaqui. Overall, average total canopy cover was higher in the burned treatment areas than in the control areas. While this treatment effect was statistically significant at the Onaqui site, higher variability in canopy cover across treatments and microsites at Marking Corral precluded statistical significance of the effects of treatment and microsite. At both sites, significant herbaceous cover increases in the burned area explained the increase in total canopy cover across microsites. Fig. 3 shows an example of a shrub-interspace plot at Onaqui in the control area (Fig. 3A) compared to the same microsite in the burned area (Fig. 3B), illustrating the higher herbaceous vegetation cover for the burned treatment. At Marking Corral, total herbaceous cover ranged from 4 to 24% in the control shrub-interspace zones and increased to 23–39% in the burned shrub-interspace zones. At Onaqui, the same trend was observed with shrub-interspace herbaceous canopy cover ranging from 5 to 12% in the control and 21–39% in the burned areas. An increasing effect of burning on herbaceous cover was also noted in the tree zones. In these microsites, herbaceous cover increased from 2–11% to 22–29% at Marking Corral and 4–16% to 34–65% at Onaqui. Herbaceous species composition was dominated by grasses at Marking Corral with a low number of forbs (0–4% across treatments and microsites) while forbs were present in higher proportion at Onaqui (0.7–14%). Burning promoted an increase in both annual and perennial herbaceous species with the highest gains in annual species at tree coppice microsites 9 years after fire. At Marking Corral, canopy cover of annual species in shrub-interspaces increased from 0 to 0.3% in the control to 3–8% in the burned areas. A similar increase was noted in tree zones where annual species made up 0–2% of canopy cover in the control and 4–19% in the burned areas. Annual species were present at Onaqui in higher amounts than at Marking Corral, but also showed an increase in the burned areas. At both sites, perennial species constituted the dominant herbaceous fraction in the control areas (98% and 77% of herbaceous cover respectively at Marking Corral and Onaqui).

Bare soil in shrub-interspace zones declined from 55.8% to 31.2% at Marking Corral and 38.2% to 15.5% at Onaqui as the result of burning (Table 2). Burning also significantly reduced bare soil in tree zones at Marking Corral (21.7% control, 4.4% burned) but had no statistical effect on bare soil in these microsites at Onaqui (~12% across treatments). Litter dominated ground cover at Marking Corral with ground cover at this site composed of 29–99% of litter and 0–39% of rock cover across microsites and treatments. At Onaqui, litter cover ranged from 6 to 87% while rock content ranged from 3 to 64% across microsites and treatments. At Onaqui, litter on tree coppices remained virtually unchanged 9 yr after burning while at Marking Corral, litter increased on the same microsite of the burned area. Burned shrub-interspaces at both sites contained more litter compared to controls.

Surface roughness estimated from 3D reconstruction of erosion plots varied from 28 mm to 52 mm at Marking Corral and 14–58 mm at Onaqui, with significant treatment effects occurring solely on shrub-interspace zones at Onaqui (Table 2). Results of the effect of burning on flow path connectivity are presented in Fig. 4. Flow path connectivity CONNECT was not statistically different across microsites in the control areas of both sites. Burning did not change flow path connectivity at Marking Corral but resulted in significantly fragmented flow path patches at Onaqui. The average functional joining between flow paths was 3.9% at Marking Corral across microsites, but was 3.1% in the burned areas at Onaqui compared to a higher average of 5.4% in unburned areas at the same site. These results are consistent with the observed significant increase in soil surface roughness observed at Onaqui, suggesting that newly established vegetation patches altered surface microtopography and effectively disaggregated flow paths.

3.2. Hydrologic and erosion response

Tables 3 and 4 summarize results of the rainfall simulation experiments at Marking Corral and Onaqui. All ten dry runs generated runoff in the control area at Marking Corral versus six out of ten in the burned area. Runoff generation was similar at Onaqui where all eight dry run events produced runoff in the control area against seven out of eight in the burned area. All wet runs generated runoff at both experimental sites. For both dry and wet runs, the time to runoff was on average 6.3 min and was similar across all treatments and microsites at both sites.

Dry-run steady-state runoff discharge and cumulative runoff were unaffected by burning and microsite (Tables 3 and 4). For the wet runs at Marking Corral, a statistically significant decrease in steady state runoff discharge (54.3 mm h^{-1} vs. 9.3 mm h^{-1}) and cumulative runoff (37.1 mm vs. 5.7 mm) was observed in burned shrub-interspace zones as compared to those in the control (Table 3). At Onaqui, burning reduced wet-run cumulative runoff in the shrub-interspace by 23 mm from 38.5 mm on control plots but did not impact wet-run steady-state runoff discharge in these microsites (Table 4). At both sites, wet-run steady-state runoff discharge and cumulative runoff were unaffected by burning on the tree coppice microsite. A higher infiltration on burned

Table 2
Biophysical surface characteristics measured on 12 m^2 rainfall simulation plots in burned and control areas at Marking Corral and Onaqui across treatments and microsites 9 years post-fire. Means sharing common lower case letters across an entire row are not statistically different ($p \geq 0.05$).

