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RESEARCH ARTICLE

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Long-term evidence for fire as an ecohydrologic threshold-reversal mechanism on woodland-encroached sagebrush shrublands

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Abstract

Encroachment of sagebrush (*Artemisia* spp.) shrublands by pinyon (*Pinus* spp.) and juniper (*Juniperus* spp.) conifers (woodland encroachment) induces a shift from biotic-controlled resource retention to abiotic-driven loss of soil resources. This shift is driven by a coarsening of the vegetation structure with increasing dominance of site resources by trees. Competition between the encroaching trees and understory vegetation for limited soil and water resources facilitates extensive bare intercanopy area between trees and concomitant increases in run-off and erosion that, over time, propagate persistence of the shrubland-to-woodland conversion. We evaluated whether tree removal by burning can decrease late-succession woodland ecohydrologic resilience by increasing vegetation and ground cover over a 9-year period after fire and whether the soil erosion feedback on late-succession woodlands is reversible by burning. To address these questions, we employed a suite of vegetation and soil measurements and rainfall simulation and concentrated overland flow experiments across multiple plot scales on unburned and burned areas at two sagebrush sites in the later stages of woodland succession. Prior to burning, tree cover was approximately 28% at the sites, and more than 70% of the area at the sites was intercanopy with depauperate understory vegetation and extensive bare ground (52–60% bare soil and rock). Burning initially increased bare ground across fine (<1 m^2) to patch (tens of metres) scales, resulting in enhanced sediment availability at the fine scale, sustained high run-off and erosion within degraded intercanopies, amplified run-off and erosion from tree canopy areas, and amplified sediment delivery across fine to patch scales. However, fire-induced increases in grass cover over nine growing seasons improved infiltration, limited run-off and sediment delivery from the fine scale, and reduced intercanopy run-off and erosion at the patch scale. These changes reflect a switch in vegetation structure, triggered by burning and subsequent vegetation re-establishment, and a shift to biotic control on run-off and erosion across spatial scales. The responses and persistence over the 9-year period postfire at the two sites demonstrate that fire can decrease woodland ecohydrologic resilience by altering plant community physiognomy and thereby can reverse the soil erosion feedback on sagebrush shrublands in the later stages of woodland encroachment.

KEYWORDS

ecohydrologic resilience, hydrology, infiltration, juniper, pattern-process, pinyon, prescribed fire, rangeland, run-off, sagebrush steppe, soil erosion feedback, structure–function, thresholds, woodland encroachment

1 | INTRODUCTION

The conversion of sagebrush (*Artemisia* spp.) shrublands to pinyon (*Pinus* spp.) and juniper (*Juniperus* spp.) woodlands has significantly altered a vast expanse of the western United States (Davies et al., 2011; Miller et al., 2011; Miller, Bates, Svejcar, Pierson, & Eddleman, 2005; Miller & Rose, 1995; Miller, Tausch, McArthur, Johnson, & Sanderson, 2008; Romme et al., 2009). Pinyon and juniper conifers extended their range 10-fold across the western United States since the mid- to late-1800s (Miller & Tausch, 2001), and much of this domain was historically sagebrush (*Artemisia* spp.) shrublands within the Great Basin Region (Davies et al., 2011; Miller et al., 2011). Range expansion and infilling of pinyon and juniper woodlands in the Great Basin are attributed to a combination of factors including climate variability, increased atmospheric CO_2, and reduced fire activity associated with intensive land use and fuel reductions following human settlement (Miller et al., 2005; Miller et al., 2008; Miller & Rose, 1995, 1999; Miller & Wigand, 1994; Romme et al., 2009). Woodland encroachment on sagebrush shrublands poses a host of negative ramifications to ecosystem services, including degradation of understory vegetation and wildlife habitat, limited forage for wild and domestic animals, and high rates of run-off and soil loss (Bates, Davies, & Sharp, 2011; Bates, Miller, & Svejcar, 2005; Coates et al., 2017; Davies et al., 2011; Miller et al., 2005, 2011; Miller, Svejcar, & Rose, 2000; Petersen & Stringham, 2008; Petersen, Stringham, & Roundy, 2009; Pierson et al., 2010; Williams, Pierson, Al-Hamdan, et al., 2014; Williams, Pierson, Robichaud, et al., 2016; Williams, Pierson, Spaeth, et al., 2016). Prescribed fire is commonly used to halt woodland encroachment on sagebrush rangelands and thereby prevent or reverse shrubland-to-woodland conversions (Bates & Davies, 2016; Bates, Sharp, & Davies, 2014; McIver et al., 2014; Miller et al., 2014; Pierson, Williams, Kormos, & Al-Hamdan, 2014; Pierson et al., 2015; Roundy, Miller, et al., 2014; Williams, Pierson, Al-Hamdan, et al., 2014; Williams, Pierson, Robichaud, et al., 2016; Williams, Pierson, Nouwakpo, et al., 2019). Wildfire activity is also increasing on woodland-encroached sagebrush shrublands throughout the Great Basin due to woody fuel loading and changing climate (Board, Chambers, Miller, & Weisberg, 2018; Keane et al., 2008; Miller & Tausch, 2001; Romme et al., 2009; Snyder et al., 2019).

Sagebrush sites in the later stages of woodland encroachment teeter along a potentially irreversible ecohydrological tipping point (Petersen et al., 2009; Williams, Pierson, Al-Hamdan, et al., 2014; Williams, Pierson, Spaeth, et al., 2016). The progression of pinyon and juniper dominance of sagebrush shrublands has been characterized into three phases (Phases I–III; see Miller et al., 2005; Miller et al., 2008). In Phase I, pinyon and juniper cover is usually ≤10%, and sagebrush and understory grasses and forbs (herbaceous cover) exert the dominant control on site ecological processes. Under these conditions, sagebrush islands and well-vegetated interspaces between shrubs and trees intercept rainfall and isolated overland flow and promote infiltration and soil retention. Conservation of these resources enhances vegetation productivity, soil stability, and accumulation of litter, organic matter, and soil nutrients. Tree cover increases with time in the absence of disturbance as trees outcompete shrubs and herbaceous plants for available water and soil resources. Trees significantly affect site-level ecological processes once tree cover approaches 10–30% (Phase II). In Phase II, the shrub layer and understory herbaceous cover substantially decline due to competition with trees; and bare ground, run-off, and erosion increase within interspaces. Continued competition for water and soil resources culminates in a tree-dominated woodland landscape (Phase III). Under these conditions, tree cover commonly exceeds 30%, more than 75% of the shrub layer is lost, native herbaceous cover is limited, and intercanopy bare ground is extensive (often >50–60%). Phase III is perpetuated by tree dominance of site resources and high rates of run-off and long-term soil loss from well-connected bare intercanopy areas (the soil erosion feedback; Miller et al., 2005; Miller et al., 2008; Petersen et al., 2009; Williams, Pierson, Al-Hamdan, et al., 2014; Williams, Pierson, Spaeth, et al., 2016). The ecological transition from Phases II to III and the associated soil erosion feedback on these landscapes are considered difficult to reverse without substantial management intervention, inclusive of seeding and tree removal (Bates et al., 2014; Roundy, Miller, et al., 2014; Williams, Pierson, Al-Hamdan, et al., 2014; Williams, Pierson, Spaeth, et al., 2016). Cover of sagebrush, native grasses, and forbs and respective seed sources are often insufficient in Phase III for re-establishing these vegetation covers simply through tree removal (Bates et al., 2014; Davies, Bates, & Boyd, 2019). Thus, sites in Phase III are particularly susceptible to long-term degradation following wildfire that removes limited existing cover and exacerbates and perpetuates high run-off and erosion rates (Davies et al., 2019; Miller, Chambers, Pyke, Pierson, & Williams, 2013; Pierson et al., 2011; Pierson & Williams, 2016; Williams, Pierson, Robichaud, & Boll, 2014).

Fire is a natural disturbance component of well-vegetated intact sagebrush communities (Wright & Bailey, 1982). Fire return periods (see Miller et al., 2011) are about 30–50 years for high-elevation productive sagebrush sites with ample annual precipitation. Fires are generally less frequent, perhaps every 100 or more years, for warmer and drier sagebrush sites at lower elevations. Fire frequencies have increased in recent years for sagebrush sites throughout much of the western United States due to invasion by the fire-prone annual cheatgrass (*Bromus tectorum* L.; Balch, Bradley, D'Antonio, & Gómez-Dans, 2013; Brooks et al., 2004; Knapp, 1996; Miller et al., 2011). Regardless, these ecosystems evolved with periodic burning and re-establishment of sagebrush and native bunchgrasses and forbs. Plot- to hillslope-scale run-off and erosion rates are low for sagebrush rangelands due to limited distribution of bare patches and generally decrease with increasing spatial scale on well-vegetated sites (Pierson et al., 2008; Pierson, Moffet, Williams, Hardegree, & Clark, 2009; Pierson, Van Vactor, Blackburn, & Wood, 1994). Removal of vegetation and ground cover by fire on these rangelands temporarily increases run-off and erosion rates (Pierson & Williams, 2016). Amplified plot- to hillslope-scale run-off and erosion rates postfire result from increased exposure of surface soils to rainfall, rapid run-off generation, formation of concentrated flow over contiguous bare areas, elevated sediment detachment and transport capacity, and connectivity of run-off and erosion processes across spatial scales (Pierson et al., 2009, 2008, 2011; Pierson, Carlson, & Spaeth, 2002; Williams, Pierson, Kormos, et al., 2016; Williams, Pierson, Robichaud, & Boll, 2014). This structural and functional process connectivity (pattern-process) declines over time as bare patches fill in with vegetation and ground cover, infiltration improves, and sediment production becomes more limited (Pierson et al., 2011; Pierson & Williams, 2016; Williams, Pierson, Al-Hamdan, et al., 2014; Williams, Pierson, Robichaud, & Boll, 2014). Sagebrush does not resprout following burning, and therefore, re-establishment of sagebrush cover postfire depends on seed availability, often taking 20 to >50 years (Harniss & Murray, 1973; Miller et al., 2013; Ziegenhagen & Miller, 2009). Herbaceous canopy cover commonly returns to prefire levels on sagebrush sites within several years postfire (Miller et al., 2013). Recovery of both the amount and distribution of ground cover can take longer and is highly variable (Miller et al., 2013). Dissipation of fire-induced increases in run-off rates often occurs within the first few years postfire (Pierson et al., 2009, 2008, 2011; Williams, Pierson, Robichaud, & Boll, 2014). Fire-induced increases in erodibility usually persist longer than those for run-off, but erosion rates for commonly occurring storms may return to prefire levels within 3 years due to limited run-off connectivity across spatial scales (Pierson & Williams, 2016; Williams, Pierson, Kormos,

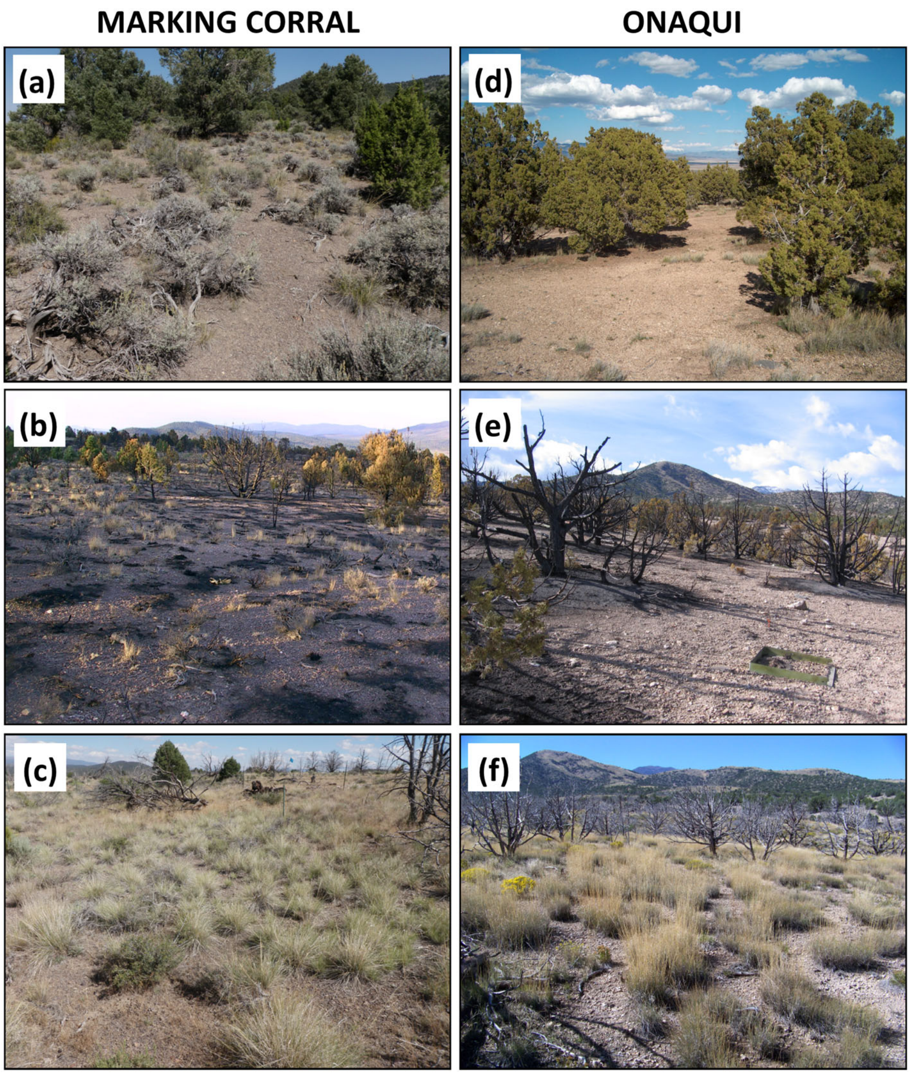

FIGURE 1 Photographs of hillslope-scale vegetation at the Marking Corral (a–c) and Onaqui (d–f) study sites showing tree islands and the intercanopy in unburned areas (a and d) and within burned treatments immediately after fire (b and e) and 8- to 9-year postfire (c and f)

et al., 2016). Overall risk of fire-induced elevated run-off and erosion during rare high-intensity or high-magnitude storms remains until vegetation and ground cover amounts and distribution return to near prefire levels (Pierson et al., 2011; Pierson & Williams, 2016; Williams, Pierson, Robichaud, & Boll, 2014). This oscillating cycle of fire, elevated run-off and erosion risks, and postfire recovery is therefore naturally occurring every 30–100+ years on intact sagebrush rangelands in the Great Basin.

Williams, Pierson, Al-Hamdan, et al., (2014) proposed that wildfire may serve as an ecohydrologic threshold-reversal mechanism on degraded sagebrush rangelands in the later stages of woodland encroachment and thus reverse the soil erosion feedback. The authors suggested that fire on these landscapes acts to reduce the ecohydrologic resilience propagating woodland persistence and resets the ecohydrologic successional trajectory. In this conceptual model, fire-induced mortality of trees frees up soil water for herbaceous vegetation (e.g., Bates, Miller, & Svejcar, 2000; Bates, Svejcar, & Miller, 2002; Roundy, Young, et al., 2014). Infilling of bare intercanopy patches with herbaceous vegetation over time postfire improves infiltration, limits run-off and sediment detachment, and disrupts hydrologic and erosion process connectivity along the hillslope, consistent with postfire recovery of sagebrush communities (Pierson et al., 2011; Pierson & Williams, 2016; Williams, Pierson, Robichaud, & Boll, 2014). Williams, Pierson, Al-Hamdan, et al., (2014) demonstrated that fire-induced mortality of western juniper (*J. occidentails* Hook.) increased herbaceous productivity in the bare intercanopy on a sagebrush site in the later stages (late Phases II–III) of woodland encroachment and thereby decreased bare ground connectivity, improved infiltration, and reduced concentrated flow erosion. That study spanned a period of two years postfire, and the authors acknowledged that the short-term nature of the study limited inferences on fire as a long-term ecohydrologic threshold-reversal mechanism on late-succession woodlands. Tree removal is commonly employed to retain or re-establish sagebrush vegetation and the associated ecohydrologic function (Pierson et al., 2013; Pierson et al., 2015; Pierson et al., 2014; Pierson, Bates, Svejcar, & Hardegree, 2007; Roundy et al., 2017; Williams, Pierson, Kormos, et al., 2019; Williams, Pierson, Al-Hamdan, et al., 2014; Williams, Pierson, Nouwakpo, et al., 2019; Williams, Pierson, Spaeth, et al., 2016), but results can vary substantially depending on the phase (vegetation conditions) of woodland encroachment at the time of treatment and initial impacts on vegetation during treatment application (Bates et al., 2014; Miller et al., 2005; Miller et al., 2013). Prescribed burning can result in mortality of the limited sagebrush and understory native bunchgrasses and prolong sparse understory conditions (Bates et al., 2011, 2014; Bates & Davies, 2016; Bates, Miller, & Davies, 2006; Miller et al., 2013), temporarily increasing run-off and erosion rates (Pierson et al., 2013; Pierson et al., 2015; Williams, Pierson, Al-Hamdan, et al., 2014; Williams, Pierson, Robichaud, et al., 2016; Williams, Pierson, Spaeth, et al., 2016). Mechanical tree-removal practices are also often used to reduce pinyon and juniper tree cover on woodland-encroached sagebrush sites (Bates et al., 2000; Bybee et al., 2016; Cline et al., 2010; Pierson et al., 2007; Pierson et al., 2013; Pierson et al., 2015; Miller et al., 2014; Roundy, Miller, et al., 2014; Roundy, Young, et al., 2014; Roundy et al., 2017; Williams,

TABLE 1 Topography, climate, soil, tree cover, and understory vegetation at the Marking Corral and Onaqui sites before prescribed burning

	Marking Corral, Nevada, USA	Onaqui, Utah, USA
Woodland community	Single-leaf pinyon[a]/Utah juniper[b]	Utah juniper[b]
Elevation (m)—Aspect	2,250—W to SW facing	1,720—N facing
Mean ann. precip. (mm)	307[c]	312[c]
Mean ann. air temp. (°C)	6.5[d]	8.9[e]
Slope (%)	10–15	10–15
Parent rock	Andesite and rhyolite[f]	Sandstone and limestone[g]
Soil association	Segura-Upatad-Cropper[f]	Borvant[g]
Depth to bedrock (m)	0.4–0.5[f]	1.0-1.5[g]
Soil surface texture	Sandy loam, 66% sand, 30% silt, 4% clay	Sandy loam, 56% sand, 37% silt, 7% clay
Tree canopy cover (%)[h,i]	21[a], 6[b]	28[b]
Trees per hectare[h,i]	465[a], 114[b]	532[b]
Mean tree height (m)[h,i]	2.3[a], 1.9[b]	2.3[b]
Common understory plants	*Artemisia tridentata* Nutt. ssp. *wyomingensis* Beetle & Young; *Artemisia nova* A. Nelson; *Artemisia tridentata* Nutt. ssp. *vaseyana* (Rydb.) Beetle; *Purshia* spp.; *Poa secunda* J. Presl; *Pseudoroegneria spicata* (Pursh) A. Löve; and various forbs	

Note. Data from Pierson et al. (2010) except where indicated by footnote.

