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Short-Term Effects of Tree Removal on Infiltration, Runoff, and Erosion in Woodland-Encroached Sagebrush Steppe

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Abstract

Land owners and managers across the western United States are increasingly searching for methods to evaluate and mitigate the effects of woodland encroachment on sagebrush steppe ecosystems. We used small-plot scale (0.5 m^2) rainfall simulations and measures of vegetation, ground cover, and soils to investigate woodland response to tree removal (prescribed fire and mastication) at two late-succession woodlands. We also evaluated the effects of burning on soil water repellency and effectiveness of aggregate stability indices to detect changes in erosion potential. Plots were located in interspaces between tree and shrub canopies and on undercanopy tree and shrub microsites. Erosion from untreated interspaces in the two woodlands differed more than 6-fold, and erosion responses to prescribed burning differed by woodland site. High-intensity rainfall (102 $mm \cdot h^{-1}$) on the less erodible woodland generated amplified runoff and erosion from tree microsites postfire, but erosion (45–75 $g \cdot m^{-2}$) was minor relative to the 3–13-fold fire-induced increase in erosion on tree microsites at the highly erodible site (240–295 $g \cdot m^{-2}$). Burning the highly erodible woodland also generated a 7-fold increase in erosion from shrub microsites (220–230 $g \cdot m^{-2}$) and 280–350 $g \cdot m^{-2}$ erosion from interspaces. High levels of runoff (40–45 mm) and soil erosion (230–275 $g \cdot m^{-2}$) on unburned interspaces at the more erodible site were reduced 4–5-fold (10 mm and 50 $g \cdot m^{-2}$) by masticated tree material. The results demonstrate that similarly degraded conditions at woodland-encroached sites may elicit differing hydrologic and erosion responses to treatment and that treatment decisions should consider inherent site-specific erodibility when evaluating tree-removal alternatives. Strong soil water repellency was detected from 0 cm to 3 cm soil depth underneath unburned tree canopies at both woodlands and its strength was not altered by burning. However, fire removal of litter exacerbated repellency effects on infiltration, runoff generation, and erosion. The aggregate stability index method detected differences in relative soil stability between areas underneath trees and in the intercanopy at both sites, but failed to provide any indication of between-site differences in erodibility or the effects of burning on soil erosion potential.

Key Words: aggregate stability, hydrophobicity, juniper, piñon, prescribed fire, rangeland, restoration, soil water repellency, tree mastication

INTRODUCTION

Ecological restoration of woodland-encroached sagebrush steppe is a primary concern for land owners and management agencies in the western United States. Piñon (*Pinus* spp.) and juniper (*Juniperus* spp.) woodlands now occupy approximately 18 million ha of rangeland in the Intermountain West (Miller and Tausch 2001), much of which was historically sagebrush steppe (Davies et al. 2011; Miller et al. 2011). Range expansion of piñon and juniper conifers in the western United States has been attributed to multiple exogenous forces including climate variability, land use, decreased fire frequency, and CO_2 fertilization (Miller and Wigand 1994; Miller and Rose 1995; Knapp and Soule 1996; Miller and Tausch 2001; Miller et al. 2005, 2008; Romme et al. 2009). The ecological impacts of woodland encroachment vary across the diverse domain in which piñon and juniper have encroached, but include decreased shrub and herbaceous cover; reduced habitat for key sagebrush obligate fauna; increased bare ground, surface runoff, and soil erosion; and a decline in ecosystem productivity and goods and services (Connelly et al. 2000; Miller et al. 2000; Aldrich et al. 2005; Miller et al. 2005; Pierson et al. 2007, 2010; Davies et al. 2011; Miller et al. 2011). Postencroachment restoration strategies commonly aim to recruit sagebrush vegetation and thereby improve site resistance and resilience to woodland encroachment (Miller et al. 2005; Davies et al. 2011; Williams et al. 2014). Resistance refers to the persistence of abiotic and biotic characteristics of a site that dictate community-sustaining ecological processes whereas resilience refers to the recovery of these attributes following disturbance (Miller et al. 2013; Chambers et al. 2014). Well vegetated sagebrush rangelands trap water and nutrient-rich soil resources (Pierson et al. 1994, 2007) that propagate plant productivity and further enhance