	Marking Corral				Onaqui			
	Control		Burned		Control		Burned	
	Shrub-interspace	Tree Coppice	Shrub-interspace	Tree Coppice	Shrub-interspace	Tree Coppice	Shrub-interspace	Tree Coppice
Surface roughness (mm)	34 c	40 bc	40 bc	33 cd	23 d	48 ab	39 bc	51 a
Slope (%)	10.9 d	10.3 d	10.9 d	12.5 cd	15.4 bc	14.7 c	20.6 a	18.7 ab
Total canopy cover (%)	29 cd	23.1 d	36.8 bc	25.6 d	13.6 e	12.7 e	41 b	65.3 a
Total Herbaceous canopy cover (%)	11.8 c	6c	30.8b	24.4 b	7.6 c	9.6 c	27.5 b	53.2 a
Shrub canopy cover (%)	16.5 a	14.8 ab	6 bc	1.1 c	6.1 bc	1.8 c	13.5 ab	9.3 abc
Grass canopy cover (%)	10.2 de	5.8 e	29.2b	24.3 bc	3.7 e	7.3 e	18.7 cd	46.5 a
Annuals herbaceous canopy cover (%)	0.1 d	0.3 d	5.5 bcd	11.9 b	2 cd	2.1 cd	10.4 bc	37.6 a
Perennial herbaceous canopy cover (%)	11.8 bcd	5.7 d	25.3 a	12.6 bc	5.5 d	7.5 cd	17.1 b	15.6 b
Litter cover (%)	39.4 d	77.1 b	53.5c	92.4 a	14.3 e	70.5 b	33.9 d	75.2 b
Rock cover (%)	1.2 bc	0.3 c	12.8 bc	4.5 bc	42.2 a	14.9 b	44.4 a	8.9 bc
Ground cover (%)	44.2 e	78.3 bc	68.8 cd	95.6 a	61.8 d	86.3 ab	84.5 ab	89.2 ab
Bare soil (%)	55.8 a	21.7 cd	31.2 bc	4.4 e	38.2 b	13.7 de	15.5 de	10.7 de
Bare ground = Bare soil + Rock (%)	57.0 b	22.1 d	44.1 c	8.9 e	80.4 a	28.6 d	59.9 b	19.7 d
Initial volumetric soil moisture content (%)	12.8 a	11.3 a	2.3 b	2.2 b	2.6 b	2.7 b	7.4 ab	5.6 ab

than control shrub-interspace zones at the sites is illustrated in Fig. 5A and C. Likewise, the infiltration trends through the wet runs were similar for burned shrub-interspace zones and both burned and unburned tree zones (Fig. 5A and C). It is important to note that infiltration rate was not directly measured but estimated as the difference between rainfall rate and runoff rate. This infiltration estimation thus

Fig. 3. Images of two plots illustrating changes in shrub-interspace herbaceous cover from the pre-burn condition in the control section (A) to 9-years post-burn condition (B) at Onaqui.

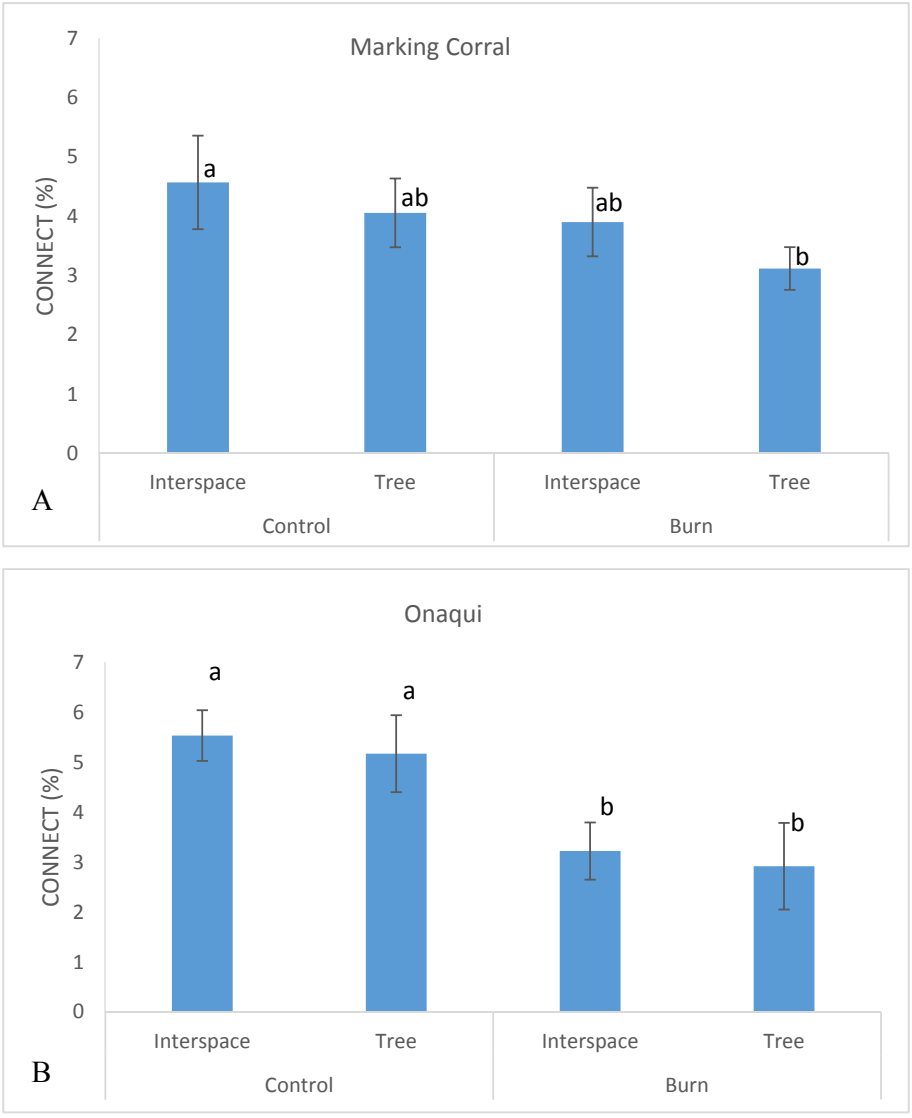

Fig. 4. Flow path Connectance Index CONNECT computed as the proportion of functional joining among flow path patches extracted from digital elevation models for Marking Corral (A) and Onaqui (B). Flow patches are connected if they are separated by a distance of less than 0.2 m. Means sharing lower case letters across treatments and microsites are not statistically different ($p \geq 0.05$).