[a]*Pinus monophylla* Torr. & Frém. [b]*Juniperus osteosperma* [Torr.] Little. [c]Estimated from 4-km grid for years 1971–2015 from Prism Climate Group (2017). Pierson et al. (2010) estimates (351 mm Marking Corral and 345 mm Onaqui) were from Prism Climate Group (2009) for years 1971–2000. Pierson et al. (2015) estimates (382 mm Marking Corral and 468 mm Onaqui) were for years 1980–2011 based on Daymet (Thornton et al., 2012). [d]Estimated from 4-km grid for years 1971–2015 from Prism Climate Group (2017). Pierson et al. (2010) estimate (7.2°C) was for years 1928–1958 from Western Regional Climate Center (WRCC), Station 264199-2, Kimberly, Nevada (WRCC, 2009). [e]Estimated from 4-km grid for years 1971–2015 from Prism Climate Group (2017). Pierson et al. (2010) estimate (7.5°C) was for years 1972–2005 from WRCC, Station 424362-3, Johnson Pass, Utah (WRCC, 2009). [f]Natural Resources Conservation Service (NRCS) (2007). [g]NRCS (2006). [h]Data from Pierson et al. (2010) but restricted to the area subsequently burned. [i]Tree data for trees ≥1 m high.

Pierson, Kormos, et al., 2019), but these practices can leave numerous residual juvenile trees that re-establish tree dominance over time (Bates et al., 2005; Bates, Svejcar, Miller, & Davies, 2017; Miller et al., 2005; Miller et al., 2013; Tausch & Tueller, 1997). Vegetation and ground cover recruitment into sparse bare intercanopy areas on sagebrush sites in the later stages of woodland succession is considered difficult with any treatment (Bates et al., 2014; Bates et al., 2017), as is the reversal of degraded ecohydrologic function and high erosion rates (Williams, Pierson, Kormos, et al., 2019).

In this study, we revisit the suggestion by Williams, Pierson, Al-Hamdan, et al. (2014) that fire can act as an ecohydrologic threshold-reversal mechanism and thereby reduce the ecohydrologic resilience propagating long-term pinyon and juniper woodland persistence on sagebrush shrublands. As in the shorter term Williams, Pierson, Al-Hamdan, et al. (2014) study, we focus on the key switch necessary for this change, a fire-induced structural shift in the plant community and an ensuing functional shift in the dominant run-off and erosion processes with postfire vegetation and ground cover recovery. The switch induces a reversal of abiotic-controlled soil erosion for the late-succession woodland state to biotic-controlled soil retention, indicative of progression to the sagebrush-dominated structural-functional state. We apply a suite of vegetation and soil measurements and rainfall simulation and concentrated overland flow experiments across multiple plot scales on unburned and burned areas at two sagebrush rangeland sites in the later stages of woodland succession to address two primary questions: (a) Can fire decrease late-succession woodland ecohydrologic resilience by increasing vegetation and ground cover over a 9-year period after fire? and (b) Is the soil erosion feedback on late-succession woodlands reversible by burning? This study follows on previous investigations at the study sites inclusive of woodland encroachment impacts on run-off and erosion (Pierson et al., 2010), initial impacts of burning on infiltration, run-off, and erosion (Pierson et al., 2014; Pierson et al., 2015; Williams, Pierson, Robichaud, et al., 2016), and the long-term impacts of burning on infiltration and erosion at the fine spatial scale (<1 m; Williams, Pierson, Nouwakpo, et al., 2019) and on run-off and erosion at the patch scale (tens of metres; Nouwakpo et al., 2019). The current study pulls together findings across these studies spanning fine to patch spatial scales and the

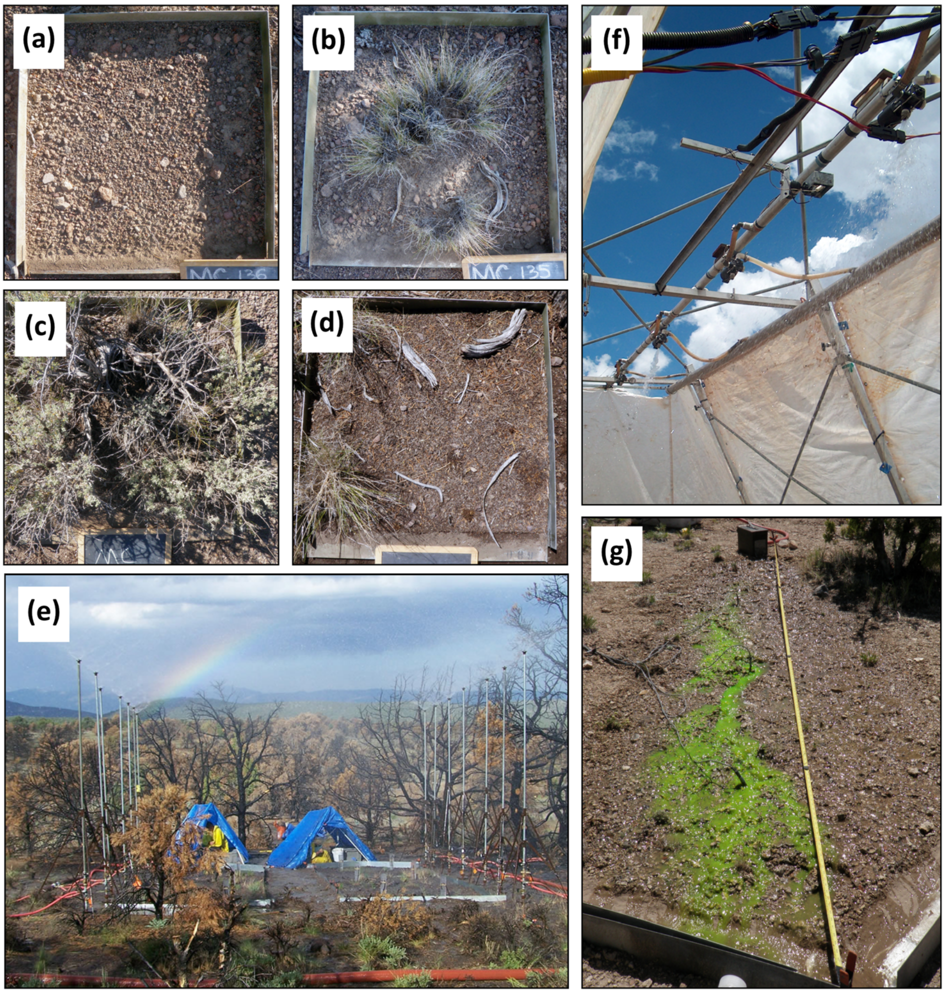

FIGURE 2 Photographs of bare interspace (a), vegetated interspace (b), shrub coppice (c), and tree coppice (d) small plots (0.5 m^2) in unburned areas at Marking Corral; paired large plots (13 m^2) with the Colorado State University type rainfall simulator within the burn (Year-1) at Marking Corral (e); the Walnut Gulch Rainfall Simulator with a four-nozzle oscillating arm and wind shields (f); and a borderless concentrated overland flow plot (9 m^2) within the unburned intercanopy at Onaqui (g)

immediate to long-term impacts of burning to present a collective examination of whether fire can serve as an ecohydrologic threshold-reversal mechanism on sagebrush sites in the later stages of woodland encroachment.

2 | METHODS

2.1 | Study area

Experiments were conducted in two woodlands (Figure 1) in the Great Basin as part of a larger study, the Sagebrush Treatment Evaluation Project (SageSTEP, McIver & Brunson, 2014; McIver et al., 2014). Our substudy was implemented in 2006 prior to tree removal at a single-leaf pinyon—Utah juniper woodland (*P. monophylla* Torr. & Frém. - *J. osteosperma* [Torr.] Little.) (Marking Corral site) and a Utah juniper woodland (Onaqui) selected from the SageSTEP study network. The Marking Corral site (Figure 1a–c; 39°27′17″N latitude, 115°06′51″W longitude) is located in the Egan Range in east central Nevada, approximately 27-km northwest of city of Ely. The Onaqui site (Figure 1d–f; 40°12′42″N latitude, 112°28′24″W longitude) is located in the Onaqui Mountains in Utah south of the Great Salt Lake, approximately 76-km southwest of Salt Lake City. The study sites are managed by the U.S. Department of the Interior, Bureau of Land Management for grazing use, but grazing has been excluded from both sites since autumn 2005 in cooperation with SageSTEP. Topography,

TABLE 2 Fine-scale canopy cover, ground cover, and surface roughness variables measured on unburned and burned small rainfall simulation plots (0.5 m^2) at the Marking Corral and Onaqui sites 1 (Year-1), 2 (Year-2), and 9 (Year-9) years following burning

	Year-1						Year-2		
	Unburned			Burned			Unburned		
	Interspace	Shrub coppice	Tree coppice	Interspace	Shrub coppice	Tree coppice	Interspace	Shrub coppice	Tree coppice
Marking Corral									
Total canopy cover (%)[a]	33.3 bc	92.8 d	6.5 a	30.0 b	53.1 c	3.5 a	17.1 a	76.2 c	9.8 a
Shrub canopy cover (%)	0.3 a	58.9 b	2.6 a	0.1 a	1.2 a	0.0 a	0.0 a	14.3 b	0.0 a
Grass canopy cover (%)	27.9 c	24.8 c	3.2 ab	12.1 b	8.6 ab	0.5 a	6.9 ab	3.3 ab	0.0 a
Forb canopy cover (%)	2.7 a	3.2 a	0.2 a	16.0 b	34.3 c	2.9 a	0.5 a	4.8 ab	0.2 a
Total ground cover (%)[b]	27.3 a	83.6 b	99.5 c	13.0 a	35.3 a	75.4 b	10.8 a	81.9 d	97.4 d
Litter cover (%)	24.6 a	79.3 b	88.1 b	11.2 a	33.6 a	74.7 b	6.8 a	77.2 bc	96.7 c
Rock cover (%)	28.6 c	4.5 ab	0.4 a	38.4 c	10.0 b	2.7 ab	36.4 cd	6.3 abc	0.8 a
Bare ground (%)[c]	72.7 c	16.4 b	0.5 a	87.0 c	64.7 c	24.6 b	89.2 d	18.1 a	2.6 a
Ash (%)	-	-	-	0.0	6.0	12.3	-	-	-
Litter depth (mm)	<1 a	2 a	40 c	<1 a	2 a	23 b	<1 a	<1 a	38 c
Surface roughness (mm)	9 a	14 b	12 ab	8 a	8 a	12 ab	8 a	14 b	10 ab
No. of plots	7	5	8	8	4	8	4	2	4
Onaqui									
Total canopy cover (%)[a]	19.4 bc	68.6 d	21.7 c	6.6 ab	27.8 c	1.7 a	11.1 a	58.4 c	16.0 a
Shrub canopy cover (%)	0.0 a	50.5 c	0.0 a	0.0 a	10.1 b	0.0 a	0.0 a	49.5 b	0.0 a
Grass canopy cover (%)	5.7 a	9.8 ab	17.4 b	2.7 a	6.9 a	1.0 a	2.9 a	1.9 a	7.4 a
Forb canopy cover (%)	7.3 b	2.9 ab	0.5 a	0.4 a	0.8 a	0.4 a	7.0 b	2.5 ab	1.0 a
Total ground cover (%)[b]	9.5 a	61.4 b	91.6 c	5.5 a	25.3 a	80.6 bc	13.5 a	69.3 c	91.7 d
Litter cover (%)	6.0 ab	57.8 c	80.6 c	4.0 a	21.9 b	80.4 c	5.6 a	66.3 c	81.1 c
Rock cover (%)	38.1 bc	20.5 b	1.4 a	55.5 c	30.1 b	2.9 a	60.8 c	18.8 ab	3.2 a
Bare ground (%)[c]	90.5 dc	38.6 b	8.4 a	94.5 c	74.7 c	19.4 ab	86.5 d	30.7 b	8.3 a
Ash (%)	-	-	-	0.8	0.8	7.4	-	-	-
Litter depth (mm)	<1 a	2 a	18 b	<1 a	1 a	19 b	<1 a	3 a	13 b
Surface roughness (mm)	11 a	13 a	12 a	9 a	11 a	12 a	11 b	12 b	9 ab
No. of plots	3	3	4	10	5	5	3	3	4

Note. Means within a row by study year (Year-1, Year-2, or Year-9) followed by a different lower case letter are significantly different ($P < 0.05$).

[a]Excludes tree canopy removed for rainfall simulation. [b]Includes cryptogam, litter, live and dead basal plant, and woody dead covers. [c]Includes ash, bare soil, and rock covers.

climate, soils, and vegetation attributes for the sites are shown in Table 1. Annual precipitation for both sites over our full 10-year study period was near or exceeded the respective long-term average (Table 1) most years, with only 2–3 years of more than 15% below normal (Williams, Pierson, Kormos, et al., 2019; Williams, Pierson, Nouwakpo, et al., 2019). Tree cover at both sites averaged near 28% prior to burning (Table 1). The vegetation and ground cover structure before prescribed fire at both sites was typical of degraded sagebrush steppe in the later stages of woodland encroachment with isolated tree cover and litter-covered tree islands and a degraded intercanopy understory with extensive bare ground (Figure 1a,d; Pierson et al., 2010). Prior to burning, litter-covered mounds around trees extended, on average, 2.5 and 2.2 m from tree bases at Marking Corral and Onaqui, respectively, and litter mass underneath trees averaged 17.4 kg m^{-2} at Marking Corral and 14.3 kg m^{-2} at Onaqui (Pierson et al., 2010). Soil bulk density over 0- to 5-cm depth prefire at Marking Corral averaged 1.03, 1.02, and 1.26 g cm^{-3}, respectively, in areas under trees, under shrubs, and in interspaces between trees and shrubs (Pierson et al., 2010). The same measures were 0.90, 1.05, and 1.08 g cm^{-3}, respectively, at Onaqui (Pierson et al., 2010). A prescribed fire was applied by the Bureau of Land Management at both sites in late summer 2006. Burn severity was not assessed after the fires, but persistence of live and scorched needle cover on trees, burned shrub skeletons, and blackened and residual litter and woody debris at both sites were indicative of low to moderate burn severities (Figure 1b,e; Parsons, Robichaud, Lewis, Napper, & Clark, 2010). Tree canopy scorch averaged 50–75% at Marking Corral and 75–99% at Onaqui (Pierson et al., 2010).

2.2 | Experimental design

Experimental plots were established to characterize fire effects on vegetation, ground surface conditions, and run-off and erosion processes over fine to hillslope scales and to quantify vegetation and soil

TABLE 2 Continued

	Year-2			Year-9					
	Burned			Unburned			Burned		
	Interspace	Shrub coppice	Tree coppice	Interspace	Shrub coppice	Tree coppice	Interspace	Shrub coppice	Tree coppice
Marking Corral									
Total canopy cover (%)[a]	23.9 a	43.6 b	11.1 a	37.5 ab	94.8 d	13.2 a	65.4 cd	56.0 bc	53.3 bc
Shrub canopy cover (%)	0.0 a	0.0 a	0.0 a	0.3 a	66.9 b	1.4 a	3.5 a	1.2 a	0.0 a
Grass canopy cover (%)	9.4 b	2.1 a	0.1 a	35.6 b	23.3 ab	5.4 a	61.6 c	50.0 bc	53.0 bc
Forb canopy cover (%)	0.1 a	18.1 b	7.0 ab	0.5 a	2.9 b	0.1 a	0.0 a	0.0 a	0.0 a
Total ground cover (%)[b]	12.6 a	34.3 b	67.9 c	13.0 a	69.8 b	82.2 b	21.8 a	39.4 a	77.0 b
Litter cover (%)	7.5 a	29.1 a	67.1 b	4.8 a	55.8 cd	79.5 d	14.9 ab	32.7 bc	70.2 d
Rock cover (%)	51.2 d	21.8 bc	10.4 ab	7.8 ab	6.1 ab	2.2 a	13.7 b	5.6 ab	5.6 ab
Bare ground (%)[c]	87.4 d	65.7 c	32.1 b	87.1 b	30.2 a	17.8 a	78.2 b	60.6 b	23.0 a
Ash (%)	0.0	0.0	1.4	-	-	-	-	-	-
Litter depth (mm)	<1 a	<1 a	17 b	1 a	6 a	34 c	3 a	12 ab	27 bc
Surface roughness (mm)	8 a	8 a	10 ab	11 a	15 a	12 a	12 a	11 a	12 a
No. of plots	8	4	8	6	4	8	6	4	8
Onaqui									
Total canopy cover (%)[a]	14.3 a	32.2 b	3.4 a	23.8 a	76.5 c	38.7 ab	63.1 c	89.1 c	58.0 bc
Shrub canopy cover (%)	0.1 a	1.0 a	0.0 a	0.0 a	47.0 b	0.5 a	1.4 a	5.4 a	1.1 a
Grass canopy cover (%)	6.1 a	9.3 a	0.6 a	9.7 a	12.4 a	21.3 ab	37.3 bc	48.6 c	34.9 bc
Forb canopy cover (%)	7.3 b	7.6 b	1.9 a	7.3 a	6.3 a	4.7 a	11.8 a	7.1 a	11.1 a
Total ground cover (%)[b]	13.0 a	33.1 b	72.0 c	13.8 a	46.5 c	72.4 d	24.9 ab	34.7 bc	42.3 c
Litter cover (%)	9.7 a	28.8 b	72.0 c	7.8 a	33.9 b	69.6 c	19.7 b	20.1 b	33.8 b
Rock cover (%)	53.2 c	28.3 b	5.3 a	40.2 b	19.8 ab	6.7 a	36.9 b	17.8 ab	5.3 a
Bare ground (%)[c]	87.0 d	66.9 c	28.0 b	86.2 b	53.5 b	27.6 a	75.1 b	65.3 b	57.7 b
Ash (%)	0.1	0.0	6.9	-	-	-	-	-	-
Litter depth (mm)	<1 a	<1 a	12 b	1 a	6 a	29 b	4 a	4 a	9 a
Surface roughness (mm)	8 a	9 ab	12 b	10 a	14 b	10 a	12 ab	14 b	14 b
No. of plots	10	5	5	6	6	8	10	5	5

effects on cross-scale run-off and erosion. Site characterization plots (30 m × 33 m) were used to quantify fire effects on hillslope-scale vegetation and ground cover. Three site characterization plots were randomly located within the burn treatment areas at both sites prior to burning (Pierson et al., 2010) and were sampled for tree cover, understory vegetation, and ground cover prefire in 2006 (Year-0, unburned) and for understory vegetation and ground cover postfire in 2007 (Year-1, burned) and 2015 (Year-9, burned). Small plots (0.7 m × 0.7 m) were used to quantify fire impacts on fine-scale vegetation, ground cover, soils, infiltration, run-off, and erosion by rainsplash and sheetflow processes. Small plots were located and installed in control and burn treatment areas at both sites in 2006 prior to burning as described in Pierson et al. (2010) and were left in place for subsequent sampling in 2007 (Year-1), 2008 (Year-2), and 2015 (Year-9). Small plots were installed in interspaces between shrub and tree canopies and in areas underneath shrub (shrub coppice) and tree (tree coppices) canopies to partition microsite cover/soil differences and respective run-off and erosion contributions to the patch scale (Figure 2a–d; Pierson et al., 2010; Williams, Pierson, Al-Hamdan, et al., 2014). The number of small plots sampled for each microsite (interspace, shrub coppice, and tree coppice) and treatment (unburned and burned) combination in each year at each site is shown in Tables 2–4. Large plots (2 m wide × 6–6.5 m long) were employed to quantify fire impacts on vegetation, soils, run-off, and erosion from combined rainsplash, sheetflow, and concentrated flow processes occurring at the patch scale (Pierson et al., 2007; Pierson et al., 2010; Williams, Pierson, Al-Hamdan, et al., 2014). Large plots were installed and sampled within untreated areas (burn area prior to treatment) in 2006 (Year-0, unburned), within the burned areas in 2007 (Year-1, burned), and within control (Year-9, unburned) and burned areas in 2015 (Year-9, burned). Large plots were installed within shrub-interspace zones (intercanopy area outside of tree canopy influence) between trees and within tree zones (areas underneath and immediately adjacent to tree canopies), with the long axis perpendicular to the hillslope contour (Figure 2e; see Nouwakpo et al., 2019; Pierson et al., 2010). The number of large plots sampled for each microsite (shrub-interspace zone and tree zone) and treatment (unburned and burned) combination in each year at each site is shown in Tables 5–8. Concentrated flow experiments (Figure 2g) were used to evaluate fire effects on vegetation, ground cover, and run-off and erosion from concentrated overland flow or rills (Al-Hamdan, Pierson, Nearing, Stone, et al., 2012; Al-Hamdan, Pierson, Nearing, Williams, et al., 2012; Al-Hamdan et al., 2013; Pierson et al., 2007; Pierson et al., 2010; Williams, Pierson, Al-Hamdan, et al., 2014). Concentrated flow experiments (2 m wide × 4.5 m long) were conducted on each large rainfall-simulation plot immediately following rainfall simulations within untreated areas (burn area prior to treatment) in 2006 (Year-0, unburned) and in burned areas in 2007 (Year-1, burned). Concentrated flow experiments were conducted as independent experiments (without rainfall simulations) in unburned control (unburned) and burned