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ecosystem structure and function (e.g., Wilcox et al. 2003; Ludwig et al. 2005; Puigdefábregas 2005). This ecohydrologic feedback is thought to increase site resistance to plant invasions and resilience of ecosystem structure and function (Briske et al. 2008; Turnbull et al. 2012; Williams et al. 2014).

Sagebrush plant community responses to tree-removal are strongly related to the pretreatment plant community and site conditions, treatment method, the prevailing soil temperature and moisture regimes, and posttreatment weather trends (Miller et al. 2013). Woodland encroachment into sagebrush steppe occurs in three phases: 1) phase I: tree cover (<1 to 3 m height) expands, but shrubs and herbaceous species remain the dominant cover; 2) phase II: tree cover increases to 10–50%, shrub and herbaceous cover decline, and trees influence key ecological processes; and 3) phase III: tree cover stabilizes, becomes the dominant cover type ($>75\%$ shrub mortality), and exerts the primary control on ecological processes (Miller et al. 2000, 2005, 2008; Johnson and Miller 2006). Sagebrush steppe restoration on late phase II to phase III woodlands (late-succession) can be difficult due to limited understory propagules and seed (Koniak and Everett 1982; Miller et al. 2000, 2005). Fires in late succession woodlands commonly burn at high severity, consume nearly 100% of sagebrush and herbaceous cover, reduce the surface soil seed bank, and cause extensive tree mortality. High severity burns that remove key native perennial species decrease resistance to weed invasions, particularly on sites with mesic-aridic soil temperature-moisture regimes ($>8°C$ annual temperature and <305 mm annual precipitation) (Young and Evans 1978; Melgoza et al. 1990; Koniak 1985; Chambers et al. 2007; Condon et al. 2011). Sagebrush does not resprout following burning and can require as long as 20 to more than 50 yr to recover postfire (Barney and Frischknecht 1974; Miller and Heyerdahl 2008; Ziegenhagen and Miller 2009). Fire surrogate treatments (e.g., mechanical tree mastication and cutting) can reduce shrub and herbaceous treatment-related mortality, but often leave residual juvenile piñon and juniper (Miller et al. 2013). Residual trees can dominate a site within as little as 15 to 60 yr following mechanical tree removal (Miller et al. 2005, 2013). Bates et al. (2006, 2007, 2011) suggested that posttreatment recruitment of desired perennial species is most likely where pretreatment perennial grass and forb densities are at least 1–2 and 5 plants per square meter respectively. The posttreatment vegetation response is also influenced by precipitation trends and can exhibit significant temporal variability due to oscillating wet/dry years regardless of pretreatment composition (West and Yorks 2002; Bates et al. 2007). Recent syntheses by Miller et al. (2005, 2013) suggest successful restoration of woodland-encroached sagebrush steppe is most likely on frigid-xeric sites and when tree-removal is applied early in the encroachment gradient (phase I–II). However, much of the woodland domain across the Intermountain West exists in aridic as well as xeric climates and is approaching late succession (Miller and Tausch 2001; Miller et al. 2008).