Table 3
Average runoff, sediment yield and volumetric soil water content obtained after the low and high intensity rainfall events on the 12 m^2 plots at Marking Corral for dry-run (70 mm h^{-1}, 45 min) and wet-run (111 mm h^{-1}, 45 min) events. For each rainfall intensity, means sharing lower case letters across treatments and microsites are not statistically different ($p \geq 0.05$).

Marking Corral	Dry run (70 mm h^{-1}, 45-min)				Wet run (111 mm h^{-1}, 45-min)			
	Control		Burned		Control		Burned	
	Shrub-interspace	Tree Coppice	Shrub-interspace	Tree Coppice	Shrub-interspace	Tree Coppice	Shrub-interspace	Tree Coppice
Total number of plots	4	6	4	6	4	6	4	6
Number of plots with runoff	4	6	3	3	4	6	4	6
Runoff duration (min)	40.1 a	36.2 a	30.9 a	31.1 a	42.9 a	42.0 a	41.1 a	38.5 a
Post-event volumetric water content (%)	30.7 a	28.7 a	27.6 a	28.4 a	> 50 a	> 50 a	31.1 a	29.9 a
Cumulative runoff (mm)	8.0 a	4.0 a	0.4 a	0.3 a	37.1 a	11.8 b	5.7 b	6.0 b
Steady-state discharge (mm h^{-1})	11.4 a	6.4 a	1.0 a	0.3 a	54.3 a	20.0 b	9.3 b	9.0 b
Cumulative sediment loss (g)	412.9 a	56.5 a	5.1 a	4.4 a	3901.6 a	245.6 b	51.1 b	53.0 b
Sediment concentration at steady-state (g L^{-1})	2.3 a	1.8 a	1.8 a	2.5 a	6.2 a	1.4 ab	0.5 b	0.6 b
Average sediment flux (g s^{-1} m^{-2})	14 a	2 a	0 a	0 a	126 a	8 b	2 b	2 b
Sediment organic matter content (%)	14.3 a	15.4 a	14.9 a	27.3 a	8.9 a	14.4 a	11.2 a	15.0 a

Table 4
Average runoff, sediment yield and volumetric soil water content obtained after the low and high intensity rainfall events on the 12 m^2 plots at Onaqui for dry-run (70 mm h^{-1}, 45 min) and wet-run (111 mm h^{-1}, 45 min) events. For each rainfall intensity, means sharing lower case letters across treatments and microsites are not statistically different ($p \geq 0.05$).

Onaqui	Dry run (70 mm h^{-1}, 45-min)				Wet run (111 mm h^{-1}, 45-min)			
	Control		Burned		Control		Burned	
	Shrub-interspace	Tree Coppice	Shrub-interspace	Tree Coppice	Shrub-interspace	Tree Coppice	Shrub-interspace	Tree Coppice
Total number of plots	4	4	4	4	4	4	4	4
Number of plots with runoff	4	4	3	4	4	4	4	4
Runoff duration (min)	40.0 a	39.0 a	26.5 a	39.9 a	43.5 a	43.0 a	42.4 a	42.4 a
Post-event volumetric water content (%)	45.9 a	43.8 a	> 50 a	43.0 a	> 50 a	> 50 a	> 50 a	> 50 a
Cumulative runoff (mm)	14.5 a	5.5 a	3.8 a	13.1 a	38.5 a	9.3 b	15.2 b	15.2 b
Steady-state discharge (mm h^{-1})	25.5 a	8.4 a	7.1 a	15.9 a	53.2 a	12.0 b	27.7 ab	19.7 b
Cumulative sediment loss (g)	651.1 a	211.1 a	229.8 a	441.1 a	2303.7 a	459.5 b	697.4 b	447.9 b
Sediment concentration at steady-state (g L^{-1})	3.2 b	3.6 ab	5.8 a	2.9 b	4.4 a	3.4 a	3.6 a	2.4 a
Average sediment flux (g s^{-1} m^{-2})	22 a	7 a	13 a	15 a	74 a	15 b	23 b	15 b
Sediment organic matter content (%)	18.7 b	21.4 ab	18.2 b	28.1 a	16.8 b	19.2 b	21.3 ab	28.4 a