TABLE 3 Fine-scale run-off, sediment, and soil water repellency response variables measured for dry- and wet-run rainfall simulations (0.5 m^2) in unburned and burned areas at the Marking Corral study site 1 (Year-1), 2 (Year-2), and 9 (Year-9) years following burning

	Year-1						Year-2		
	Unburned			Burned			Unburned		
Marking Corral	Interspace	Shrub coppice	Tree coppice	Interspace	Shrub coppice	Tree coppice	Interspace	Shrub coppice	Tree coppice
Dry run simulation (64 mm hr^{-1}, 45 min)									
Cumulative run-off (mm)	6 b	0 a	0 a	3 ab	0 a	7 b	6 ab	0 a	0 a
Cumulative sediment (g m^{-2})[a]	7 a	-	-	6 a	-	17 a	10 a	-	-
Sediment/run-off (g $m^{-2} \cdot mm^{-1}$)[a]	0.89 a	-	-	0.95 a	-	1.32 a	0.88 a	-	-
Mean soil water repellency (WDPT; s)[b]	-	-	48 a	-	-	65 a	-	-	34 a
Maximum strength of soil water repellency (s)[c]	-	-	80	-	-	91	-	-	98
Depth of max soil water repellency (cm)[d]	-	-	1	-	-	3	-	-	0
Percent of plots with run-off	57	0	0	38	0	63	50	0	0
Wet run simulation (102 mm hr^{-1}, 45 min)									
Cumulative run-off (mm)	31 bc	3 a	0 a	35 c	8 a	21 b	41 c	0 a	0 a
Cumulative sediment (g m^{-2})[a]	23 a	6 a	-	41 a	48 a	46 a	42 a	-	-
Sediment/run-off (g $m^{-2} \cdot mm^{-1}$)[a]	0.66 a	1.01 a	-	1.10 a	2.07 a	1.96 a	0.90 a	-	-
Percent of plots with run-off	100	40	0	100	50	88	100	0	0
No. of plots	7	5	8	8	4	8	4	2	4

Note. Means within a row by study year (Year-1, Year-2, or Year-9) followed by a different lower case letter are significantly different ($P < 0.05$).

[a]Means based solely on plots that generated run-off. [b]Mean soil water repellency for 0- to 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, Dekker, & Schoute, 1993). [c]Average persistence of soil water repellency measured at the most water repellent soil layer over 0- to 5-cm soil depth assessed as WDPT. [d]Soil depth with the highest average soil water repellency assessed as WDPT.

(burned) areas in 2008 (Year-2) and 2015 (Year-9). Concentrated flow experiments in Year-0 and Year-1 had plot borders given the experiments were conducted on rainfall simulation plots (Figure 2e) after rainfall experiments concluded (Pierson et al., 2015, 2010). The concentrated flow plots conducted in Year-2 and Year-9, independent of rainfall simulation plots, did not include plot borders (Figure 2g; Pierson et al., 2015; Williams, Pierson, Nouwakpo, et al., 2019). Plot installation methods for concentrated flow plots conducted in Year-0, Year-1, and Year-2 are provided in Pierson et al. (2015, 2010) and installations methods for those in Year-9 are provided in Williams, Pierson, Nouwakpo, et al., (2019). The number of concentrated flow plots (Table 9) sampled for each microsite (shrub-interspace zone and tree zone) and treatment (unburned and burned) combination in each year at each site is consistent with the number of large rainfall simulation or concentrated flow plots as shown in Tables 5 and 6. Trees were trimmed or removed from all rainfall simulation and concentrated flow plots immediately preceding experiments to minimize canopy interference with rainfall and plot sampling. Shrubs were retained on plots but were trimmed along plot boundaries to prevent stemflow from exiting or entering the plot during rainfall simulations.

2.3 | Site characterization: Hillslope scale

Hillslope-scale tree cover and understory canopy and ground cover were estimated from measurements on the 30 × 33-m site characterization plots. All live pinyon and juniper trees >0.5-m height on each site characterization plot were tallied in Year-0 prior to burning. Each tree was measured for height and live crown radius. The live crown radius for each tree was calculated as one-half the average of measured minimum and maximum crown diameters for the respective tree. Individual tree crown area (tree cover) was assumed equivalent to the area of a circle and was calculated as such for each tree using the respective derived crown radius. Total tree cover for each plot was obtained as the sum of individual tree cover values on the respective plot. Understory canopy and ground cover were quantified on each site characterization plot using line-point intercept methods along five 30-m transects located 5–8 m apart and oriented perpendicular to the hillslope contour (Herrick, Van Zee, Havstad, Burkett, & Whitford, 2005). Canopy (foliar) and ground cover on each plot were recorded for 60 points, each spaced 50 cm apart, along each of the five transects for a total of 300 sample points per plot. Percent cover for each sampled cover type was derived for each plot as the frequency of respective cover type hits divided by the total number of points sampled.

2.4 | Small plot: Fine scale

Canopy (foliar) cover, ground cover, and ground surface roughness were measured on small plots using point frame methods (Mueller-Dombois & Ellenberg, 1974). Canopy and ground cover for each plot were sampled at 15 points, spaced 5 cm apart, along each of seven transects oriented parallel to the hillslope contour and spaced 10 cm apart, for a total of 105 sample points per plot. Percent cover for each sampled cover type on a plot was derived from the frequency of respective cover type hits divided by the total number of points sampled within the plot. A relative ground surface height for each sample point on each plot was measured by ruler as the distance between the ground surface and a point frame level line. Ground surface roughness for each plot was calculated as the arithmetic average of the standard deviations of the measured ground surface heights for each of the

TABLE 3 Continued

	Year-2			Year-9					
	Burned			Unburned			Burned		
Marking Corral	Interspace	Shrub coppice	Tree coppice	Interspace	Shrub coppice	Tree coppice	Interspace	Shrub coppice	Tree coppice
Dry run simulation (64 mm hr^{-1}, 45 min)									
Cumulative run-off (mm)	4 ab	0 a	9 b	18 b	-	8 a	7 a	8 a	15 b
Cumulative sediment (g m^{-2})[a]	9 a	-	20 a	18 bc	-	9 ab	8 a	23 c	18 bc
Sediment/run-off (g m^{-2} · mm^{-1})[a]	0.85 a	-	1.37 a	1.02 ab	-	0.83 a	0.76 a	2.30 b	1.06 ab
Mean soil water repellency (WDPT; s)[b]	-	-	72 a	-	-	80 a	-	-	62 a
Maximum strength of soil water repellency (s)[c]	-	-	176	-	-	145	-	-	92
Depth of max soil water repellency (cm)[d]	-	-	0	-	-	0	-	-	5
Percent of plots with run-off	38	0	75	100	25	88	83	75	88
Wet run simulation (102 mm hr^{-1}, 45 min)									
Cumulative run-off (mm)	35 c	5 a	22 b	44 d	3 a	12 ab	20 bc	23 bc	31 cd
Cumulative sediment (g m^{-2})[a]	35 a	27 a	75 a	59 c	3 a	10 a	26 ab	54 bc	27 ab
Sediment/run-off (g m^{-2} · mm^{-1})[a]	0.92 a	2.49 a	2.14 a	1.35 ab	0.41 a	0.67 a	1.05 ab	1.93 b	0.72 a
Percent of plots with run-off	100	50	88	100	75	88	100	100	100
No. of plots	8	4	8	6	4	8	6	4	8

seven transects sampled on the respective plot. Litter depth on each plot was quantified to the nearest 1 mm at four evenly spaced points (~15-cm spacing) along the outside edge of each of the two plot borders located perpendicular to the hillslope contour. An average litter depth was estimated for each plot as the mean of the eight measured litter depths.

Surface soils for each plot were sampled for soil moisture and soil water repellency. Soil moisture for each plot was derived from a sample obtained over 0- to 5-cm depth immediately adjacent to each small plot before rainfall simulations. All samples were analysed gravimetrically in the laboratory for soil water content. Water repellency of surface soils was quantified immediately adjacent (within ~50 cm) to each small plot prior to rainfall simulations using the water drop penetration time (WDPT) procedure (DeBano, 1981). After carefully removing litter, eight water drops (~3-cm spacing) were applied to the mineral soil surface, and the time required for infiltration of each drop was recorded up to a 300-s maximum time. Following this procedure, 1 cm of soil was excavated immediately underneath the previously sampled area, and the WDPT method was continued for an additional eight drops. This process was repeated until the 5-cm soil depth was reached and sampled. A mean WDPT at 0-, 1-, 2-, 3-, 4-, and 5-cm soil depths was recorded for each plot as the average of the eight WDPT (s) samples at the respective depth. A plot mean soil water repellency across all sample depths was derived as the arithmetic average of the means from each of the 1-cm depths sampled. Soils were classified as wettable when WDPT <5 s, slightly water repellent when WDPT ranged 5 to 60 s, and strongly water repellent when WDPT >60 s (Bisdom et al., 1993).

Rainfall simulations were conducted on each small plot at target intensities of 64 (dry run) and 102 mm hr^{-1} (wet run) for 45 min each using a Meyer and Harmon-type portable oscillating-arm rainfall simulator (Meyer & Harmon, 1979). The simulator was fitted with a single 80–100 Vee-jet nozzle positioned 3 m above the ground surface. Simulator configuration/design, rainfall attributes, and calibration procedures are described in detail in Pierson et al. (2008, 2009, 2010). Dry run simulations were conducted on dry antecedent-soil moisture conditions (<12% gravimetric). The wet run simulation on each plot was applied approximately 30 min following the dry run, on wet soil conditions. Mean rainfall intensity and cumulative rainfall applied by run type were similar (P > 0.05) across unburned and burned conditions at both sites. For both study sites, the dry run intensity applied for 5-, 10-, and 15-min durations is equivalent to local storm return intervals of 7, 15, and 25 years, and the wet run intensity over the same durations is equivalent to local storm return intervals of 25, 60, and 120 years (Bonnin et al., 2006).

Timed plot run-off samples were collected at 1- to 3-min time intervals throughout each 45-min rainfall simulation and were analysed in

TABLE 4 Fine-scale run-off, sediment, and soil water repellency response variables measured for dry- and wet-run rainfall simulations (0.5 m^2) in unburned and burned areas at the Onaqui study site 1 (Year-1), 2 (Year-2), and 9 (Year-9) years following burning

	Year-1						Year-2		
	Unburned			Burned			Unburned		
Onaqui	Interspace	Shrub coppice	Tree coppice	Interspace	Shrub coppice	Tree coppice	Interspace	Shrub coppice	Tree coppice
Dry run simulation (64 mm hr^{-1}, 45 min)									
Cumulative run-off (mm)	10 bc	0 a	7 bc	12 c	2 a	5 ab	12 cd	0 a	2 ab
Cumulative sediment (g m^{-2})[a]	69 a	-	33 a	64 a	18 a	57 a	70 b	-	8 a
Sediment/run-off (g m^{-2} · mm^{-1})[a]	4.80 a	-	4.02 a	5.15 a	6.79 a	6.84 a	3.72 ab	-	1.72 a
Mean soil water repellency (WDPT; s)[b]	-	-	88	-	-	125	-	-	47
Maximum strength of soil water repellency (s)[c]	-	-	160	-	-	185	-	-	151
Depth of max soil water repellency (cm)[d]	-	-	1	-	-	1	-	-	0
Percent of plots with run-off	67	0	75	100	60	60	67	0	50
Wet run simulation (102 mm hr^{-1}, 45 min)									
Cumulative run-off (mm)	41 bc	6 a	16 a	49 c	22 ab	13 a	45 d	2 a	9 ab
Cumulative sediment (g m^{-2})[a]	233 bc	33 a	98 ab	351 c	220 bc	294 c	274 b	-	19 a
Sediment/run-off (g m^{-2} · mm^{-1})[a]	5.53 bc	4.99 ab	4.65 ab	7.11 bc	7.90 c	10.40 c	6.11 b	-	1.52 a
Percent of plots with run-off	100	100	75	100	80	80	100	33	75
No. of plots	3	3	4	10	5	5	3	3	4

Note. Means within a row by study year (Year-1, Year-2, or Year-9) followed by a different lower case letter are significantly different (P < 0.05).

[a]Means based solely on plots that generated run-off. [b]Mean soil water repellency for 0- to 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]Average persistence of soil water repellency measured at the most water repellent soil layer over 0- to 5-cm soil depth assessed as WDPT. [d]Soil depth with the highest average soil water repellency assessed as WDPT.

the laboratory for run-off volume and sediment concentration. The volume of run-off and sediment concentration for each sample were obtained by weighing the sample before and after oven drying at 105°C. Multiple hydrologic and erosion response variables were derived for each rainfall simulation using the timed run-off samples. An average run-off rate (mm hr^{-1}) for each sample interval was calculated as the sample cumulative run-off divided by interval time. Cumulative run-off (mm) for each simulation was derived from the integration of run-off rates over the total time of run-off. An average infiltration rate (mm hr^{-1}) for each sample interval was computed as the difference between applied rainfall and measured run-off divided by duration of the sample interval. Sediment discharge (g s^{-1}) for each sample interval was derived as the sample cumulative sediment divided by the interval time. Cumulative sediment yield (g m^{-2}) for each simulation was computed as the integrated sum of sediment collected during run-off and was extrapolated to a unit area by dividing by the 0.5-m^2 plot area. A sediment-to-run-off ratio (g m^{-2}·mm^{-1}), a surrogate for erodibility, was derived for each simulation by dividing cumulative sediment yield by cumulative run-off.

2.5 | Large plot: Patch scale

Canopy and ground cover by cover type and distances between plant bases (basal gaps) and canopies (canopy gaps) were quantified on large plots and on concentrated flow plots using line-point intercept and gap-intercept methods (Herrick et al., 2005). Canopy and ground cover on Year-0 and Year-1 combined rainfall simulation/concentrated flow plots were recorded for 59 points with 10-cm spacing, along each of five transects 6 m in length, spaced 40 cm apart, and oriented perpendicular to the hillslope contour, for a total of 295 points per plot. In Year-2 and Year-9, canopy and ground cover at the patch scale were assessed on concentrated flow plots. Canopy and ground cover on these plots were recorded for 24 points with 20-cm spacing, along each of nine line-point transects 4.6 m in length, spaced 20 cm apart, and oriented perpendicular to the hillslope contour, for a total of 216 points per plot. Percent cover for each sampled cover type on each plot was derived from the frequency of respective cover type hits divided by the total number of points sampled for the plot. Plant basal and canopy gaps exceeding 20 cm were recorded along each line-point transect. Average basal and canopy gap sizes were determined for each plot as the average of all respective gaps measured in excess of 20 cm. A relative ground-surface height at each line-point sample location was calculated as the distance between the ground surface and a survey transit level-line above the respective sample point. Ground surface roughness for each plot was derived as the arithmetic average of the standard deviations of the ground surface heights across the plot line-point transects.

Rainfall on large plots in each year was applied at the same durations and for the same dry-run and wet-run sequences as described above for the small plot rainfall simulations. As with those plots, the dry run was conducted on each plot during dry antecedent-soil moisture conditions (<12% gravimetric), and the wet run was applied to each plot within 30 min after the dry run. Rainfall simulations were conducted on large plot pairs (two plots, each 2 m wide × 6.5 m long) in Year-0 and Year-1 using a Colorado State University (CSU) type rainfall simulator (Holland,

TABLE 4 Continued

	Year-2			Year-9					
	Burned			Unburned			Burned		
Onaqui	Interspace	Shrub coppice	Tree coppice	Interspace	Shrub coppice	Tree coppice	Interspace	Shrub coppice	Tree coppice
Dry run simulation (64 mm hr^{-1}, 45 min)									
Cumulative run-off (mm)	16 d	3 ab	6 bc	18 c	1 a	8 ab	8 ab	1 a	15 bc
Cumulative sediment (g m^{-2})[a]	60 b	48 ab	30 a	126 c	9 a	53 ab	52 ab	9 a	71 bc
Sediment/run-off (g m^{-2} · mm^{-1})[a]	3.51 ab	6.38 b	2.43 a	6.76 bc	5.55 abc	7.39 c	5.51 ab	4.10 a	4.64 a
Mean soil water repellency (WDPT; s)[b]	-	-	127	-	-	98 a	-	-	88 a
Maximum strength of soil water repellency (s)[c]	-	-	194	-	-	148	-	-	124
Depth of max soil water repellency (cm)[d]	-	-	0	-	-	0	-	-	1
Percent of plots with run-off	100	60	60	100	60	100	100	60	100
Wet run simulation (102 mm hr^{-1}, 45 min)									
Cumulative run-off (mm)	48 d	30 cd	23 bc	46 b	6 a	23 a	22 a	8 a	22 a
Cumulative sediment (g m^{-2})[a]	280 b	230 b	242 b	381 b	48 a	174 a	152 a	61 a	108 a
Sediment/run-off (g m^{-2} · mm^{-1})[a]	5.85 b	6.85 b	6.28 b	8.05 d	5.44 ab	7.30 cd	6.29 bc	6.06 abc	4.90 a
Percent of plots with run-off	100	100	80	100	80	100	100	80	100
No. of plots	10	5	5	4	5	5	10	5	5

TABLE 5 Patch-scale canopy cover, ground cover, and surface roughness measured on unburned and burned large rainfall simulation plots (13 m^2, Year-0 and Year-1) and concentrated flow plots (9 m^2, Year-2 and Year-9) at the Marking Corral site prior to burning (Year-0) and 1 (Year-1), 2 (Year-2), and 9 (Year-9) years following burning

	Year-0/Year-1				Year-2				Year-9			
	Unburned		Burned		Unburned		Burned		Unburned		Burned	
Marking Corral	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Tree zone
Total canopy cover (%)[a]	34.7 c	15.9 b	23.0 b	6.2 a	48.7 b	37.4 ab	22.0 a	12.9 a	56.0 ab	42.8 a	68.2 b	78.3 b
Shrub canopy cover (%)	20.6 c	1.5 b	0.4 a	0.0 a	16.0 b	12.3 b	1.0 a	0.0 a	17.8 b	12.7 b	1.9 a	0.2 a
Grass canopy cover (%)	8.1 c	2.8 ab	5.1 bc	1.5 a	18.2 c	17.3 c	9.8 b	2.0 a	33.1 a	21.8 a	61.8 b	72.8 b
Forb canopy cover (%)	0.4 a	0.2 a	14.1 b	4.2 a	2.5 ab	0.2 a	9.7 bc	10.8 c	1.1 a	0.1 a	0.3 a	0.2 a
Total ground cover (%)[b]	31.7 b	93.6 d	14.5 a	72.8 c	52.7 b	81.7 c	18.9 a	69.8 bc	31.5 a	88.5 c	49.4 b	77.9 c
Litter cover (%)	28.5 b	87.8 d	10.4 a	66.9 c	39.7 b	68.6 c	12.7 a	66.2 c	15.3 a	83.4 c	35.7 b	68.9 c
Rock cover (%)	46.8 c	3.6 a	15.6 b	3.6 a	9.9 b	1.6 a	19.6 c	1.8 a	18.1 b	2.2 a	23.0 b	10.5 a
Bare ground (%)[c]	68.3 c	6.4 a	85.5 d	27.2 b	47.3 b	18.3 a	81.1 c	30.2 ab	68.5 c	11.5 a	50.6 b	22.1 a
Ash (%)	-	-	0.2	4.8	-	-	-	-	-	-	-	-
Surface roughness (mm)	17 ab	22 b	15 a	13 a	18 ab	22 b	16 ab	10 a	18 a	18 a	16 a	15 a
Average canopy gap (cm)	76.8 a	191.5 b	73.6 a	234.5 b	43.0 a	56.3 a	67.8 a	117.2 b	45.5 a	66.5 b	42.5 a	34.1 a
Average basal gap (cm)	136.8 a	331.8 b	170.1 a	368.9 b	62.2 a	79.8 a	105.6 b	183.5 b	60.2 b	72.0 b	51.5 ab	40.3 a
Number of plots	6	6	6	6	3	3	6	6	5	5	5	5

Note. Means within a row by study year combination (Year-0/Year-1) or year (Year-2 or Year-9) followed by a different lower case letter are significantly different ($P < 0.05$).