Knowledge regarding linkages in vegetation and hydrologic responses to the various tree removal treatments is limited. The general premise is that favorable canopy and ground cover recruitment following tree removal will reduce runoff and erosion and enhance site productivity. Amplified soil loss from late-succession woodlands occurs primarily due to interconnected runoff source areas on degraded surface soils (Davenport et al. 1998; Pierson et al. 2007, 2010, 2013; Williams et al. 2014). Poor infiltration in bare interspaces (area between tree and shrub canopies) promotes runoff generation that concentrates into high-velocity flow paths through the intercanopy. The high-velocity flow incises degraded surface soils and becomes the primary conduit for downslope movement of rainsplash- and flow-detached sediment during runoff events (Pierson et al. 2010; Al-Hamdan et al. 2012a; Williams et al. 2014). Pierson et al. (2007) found that enhanced intercanopy herbaceous cover 10 yr following tree cutting in a western juniper (*J. occidentalis* Hook.) woodland significantly reduced runoff generation and soil erosion from simulated rainfall. The study measured negligible soil loss from simulated storms (55 $mm \cdot h^{-1}$, 60 min, 32.5 m^2 plots) in well-vegetated intercanopy areas of the cut woodland and 118 $g \cdot m^{-2}$ soil erosion from simulations in the uncut woodland. Overland flow simulations in the study produced 15-fold more erosion from the uncut than cut site. Pierson et al. (2007) attributed the higher rates of soil loss at the uncut woodland to formation of concentrated flow within the bare intercanopy. Cline et al. (2010) found placement of masticated tree material in bare interspaces of a Utah juniper (*J. osteosperma* [Torr.] Little) woodland improved infiltration of artificial rainfall (102 $mm \cdot h^{-1}$, 45 min, 0.5 m^2 plots) by 3-fold and resulted in an 8-fold decrease in soil erosion. Williams et al. (2014) found burning generated a 35-fold increase in erosion from simulated high-intensity rainfall (102 $mm \cdot h^{-1}$, 45 min, 0.5 m^2 plots) in tree canopy areas of a western juniper woodland 1 yr postfire. However, runoff from a lower intensity simulated storm (64 $mm \cdot h^{-1}$, 45 min, 0.5 m^2 plots) and erosion from overland flow simulations (15–45 $L \cdot min^{-1}$, 8 min) were both significantly reduced (2- to nearly 15-fold) 2 yr following burning of intercanopy areas at the study site. The intercanopy represented approximately 74% of the study area. Williams et al. (2014) attributed the improved intercanopy hydrologic function to fire-induced increases in herbaceous vegetation and suggested that burning may provide an ecohydrologic restoration pathway for woodland-encroached sagebrush steppe where fire promotes intercanopy herbaceous production.

Rangeland managers and policymakers are increasingly relying on rapid field assessment protocols, ecological (e.g., state-and-transition models) models, and predictive technologies to prioritize and evaluate the need for restoration treatments, as well as to quantify posttreatment improvements in rangeland health (Pyke et al. 2002; Briske et al. 2008; Weltz et al. 2008; Petersen et al. 2009; Herrick et al. 2010). The quality of these approaches depends in part on knowledge of key indicator variables to measure and the ability of selected conceptual and quantitative models to accurately predict ecosystem processes of interest. Rangeland management agencies and researchers in the United States have sought to improve and standardize protocols for assessing rangeland health (Pyke et al. 2002; Herrick et al. 2010) and now include physical process-based ecological information in conceptual and quantitative models (e.g., Petersen et al. 2009; Nearing et al. 2011; Al-Hamdan et al. 2012b). Although these efforts have advanced assessment approaches, identification of indicator variables is often undertaken without well-replicated quantification of the processes that they are inferred to drive. This is particularly true relative to woodland encroachment and evaluation of tree-removal restoration treatments. The magnitude of vegetation change and soil loss following piñon and juniper encroachment, as well as tree removal, can vary substantially with ecological site attributes (Wilcox et al. 1996; Davenport et al. 1998; Wilcox et al. 2003; Pierson et al. 2010; Miller et al. 2013; Pierson et al. 2013; Williams et al. 2014). Quantitative data are needed across a wide range of ecological sites in order to advance ecological process understanding and model developments (Al-Hamdan et al. 2012a, 2012b). Furthermore, determination of key indicator variables for rangeland health protocols merits additional study. For example, the validity of rapidly acquired aggregate stability measures (i.e., Herrick et al. 2001, 2005) to represent soil erosion potential has rarely been evaluated for woodland sites, and naturally occurring soil water repellency and its influence on vegetation and hydrologic responses to tree-removal treatments have received only minor attention in the literature (Rau et al. 2005; Pierson et al. 2010, 2011; Madsen et al. 2011, 2012; Pierson et al. 2013; Williams et al. 2014).