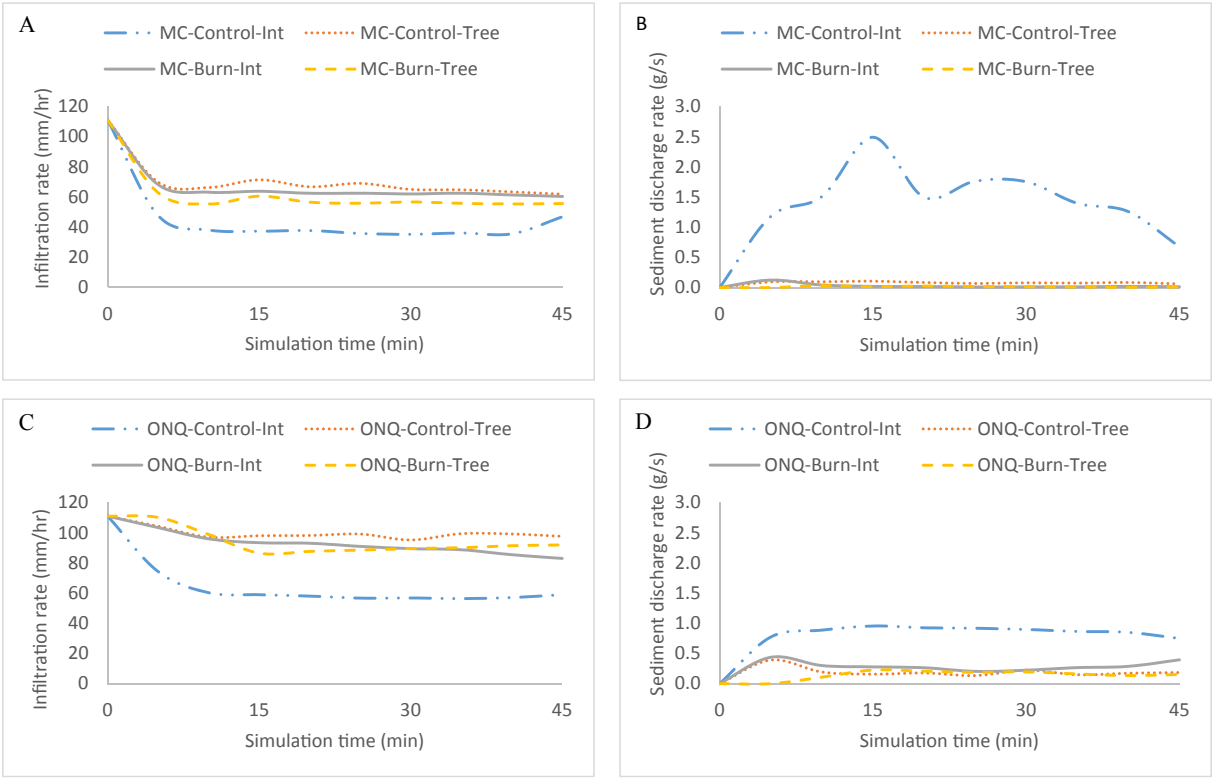

Fig. 5. Average infiltration rate (A and C) and sediment discharge rate (B and D) observed every 5 min during the wet runs (111 mm·h^{-1}, 45 min) at Marking Corral (MC) (A and B) and Onaqui (ONQ) (C and D) in shrub-interspace and tree microsites of burn and control areas.

encompasses the fraction of rainfall stored in soil surface depressions (depressional storage) and plant canopy.

Table 3 shows that for the dry-run erosion response at Marking Corral, there was no difference in sediment concentration, cumulative sediment and average sediment flux between burned and unburned areas of both microsites. At Onaqui, dry-run sediment concentration was greater on burned shrub-interspaces (averaging 5.8 g L^{-1}) than their unburned counterparts (3.2 g L^{-1} average) but dry-run cumulative sediment loss and average sediment flux were similar across microsites and treatments (Table 4). For wet runs, burning had a decreasing effect on cumulative sediment loss and sediment flux from shrub-interspaces at both sites (Tables 3 and 4). At Marking Corral, wet-run cumulative sediment loss was reduced from 3901 g to 51 g in the shrub-interspace areas whereas at Onaqui, cumulative sediment loss was on average 2303 g and 697 g on control and burned plots respectively. Fig. 5B and D show that for the burned plots, the steady-state sediment discharge rate in shrub-interspaces were of comparable levels to the well-protected tree coppices plots.

Figs. 6 and 7 show dot matrices of the effect of biophysical variables on hydrologic and erosion responses at Marking Corral and Onaqui for both events. Detailed results on the multiple regressions and general dominance values used to make these dot matrices have been added as supplemental materials. At Marking Corral (Fig. 6), litter cover and the burn treatment predominantly controlled hydrology and soil erosion

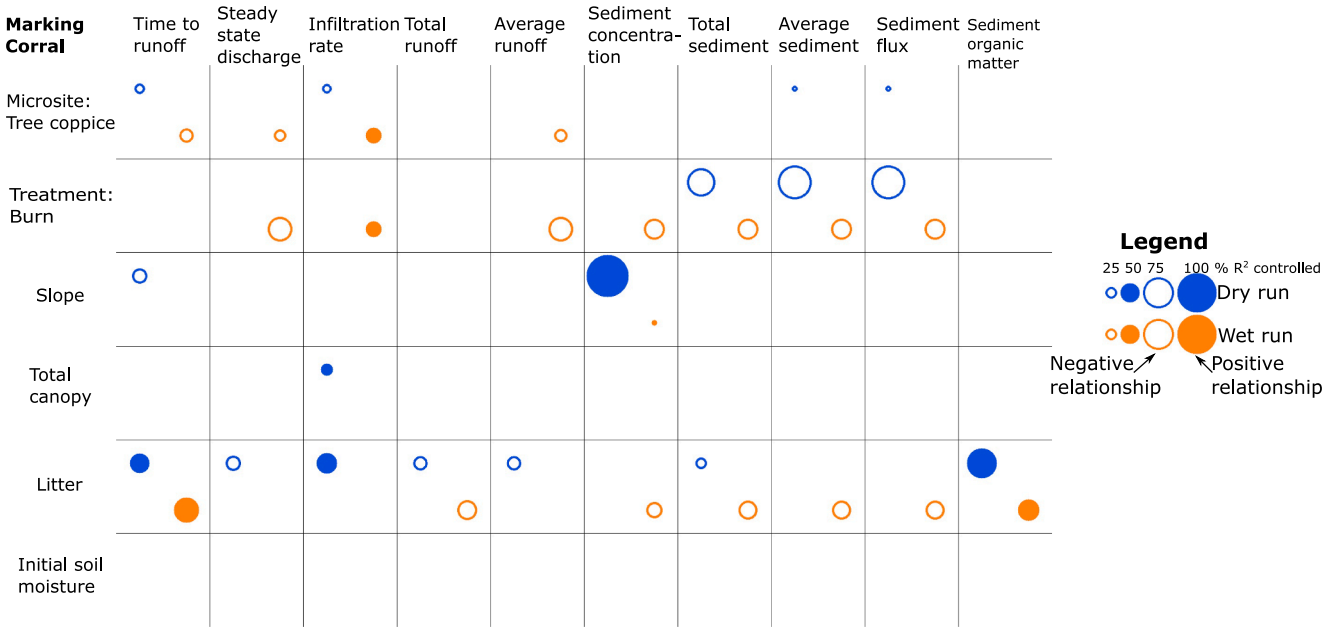

Fig. 6. Dot matrix of explanatory variables (rows) with significant effects on hydrology and erosion metrics (columns) for the dry-run (70 mm h^{-1}, 45 min) events (blue color top left of each cell) and the wet-run (111 mm h^{-1}, 45 min) events (orange color lower right of each cell) at Marking Corral. Circle sizes are proportional to the dominance of the explanatory variables on the coefficient of determination (R^2) of the explained variable in a column (i.e. sum of proportions add to one along each column for each event). Filled circles symbolize positive effects while hollow circles indicate negative effects. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

metrics. Under both dry and wet runs at this site, litter delayed runoff initiation, reduced total runoff and total soil loss. The burn treatment was associated with reduced steady state and average runoff discharges under the wet runs at this site. Total and average sediment as well as sediment flux were all lower on the burned plots. Slope had a strong control on sediment concentration under the dry run at Marking Corral.