[a]Excludes tree canopy removed immediately prior to rainfall simulation. [b]Includes cryptogam, litter, live and dead basal plant, and woody dead cover. [c]Includes ash, bare soil, and rock covers.

TABLE 6 Patch-scale canopy cover, ground cover, and surface roughness measured on unburned and burned large rainfall simulation plots (13 m^b, Year-0 and Year-1) and concentrated flow plots (9 m^2, Year-2 and Year-9) at the Onaqui site prior to burning (Year-0) and 1 (Year-1), 2 (Year-2), and 9 (Year-9) years following burning

	Year-0/Year-1				Year-2				Year-9			
	Unburned		Burned		Unburned		Burned		Unburned		Burned	
Onaqui	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Tree zone
Total canopy cover (%)[a]	12.7 a	20.8 b	17.2 b	3.3 a	5.8 a	10.3 a	22.3 b	4.5 a	19.2 a	51.6 b	60.2 b	87.2 c
Shrub canopy cover (%)	0.5 b	0.0 a	0.2 a	0.1 a	0.5 a	0.8 a	0.1 a	0.1 a	0.1 a	0.2 a	5.4 b	9.8 b
Grass canopy cover (%)	5.7 b	12.3 c	6.1 b	0.7 a	1.1 a	6.9 ab	9.3 b	1.6 a	4.4 a	28.9 b	28.5 b	60.5 c
Forb canopy cover (%)	4.7 b	2.8 ab	1.8 a	0.5 a	2.9 a	0.5 a	11.9 b	2.2 a	10.6 a	8.5 a	20.0 a	12.1 a
Total ground cover (%)[b]	15.8 a	88.2 c	19.0 a	31.9 b	7.1 a	94.7 c	20.6 a	41.5 b	12.3 a	88.4 c	23.3 a	66.7 b
Litter cover (%)	7.3 a	78.8 c	15.1 a	30.1 b	5.3 a	88.9 c	16.0 a	39.2 b	8.2 a	79.3 c	7.7 a	48.9 b
Rock cover (%)	58.0 c	7.7 a	38.1 b	9.7 a	45.5 b	2.4 a	36.7 b	8.5 a	51.8 b	4.4 a	31.5 b	9.5 a
Bare ground (%)[c]	84.2 c	11.8 a	81.0 c	68.1 b	92.9 c	5.3 a	79.4 c	58.5 b	87.7 c	11.6 a	76.7 c	33.3 b
Ash (%)	-	-	0.2	17.4	-	-	-	-	-	-	-	-
Surface roughness (mm)	31 a	35 a	26 a	26 a	16 a	19 a	21 a	19 a	19 a	18 a	20 a	17 a
Average canopy gap (cm)	88.0 a	104.9 a	110.1 a	356.0 b	128.2 b	129.8 b	55.5 a	186.0 b	62.1 b	45.0 a	41.6 a	36.4 a
Average basal gap (cm)	120.5 a	147.8 a	167.6 b	367.0 c	176.2 b	159.3 b	87.0 a	244.7 b	71.6 b	54.6 a	55.1 a	44.3 a
Number of plots	6	6	6	6	2	2	6	6	5	5	5	5

Note. Means within a row by study year combination (Year-0/Year-1) or year (Year-2 or Year-9) followed by a different lower case letter are significantly different ($P < 0.05$).

[a]Excludes tree canopy removed immediately prior to rainfall simulation. [b]Includes cryptogam, litter, live and dead basal plant, and woody dead cover. [c]Includes ash, bare soil, and rock covers.

TABLE 7 Patch-scale run-off and sediment response variables measured on unburned and burned large rainfall simulation plots (13 m^2 in Year-0/Year-1; 12 m^2 in Year-9) at the Marking Corral site prior to burning (Year-0) and 1 (Year-1) and 9 (Year-9) years following burning

	Year-0/Year-1				Year-9			
	Unburned		Burned		Unburned		Burned	
Marking Corral	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Tree zone
Dry run simulation (64 mm hr^{-1} [Year-0/Year-1] or 70 mm hr^{-1} [Year-9], 45 min)								
Cumulative run-off (mm)	12 b	1 a	3 a	1 a	8 a	4 a	0 a	0 a
Cumulative sediment (g m^{-2})[a]	45 c	17 b	25 b	6 a	34 a	5 a	-	-
Sediment/run-off (g m^{-2}·mm^{-1})[a]	3.80 a	15.72 bc	7.54 ab	21.94 c	4.29 a	1.18 a	-	-
Percent of plots with run-off	100	33	80	67	100	100	75	50
No. of plots	5	6	5	6	4	6	4	6
Wet run simulation (102 mm hr^{-1} [Year-0/Year-1] or 111 mm hr^{-1} [Year-9], 45 min)								
Cumulative run-off (mm)	36 b	3 a	34 b	11 a	37 b	12 a	6 a	6 a
Cumulative sediment (g m^{-2})[a]	154 b	43 a	346 c	78 ab	325 b	20 a	4 a	4 a
Sediment/run-off (g m^{-2}·mm^{-1})[a]	4.21 a	5.40 a	9.56 a	7.15 a	8.76 b	1.73 ab	0.75 a	0.74 a
Percent of plots with run-off	100	67	100	100	100	100	100	100
No. of plots	6	6	6	6	4	6	4	6

Note. Means within a row by study year combination (Year-0/Year-1) or year (Year-9) followed by a different lower case letter are significantly different ($P < 0.05$).

[a]Means based solely on plots that generated run-off.

TABLE 8 Patch-scale run-off and sediment response variables measured on unburned and burned large rainfall simulation plots (13 m^2 in Year-0/Year-1; 12 m^2 in Year-9) at the Onaqui site prior to burning (Year-0) and 1 (Year-1) and 9 (Year-9) years following burning

	Year-0/Year-1				Year-9			
	Unburned		Burned		Unburned		Burned	
Onaqui	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Tree zone
Dry run simulation (64 mm hr^{-1} [Year-0/Year-1] or 70 mm hr^{-1} [Year-9], 45 min)								
Cumulative run-off (mm)	8 ab	2 a	3 a	11 b	14 a	6 a	4 a	13 a
Cumulative sediment (g m^{-2})[a]	60 a	18 a	41 a	448 b	54 a	18 a	19 a	37 a
Sediment/run-off (g m^{-2}·mm^{-1})[a]	14.06 a	9.34 a	10.52 a	35.70 b	3.74 a	3.20 a	5.04 a	2.81 a
Percent of plots with run-off	100	83	60	100	100	100	75	100
No. of plots	6	6	5	5	4	4	4	4
Wet run simulation (102 mm hr^{-1} [Year-0/Year-1] or 111 mm hr^{-1} [Year-9], 45 min)								
Cumulative run-off (mm)	47 b	11 a	31 b	43 b	39 b	9 a	15 a	15 a
Cumulative sediment (g m^{-2})[a]	401 b	78 a	491 b	1893 c	192 b	38 a	58 a	37 a
Sediment/run-off (g m^{-2}·mm^{-1})[a]	9.01 b	6.09 a	16.01 c	44.67 d	4.99 a	4.12 a	3.82 a	2.46 a
Percent of plots with run-off	100	100	100	100	100	100	100	100
No. of plots	6	6	5	5	4	4	4	4

Note. Means within a row by study year combination (Year-0/Year-1) or year (Year-9) followed by a different lower case letter are significantly different ($P < 0.05$).

[a]Means based solely on plots that generated run-off.

1969; Pierson et al., 2007; Pierson et al., 2009; Pierson et al., 2010; Pierson et al., 2013; Pierson et al., 2015; Williams, Pierson, Al-Hamdan, et al., 2014) and the same dry- and wet-run target intensities as on small rainfall simulations. The CSU simulator consists of seven stationary sprinklers elevated 3.05 m above the ground surface and evenly spaced along each of the outermost borders of the respective rainfall-plot pair (Figure 2e). The CSU simulator design, rainfall attributes, and calibration procedures are described in detail in Pierson et al. (2013, 2015, 2010).

TABLE 9 Patch-scale run-off and sediment response variables by flow release rate on unburned and burned concentrated flow experiments (9 m^2) at the Marking Corral and Onaqui sites prior to burning (Year-0) and 1 (Year-1), 2 (Year-2), and 9 (Year-9) years following burning

		Year-0/Year-1				Year-2				Year-9			
		Unburned		Burned		Unburned		Burned		Unburned		Burned	
Marking Corral	Release Rate (L min^{-1})	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Tree zone
Marking Corral													
Cumulative run-off (L)	15	53 b	5 a	57 b	28 ab	11 a	0 a	20 a	28 a	43 c	0 a	0 a	9 b
	30	174 b	75 a	177 b	97 a	109 b	9 a	94 b	90 b	171 b	30 a	26 a	52 a
	45	283 b	213 a	300 b	234 a	217 b	69 a	197 b	176 b	287 a	106 a	153 a	167 a
Cumulative sediment (g)[a]	15	25 a	-	142 b	499 c	34 a	-	45 a	1,227 a	82 b	-	-	9 a
	30	176 a	128 a	2,240 c	972 b	1,970 b	19 a	254 b	1,496 b	718 b	203 a	31 a	29 a
	45	371 a	136 a	4,845 c	1,875 b	2,410 b	375 a	911 ab	1,158 ab	1,542 b	220 a	84 a	166 a
Flow velocity (m s^{-1})[a]	15	0.06 a	-	0.07 a	0.08 a	0.08 a	-	0.07 a	0.11 a	0.10 b	-	-	0.03 a
	30	0.08 b	0.04 a	0.12 b	0.12 b	0.14 b	0.05 a	0.11 ab	0.15 b	0.15 b	0.04 a	0.07 a	0.05 a
	45	0.10 b	0.04 a	0.17 b	0.12 b	0.18 b	0.07 a	0.16 ab	0.15 ab	0.22 b	0.05 a	0.09 a	0.07 a
Onaqui													
Cumulative run-off (L)	15	72 b	13 a	55 b	70 b	5 a	0 a	7 a	68 b	20 a	1 a	17 a	21 a
	30	164 b	75 a	161 b	176 b	85 a	29 a	59 a	163 b	106 b	13 a	73 b	93 b
	45	272 b	184 a	279 b	287 b	224 ab	145 a	208 ab	255 b	245 c	89 a	175 b	219 bc
Cumulative sediment (g)[a]	15	173 b	90 a	243 b	2,800 c	34 a	-	102 a	3,411 b	135 b	8 a	150 b	224 b
	30	587 b	287 a	1,372 c	3,422 d	1,109 ab	1,237 ab	702 a	3,904 b	694 b	111 a	533 b	599 b
	45	1,588 b	951 a	2,398 bc	3,840 c	4,093 a	2,426 a	1,920 a	3,571 a	2,186 b	500 a	1,300 b	1,348 b
Flow velocity (m s^{-1})[a]	15	0.07 ab	0.05 a	0.07 ab	0.11 b	0.10 ab	-	0.07 a	0.16 b	0.07 a	-	0.06 a	0.05 a
	30	0.11 ab	0.06 a	0.13 bc	0.19 c	0.12 ab	0.10 a	0.11 a	0.19 b	0.10 bc	0.04 a	0.14 c	0.08 b
	45	0.17 ab	0.14 a	0.17 ab	0.26 b	0.20 b	0.14 a	0.13 a	0.21 b	0.12 b	0.07 a	0.15 b	0.10 ab

Note. Means within a row by study year combination (Year-0/Year-1) or year (Year-2 or Year-9) followed by a different lower case letter are significantly different ($P < 0.05$).

[a]Mean based solely on plots that generated run-off.

Large plot rainfall simulations in Year-9 were conducted on individual plots (2 m wide × 6 m long) using a Walnut Gulch Rainfall Simulator (WGRS; Paige, Stone, Smith, & Kennedy, 2004) and target intensities of 70 mm hr^{-1} for the dry run and 111 mm hr^{-1} for the wet run. The WGRS consists of an oscillating central boom fitted with four 80–100 Vee-jet nozzles positioned 2.44 m above the ground surface (Figure 2 f). The WGRS simulator design, rainfall attributes, and calibration procedures are described in (Nouwakpo et al., 2017, 2019; Paige et al., 2004). For each simulator, mean rainfall intensity and cumulative rainfall applied by run type (dry and wet) were similar ($P > 0.05$) across unburned and burned conditions at each site (Nouwakpo et al., 2019; Pierson et al., 2015, 2010).

Timed samples of plot run-off and sediment for Year-0 and Year-1 CSU-type rainfall simulation experiments were collected by direct bottle samples of the discharge at the plot outlet at 1- to 3-min intervals throughout each 45-min rainfall simulation (see Pierson et al., 2015, 2010). For the WGRS simulations, run-off was conveyed into a supercritical flume at the downslope end of the plot where a Teledyne 4230 flow meter (Isco, Inc., Lincoln, NE) measured discharge at a rate of four samples per minute. Manual timed run-off samples were also collected periodically during WGRS simulations to validate automatically sensed run-off rate values (see Nouwakpo et al., 2019). Run-off volume and sediment concentration for each run-off sample collected were obtained in the laboratory by weighing the sample before and after drying at 105°C. Hydrologic response variables were derived for each large-plot rainfall simulation consistent with those described above for small-plot rainfall simulations.

Overland flow was applied to all concentrated flow plots using datalogger-controlled flow regulators and methodologies described by Pierson et al. (2010, 2015). All plots were prewet before simulations with a gently misting sprinkler to create wet soil conditions (~20% gravimetric) similar to those under which run-off occurs, but without detaching and eroding sediment. Concentrated overland flow was released at rates of 15, 30, and 45 L min^{-1} to each plot. The flow release sequence for each simulation was 12 min at 15 L min^{-1}, immediately followed by 12 min at 30 L min^{-1}, and immediately followed by 12 min at 45 L min^{-1}. Flow was released on each plot from a single location ~4-m upslope of the plot outlet. Flow passing through regulators was directly routed into a metal box filled with styrofoam pellets and was released through a 10-cm-wide mesh-screened opening at the base of the box (Figure 2g). Overland flow velocity was measured for each flow release rate on each plot by releasing a concentrated salt solution ($CaCl_2$, ~50 ml) into the flow and applying electrical conductivity probes to track the mean transit time of the salt over a 2-m flowpath length (Figure 2g; Pierson et al., 2009, 2008, 2015, 2010). Flow velocity (m s^{-1}) was computed by dividing the flowpath length (2 m) by the mean of multiple sampled salt travel times (n = 2 to 3 per rate per plot) in seconds.

Run-off samples for concentrated flow plots were collected at the plot outlet at 1- to 2-min intervals for each 12-min flow rate simulation and were processed in the laboratory for run-off and sediment concentration as described for rainfall simulation plots. Run-off and erosion response variables for each flow release rate were derived for an 8-min time period beginning at run-off initiation. A mean run-off rate (L min^{-1}) was computed for each sample interval as the cumulative run-off divided by the interval time. Cumulative run-off (L) by release rate for each plot was derived as the integration of run-off rates over the respective 8-min time of run-off. An averaged sediment concentration (g L^{-1}) was calculated for each sample interval as the cumulative sediment divided by the interval cumulative run-off. The mean sediment concentration for each flow release on each plot was computed as the average of all sediment concentrations for the respective rate. Cumulative sediment (g) by release rate for each plot was derived as the integrated sum of sediment collected during the 8-min run-off period.

2.6 | Data analysis

Statistical analyses were conducted using SAS software, version 9.4 (SAS Institute Inc, 2013). Statistical analyses for all spatial scales were restricted to within-site comparisons except where explicitly stated in results. Hillslope-scale vegetation and ground cover data collected on 30 × 33-m site characterization plots were analysed using a repeated measures mixed model with three treatment levels: Year-0 unburned, Year-1 burned, and Year-9 burned. Sample year was the repeated measure, with three levels: 2006, 2007, and 2015. The covariance structure was evaluated using fit statistics suggested by Littell, Milliken, Stroup, Wolfinger, and Schabenberger (2006), and the best fit model was applied. Analyses of measures from small plots were restricted to within-year comparisons at a site. Vegetation, ground cover, soil, and hydrologic and erosion response variables from small plots were analysed using a mixed model with two treatment levels (unburned and burned) and three microsite levels: interspace, shrub coppice, and tree coppice. For Year-0 and Year-1, large plot vegetation, ground cover, and hydrologic and erosion response variables and concentrated flow velocity data were analysed using a mixed model with two treatment levels (Year-0 unburned and Year-1 burned) and two microsite levels (shrub-interspace zone and tree zone). Hydrologic and erosion response variables for large plots in Year-9 were analysed using a mixed model with two treatment levels (unburned and burned) and two microsite levels (shrub-interspace zone and tree zone). Vegetation, ground cover, and flow velocity data for concentrated flow plots in Year-2 and Year-9 were analysed using a mixed model with two treatment levels (unburned and burned) and two microsite levels (shrub-interspace zone and tree zone). Concentrated flow run-off and erosion data for each site were analysed with a repeated measures mixed model using the treatment and microsite levels specified above for respective large-plot or concentrated flow-plot cover data. Flow release rate was the repeated measure for concentrated-flow run-off and erosion analyses, with three levels: 15, 30, and 45 L min^{-1}. Carry-over effects of concentrated flow releases were modelled with an autoregressive order one covariance structure (Littell et al., 2006). Plot location was considered a random effect, and site, treatment, and microsite were considered fixed effects in all analyses. Normality was tested prior to analysis of variance using the Shapiro–Wilk test, and data transformation was applied where necessary to address deviance. Where required, arcsine-square root transformations were used to normalize proportion data (e.g., canopy and ground cover), and logarithmic transformations were used to normalize run-off and erosion data. Back transformed means are reported. All reported significant effects were tested at the $P < 0.05$ level.