This study uses small-plot scale (0.5 m^2) rainfall simulations and measures of vegetation, ground cover, and soils to evaluate woodland response to tree removal (by prescribed fire and mastication) at two late-succession woodland sites in the Great Basin, USA. We address three basic research questions: 1) What are the short-term (1–2 yr posttreatment) impacts of prescribed fire and tree mastication on small-plot scale vegetation, soils, and hydrologic and erosion responses? 2) Does burning enhance the effects of soil water repellency on infiltration and runoff generation in woodlands? and 3) How well do rapidly acquired measures of aggregate stability accurately depict soil erosion potential following tree removal? The study results advance process level understanding of the vegetation, hydrologic, and erosion responses to tree-removal on woodland-encroached sagebrush rangelands and provide quantitative data for improving conceptual models and predictive tools (e.g., Petersen et al. 2009; Wei et al. 2009; Herrick et al. 2010; Nearing et al. 2011; Al-Hamdan et al. 2012a, 2012b). This research is part of the larger Sagebrush Steppe Treatment Evaluation Project (SageSTEP, www.sagestep.org) aimed at investigating the ecological impacts of invasive species and woodland encroachment into sagebrush steppe ecosystems in the Great Basin and the effects of various sagebrush steppe restoration methods including tree removal (McIver et al. 2010; McIver and Brunson 2014).

METHODS

Study Sites and Treatments

A single leaf piñon-Utah juniper site (*P. monophylla* Torr. and Frém, *J. osteosperma* [Torr.] Little) (Marking Corral, Nevada, USA) and a Utah juniper site (Onaqui, Utah, USA) were selected for investigation within the SageSTEP study network (McIver and Brunson 2014). The Marking Corral site (lat 39°27′17″N, long 115°06′51″W) is located in the Egan Range, approximately 27 km northwest of Ely, Nevada. The Onaqui site (lat 40°12′42″N, long 112°28′24″W) is located in the Onaqui Mountains, 76 km southwest of Salt Lake City, Utah. Both sites are managed by the Bureau of Land Management (BLM) for grazing use but have been excluded from grazing since autumn 2005. The study sites were subject of previous hydrologic research by Pierson et al. (2010) and Cline et al. (2010). Pierson et al. (2010) evaluated runoff and erosion from the study sites in 2006 prior to tree removal. Cline et al. (2010) evaluated the impacts of tree mastication on compaction, runoff, and erosion at Onaqui in 2007. This study expands the previous research through additional data presentation and evaluation of linkages in changes to vegetation, soils, and hydrologic and erosion function following tree removal.

Detailed characteristics for the study sites prior to tree-removal treatments are provided in Table 1, as reported by Pierson et al. (2010). Precipitation in years 2006, 2007, and 2008 was approximately 322 mm, 269 mm, and 229 mm at Marking Corral and 527 mm, 379 mm, and 381 mm at Onaqui (Thornton et al. 2012). Annual precipitation during the study period was on average 71% and 92% of mean annual estimates for Marking Corral and Onaqui, respectively (Table 1). Soil temperature-moisture regimes for the sites are at the fringe between mesic-aridic and frigid-xeric classifications (McIver and Brunson 2014). Both sites are in late phase II to early phase III woodland encroachment and historically consisted of sagebrush steppe vegetation (Pierson et al. 2010). More than 70% of the area at each site is degraded intercanopy (Table 1). Prior to tree-removal treatments, litter mounds (coppices) underneath trees extended, on average, 2.5 m and 2.2 m from tree bases at Marking Corral and Onaqui, respectively. Pretreatment litter mass underneath tree canopies averaged 17.4 $kg \cdot m^{-2}$ at Marking Corral and 14.3 $kg \cdot m^{-2}$ at Onaqui (Pierson et al. 2010). Most shrub coppices pretreatment did not exceed the 0.5 m^2 scale of small plots used in this study. Pretreatment shrub cover in the intercanopy was 21% at Marking Corral and 5% at Onaqui (Table 1).