At Onaqui (Fig. 7), litter cover was the most dominant factor, controlling hydrology and erosion metrics under the wet runs. Steady state runoff discharge rate, total and average runoff as well as all the erosion metrics were all inversely related to litter at this site. Microsite controlled average sediment loss and sediment flux (lower on tree coppices compared to shrub-interspaces) under the wet runs. The time needed for runoff initiation was primarily controlled at this site by initial soil moisture content of plots before the dry runs. Vegetation canopy cover at Onaqui and litter cover at Marking Corral were the primary key indicators of organic matter in the sediments.

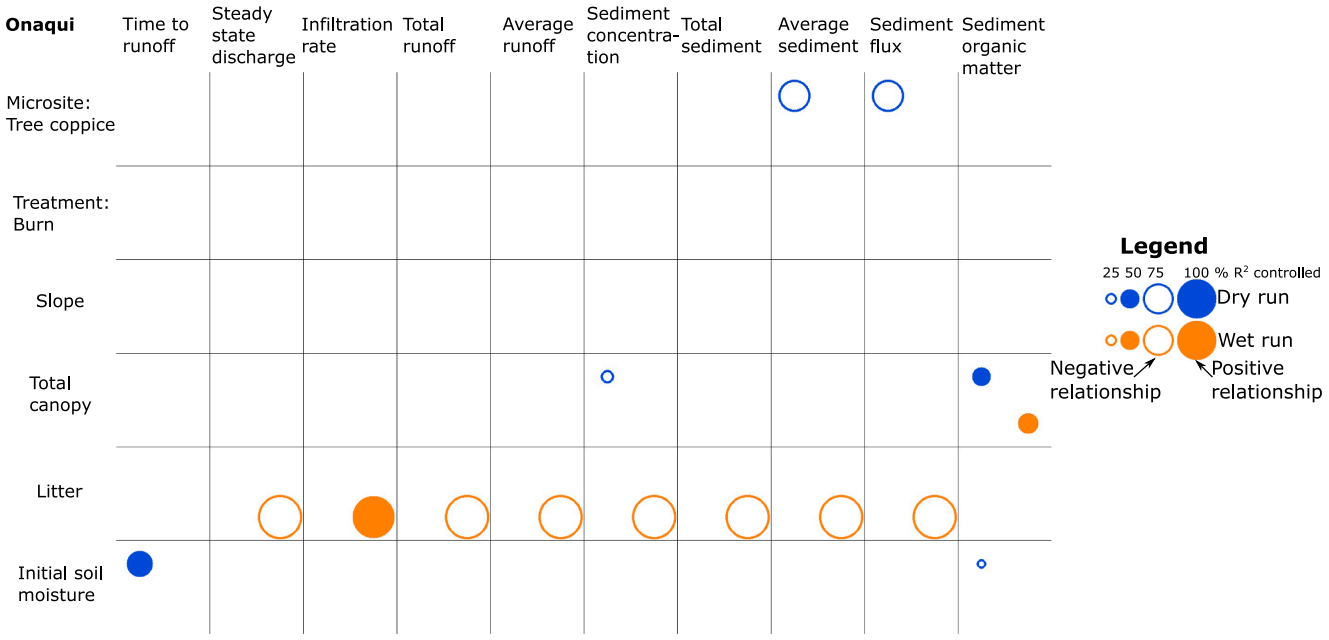

Fig. 7. Dot matrix of explanatory variables (rows) with significant effects on hydrology and erosion metrics (columns) for the 70 mm h^{-1} event (blue color top left of each cell) and the 111 mm h^{-1} event (orange color lower right of each cell) at Onaqui. Circle sizes are proportional to the dominance of the explanatory variables on the coefficient of determination (R^2) of the explained variable in a column (i.e. sum of proportions add to one along each column for each event). Filled circles symbolize positive effects while hollow circles indicate negative effects. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