3 | RESULTS

3.1 | Hillslope-scale vegetation and ground cover

The limited understory canopy cover, degraded shrub layer, and low ground cover at both sites prior to burning reflect understory competition with pinyon and juniper for water and soil resources and indicate the sites were well into the later stages of woodland encroachment at initiation of this study (Figures 1a,d and 3). Hillslope-scale vegetation structure prior to burning at both sites was composed of tree islands (27–28% canopy cover) with a sparsely vegetated intercanopy and extensive well-connected bare interspaces (52–60% bare ground; Figure 1a,d). Understory canopy cover averaged 20–27% across the two sites and was mainly shrub cover (18%) at Marking Corral and grass (6%) and forb (3%) cover at Onaqui (Figure 3a,b). Shrub canopy cover was less than 1% at Onaqui. Dead shrubs made up approximately 12% of the shrub layer at Marking Corral and nearly half of the more limited shrub layer at Onaqui. Ground cover at both sites consisted primarily of litter (34–47%; Figure 3c,d), with most of the litter occurring underneath trees. Basal plant cover averaged less than 1% at both sites prefire. Bare ground was composed of approximately 27–31% bare soil and 25–29% rock cover (fragments >5 mm) prior to burning at the sites (Figure 3c,d).

The effect of burning on hillslope-scale understory canopy cover was minor in Year-1 due in part to the degraded prefire conditions, but burning substantially enhanced grass canopy cover and distribution of basal plant cover as measured nine growing seasons postfire. In Year-1 at Marking Corral, shrub cover was less and forb cover was greater for burned conditions relative to Year-0 unburned plots at that site (Figure 3a). All understory canopy cover measures at Onaqui were similar for unburned Year-0 and burned Year-1 plots (Figure 3b). Burning reduced ground cover and increased bare ground at Marking Corral and the fire-induced reductions in ground cover persisted for that site in Year-1 (Figure 3c). Burning had no effect on ground cover by Year-1 at Onaqui (Figure 3d). Fire-induced increases in grass cover at Marking Corral over nine growing seasons improved total canopy cover, but shrub cover in Year-9 was about 50% of that measured prior to burning (Figure 3a). In contrast, shrub, grass, and forb canopy cover all increased substantially (4- to 12-fold) over nine growing seasons postfire at Onaqui (Figure 3b). Bare ground remained high (near 50%) at both sites in Year-9 (Figure 3c,d), but burning stimulated increases in basal plant cover of more than 20-fold at Marking Corral and 15-fold at Onaqui over study period. Prescribed fire was effective at reducing pinyon and juniper cover at both sites. Residual density of these species for the 5- to 50-cm height size class in Year-9 at both sites was approximately 90% less than that measured in Year-0 prefire. Live canopy cover of pinyon and juniper trees ≥1 m was less than 4% at Marking Corral and less than 1% at Onaqui in Year-9, with densities for this size

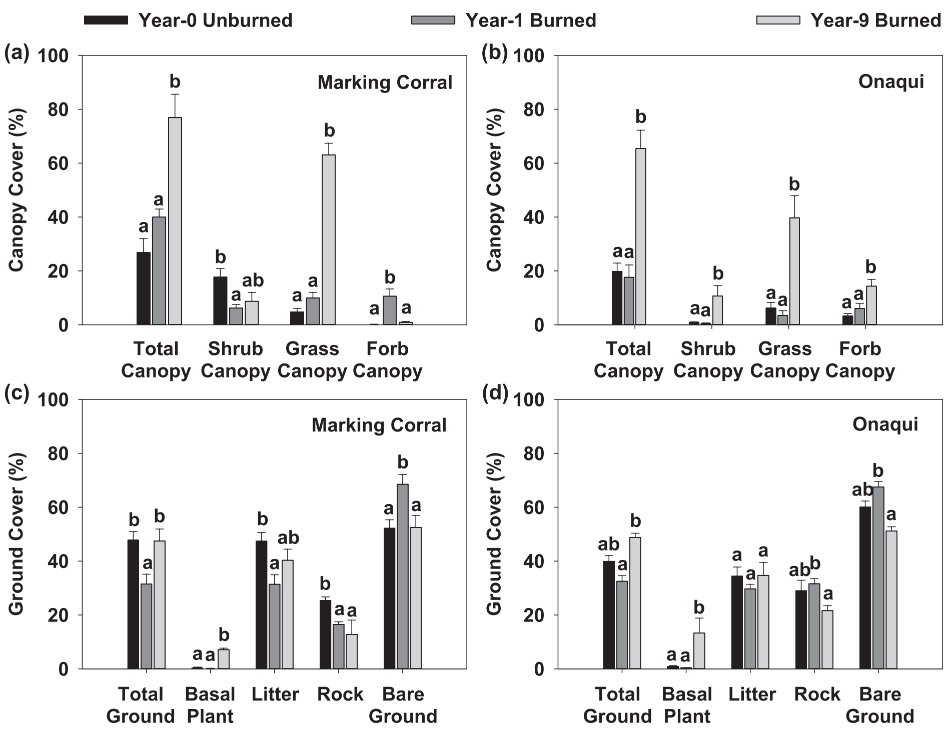

FIGURE 3 Hillslope-scale canopy (a and b) and ground (c and d) cover measured on site characterization plots (990 m^2) at the Marking Corral (a and c) and Onaqui (b and d) study sites immediately prefire (Year-0 unburned) and 1 (Year-1 burned) and 9 (Year-9 burned) years postfire. Means within a cover type across years/treatments (Year-0 Unburned, Year-1 Burned, and Year-9 Burned) for a given site (Marking Corral or Onaqui) with different lower case letters are significantly different ($P < 0.05$)

class reduced more than 70% from prefire levels. The primary impact of burning on hillslope-scale vegetation and ground cover at the sites was eradication of tree dominance and conversion of the understory to a grass-dominated structure with enhanced basal plant cover (Figure 1).

3.2 | Fine-scale vegetation and ground cover

First year responses of fine-scale vegetation and ground cover to burning were highly variable across the two sites (Table 2). The fires exerted minimal impact on the pre-existing sparse canopy and ground cover in interspaces at the sites over the first growing season postfire. Burning reduced grass cover by twofold on interspaces at Marking Corral but had no impact on grass cover as measured in Year-1 on interspaces at Onaqui (Table 2). Interspace forb cover at the two sites exhibited contrasting responses to burning, increasing at Marking Corral, and declining at Onaqui in Year-1 (Table 2). Total ground cover was low for unburned (10–27%) and burned (5–13%) interspaces at both sites in Year-1, and interspace bare ground in Year-1 averaged about 80% and 93% at Marking Corral and Onaqui, respectively, across burned and unburned conditions. The primary effects of burning on shrub coppices as measured in Year-1 included near complete removal of shrub canopy cover and a twofold to threefold reduction of litter cover, resulting in greater bare ground for burned (65–75%) versus unburned (16–39%) shrub coppices at the sites. With exception of grass cover at Onaqui, prescribed fire minimally altered canopy cover underneath trees as measured in Year-1 (Table 2). Likewise, ground cover percentages were largely unchanged by burning on tree coppices. Burning reduced litter depth nearly 2-fold at Marking Corral. The reduction in litter depth for tree plots at Making Corral without reduction in spatial litter coverage is likely due to minor amounts of needle fall from burned trees over the first year postfire. Fire effects on fine-scale canopy and ground cover in Year-2 were restricted to shrub and tree coppice microsites (Table 2). Shrub canopy cover and litter ground cover were lower on burned than unburned shrub coppice plots at both sites in Year-2. Likewise, bare ground was greater on burned than unburned shrub and tree coppice plots at both sites in Year-2. Over Year-1 and Year-2, the most persistent effects of burning were reduced shrub cover

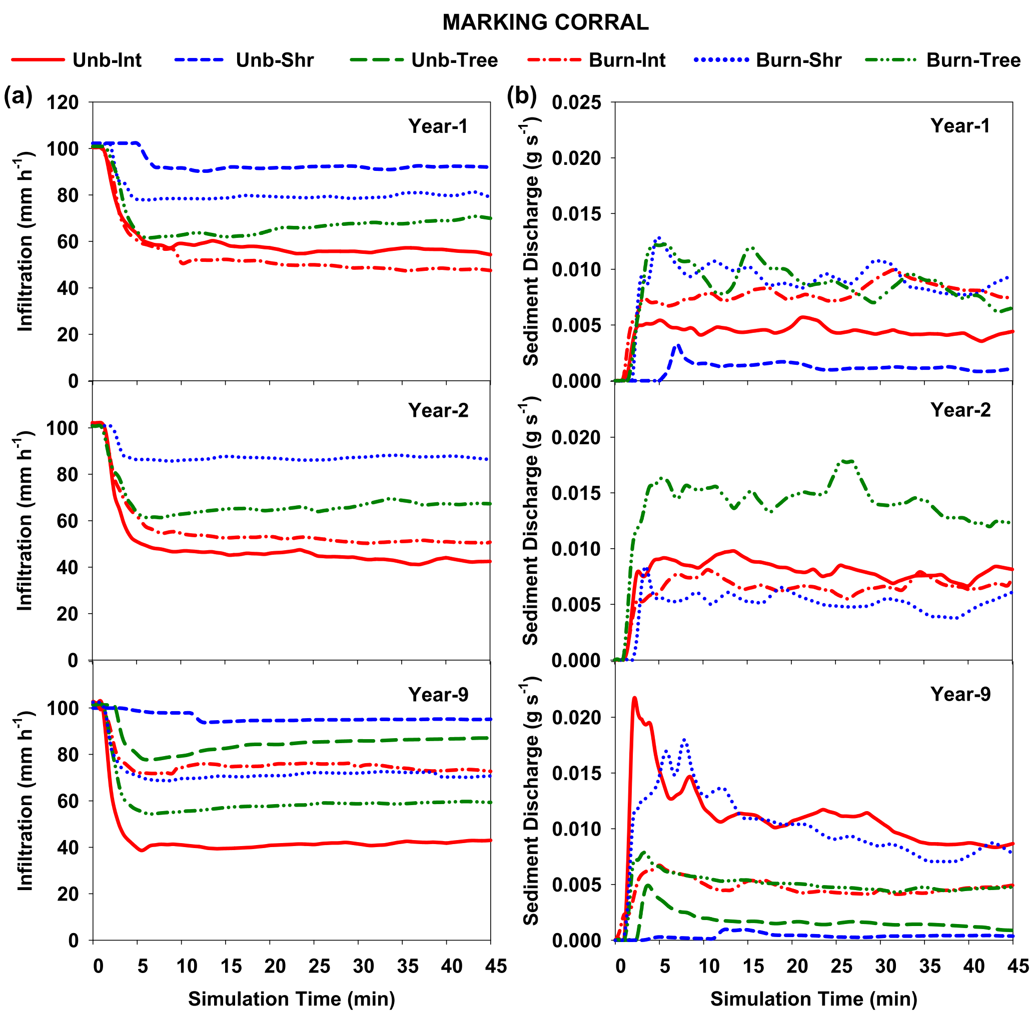

FIGURE 4 Fine-scale infiltration (a) and sediment discharge (b) for wet-run (102 mm hr^{-1}, 45 min) rainfall simulations on unburned (Unb) and burned (Burn) interspace (Int), shrub coppice (Shr), and tree coppice (Tree) small plots (0.5 m^2) at the Marking Corral site 1 (Year-1), 2 (Year-2), and 9 (Year-9) years after prescribed fire. For a given year, microsite (Int, Shr, or Tree) responses in a given treatment (Unb or Burn) are only shown for microsites that generated run-off

on shrub coppices and increased bare ground on shrub and tree plots at both sites (Table 2). Fire effects on surface roughness in the first 2 years postfire were limited to shrub coppices at Marking Corral (Year-1 and Year-2) and interspaces at Onaqui (Year-2). Burning reduced surface roughness by 3–6 mm in both cases (Table 2).

Grass cover increased dramatically at the fine-spatial scale over the 9-year period postfire (Table 2). Grass cover increased twofold to fourfold in interspaces over 9 years following burning at both sites. By Year-9, total canopy cover exceeded 60% in burned interspaces at both sites, and most of that cover was live and dead grasses. Although grass cover increased in interspaces after burning, bare ground remained more than 75% for burned interspaces in Year-9. Shrub canopy cover remained lower for burned than unburned conditions on shrub coppices in Year-9, and bare ground remained greater for burned than unburned shrub plots at Marking Corral after nine growing seasons (Table 2). Fire impacts on fine-scale canopy and ground cover on tree plots in Year-9 were highly variable across the two sites (Table 2). The main effects were increased grass canopy cover on burned tree plots at Marking Corral and reduced litter cover and increased bare ground on tree plots at Onaqui. Overall, prescribed burning at both sites reduced the coarseness of fine-scale vegetation structure via increased grass productivity and recruitment, homogenizing the landscape through grass dominance across interspace, shrub coppice, and tree coppice microsites (Table 2; Figure 1). In Year-9, grass canopy cover made up more than 90% and nearly 60% of total canopy cover on burned plots at Marking Corral and Onaqui, respectively. For unburned conditions in Year-9, grass canopy cover made up much (41–95%) of the total canopy cover on interspace and tree plots, but composed <25% of total canopy cover on shrub coppices. Grass canopy cover ranged 35% to 62% across all burned plots and 5% to 36% across all unburned plots at the two sites in Year-9.

3.3 | Fine-scale infiltration, run-off, and erosion

Interspaces were the primary source of run-off and sediment delivery from the fine-spatial scale prior to burning, and therefore initial fire effects on fine-scale infiltration, run-off, and erosion were mainly

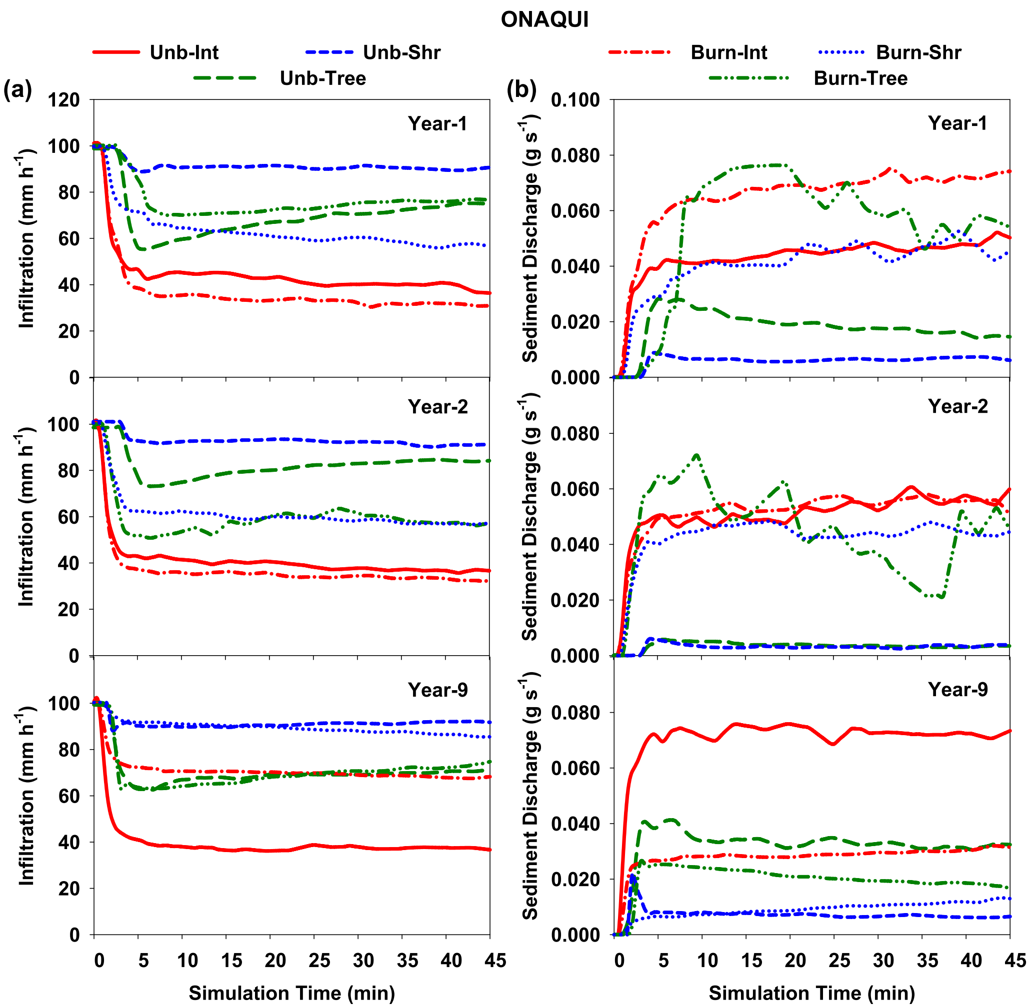

FIGURE 5 Fine-scale infiltration (a) and sediment discharge (b) for wet-run (102 mm hr^{-1}, 45 min) rainfall simulations on unburned (Unb) and burned (Burn) interspace (Int), shrub coppice (Shr), and tree coppice (Tree) small plots (0.5 m^2) at the Onaqui site 1 (Year-1), 2 (Year-2), and 9 (Year-9) years after prescribed fire. For a given year, microsite (Int, Shr, or Tree) responses in a given treatment (Unb or Burn) are only shown for microsites that generated run-off

limited to shrub and tree coppices (Tables 3, 4). Run-off and erosion were generally low for dry run rainfall simulations in each year with exception of erosion from unburned and burned interspaces and from burned coppice plots at Onaqui (Tables 3, 4). In general sediment yield and the sediment-to-run-off ratio were greater for rainfall simulations at Onaqui relative Marking Corral, indicating greater erodibility at that site (Tables 3, 4). For unburned conditions in Year-1 and Year-2, the highest amounts of wet-run run-off and erosion were generated from interspaces. These plots also exhibited the lowest wet-run steady state infiltration and highest wet-run sediment discharge rates for unburned plots each year (Figures 4, 5). The highest infiltration rates occurred on unburned shrub coppices, followed by tree coppices (Figures 4a and 5a). Soils were hydrophobic on unburned tree coppices (Tables 3, 4), but thick litter layers underneath trees captured and stored rainfall and allowed time for infiltration through the repellent layer. The effect of the litter layer in facilitating gradual increased infiltration through water repellent soils is seen in the infiltration curves for tree coppices, showing a gradual increase in infiltration rate throughout rainfall simulation (with increased wetting; Figures 4a and 5a). Sediment discharge was generally low from unburned shrub and tree coppices (Figures 4b and 5b), owing, respectively, to surface protection by high levels of canopy cover and litter ground cover. Fire impacts on wet-run run-off and erosion in Year-1 and Year-2 varied across the sites and by year but generally were greater for shrub and tree coppices given inherently high run-off and erosion from interspaces. At Marking Corral, burning facilitated run-off generation and sediment delivery from shrub coppices and increased run-off and erosion on tree coppices during wet-run simulations (Table 3; Figure 4). At Onaqui, fire impacts on wet-run run-off generation the first few years postfire were limited to shrub coppices in Year-2 (Table 4). However, burning increased wet-run sediment discharge and cumulative erosion from shrub and tree coppices at that site in both Year-1 and Year-2 (Figure 5b; Table 4). Wet-run erosion levels from unburned and burned interspaces in Year-1 and Year-2 at Onaqui were similar to erosion levels for burned shrub and tree coppices at that site (Table 4) and were greater than the same measures at Marking Corral (Table 3). Microsite hydrologic and erosion responses to burning the first few years postfire reflect initially high run-off and erosion from interspaces, fire effects on cover for coppice plots, persistence of soil water repellency on tree coppices, and enhanced sediment availability on shrub and tree plots after burning.