Tree-removal treatments were administered by the BLM at both study sites. Prescribed fire was applied at the Marking Corral site in August 2006, and prescribed fire and mastication treatments were applied at Onaqui in September 2006. Burn severity was not quantified, but site conditions after the burns were consistent with those of a low to moderate severity wildfire (Parsons et al. 2010). The prescribed fire reduced litter cover underneath trees at both sites from nearly 100% prefire to approximately 50% immediately postfire and reduced litter cover under shrubs from 65% prefire to 10% immediately postfire at Marking Corral and from 35% prefire to 20% immediately postfire at Onaqui. Burned shrub skeletons were present at both sites. Individual tree canopy scorch averaged 50–75% at Marking Corral and 75–99% at Onaqui. Tree mastication at Onaqui was conducted using a rubber-tired Tigercat M726E Mulcher (see Cline et al. 2010). The mastication treatment uniformly removed overstory tree cover in the respective treatment area, but resulted in a patchy ground cover of masticated material or mulch. Posttreatment, bare ground (bare soil and rock) in the mastication area was approximately 30%, and mulch cover was 40%, with an average depth of 56 mm where it occurred (Cline et al. 2010). None of the study domain was seeded in this study.

Experimental Design

A suite of small-plot scale (0.7 m$\times$0.7 m) vegetation, soil, hydrology, and erosion measurements were collected in

Table 1. Site descriptions for the Marking Corral and Onaqui study sites immediately prior to prescribed fire and tree mastication treatments. Data from Pierson et al. (2010).

Site characteristic	Marking Corral, Nevada	Onaqui, Utah
Woodland community	Single-leaf piñon[1]/Utah juniper[2]	Utah juniper[2]
Elevation (m)	2 250	1 720
Mean annual precipitation (mm)	382[3]	468[3]
Mean annual air temperature (°C)	7.2[4]	7.5[5]
Slope (%)	10–15	10–15
Parent rock	Andesite and rhyolite[6]	Sandstone and limestone[7]
Soil association	Sequra-Upatad-Cropper[6]	Borvant[7]
Soil surface texture	Sandy loam, 66% sand, 30% silt, 4% clay	Sandy loam, 56% sand, 37% silt, 7% clay
Soil profile texture	Gravelly clay to clay loam[6]	Gravelly loam[7]
Depth to bedrock (m)	0.4–0.5[6]	1.0–1.5[7]
Depth to restrictive layer (m)	0.4–0.5[6]	0.3–0.5[7]
Tree canopy cover (%)	15,[1] 10[2]	26[2]
Trees per hectare	329,[1] 150[2]	476[2]
Mean tree height (m)	2.3,[1] 2.4[2]	2.4[2]
Dead shrubs per hectare	2 065	957
Intercanopy shrub canopy cover (%)	20.9	5.2
Intercanopy herbaceous canopy cover (%)	13.1[8]	10.6[8]
Intercanopy bare soil and rock cover (%)	63.9	79.3
Common understory plants	*Artemisia tridentata* Nutt. ssp. *wyomingensis* Beetle & Young; *Artemisia nova* A. Nelson; *Purshia* spp.; *Poa secunda* J. Presl; *Pseudoroegneria spicata* (Pursh) A. Löve; and various forbs	

[1]*Pinus monophylla* Torr. & Frém.
[2]*Juniperus osteosperma* [Torr.] Little.
[3]Estimated for years 1980–2011 (Thornton et al. 2012). Pierson et al. (2010) estimate (351 mm Marking Corral, 345 mm Onaqui) was based on data from Prism Group (2009) for years 1971–2000.
[4]Western Regional Climate Center (WRCC), Station 264199-2, Kimberly, Nevada (WRCC 2009).
[5]WRCC, Station 424362-3, Johnson Pass, Utah (WRCC 2009).
[6]Natural Resources Conservation Service (NRCS) 2007.
[7]NRCS 2006.
[8]Intercanopy grass and forb canopy cover.