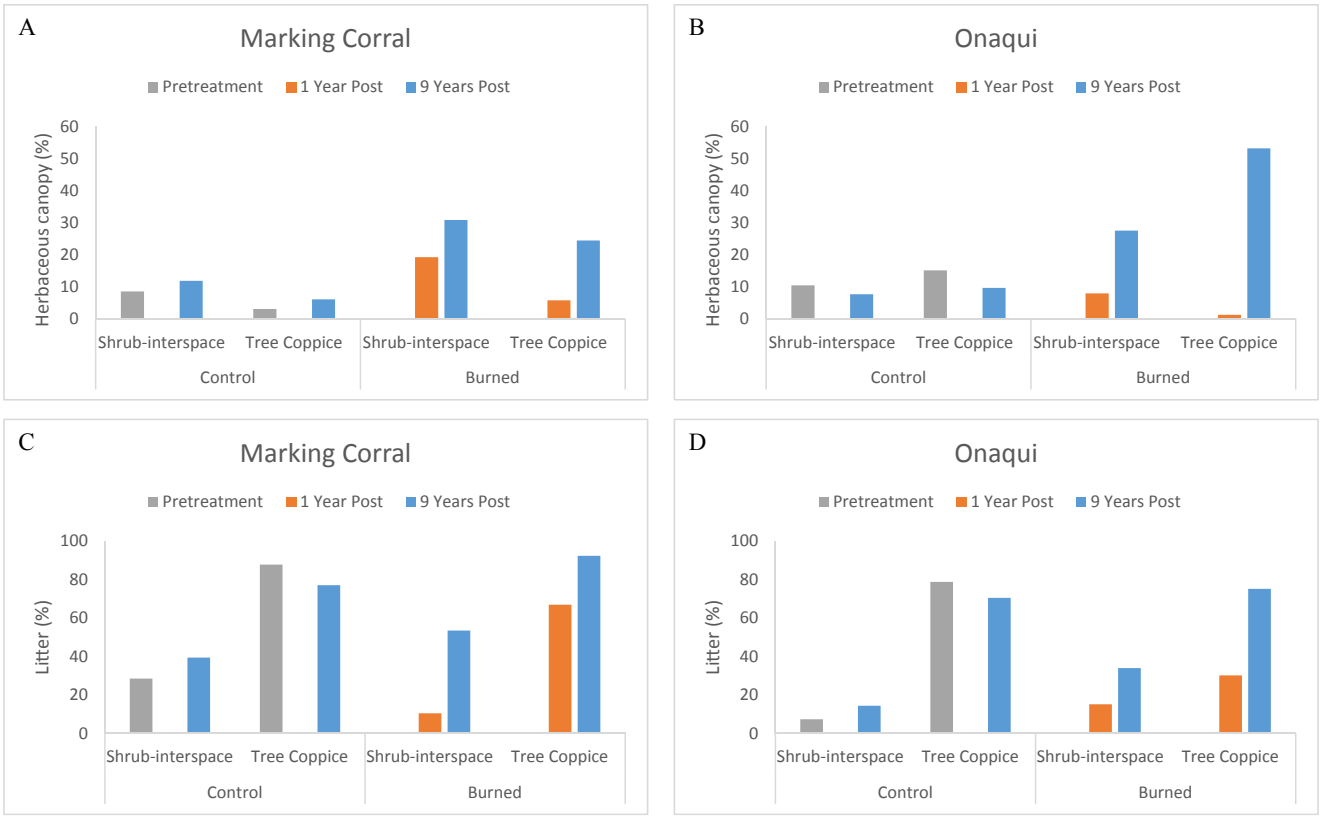

Fig. 8. Herbaceous and litter cover for pretreatment, 1-year-post and 9-years-post burn conditions at Marking Corral (A, C) and Onaqui (B, D). Pretreatment and 1-year post burn data from Williams et al. (2016a).

In summary, runoff generation under the wet runs was reduced on shrub-interspaces of burned areas at both sites. The total runoff collected after 45 min for the wet runs was reduced 6.5-fold at Marking Corral and 2.5-fold at Onaqui in the shrub-interspace after burning. Cumulative soil loss for the same rainfall intensity and duration was reduced 76-fold and 3-fold respectively at Marking Corral and Onaqui. Key factors controlling runoff and erosion processes at Marking Corral were microsite, burn treatment, slope and litter while at Onaqui key factors were microsite, litter and initial soil moisture. Soil organic matter transported with sediments was a function of vegetative material (canopy cover and litter) present on the soil surface.

3.3. Long-term prescribed burn effect

Figs. 8 and 9 show changes in vegetation and ground cover (Fig. 8) and hydrology and erosion (Fig. 9) from pre-fire conditions, 1-year post-fire and 9-years post-fire by combining our data with that of the Williams et al. (2016a) study. Vegetation and ground cover measurement in the control area showed some variations between pretreatment conditions and 9-year post-treatment conditions. These variations were however modest in magnitude compared to observed changes in the burned areas. On burned plots, herbaceous canopy cover and litter were improved on both microsites 9 years post-fire compared to the 1-year post-fire assessment (Fig. 8). At Marking Corral, herbaceous canopy cover gains were 11.6% and 18.7% respectively on shrub-interspace and tree coppices 9 years post-fire compared to conditions prevailing 1 year after burn (Fig. 8A). At this site, litter cover gains were 43.1% and 25.5% on shrub-interspace and tree coppices respectively (Fig. 8C). At Onaqui, herbaceous canopy cover gains were 19.6% and 52% on shrub-interspace and tree coppices (Fig. 8B). At the same site, litter gains were 18.8% and 45.1% respectively on shrub-interspace and tree coppice microsites (Fig. 8D).

At both sites, hydrology and erosion response in the control area has mostly remained unchanged between the pretreatment year and the 9-year post-treatment assessment (Fig. 9A and B). Average runoff ratios for controls were 0.45 and 0.48 in shrub-interspaces at Marking Corral and Onaqui respectively and 0.09 and 0.11 in tree coppices for the respective sites. Fire treatment benefits on shrub-interspace hydrologic response were not observed in year 1, but were clearly apparent at year 9, resulting in runoff to rainfall ratios lower than pretreatment conditions in these microsites (Fig. 9A and B). When both sites are combined, average runoff ratio in the shrub-interspaces 9 years post-burn was respectively $^{1}/_{6}$ and ½ that of year 1 post-treatment and $^{1}/_{7}$ and $^{1}/_{3}$ that of pre-burn level. Tree coppices at Marking Corral were generally unaffected by burning with runoff ratios on burned tree coppices of similar magnitude as control plots at year 9 and pretreatment. At Onaqui, an average pretreatment runoff ratio on tree coppices of 0.12 experienced a 370% increase at year 1 but returned to pretreatment levels (runoff ratio = 0.18) at year 9.

Overall, sediment yield per unit runoff did not show any consistent pattern over time in the control areas (Fig. 9C and D). Sediment yield per unit runoff was exacerbated 1 year after treatment but, for burned conditions, was reduced 9 years post-fire by a factor of 13 in the shrub-interspace zone at Marking Corral and a factor of 4 for that microsite at Onaqui. On the tree coppice microsite, reductions in sediment yield per unit runoff between the 1-year and 9-year time frame were 10- and 18-fold at Marking Corral and Onaqui respectively.