The primary effect of burning on fine-scale hydrologic and erosion processes after nine growing seasons postfire was enhanced infiltration in interspaces (Figures 4a and 5a). Wet-run infiltration rates for burned interspace plots in Year-9 were, on average, 75% greater than the same measures in unburned interspaces at both sites. The enhanced infiltration in interspaces after burning reduced interspace wet-run run-off more than twofold and thereby limited sediment delivery from the primary source of run-off and sediment yield prefire (Figures 4 and 5). In Year-9, cumulative sediment yield from wet-run simulations on burned interspaces was, on average, near 60% less than from unburned interspaces at the sites (Tables 3, 4). Fire impacts on wet-run run-off and erosion from shrub and tree coppices persisted at Marking Corral solely in Year-9 (Table 3), and as in Year-1 and Year-2, erosion rates for all microsites were generally greater at Onaqui than Marking Corral for unburned and burned conditions in Year-9 (Figures 4b and 5b).

3.4 | Patch-scale vegetation and ground cover

The initial impact of burning on patch-scale understory vegetation and ground cover was persistence of bare shrub-interspace zones and reduced ground cover in tree zones. Bare ground was 68% and 84% in shrub-interspace zones at Marking Corral and Onaqui, respectively, in Year-0 prior to burning (Tables 5, 6). Total understory canopy cover declined with burning in shrub-interspace zones at Marking Corral due to fire reduction of shrub canopy cover from 21% in Year-0 to <1% in Year-1 (Table 5). Forb canopy cover increased more than 30-fold from Year-0 to Year-1 on shrub interspaces at Marking Corral due to burning but was not enough to counter the effect of shrub cover loss on total canopy cover at that site. Burning reduced both total ground cover and litter cover and increased bare ground from 68% in Year-0 to 86% in Year-1 at Marking Corral (Table 5). At Onaqui, understory total canopy cover (13–17%) and ground cover (16–19%) in shrub-interspace zones were generally low prior to (Year-0) and the first year (Year-1) after burning (Table 6). Fire reduced total canopy cover in tree zones at the sites by about threefold to sixfold over the first year and increased tree-zone bare ground by fourfold to sixfold from Year-0 to Year-1. Increased bare ground in tree zones resulted from fire consumption of the litter underneath and adjacent to trees, creating 27% bare ground in burned tree zones at Marking Corral and 68% in burned tree zones at Onaqui in Year-1. The extensive bare ground in burned shrub-interspace and tree zones at Onaqui in Year-1 also resulted in greater average basal gap distances for burned conditions at that site (Table 6). In Year-2, shrub canopy cover remained low on burned shrub-interspace zone plots at both sites (Tables 5, 6). Grass canopy cover in Year-2 was lower for burned than unburned shrub interspaces at Marking Corral (Table 5) but was enhanced by burning on shrub interspaces at Onaqui (Table 6). Bare ground remained high (>80%) on burned shrub-interspace zones in Year-2 at Marking Corral (Table 5) and was similar for burned and unburned shrub-interspace plots at Onaqui (Table 6). Fire effects on patch-scale cover in tree zones in Year-2 were highly variable across the two sites, with reduced grass and shrub cover at Marking Corral (Table 5) and reduced total ground cover and litter cover at Onaqui (Table 6).

By Year-9, grass canopy cover substantially increased across burned plots at both sites relative to unburned conditions (Tables 5, 6; Figure 1). Grass canopy cover was twofold to more than sixfold greater on burned than unburned shrub-interspace and tree zone plots in Year-9 (Tables 5, 6). Grass canopy cover ranged 62% to 73% across all burned plots at Marking Corral and 29% to 61% across all burned plots at Onaqui in Year-9. These cover increases contributed to greater litter cover and reduced bare ground on shrub interspaces at Marking Corral (Table 5) and reduced basal and canopy gap distances in shrub-interspace zones at Onaqui (Table 6). Fire-induced increases in grass cover in tree zones in Year-9 reduced tree-zone canopy and basal gap distances at Marking Corral (Table 5) but did not counteract persistent fire effects on ground cover recruitment in burned tree zones at Onaqui. As with the fine scale, burning effectively enhanced grass cover across microsites and either reduced bare ground or improved the distribution of cover within shrub-interspace zones (Figure 1). Bare ground remained >50% on burned shrub-interspace plots at both sites in Year-9 (Tables 5, 6), indicating ground cover recruitment requires patience in these systems.

3.5 | Patch-scale run-off and erosion from combined processes

Prescribed fire was effective in reducing patch-scale run-off and erosion from degraded intercanopy areas over a 9-year period postfire. Prior to burning, run-off and erosion rates were high for the mostly bare shrub-interspace zones and were minimal for the well-protected tree zones at the sites (Figures 6, 7). Burning had limited negative impact on run-off and erosion from the patch scale at Marking Corral over the first year postfire (Table 7; Figure 6). Likewise, burning had minimal initial impact on run-off and erosion from shrub-interspace zones at Onaqui (Table 8). However, fire removal of cover in tree zones at Onaqui facilitated fourfold to fivefold increases in run-off and more than 20-fold increases in erosion in Year-1 (Table 8; Figure 7). By Year-9, enhanced cover in shrub-interspace zones limited run-off during dry- and wet-run rainfall simulations at both sites and thereby reduced sediment yield from shrub-interspace zone plots (Figures 6, 7). In Year-9, wet-run run-off from burned shrub-interspace zone plots was, on average, threefold to sixfold less than on unburned shrub-interspace zone plots (Tables 7, 8). Cumulative sediment yield from burned shrub-interspace zone plots in Year-9 was reduced substantially in comparison with unburned shrub-interspace plots (Tables 7, 8). Run-off and sediment yield were low for unburned and burned tree zones at the sites in Year-9 and were consisted with measures from Year-9 burned shrub-interspace plots (Tables 7, 8; Figure 6, 7).

3.6 | Patch-scale run-off and erosion from concentrated flow processes

Shrub-interspace zones at both sites delivered high amounts of concentrated overland flow and sediment under unburned conditions, and burning was effective at reducing both for the Marking Corral site by Year-9. In most cases, run-off and sediment generated by released concentrated flow was greater for degraded shrub-interspace zones relative to tree zones for unburned conditions, although these relationships exhibited some temporal variability by site (Table 9). Burning had minimal impact on run-off and sediment delivery from shrub-interspace zones the first few years postfire with exception of increased sediment yield for burned conditions at Marking Corral. In contrast, burning increased tree zone sediment delivery by 8- to 14-fold at Marking Corral and by 4- to more than 30-fold at the more erodible Onaqui site over the first year postfire (Table 9). Run-off for the burned condition on tree zones in Year-1 was sustained in high-velocity concentrated flow paths with high sediment detachment and transport energy (Table 9). Fire

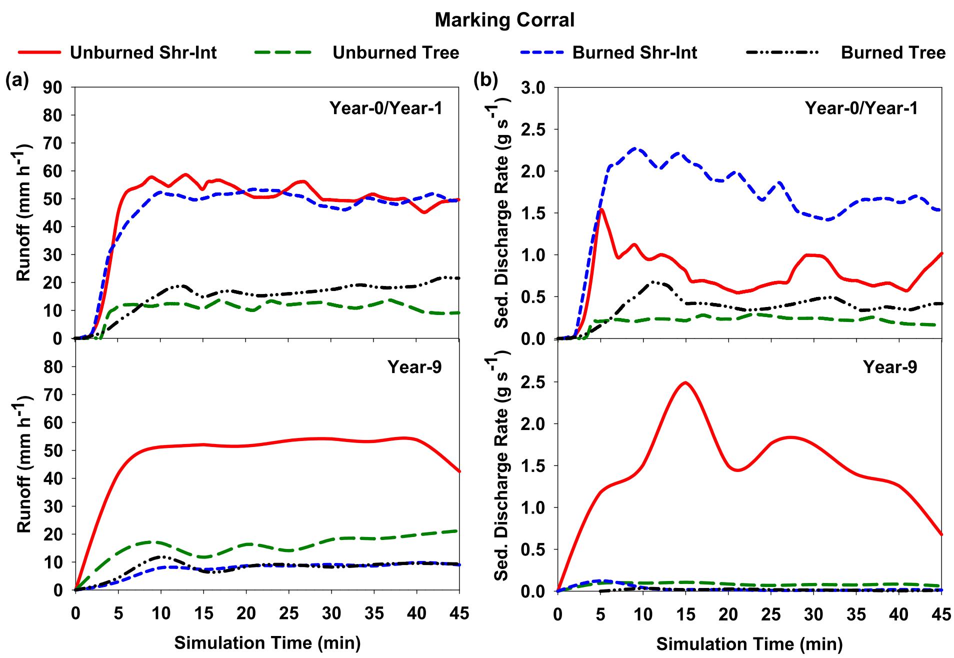

FIGURE 6 Patch-scale run-off rate (a) and sediment discharge (b) for wet-run (102 mm hr^{-1} [Year-0/Year1] to 111 mm hr^{-1} [Year-9], 45 min duration) rainfall simulations on unburned and burned shrub-interspace zone (Shr–Int) and tree zone (Tree) plots at the Marking Corral site. Plots were in unburned areas only in the year prior to burning (Year-0; 13 m^2), in burned areas only the first year postfire (Year-1; 13 m^2), and in unburned and burned areas 9 years postfire (Year-9; 12 m^2)

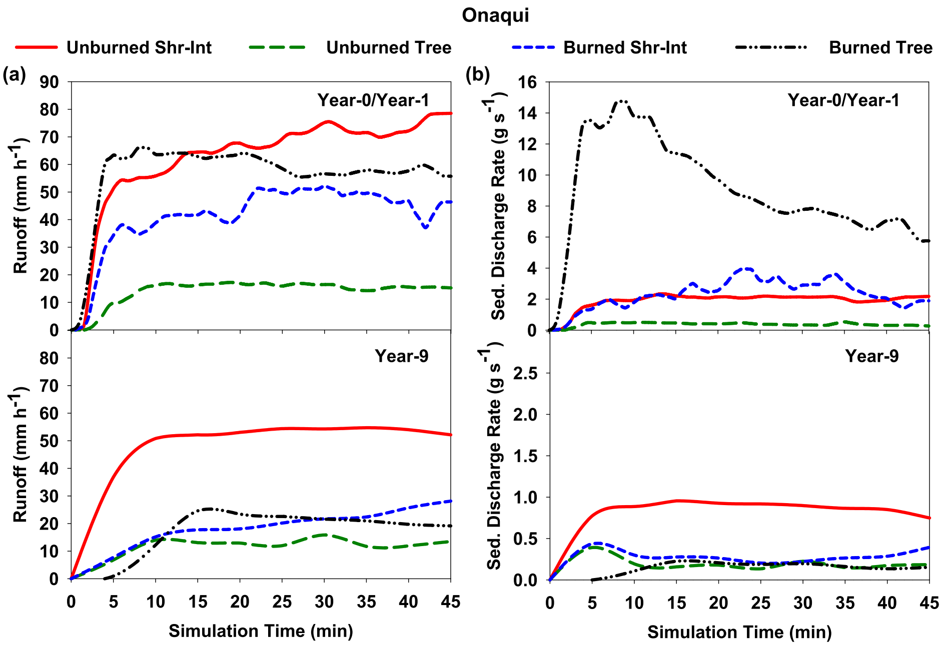

FIGURE 7 Patch-scale run-off rate (a) and sediment discharge (b) for wet-run (102 mm hr^{-1} [Year-0/Year-1] to 111 mm hr^{-1} [Year-9], 45 min duration) rainfall simulations on unburned and burned shrub-interspace zone (Shr–Int) and tree zone (Tree) plots at the Onaqui site. Plots were in unburned areas only in the year prior to burning (Year-0; 13 m^2), in burned areas only the first year postfire (Year-1; 13 m^2), and in unburned and burned areas 9 years postfire (Year-9; 12 m^2)

impacts on run-off and sediment delivery from tree zones persisted in Year-2, with both cumulative run-off and sediment on burned tree zones generally exceeding that of unburned tree zones. By Year-9, changes in cover on burned shrub-interspace at Marking Corral buffered concentrated flow releases and reduced both run-off and sediment delivery. Increased cover on tree zones at that site also resulted in similar run-off and sediment delivery across burned and unburned tree zones in Year-9 (Table 9). Run-off and sediment from shrub interspaces at Onaqui was similar for burned and unburned conditions in Year-9, but both measures were greater for burned than unburned tree zones at that site for most flow release rates (Table 9). The contrasting site responses by microsite in Year-9 are attributed to more bare conditions for the respective microsites at Onaqui relative to Marking Corral.

4 | DISCUSSION

4.1 | Can fire decrease late-succession woodland ecohydrologic resilience by increasing vegetation and ground cover over a 9-year period after fire?

Our study of two woodlands over a 9-year period postfire demonstrates fire can decrease late-succession woodland structure and function by increasing vegetation cover and altering plant community physiognomy. The prescribed fires in this study were effective at reducing pinyon and juniper cover to less than 4% and in substantially reducing tree seedlings (90% reduction in density; Figure 1). Enhanced grass cover over nine growing seasons at both sites indicates the tree reductions increased soil water availability for understory cover and thereby induced a switch in the dominant vegetation type and vegetation structure (Figures 1 and 3a,b; Tables 5, 6). Roundy, Miller, et al. (2014) and Roundy, Young, et al. (2014) assessed the effects of pinyon and juniper removal on soil water and vegetation at multiple woodland-encroached sagebrush sites in the Great Basin over a 4-year period posttreatment as part of the SageSTEP study. Sites in that study included experimental plots at our study sites, in adjacent areas to our experimental domain. Roundy, Young, et al. (2014) found that tree removal by both prescribed fire and mechanical methods increased the number of days that soil water was available to plants in the spring and that the increased time of soil water availability was greatest on sites with high tree dominance. Increases in spring season soil water availability declined with time posttreatment as plant cover increased, but treatment effects on soil water availability persisted in the fourth year postfire. Roundy, Miller, et al. (2014) found total perennial grass and forb cover was similar for burned and unburned treatments 2 years after burning but was greater for the burned than unburned treatments in the third year postfire. In a follow-up study of the same plots in Roundy, Miller, et al. (2014) and Roundy, Young, et al. (2014), Williams et al. (2017) found tall perennial grass cover was similar for burned and unburned treatments 3-year postfire but was greater in the burned than unburned treatments by the sixth year after fire. The increases in days of spring soil water availability across the Marking Corral and Onaqui sites in Roundy, Young, et al. (2014) averaged about 6–36 days the second year posttreatment and about 6–30 days the fourth year posttreatment for Phases II–III woodland plots (Roundy, Young, et al., 2014). In our study plots at Marking Corral and Onaqui, grass cover at the hillslope and patch scales was similar on unburned and burned plots the first year postfire, was generally low on unburned and burned plots in Year-2, and increased substantially in the burned treatment by Year-9 (Figure 3a,b; Tables 5, 6). In Year-9, much of the total grass canopy cover across the sites (40–63%) was tall perennial grasses (21–33% cover) located within the intercanopy (Figure 1c,f), with the remainder primarily as cheatgrass islands on burned tree coppices (16–30% cover). Burning consumed the shrub cover in our study, as expected, and much of the shrub cover in Year-9 was of root-sprouting species, mainly yellow rabbitbrush (*Chrysothamnus viscidiflorus* [Hook.] Nutt.). Sagebrush does not resprout after fire, and we anticipate it may take as many as 20–50 years to re-establish dominance on the sites (Miller et al., 2013). Sagebrush cover was about 6% at Marking Corral and remained <1% at Onaqui in Year-9. Although ground cover remained low across our study sites 9-year postfire (Figure 3c,d), fire-induced tree mortality and the subsequent increases in total cover (from 20% to 27% prefire to 65–77% 9-year postfire) and grass cover (from ~5% prefire to 40–63% 9-year postfire) clearly altered the vegetation and ground cover structure at the sites (Figure 1). Burning reduced bare ground in shrub-interspace zones at Marking Corral and reduced the distance between plant bases in shrub interspaces at Onaqui as measured in Year-9 (Tables 5, 6). We anticipate ground cover will increase and bare ground will decline across both sites over time as grass cover persists and shrub cover increases in the absence of competition with pinyon and juniper. Overall, the initial shift to grass cover as the dominant vegetation type, short-term decline in shrub cover postfire, and delayed recruitment in ground cover are typical of degraded sagebrush steppe following pinyon and juniper removal (Bates et al., 2005; Bates et al., 2014; Bates et al., 2017; Bates & Davies, 2016; Chambers et al., 2014; Miller et al., 2013; Roundy, Miller, et al., 2014; Williams et al., 2017). However, our long-term data clearly indicate burning can suppress the effects of woodland encroachment on understory cover even in the later stages of woodland encroachment and can thereby increase the amount and distribution of tall perennial grass cover over the first decade postfire.