untreated and tree-removed treatment areas at both study sites 1 (year 1: summer 2007) and 2 (year 2: summer 2008) yr posttreatment. Plot locations were selected by a stratified random approach. Plots were randomly selected within interspaces between tree and shrub canopies and for individual tree and shrub coppice locations. Tree plots were located such that the entire 0.5-m^2 plot was within a tree coppice mound. Shrub plots were centered on a shrub coppice mound. Plot installation methods are described in detail in Pierson et al. (2010). Small plots at Marking Corral were installed in 2006 and received rainfall simulation in summer 2006, prior to prescribed fire (Pierson et al. 2010). Small plots for burned, unburned, and mulch-free (interspace only) treatments at Onaqui were also installed in 2006 and likewise received rainfall simulation in summer 2006 prior to the prescribed burn and tree mastication applications (Pierson et al. 2010). Mulch-free plots at Onaqui were installed in interspaces within the area subsequently treated by mastication, but did not receive any mulch during the mastication process. In summer 2007, 10 additional small plots were installed on mulch-covered interspaces within the mastication treatment area at Onaqui. These plots are designated as the "mulch" treatment and did not receive artificial rainfall prior to this study. For this study, rainfall simulations were conducted on unburned, burned, mulch-free, and mulch treatments in year 1 and on burned and unburned treatments in year 2. The number of small plots for each year × treatment × microsite combination is shown in Table 2 for Marking Corral and Table 3 for Onaqui.

Vegetation and Soils Characterization

Ground and canopy cover by life form and surface roughness on each small plot were measured using point-frame methods. Point measurements were recorded for 15 points with 5-cm spacing, along each of 7 transects oriented parallel to the hillslope contour and spaced 10 cm apart. Percent ground and canopy cover by life form on each plot was derived from the frequency of respective cover hits divided by the total number of sample points per plot (105). The relative ground-surface height at each sample point was calculated as the distance between the point-frame level line and the ground surface. Ground surface roughness on each plot was estimated as the arithmetic average of the standard deviations of the ground surface heights for each of the seven transects sampled on the respective plot. The depth of litter on the ground surface was measured adjacent to each small plot at four evenly spaced points along each of the two small plot borders oriented perpendicular to the hillslope contour.

The strength of soil water repellency was measured in situ over 0–5-cm soil depth before rainfall simulation immediately adjacent to each plot using the water drop penetration time (WDPT) procedure (DeBano 1981). The time required for water drop infiltration (up to 300 s) was recorded for eight

Table 2. Average surface roughness, aggregate stability, and cover variables measured on burned and unburned rainfall simulation plots (0.5 m^2) at Marking Corral 1 (year 1) and 2 (year 2) yr following burning. Means within a row by study year (year 1 or year 2) followed by a different lowercase letter are significantly different ($P < 0.05$).