4. Discussion

This study was part of a broader project to understand the hydrological effects of prescribed fire on shrubland hillslopes that have been invaded by woody species. Since fire treatments were applied in 2006, the two sites Marking Corral and Onaqui have been the object of multiple studies focused at the small plot ($< 1\ m^2$) as well as large plot ($12–13\ m^2$) scales (Pierson et al., 2010). One year after burn treatment,

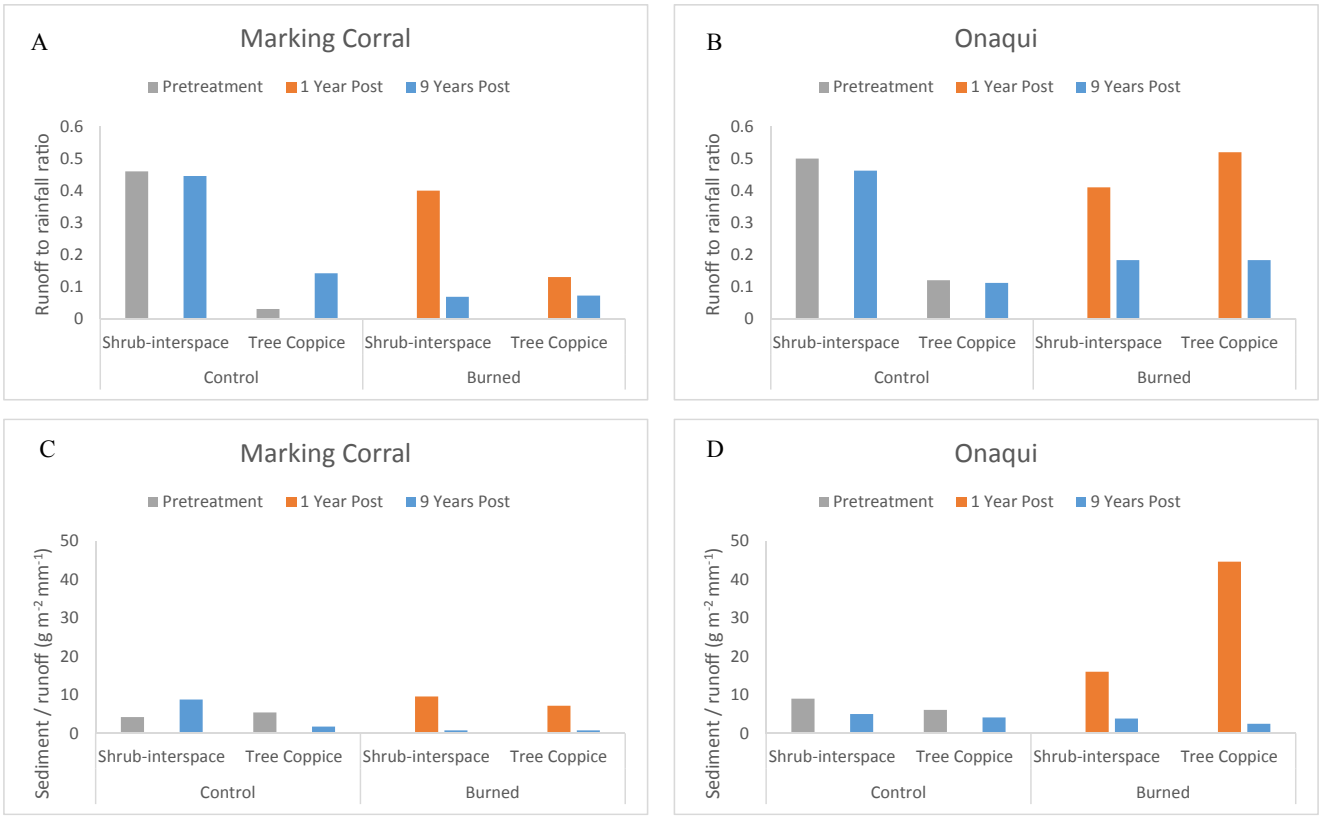

Fig. 9. Runoff ratio and soil loss per runoff depth for pretreatment, 1-year-post and 9-years-post burn conditions after the wet run (111 mm/h) at Marking Corral (A, C) and Onaqui (B, D). Pretreatment and 1-year post burn data are wet runs (102 mm/h) from (Williams et al., 2016a) normalized for comparison in this study.

shrub-interspaces showed a modest herbaceous increase at Marking Corral but not at Onaqui, resulting in no erosion benefit at either site (Pierson et al., 2015). In our study, herbaceous cover was dramatically increased in the shrub-interspaces 9 years after burn treatment. Litter cover at Marking Corral declined the first year post-treatment due to burning, but was abundantly replaced at this site 9 years after treatment. Pierson et al. (2015) noted that an increase in herbaceous vegetation 2 years post-burn at the same site was associated with a reduction in concentrated flow erosion. A similar reduction in concentrated flow erosion likely contributed to the decrease in erosion observed in this study between control and burned shrub-interspaces. Williams et al. (2020) conducted rainfall simulations on small plots (0.5 m^2) and runoff release experiments on concentrated flow paths at the same sites and the same year. Results from the Williams et al. (2020) study suggest that the ability of interspace areas to produce runoff and sediment from rainsplash and sheet flow has been significantly reduced 9 years after burning at both sites while detachment and transport by concentrated flow in the shrub-interspace zones were reduced at Marking Corral and unaffected at Onaqui. Rainfall simulation experiments on large plots (> 12 m^2) as was the case in our study are designed to integrate both runoff and sediment production by rainsplash and sheetflow and concentrated flow processes (Pierson et al., 2010). As illustrated in our cumulative runoff and sediment results, both Marking Corral and Onaqui showed benefit of fire 9 years post-burn in the shrub-interspace areas, suggesting that a lesser degree of flow concentration was achieved on our 12 m^2 plots at Onaqui compared to the Williams et al. (2020) concentrated flow experiments. These results suggest that while concentrated flow experiments may not have revealed benefits of burning at Onaqui, conditions leading to well-connected concentrated flow networks are not likely to occur during rainfall events similar in magnitudes to the ones simulated in our study.