The hydrologic and erosion responses to vegetation changes over 9-year postfire in this study demonstrate burning of late-succession woodlands can reduce woodland ecohydrologic resilience over time. Pinyon and juniper woodlands with degraded intercanopies commonly exist as "leaky" or "nonconserving" systems in which subtle further reductions in ground cover enhance run-off and soil loss across spatial scales and propagate long-term site degradation (Davenport, Breshears, Wilcox, & Allen, 1998; Wilcox et al., 1996; Wilcox, Breshears, & Allen, 2003; Wilcox, Pitlick, Allen, & Davenport, 1996; Williams, Pierson, Al-Hamdan, et al., 2014; Williams, Pierson, Spaeth, et al., 2016). For these landscapes, run-off and erosion are commonly well correlated at the patch scale, and the magnitude of run-off and sediment yield at that scale are strongly regulated by the amount and connectivity of bare ground and the water input rate (Pierson et al., 2013, 2010; Williams, Pierson, Al-Hamdan, et al., 2014). At the hillslope scale, connectivity of run-off and sediment sources and hydrologic and erosion processes, as affected by the amount and distribution of cover, dictate the magnitude of soil loss (Wilcox et al., 2003; Williams, Pierson, Robichaud, et al., 2016). These relationships have been well demonstrated in the literature for degraded woodland-encroached sagebrush sites in the Great Basin (Petersen et al., 2009; Petersen & Stringham, 2008; Pierson et al., 2007; Pierson et al., 2013, 2010; Williams, Pierson, Al-Hamdan, et al., 2014; Williams, Pierson, Robichaud, et al., 2016; Williams, Pierson, Spaeth, et al., 2016). The reversal of such abiotic-controlled soil loss requires a trigger, such as a disturbance, that induces a switch in the dominant vegetation type and structure and long-term persistence of that change (Pierson et al., 2007; Roundy et al., 2017; Williams, Pierson, Al-Hamdan, et al., 2014). Both sites in this study were in the later stages of woodland encroachment and exhibited high rates of run-off and erosion across the fine to patch scales (Tables 3, 4 and 7, 8). Prior to burning, the extensive bare ground in the intercanopies at both sites promoted run-off generation from bare interspaces during high-intensity rainfall simulations that accumulated downslope into concentrated flow paths with high flow velocity and sediment detachment and transport capacity (Pierson et al., 2010; Williams, Pierson, Robichaud, et al., 2016). Pierson et al. (2010) found that run-off was generally similar across small-plot to large-plot scales at both sites but that erosion rates increased with increasing plot scale due to run-off as concentrated flow at the larger spatial scale. Pierson et al. (2010) also reported run-off and erosion were well correlated for the sites, typical for degraded systems. Burning initially increased bare ground across scales (Figure 3; Tables 2, 5, and 6) resulting in enhanced sediment availability in shrub and tree coppice plots (Figures 4b and 5b), sustained high run-off and sediment discharge rates from shrub-interspace zones (Figures 6 and 7), and amplified run-off and sediment delivery from tree zones (Tables 8, 9). Collectively, these fire impacts propagated an increase in sediment delivery with increasing plot scale for all but burned tree zones at Marking Corral the first year postfire (Tables 3, 4 and 7, 8; Williams, Pierson, Robichaud, et al., 2016). By Year-9, fire-induced increases in grass cover within interspaces improved infiltration (Figures 4a and 5a), limited run-off and sediment delivery from the fine-spatial scale (Tables 3, 4), and reduced run-off and erosion from shrub-interspace zones at the patch scale (Tables 7, 8). These changes reflect a switch in vegetation structure, triggered by burning and subsequent vegetation re-establishment, and the onset of biotic control on run-off and erosion across spatial scales at both sites (Wilcox et al., 2003; Williams, Pierson, Al-Hamdan, et al., 2014). The reduced run-off and erosion in shrub-interspace zones suggests hydrologic function improved throughout most of the domain at each site, as the intercanopy made up more 70% of total area at both sites prior to treatment (Table 1). Collectively, the responses over the nine years postfire at the two sites clearly support the inference by Williams, Pierson, Al-Hamdan, et al. (2014) that fire can act as an ecohydrologic threshold reversal mechanism on sagebrush shrublands in the later stages of woodland encroachment.

4.2 | Is the soil erosion feedback on late-succession woodlands reversible by burning?

Substantial declines in run-off and erosion across the fine to patch scales at the two woodlands in this study in association with a shift to biotic control on run-off and erosion rates demonstrate that the soil erosion feedback is reversible on late-succession woodlands through burning. The shift to biotic control on cross-scale run-off and erosion in this study is indicative of a change from a "nonconserving" to a "resource conserving" system (Wilcox et al., 2003). In "resource conserving" woodlands, vegetated patches and ground cover capture rainfall and isolated run-off and limit soil loss, and the retention of these resources further enhances vegetation and soil productivity (Ludwig, Wilcox, Breshears, Tongway, & Imeson, 2005; Reid, Wilcox, Breshears, & MacDonald, 1999; Wilcox et al., 2003). "Resource conserving" conditions are typical for sagebrush steppe in good ecological condition (Pierson et al., 1994). For these ecosystems, infiltration rates are typically high, and sediment delivery rates are low for shrub coppices and vegetated interspaces (Pierson et al., 1994; Pierson et al., 2009, 2008; Williams, Pierson, Kormos, et al., 2016). Run-off and erosion generated in bare interspaces is readily captured downslope in vegetated and litter-covered microsites (Pierson & Williams, 2016). Burning in our study re-established grass as the dominant cover and the fire-induced recruitment and expansion of grass cover filled in bare patches through the bare intercanopies at both sites (Figure 1c,f). Sagebrush recruitment continues at both sites and, overall, burning set forth a trajectory towards re-establishment of the sagebrush steppe "resource conserving" vegetation structure typical for these ecosystems. We anticipate the current "resource conserving" ecohydrologic function at both sites will be further enhanced as the sagebrush steppe vegetation structure continues to develop, as is common for recovery of sagebrush steppe following burning (Williams, Pierson, Spaeth, et al., 2016). The recovery rate is of course prolonged for degraded pretreatment conditions such as those at Marking Corral and Onaqui, particularly for ground cover recruitment and soil development (Bates et al., 2014; Bates et al., 2017; Miller et al., 2013; Williams, Pierson, Kormos, et al., 2019), but a "resource conserving" structure already exists at both sites, clearly depicting a reverse of the soil erosion feedback (Williams, Pierson, Al-Hamdan, et al., 2014). The persistence of high run-off and erosion from concentrated flow releases in burned plots at Onaqui nine years postfire reflects the effect of the delayed ground cover recruitment at that site (Table 9). Those experiments are intended to stress the system and represent more extreme events. The minimal run-off and erosion rates from the patch-scale high-intensity rainfall simulation experiments in burned plots at Onaqui (Figure 7) suggest formation of concentrated flow during all but the most extreme events is not likely at Onaqui given cover conditions in Year-9. Therefore, the persistent high erosion rates for concentrated flow experiments on burned conditions nine years postfire at that site do not negate our assertion of the shift to resource-conserving conditions for that site overall.

Our assertion that burning can reverse the soil erosion feedback on late-succession woodlands by reducing ecohydrologic resilience of the woodland conditions (promoting ecohydrologic resilience of new state) comes with some caveats. First, our study does not define a timeline for reversal of a soil erosion feedback or the shift to "resource-conserving" conditions in these systems. Re-establishment of sagebrush steppe vegetation and associated surface soil conditions on degraded woodlands is highly variable, is driven by conditions at the time of treatment, the type of treatment, and posttreatment weather, and may require seeding (Bates et al., 2005; Bates et al., 2014; Bates et al., 2017; Bates & Davies, 2016; Chambers et al., 2014; Davies et al., 2019; Miller et al., 2005; Miller et al., 2013; Miller et al., 2014; Roundy, Miller, et al., 2014). However, our study does demonstrate tree removal on late-succession woodlands can effectively re-establish a "resource-conserving" vegetation structure and reverse the soil erosion feedback. Similar results were reported by Pierson et al. (2007) following tree removal by cutting in a western juniper (*J. occidentalis* Hook.) woodland in the northwestern portion of the Great Basin. Pierson et al. (2007) reported increased intercanopy perennial herbaceous cover and litter cover 10 years after cutting juniper on a sagebrush shrubland in the later stages of woodland encroachment. Patch-scale rainfall simulations in bare (84% bare ground) intercanopy areas in an adjacent uncut woodland generated run-off and erosion levels 14- to more than 85-fold greater, respectively, than in the cut woodland. Run-off and erosion from patch-scale rainfall simulations in the cut woodland were negligible. In companion experiments to our study, Williams, Pierson, Kormos, et al. (2019) found that mechanical removal (cutting and mastication) of pinyon and juniper at Marking Corral and Onaqui effectively re-established a successional trajectory towards sagebrush steppe vegetation over a 9-year period posttreatment, but vegetation responses to treatment had minimal impact on patch-scale infiltration and sediment delivery with respect to prescribed fire treatments in this study. In another Great Basin study, Roundy et al. (2017) found that pinyon and juniper removal by chaining paired with a seeding treatment increased intercanopy vegetation cover from 5% to 24% 1 year after treatment and to more than 40% 3 years after treatment. Litter cover was near 15% in untreated areas and exceeded 50% in treated areas within 3 years after chaining and seeding. Intercanopy run-off and sediment delivery during natural rainfall at the patch scale were 5-fold and 10-fold less the fifth year after treatment relative to 5-year averages for untreated conditions. The studies noted here demonstrate tree removal can be effective at recruiting understory cover and reversing the soil erosion feedback but that responses can be prolonged for substantially degraded cover conditions depending on the method of treatment (Pierson et al., 2007; Roundy et al., 2017; Williams, Pierson, Kormos, et al., 2019). Furthermore, numerous studies have demonstrated that residual and newly recruited pinyon and juniper can re-establish tree dominance over time following treatment and that understory cover declines as tree cover expands (Barney & Frischknecht, 1974; Bates et al., 2005; Bates et al., 2017; Miller et al., 2005; Tausch & Tueller, 1997). Therefore, follow-up treatment may be required to maintain a "resource-conserving" vegetation structure and ecohydrologic function after tree removal on woodland-encroached sagebrush sites, particularly after mechanical treatments. Lastly, substantial increases in the fire-prone annual cheatgrass at Marking Corral and Onaqui have increased the risk of wildfire and the potential for more frequent burning at the sites (Balch et al., 2013; Brooks et al., 2004; Link, Keeler, Hill, & Hagen, 2006). Soil loss could increase at either site over time with more frequent burning in association with long-term conversion of the vegetation type to cheatgrass (Pierson et al., 2011; Wilcox et al., 2012; Williams, Pierson, Robichaud, & Boll, 2014). Conversion of either site to a cheatgrass monoculture over time, however, is unlikely given ample coverage of perennial bunchgrasses (Bates et al., 2014; Chambers, Roundy, Blank, Meyer, & Whittaker, 2007; Miller et al., 2013; Miller et al., 2014). Conversion of the vegetation type to a cheatgrass monoculture would most likely require a broad-scale high severity fire with enough heat to kill the existing native perennial bunchgrasses (Bates et al., 2006; Bates et al., 2011, 2014; Bates & Davies, 2016). Marking Corral and Onaqui are geographically positioned along a climatic gradient whereby risk of conversion to a cheatgrass-dominated cover type and fire-related long-term soil loss are unlikely, but woodland-encroached sites at lower and warmer elevations may be more susceptible to cheatgrass invasion and posttreatment dominance following tree removal by prescribed fire (Chambers et al., 2014; Miller et al., 2013).

5 | SUMMARY AND CONCLUSIONS

Our study of two woodlands over a 9-year period postfire demonstrates fire can act as an ecohydrologic threshold reversal mechanism on late-succession woodland-encroached sagebrush shrublands. We measured extensive bare ground and depauperate vegetation across fine to hillslope scales within the intercanopies of two woodland-encroached sagebrush sites in the Great Basin. The degraded cover conditions were associated with competition between encroaching trees and understory vegetation for limited water and soil resources. The extensive bare conditions promoted high levels of run-off and erosion at the fine spatial scale that accumulated into concentrated flow paths with high flow velocity and sediment detachment and transport capacity over larger spatial scales. Burning initially increased bare conditions, exacerbated pre-existing high run-off and erosion rates in degraded intercanopy areas, and amplified run-off and soil loss from areas previously vegetated by shrub and trees. Over a period of 9 years, the eradication of tree dominance increased cover and the spatial distribution of grasses throughout previously bare intercanopy patches and thereby enhanced infiltration at the fine spatial scale, limited run-off and sediment delivery to the patch scale, and decreased run-off and erosion across all spatial scales. The shift in plant community physiognomy with tree removal indicates fire can effectively decrease late-succession woodland ecohydrologic resilience by increasing vegetation and ground cover, over a 9-year period postfire in this case. The shift from an abiotic-driven run-off and long-term soil loss degraded state to a “resource-conserving” state of biotic-controlled resource retention at the two sites in this study demonstrates fire can reverse the soil erosion feedback on late-succession woodland encroached sagebrush sites. This assertion is of course based on the sites studied here, and we acknowledge that the long-term effects of burning on vegetation, ground cover, and run-off and erosion processes likely vary with initial site conditions, burn severity, and climate trends and land management in the postfire recovery period. Furthermore, persistence of favourable responses as measured in this study likely requires follow-up treatment to prevent re-establishment of tree dominance and degradation of understory cover over time.

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REFERENCES

Al-Hamdan, O. Z., Pierson, F. B., Nearing, M. A., Stone, J. J., Williams, C. J., Moffet, C. A., … Weltz, M. A. (2012). Characteristics of concentrated flow hydraulics for rangeland ecosystems: Implications for hydrologic modeling. *Earth Surface Processes and Landforms*, *37*(2), 157–168. https://doi.org/10.1002/esp.2227

Al-Hamdan, O. Z., Pierson, F. B., Nearing, M. A., Williams, C. J., Stone, J. J., Kormos, P. R., … Weltz, M. A. (2012). Concentrated flow erodibility for physically based erosion models: Temporal variability in disturbed and undisturbed rangelands. *Water Resources Research*, *48*(7), W07504.

Al-Hamdan, O. Z., Pierson, F. B., Nearing, M. A., Williams, C. J., Stone, J. J., Kormos, P. R., … Weltz, M. A. (2013). Risk assessment of erosion from concentrated flow on rangelands using overland flow distribution and shear stress partitioning. *Transactions of the ASABE*, *56*(2), 539–548. https://doi.org/10.13031/2013.42684

Balch, J. K., Bradley, B. A., D'Antonio, C. M., & Gómez-Dans, J. (2013). Introduced annual grass increases regional fire activity across the arid western USA (1980-2009). *Global Change Biology*, *19*(1), 173–183. https://doi.org/10.1111/gcb.12046

Barney, M. A., & Frischknecht, N. C. (1974). Vegetation changes following fire in the pinyon-juniper type of west-central Utah. *Journal of Range Management*, *27*(2), 91–96. https://doi.org/10.2307/3896738

Bates, J. D., & Davies, K. W. (2016). Seasonal burning of juniper woodlands and spatial recovery of herbaceous vegetation. *Forest Ecology and Management*, *361*, 117–130. https://doi.org/10.1016/j.foreco.2015.10.045

Bates, J. D., Davies, K. W., & Sharp, R. N. (2011). Shrub-steppe early succession following juniper cutting and prescribed fire. *Environmental*

Management, *47*(3), 468–481. https://doi.org/10.1007/s00267-011-9629-0

Bates, J. D., Miller, R. F., & Davies, K. W. (2006). Restoration of quaking aspen woodlands invaded by western juniper. *Rangeland Ecology and Management*, *59*(1), 88–97. https://doi.org/10.2111/04-162R2.1

Bates, J. D., Miller, R. F., & Svejcar, T. (2005). Long-term successional trends following western juniper cutting. *Rangeland Ecology and Management*, *58*(5), 533–541. https://doi.org/10.2111/1551-5028(2005)58[533:LSTFWJ]2.0.CO;2

Bates, J. D., Miller, R. F., & Svejcar, T. J. (2000). Understory dynamics in cut and uncut western juniper woodlands. *Journal of Range Management*, *53*(1), 119–126. https://doi.org/10.2307/4003402

Bates, J. D., Sharp, R. N., & Davies, K. W. (2014). Sagebrush steppe recovery after fire varies by development phase of *Juniperus occidentalis* woodland. *International Journal of Wildland Fire*, *23*(1), 117–130. https://doi.org/10.1071/WF12206

Bates, J. D., Svejcar, T., Miller, R., & Davies, K. W. (2017). Plant community dynamics 25 years after juniper control. *Rangeland Ecology and Management*, *70*(3), 356–362. https://doi.org/10.1016/j.rama.2016.11.003

Bates, J. D., Svejcar, T. J., & Miller, R. F. (2002). Effects of juniper cutting on nitrogen mineralization. *Journal of Arid Environments*, *51*(2), 221–234. https://doi.org/10.1006/jare.2001.0948

Bisdom, E. B. A., Dekker, L. W., & Schoute, J. F. T. (1993). Water repellency of sieve fractions from sandy soils and relationships with organic material and soil structure. *Geoderma*, *56*(1–4), 105–118. https://doi.org/10.1016/0016-7061(93)90103-R

Board, D. I., Chambers, J. C., Miller, R. F., & Weisberg, P.J. (2018). Fire patterns in piñon and juniper land cover types in the semiarid western United States from 1984 through 2013. RMRS-GTR-372. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Bonnin, G. M., Martin, D., Lin, B., Parzybok, T., Yekta, M., & Riley, D. (2006). *Precipitation frequency atlas of the United States, NOAA Atlas 14, Volume 1, Version 4.0*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Weather Service.

Brooks, M. L., D'Antonio, C. M., Richardson, D. M., Grace, J. B., Keeley, J. E., DiTomaso, J. M., … Pyke, D. (2004). Effects of invasive alien plants on fire regimes. *Bioscience*, *54*(7), 677–688. https://doi.org/10.1641/0006-3568(2004)054[0677:EOIAPO]2.0.CO;2

Bybee, J., Roundy, B. A., Young, K. R., Hulet, A., Roundy, D. B., Crook, L., … Cline, N. L. (2016). Vegetation response to piñon and juniper tree shredding. *Rangeland Ecology and Management*, *69*(3), 224–234. https://doi.org/10.1016/j.rama.2016.01.007

Chambers, J. C., Miller, R. F., Board, D. I., Pyke, D. A., Roundy, B. A., Grace, J. B., … Tausch, R. J. (2014). Resilience and resistance of sagebrush ecosystems: Implications for state and transition models and management treatments. *Rangeland Ecology and Management*, *67*(5), 440–454. https://doi.org/10.2111/REM-D-13-00074.1

Chambers, J. C., Roundy, B. A., Blank, R. R., Meyer, S. E., & Whittaker, A. (2007). What makes Great Basin sagebrush ecosystems invasible by *Bromus tectorum*? *Ecological Monographs*, *77*(1), 117–145. https://doi.org/10.1890/05-1991

Cline, N. L., Roundy, B. A., Pierson, F. B., Kormos, P., & Williams, C. J. (2010). Hydrologic response to mechanical shredding in a juniper woodland *Rangeland Ecology and Management*, *63*(4), 467–477. https://doi.org/10.2111/rem-d-09-00196.1

Coates, P. S., Prochazka, B. G., Ricca, M. A., Gustafson, K. B., Ziegler, P., & Casazza, M. L. (2017). Pinyon and juniper encroachment into sagebrush ecosystems impacts distribution and survival of greater sage-grouse. *Rangeland Ecology and Management*, *70*(1), 25–38. https://doi.org/10.1016/j.rama.2016.09.001

Davenport, D. W., Breshears, D. D., Wilcox, B. P., & Allen, C. D. (1998). Viewpoint: Sustainability of pinon-juniper ecosystems—A unifying perspective of soil erosion thresholds. *Journal of Range Management*, *51*(2), 231–240. https://doi.org/10.2307/4003212

Davies, K. W., Bates, J. D., & Boyd, C. S. (2019). Postwildfire seeding to restore native vegetation and limit exotic annuals: An evaluation in juniper-dominated sagebrush steppe. *Restoration Ecology*, *27*(1), 120–127. https://doi.org/10.1111/rec.12848

Davies, K. W., Boyd, C. S., Beck, J. L., Bates, J. D., Svejcar, T. J., & Gregg, M. A. (2011). Saving the sagebrush sea: An ecosystem conservation plan for big sagebrush plant communities. *Biological Conservation*, *144*(11), 2573–2584. https://doi.org/10.1016/j.biocon.2011.07.016

DeBano, L. F. (1981). Water repellent soils: A state-of-the-art. General Technical Report PSW-46. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station.

Harniss, R. O., & Murray, R. B. (1973). 30 years of vegetal change following burning of sagebrush-grass range. *Journal of Range Management*, *26*(5), 322–325. https://doi.org/10.2307/3896846

Herrick, J. E., Van Zee, J. W., Havstad, K. M., Burkett, L. M., & Whitford, W. G. (2005). *Monitoring manual for grassland, shrubland, and savanna ecosystems, Volume I: Quick Start*. Las Cruces, NM: U.S. Department of Agriculture, Agricultural Research Service, Jornada Experimental Range.