Marking Corral	Year 1						Year 2					
	Burned			Unburned			Burned			Unburned		
Plot characteristic	Interspace	Tree coppice	Shrub coppice	Interspace	Tree coppice	Shrub coppice	Interspace	Tree coppice	Shrub coppice	Interspace	Tree coppice	Shrub coppice
Surface roughness (mm)	8 a	12 ab	8 a	9 a	12 ab	14 b	8 a	10 ab	8 a	8 a	10 ab	14 b
Aggregate stability class (1–6)[1]	2 a	5 b	4 b	2 a	5 b	2 a	1 ab	5 c	1 a	2 b	5 c	2 b
Total canopy cover (%)[2]	30.0 b	3.5 a	53.1 c	33.3 bc	6.5 a	92.8 d	23.9 a	11.1 a	43.6 b	17.1 a	9.8 a	76.2 c
Total herbaceous canopy cover (%)[3]	28.1 b	3.3 a	42.9 b	30.6 b	3.5 a	28.0 b	9.5 bc	7.1 ab	20.2 c	7.4 abc	0.2 a	8.1 abc
Shrub canopy cover (%)	0.1 a	0.0 a	1.2 a	0.3 a	2.6 a	58.9 b	0.0 a	0.0 a	0.0 a	0.0 a	0.0 a	14.3 b
Grass canopy cover (%)	12.1 b	0.5 a	8.6 ab	27.9 c	3.2 ab	24.8 c	9.4 b	0.1 a	2.1 a	6.9 ab	0.0 a	3.3 ab
Litter cover (%)	11.2 a	74.7 b	33.6 a	24.6 a	88.1 b	79.3 b	7.5 a	67.1 b	29.1 a	6.8 a	96.7 c	77.2 bc
Rock cover (%)	38.4 c	2.7 ab	10.0 b	28.6 c	0.4 a	4.5 ab	51.2 d	10.4 ab	21.8 bc	36.4 cd	0.8 a	6.3 abc
Total ground cover (%)[4]	51.4 a	90.5 b	51.3 a	55.9 a	99.9 c	88.1 b	63.8 ab	79.8 b	56.1 a	47.1 a	98.2 c	88.2 bc
Bare soil (%)	48.6 c	9.5 b	48.7 c	44.1 c	0.1 a	11.9 b	36.2 bc	20.2 b	43.9 bc	52.9 c	1.8 a	11.8 ab
Litter depth (mm)	< 1 a	23 b	2 a	< 1 a	40 c	2 a	< 1 a	17 b	< 1 a	< 1 a	38 c	< 1 a
Ash (%)	0.0 a	12.3 b	6.0 a	-	-	-	0.0 a	1.4 a	0.0 a	-	-	-
No. of plots	8	8	4	7	8	5	8	8	4	4	4	2

[1]Stability classes: (1) less than 10% stable aggregates, 50% structural integrity lost within 5 s; (2) less than 10% stable aggregates, 50% structural integrity lost within 5–30 s; (3) less than 10% stable aggregates, 50% structural integrity lost within 30–300 s; (4) 10–25% stable aggregates; (5) 25–75% stable aggregates; (6) 75–100% stable aggregates (Herrick et al. 2001, 2005).
[2]Excludes tree canopy removed for rainfall simulation.
[3]Grass and forb canopy cover.
[4]Includes ash, cryptogam, litter, live and dead basal plant, rock, and woody dead cover.

water drops (3-cm spacing) applied at the mineral soil surface (ash, litter, and mulch removed). Following this procedure, one centimeter of soil was excavated immediately underneath the previously sampled area and the WDPT procedure was repeated with eight additional drops. WDPT sample iterations continued until a depth of 5 cm was reached. The mean strength of soil water repellency at each 1-cm depth for each plot was recorded as the average of the eight WDPT (s)

Table 3. Average surface roughness, aggregate stability, and cover variables measured on burned, unburned, mulch, and mulch-free rainfall simulation plots (0.5 m^2) at Onaqui 1 (year 1) and 2 (year 2) yr following treatments. Means within a row by study year (year 1 or year 2) followed by a different lowercase letter are significantly different ($P < 0.05$).