This study showed that changes in vegetation amount stimulated by prescribed fire on woodland-encroached shrublands translated into long-term hydrologic and erosion benefits. Nine years post-burn, shrub-interspaces at both sites dramatically increased in spatial heterogeneity with small bare areas disconnected by well-established herbaceous vegetation patches. The use of the landscape metric CONNECT to characterize the degree of connectivity of flow paths on each plot provided a unique insight into the interactions between biotic processes and hydrologic connectivity. Other connectivity metrics have been used to study fire effect on hydrology and erosion processes. These include the functional hydraulic connectivity used by Moody et al. (2008) to describe spatially varying patterns of infiltration associated with non-uniform burn severity across the hillslope and the index of connectivity (Borselli et al., 2008) applied by Ortiz-Rodriguez et al. (2019) to evaluate changes in connectivity as a result of fire at the landscape scale. Most of these studies focused however on the enhancing effect of fire on connectivity in the short term. Our study revealed that flow path connectivity decreased (particularly at Onaqui) as the result of post-fire vegetation recovery in the long term (9 years). Post-fire recovery resulted in an increase in vegetation life form diversity as herbaceous plants were selectively promoted compared to woody plants. With this increase in vegetation diversity, new herbaceous vegetation patches established in previously bare areas and disrupted flow path continuity. These results are consistent with other studies (e.g., Hueso-González et al., 2018; Ludwig et al., 2004, 2005) showing a relationship between biodiversity and the spatial structure of vegetation and bare ground. Greater runoff and erosion benefits from post-fire herbaceous species recruitment was also observed by Cerdá and Doerr (2005) who noted herbs and shrubs produced negligible amounts of runoff and erosion 2 years post-fire while trees and dwarf shrubs maintained high runoff and erosion rates even after 5 years of recovery. In our study, plots in both shrub-interspace and tree coppice areas encompassed herbs, shrubs and forbs. The hydrologic response obtained on each plot was therefore an aggregate of runoff and erosion responses on individual vegetated and bare patches. Much of the decrease in runoff and erosion

post-fire can be attributed to the development of patches dominated by herbaceous cover while response on shrub patches was likely similar to that observed by Cerdá and Doerr (2005). Over time, it is expected that newly established herbaceous patches will further develop into "resource islands" where favorable conditions for plant growth feedback into surface processes by enhancing runoff, sediment and nutrient capture (e.g., Cammeraat and Imeson, 1999; Puigdefabregas, 2005).

Plots at Marking Corral had higher vegetation cover in the control areas compared to Onaqui and this finding is in agreement with observations from previous studies at these two sites (Pierson et al., 2010, 2014). Flow path connectivity at Marking Corral was not significantly affected while water and sediment loss at this site were dramatically reduced 9 years after burn. At Onaqui however, flow path connectivity was significantly reduced 9 years after burning, with associated reductions in runoff and soil loss. It is important to note that bare ground 9 yr post-burn was still nearly 60% in the shrub-interspaces at Onaqui while it had dropped to 44% in the same microsite at Marking Corral. Many studies have suggested a ground cover of 50% as a threshold below which runoff (e.g., Cerdá and Doerr, 2005) and erosion (e.g., Al-Hamdan et al., 2017; Pierson et al., 2013; Weltz et al., 1998) reduction benefits are appreciable. At Onaqui, it is likely that the flow path connectivity has been significantly disrupted by new herbaceous growth in the interspaces, but these newly established patches need more than 9 years to produce enough ground coverage to pass the 50% threshold for hydrologic benefits to match levels observed at the less degraded Marking Corral site.

5. Conclusions

Nine years after prescribed fire, changes to vegetation indicate that both Marking Corral and Onaqui sites are on a trajectory to recovery. The sites were transformed from a woodland-dominated vegetation type with extensive bare ground to sites dominated by herbaceous vegetation and limited shrub cover. Perennial herbaceous vegetation substantially increased and became the dominate cover type in the shrub-interspace zones at both sites over 9 growing seasons. Annual grasses increased at both sites and were either co-dominate with perennial vegetation or the dominate vegetation cover in tree zones 9 yr post-fire. Increases in vegetation on burned plots compared to their unburned counterpart altered soil surface microtopography, disrupting flow paths and creating favorable hydrologic and erosion-protecting conditions. Reduction in flow path connectivity in the shrub-interspace zones was more dramatic and significant at the initially more degraded site, Onaqui. Total runoff and soil loss were systematically reduced for shrub-interspace areas at both sites on burned compared to unburned conditions under the high ($111\ mm\ h^{-1}$) intensity of the wet runs. These locations comprise more than 70% of the total area at both sites and generated high levels of runoff and sediment yield prior to treatment. The burned treatment reduced shrub-interspace cumulative runoff under 45 min of wet run by 6.5-fold at Marking Corral and 2.5-fold at Onaqui. Cumulative sediment delivered for the same experiments was reduced 76-fold at Marking Corral and 3-fold at Onaqui. Additional factors such as slope, litter cover and canopy cover further modulated sediment generation. These results provide insight into the long-term ecohydrologic trajectory of rangeland ecosystems where prescribed fire has been used to combat woody species encroachment. This study shows that often documented elevated erosion and runoff risks in the short term post-fire seem to be generally balanced by long-term benefits in vegetation diversity and spatial structure which confer favorable hydrologic response to the burned landscape. Nevertheless, slow ecohydrologic recovery expected on initially degraded sites pre-fire may warrant additional site treatments to reduce vulnerability to elevated runoff and erosion during recovery.

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Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.catena.2019.104301.

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