Holland, M. E. (1969). Colorado State University experimental rainfall-runoff facility, design and testing of a rainfall system. Report CER 69-70 MEH. Fort Collins, CO: Colorado State University, Colorado State University Experimental Station.

Keane, R. E., Agee, J. K., Fulé, P., Keeley, J. E., Key, C., Kitchen, S. G., … Schulte, L. A. (2008). Ecological effects of large fires on US landscapes: Benefit or catastrophe? *International Journal of Wildland Fire*, *17*(6), 696–712. https://doi.org/10.1071/WF07148

Knapp, P. A. (1996). Cheatgrass (*Bromus tectorum* L) dominance in the Great Basin desert. History, persistence, and influences to human activities. *Global Environmental Change*, *6*(1), 37–52. https://doi.org/10.1016/0959-3780(95)00112-3

Link, S. O., Keeler, C. W., Hill, R. W., & Hagen, E. (2006). *Bromus tectorum* cover mapping and fire risk. *International Journal of Wildland Fire*, *15*(1), 113–119. https://doi.org/10.1071/WF05001

Littell, R. C., Milliken, G. A., Stroup, W. W., Wolfinger, R. D., & Schabenberger, O. (2006). *SAS for mixed models*. Cary, NC: SAS Institute, Inc.

Ludwig, J. A., Wilcox, B. P., Breshears, D. D., Tongway, D. J., & Imeson, A. C. (2005). Vegetation patches and runoff-erosion as interacting ecohydrological processes in semiarid landscapes. *Ecology*, *86*(2), 288–297. https://doi.org/10.1890/03-0569

McIver, J., & Brunson, M. (2014). Multidisciplinary, multisite evaluation of alternative sagebrush steppe restoration treatments: The SageSTEP project. *Rangeland Ecology and Management*, *67*(5), 435–439.

McIver, J., Brunson, M., Bunting, S., Chambers, J., Doescher, P., Grace, J., … Williams, J. (2014). A synopsis of short-term response to alternative restoration treatments in Sagebrush-Steppe: The SageSTEP project. *Rangeland Ecology and Management*, *67*(5), 584–598. https://doi.org/10.2111/REM-D-14-00084.1

Meyer, L. D., & Harmon, W. C. (1979). Multiple-intensity rainfall simulator for erosion research on row sideslopes. *Transactions American Society of Agricultural Engineers*, *22*(1), 100–103. https://doi.org/10.13031/2013.34973

Miller, R. E., & Rose, J. A. (1995). Historic expansion of *Juniperus occidentalis* (western juniper) in southeastern Oregon. *Great Basin Naturalist*, *55*(1), 37–45.

Miller, R. F., Bates, J. D., Svejcar, T. J., Pierson, F. B., & Eddleman, L. E. (2005). *Biology, ecology, and management of western juniper*. Oregon

State University Agricultural Experiment Station Bulletin 152. Corvallis, OR: Oregon State University, Oregon State University Agricultural Experiment Station.

Miller, R. F., Chambers, J. C., Pyke, D. A., Pierson, F. B., & Williams, C. J. (2013). A review of fire effects on vegetation and soils in the Great Basin Region: Response and ecological site characteristics. General Technical Report RMRS-GTR-308. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Miller, R. F., Knick, S. T., Pyke, D. A., Meinke, C. W., Hanser, S. E., Wisdom, M. J., & Hild, A. L. (2011). Characteristics of sagebrush habitats and limitations to long-term conservation. In S. T. Knick, & J. W. Connelly (Eds.), *Greater sage-grouse, ecology and conservation of a landscape species and its habitats, studies in Avian Biology, book 38* (pp. 145–184). Berkeley, CA, USA: University of California Press.

Miller, R. F., Ratchford, J., Roundy, B. A., Tausch, R. J., Hulet, A., & Chambers, J. (2014). response of conifer-encroached shrublands in the Great Basin to prescribed fire and mechanical treatments. *Rangeland Ecology and Management*, *67*(5), 468–481. https://doi.org/10.2111/REM-D-13-00003.1

Miller, R. F., & Rose, J. A. (1999). Fire history and western juniper encroachment in sagebrush steppe. *Journal of Range Management*, *52*(6), 550–559. https://doi.org/10.2307/4003623

Miller, R. F., Svejcar, T. J., & Rose, J. A. (2000). Impacts of western juniper on plant community composition and structure. *Journal of Range Management*, *53*(6), 574–585. https://doi.org/10.2307/4003150

Miller, R. F., & Tausch, R. J. (2001). The role of fire in juniper and pinyon woodlands: A descriptive analysis. In K. E. M. Galley, & T. P. Wilson (Eds.), *Proceedings of the invasive species workshop: The role of fire in the control and spread of invasive species, fire conference 2000: The first National Congress on fire ecology, prevention, and management* (pp. 15–30). Tallahassee, FL, USA: Tall Timbers Research Station.

Miller, R. F., Tausch, R. J., McArthur, E. D., Johnson, D. D., & Sanderson, S. C. (2008), Age structure and expansion of pinon-juniper woodlands: A regional perspective in the Intermountain West. Research Paper RMRS-RP-69. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Miller, R. F., & Wigand, P. E. (1994). Holocene changes in semiarid pinyon-juniper woodlands. Response to climate, fire, and human activities in the US Great Basin. *Bioscience*, *44*(7), 465–474. https://doi.org/10.2307/1312298

Mueller-Dombois, D., & Ellenberg, H. (1974). *Aims and methods of vegetation ecology*. New York, NY: John Wiley and Sons.

Nouwakpo, S. K., Weltz, M. A., McGwire, K. C., Williams, C. J., Al-Hamdan, O. Z., & Green, C. H. M. (2017). Insight into sediment transport processes on saline rangeland hillslopes using three-dimensional soil microtopography changes. *Earth Surface Processes and Landforms*, *42*, 681–696. https://doi.org/10.1002/esp.4013

Nouwakpo, S. K., Williams, C. J., Pierson, F. B., Weltz, M. A., Arslan, A., & Al-Hamdan, O. Z. (2019). 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. *Catena*, *xx*, xx–xx. In Review

NRCS (Natural Resources Conservation Service). (2006). *Soil Survey Geographic (SSURGO) database for Tooele Area, Utah—Tooele County and Parts of Box Elder, Davis, and Juab Counties, Utah, White Pine and Elko Counties*. US Department of Agriculture, Natural Resources Conservation Service, Fort Worth, TX, USA: Nevada.

NRCS (Natural Resources Conservation Service). (2007). *Soil Survey Geographic (SSURGO) database for Western White Pine County Area, Nevada, Parts of White Pine and Eureka counties*. Fort Worth, TX, USA: US Department of Agriculture, Natural Resources Conservation Service.

Paige, G. B., Stone, J. J., Smith, J. R., & Kennedy, J. R. (2004). The Walnut Gulch Rainfall Simulator: A computer-controlled variable intensity rainfall simulator. *Applied Engineering in Agriculture*, *20*, 25–31. https://doi.org/10.13031/2013.15691

Parsons, A., Robichaud, P. R., Lewis, S. A., Napper, C., & Clark, J. T. (2010), Field guide for mapping post-fire soil burn severity. General Technical Report RMRS-GTR-243. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Petersen, S. L., & Stringham, T. K. (2008). Infiltration, runoff, and sediment yield in response to western juniper encroachment in southeast Oregon. *Rangeland Ecology and Management*, *61*(1), 74–81. https://doi.org/10.2111/07-070R.1

Petersen, S. L., Stringham, T. K., & Roundy, B. A. (2009). A process-based application of state-and-transition models: A case study of western juniper (*Juniperus occidentalis*) encroachment. *Rangeland Ecology and Management*, *62*(2), 186–192. https://doi.org/10.2111/06-171.1

Pierson, F. B., Bates, J. D., Svejcar, T. J., & Hardegree, S. P. (2007). Runoff and erosion after cutting western juniper. *Rangeland Ecology and Management*, *60*(3), 285–292. https://doi.org/10.2111/1551-5028(2007)60[285:RAEACW]2.0.CO;2

Pierson, F. B., Carlson, D. H., & Spaeth, K. E. (2002). Impacts of wildfire on soil hydrological properties of steep sagebrush-steppe rangeland. *International Journal of Wildland Fire*, *11*(2), 145–151. https://doi.org/10.1071/WF02037

Pierson, F. B., Moffet, C. A., Williams, C. J., Hardegree, S. P., & Clark, P. E. (2009). Prescribed-fire effects on rill and interrill runoff and erosion in a mountainous sagebrush landscape. *Earth Surface Processes and Landforms*, *34*(2), 193–203. https://doi.org/10.1002/esp.1703

Pierson, F. B., Robichaud, P. R., Moffet, C. A., Spaeth, K. E., Hardegree, S. P., Clark, P. E., & Williams, C. J. (2008). Fire effects on rangeland hydrology and erosion in a steep sagebrush-dominated landscape. *Hydrological Processes*, *22*(16), 2916–2929. https://doi.org/10.1002/hyp.6904

Pierson, F. B. Jr., Van Vactor, S. S., Blackburn, W. H., & Wood, J. C. (1994). Incorporating small scale spatial variability into predictions of hydrologic response on sagebrush rangelands. In W. H. Blackburn, F. B. Pierson, G. E. Schuman, & R. Zartman (Eds.), *Variability in rangeland water erosion processes, Soil Science Society of America special publication 38* (pp. 23–34). Madison, WI, USA: Soil Science Society of America.

Pierson, F. B., & Williams, C. J. (2016), Ecohydrologic impacts of rangeland fire on runoff and erosion: A literature synthesis. General Technical Report RMRS-GTR-351. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Pierson, F. B., Williams, C. J., Hardegree, S. P., Clark, P. E., Kormos, P. R., & Al-Hamdan, O. Z. (2013). Hydrologic and erosion responses of sagebrush steppe following juniper encroachment, wildfire, and tree cutting. *Rangeland Ecology and Management*, *66*(3), 274–289. https://doi.org/10.2111/REM-D-12-00104.1

Pierson, F. B., Williams, C. J., Hardegree, S. P., Weltz, M. A., Stone, J. J., & Clark, P. E. (2011). Fire, plant invasions, and erosion events on western rangelands. *Rangeland Ecology and Management*, *64*(5), 439–449. https://doi.org/10.2111/REM-D-09-00147.1

Pierson, F. B., Williams, C. J., Kormos, P. R., & Al-Hamdan, O. Z. (2014). Short-term effects of tree removal on infiltration, runoff, and erosion in woodland-encroached sagebrush steppe. *Rangeland Ecology and Management*, *67*(5), 522–538. https://doi.org/10.2111/REM-D-13-00033.1

Pierson, F. B., Williams, C. J., Kormos, P. R., Al-Hamdan, O. Z., Hardegree, S. P., & Clark, P. E. (2015). Short-term impacts of tree removal on runoff and erosion from pinyon- and juniper-dominated sagebrush hillslopes. *Rangeland Ecology and Management*, *68*(5), 408–422. https://doi.org/10.1016/j.rama.2015.07.004

Pierson, F. B., Williams, C. J., Kormos, P. R., Hardegree, S. P., Clark, P. E., & Rau, B. M. (2010). Hydrologic vulnerability of sagebrush steppe following pinyon and juniper encroachment. *Rangeland Ecology and Management*, *63*(6), 614–629. https://doi.org/10.2111/REM-D-09-00148.1

Prism Climate Group. (2009). Oregon State University. Available at: Prism Climate Group. http://www.prism.oregonstate.edu, Accessed date: 23 September 2009.

Prism Climate Group. (2017). Oregon State University. Available at: Prism Climate Group. http://www.prism.oregonstate.edu, Accessed date: 14 June 2017.

Reid, K. D., Wilcox, B. P., Breshears, D. D., & MacDonald, L. (1999). Runoff and erosion in a pinon-juniper woodland: Influence of vegetation patches. *Soil Science Society of America Journal*, *63*(6), 1869–1879. https://doi.org/10.2136/sssaj1999.6361869x

Romme, W. H., Allen, C. D., Bailey, J. D., Baker, W. L., Bestelmeyer, B. T., Brown, P. M., … Weisberg, P. J. (2009). Historical and modern disturbance regimes, stand structures, and landscape dynamics in pinon-juniper vegetation of the western United States. *Rangeland Ecology and Management*, *62*(3), 203–222. https://doi.org/10.2111/08-188R1.1

Roundy, B. A., Farmer, M., Olson, J., Petersen, S., Nelson, D. R., Davis, J., & Vernon, J. (2017). Runoff and sediment response to tree control and seeding on a high soil erosion potential site in Utah: Evidence for reversal of an abiotic threshold. *Ecohydrology*, *10*(1). https://doi.org/10.1002/eco.1775

Roundy, B. A., Miller, R. F., Tausch, R. J., Young, K., Hulet, A., Rau, B., … Eggett, D. (2014). Understory cover responses to piñon–juniper treatments across tree dominance gradients in the Great Basin. *Rangeland Ecology & Management*, *67*(5), 482–494. https://doi.org/10.2111/REM-D-13-00018.1

Roundy, B. A., Young, K., Cline, N., Hulet, A., Miller, R. F., Tausch, R. J., … Rau, B. (2014). Piñon-juniper reduction increases soil water availability of the resource growth pool. *Rangeland Ecology and Management*, *67*(5), 495–505. https://doi.org/10.2111/REM-D-13-00022.1

SAS Institute Inc. (2013). *SAS System Software Version 9.4*. Cary, NC: SAS Institute Inc.

Snyder, K. A., Evers, L., Chambers, J. C., Dunham, J., Bradford, J. B., & Loik, M. E. (2019). Effects of changing climate on the hydrological cycle in cold desert ecosystems of the Great Basin and Columbia Plateau. *Rangeland Ecology and Management*, *72*, 1–12. https://doi.org/10.1016/j.rama.2018.07.007

Tausch, R. J., & Tueller, P. T. (1997). Plant succession following chaining of pinyon-juniper woodlands in eastern Nevada. *Journal of Range Management*, *30*(1), 44–48.

Thornton, P. E., Thornton, M. M., Mayer, B. W., Wilhelmi, N., Wei, Y., & Cook, R. B. (2012). *Daymet: Daily surface weather on a 1 km grid for North America, 1980–2011*. Oak Ridge National Laboratory Distributed Active Archive Center, Oak Ridge, TN, USA. Available at: http://daymet.ornl.gov, Accessed date: 14 February 2013.

Wilcox, B. P., Breshears, D. D., & Allen, C. D. (2003). Ecohydrology of a resource-conserving semiarid woodland: Effects of scale and disturbance. *Ecological Monographs*, *73*(2), 223–239. https://doi.org/10.1890/0012-9615(2003)073[0223:EOARSW]2.0.CO;2

Wilcox, B. P., Newman, B. D., Allen, C. D., Reid, K. D., Brandes, D., Pitlick, J., & Davenport, D. W. (1996). Runoff and erosion on the Pajarito Plateau: Observations from the field, paper presented at The Jemez Mountain Region: New Mexico Geological Society, Forth-Seventh Field Conference, New Mexico Geological Society, Albuquerque, NM, September 25-28, 1996.

Wilcox, B. P., Pitlick, J., Allen, C. D., & Davenport, D. W. (1996). Runoff and erosion from a rapidly eroding pinyon-juniper hillslope. In M. G. Anderson, & S. M. Brooks (Eds.), *Advances in hillslope processes* (pp. 61–71). New York, NY: John Wiley and Sons.

Wilcox, B. P., Turnbull, L., Young, M. H., Williams, C. J., Ravi, S., Seyfried, M. S., … Wainwright, J. (2012). Invasion of shrublands by exotic grasses: Ecohydrological consequences in cold versus warm deserts. *Ecohydrology*, *5*(2), 160–173. https://doi.org/10.1002/eco.247

Williams, C. J., Pierson, F. B., Al-Hamdan, O. Z., Kormos, P. R., Hardegree, S. P., & Clark, P. E. (2014). Can wildfire serve as an ecohydrologic threshold-reversal mechanism on juniper-encroached shrublands. *Ecohydrology*, *7*(2), 453–477. https://doi.org/10.1002/eco.1364

Williams, C. J., Pierson, F. B., Kormos, P. R., Al-Hamdan, O. Z., Hardegree, S. P., & Clark, P. E. (2016). Ecohydrologic response and recovery of a semi-arid shrubland over a five year period following burning. *Catena*, *144*, 163–176. https://doi.org/10.1016/j.catena.2016.05.006

Williams, C. J., Pierson, F. B., Kormos, P. R., Al-Hamdan, O. Z., Nouwakpo, S. K., & Weltz, M. A. (2019). Vegetation, hydrologic, and erosion responses of sagebrush steppe 9 years following mechanical tree removal. *Rangeland Ecology and Management*, *72*, 47–68. https://doi.org/10.1016/j.rama.2018.07.004

Williams, C. J., Pierson, F. B., Nouwakpo, S. K., Al-Hamdan, O. Z., Kormos, P. R., & Weltz, M. A. (2019). Effectiveness of prescribed fire to re-establish sagebrush steppe vegetation and ecohydrologic function on woodland-encroached sagebrush rangelands, Great Basin, USA: Part I: Vegetation, hydrology, and erosion responses. *Catena*, *xx*, xx–xx. https://doi.org/10.1016/j.catena.2018.02.027

Williams, C. J., Pierson, F. B., Robichaud, P. R., Al-Hamdan, O. Z., Boll, J., & Strand, E. K. (2016). Structural and functional connectivity as a driver of hillslope erosion following disturbance. *International Journal of Wildland Fire*, *25*(3), 306–321. https://doi.org/10.1071/WF14114

Williams, C. J., Pierson, F. B., Robichaud, P. R., & Boll, J. (2014). Hydrologic and erosion responses to wildfire along the rangeland-xeric forest continuum in the western US: A review and model of hydrologic vulnerability. *International Journal of Wildland Fire*, *23*(2), 155–172. https://doi.org/10.1071/WF12161

Williams, C. J., Pierson, F. B., Spaeth, K. E., Brown, J. R., Al-Hamdan, O. Z., Weltz, M. A., … Nichols, M. H. (2016). Incorporating hydrologic data and ecohydrologic relationships into ecological site descriptions. *Rangeland Ecology & Management*, *69*(1), 4–19. https://doi.org/10.1016/j.rama.2015.10.001

Williams, R. E., Roundy, B. A., Hulet, A., Miller, R. F., Tausch, R. J., Chambers, J. C., … Eggett, D. (2017). Pretreatment tree dominance and conifer removal treatments affect plant succession in sagebrush communities. *Rangeland Ecology and Management*, *70*(6), 759–773. https://doi.org/10.1016/j.rama.2017.05.007

WRCC (Western Regional Climate Center). (2009). Western US climate historical summaries (individual stations). Available at: http://www.wrcc.dri.edu/Climsum.html, Accessed date: 23 September 2009.

Wright, H. A., & Bailey, A. W. (1982). *Fire ecology United States and Canada* (p. 501). New York: John Wiley and Sons.

Ziegenhagen, L. L., & Miller, R. F. (2009). Postfire recovery of two shrubs in the interiors of large burns in the Intermountain West, USA. *Western North American Naturalist*, *69*(2), 195–205. https://doi.org/10.3398/064.069.0208

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