Onaqui	Year 1								Year 2					
	Burned			Unburned			Mulch	Mulch-free	Burned			Unburned		
Plot characteristic	Interspace	Tree coppice	Shrub coppice	Interspace	Tree coppice	Shrub coppice	Interspace	Interspace	Interspace	Tree coppice	Shrub coppice	Interspace	Tree coppice	Shrub coppice
Surface roughness (mm)	9 a	12 a	11 a	11 a	12 a	13 a	12 a	12 a	8 a	12 b	9 ab	11 b	9 ab	12 b
Aggregate stability class (1–6)[1]	2 a	6 b	3 ab	2 a	5 b	3 a	2 a	2 a	2 a	6 b	3 a	2 a	5 b	3 a
Total canopy cover (%)[2]	6.6 a	1.7 a	27.8 b	19.4 ab	21.7 ab	68.6 c	15.3 ab	27.1 b	14.3 a	3.4 a	32.2 b	11.1 a	16.0 a	58.4 c
Total herbaceous canopy cover (%)[3]	3.0 ab	1.3 a	7.6 abc	13.0 bc	17.9 c	12.7 bc	12.3 bc	22.5 c	13.4 b	2.5 a	17.0 b	9.8 ab	8.3 ab	4.4 ab
Shrub canopy cover (%)	0.0 a	0.0 a	10.1 b	0.0 a	0.0 a	50.5 c	0.0 a	0.1 a	0.1 a	0.0 a	1.0 a	0.0 a	0.0 a	49.5 b
Grass canopy cover (%)	2.7 a	1.0 a	6.9 ab	5.7 ab	17.4 b	9.8 ab	9.7 ab	17.3 b	6.1 a	0.6 a	9.3 a	2.9 a	7.4 a	1.9 a
Litter cover (%)	4.0 a	80.4 b	21.9 a	6.0 a	80.6 b	57.8 b	73.5 b	13.5 a	9.7 a	72.0 c	28.8 b	5.6 a	81.1 c	66.3 c
Rock cover (%)	55.5 d	2.9 ab	30.1 c	38.1 cd	1.4 a	20.5 bc	8.4 ab	38.6 cd	53.2 c	5.3 a	28.3 b	60.8 c	3.2 a	18.8 ab
Total ground cover (%)[4]	61.8 ab	90.9 c	56.2 ab	47.6 a	93.0 c	81.9 bc	83.5 c	56.2 a	66.4 a	84.3 bc	61.4 a	74.3 ab	94.9 c	88.1 bc
Bare soil (%)	38.2 bc	9.1 a	43.8 c	52.4 c	7.0 a	18.1 ab	16.5 a	43.8 c	33.6 c	15.7 ab	38.6 c	25.7 bc	5.1 a	11.9 ab
Litter depth (mm)	< 1 a	19 b	1 a	< 1 a	18 b	2 a	22 b	1 a	< 1 a	12 b	< 1 a	< 1 a	13 b	3 a
Ash (%)	0.8 a	7.4 b	0.8 a	-	-	-	-	-	0.1 a	6.9 b	0.0 a	-	-	-
No. of plots	10	5	5	3	4	3	10	10	10	5	5	3	4	3

[1]Stability classes: (1) less than 10% stable aggregates, 50% structural integrity lost within 5 s; (2) less than 10% stable aggregates, 50% structural integrity lost within 5–30 s; (3) less than 10% stable aggregates, 50% structural integrity lost within 30–300 s; (4) 10–25% stable aggregates; (5) 25–75% stable aggregates; (6) 75–100% stable aggregates (Herrick et al. 2001, 2005).
[2]Excludes tree canopy removed for rainfall simulation.
[3]Grass and forb canopy cover.
[4]Includes ash, cryptogam, litter, live and dead basal plant, rock, and woody dead cover.

samples. The strength of soil water repellency was assigned as slightly water repellent when WDPT ranged from 5 to 60 s and strongly water repellent when WDPT exceeded 60 s (Bisdom et al. 1993). Soil samples were obtained from 0–5-cm depth adjacent to WDPT measurements and were analyzed gravimetrically in the lab for soil water content.

An index of surface soil aggregate stability for each plot was determined using the aggregate stability test described by Herrick et al. (2001, 2005). Six surface soil aggregates/peds (2–3 mm thick, 6–8 mm diameter) were sampled immediately adjacent to each plot and were immersed in water for 5 min. Aggregates persisting after immersion for 5 min were subjected to a sequence of five additional 1-s immersions. Each aggregate was assigned a stability rating as follows: 1) 50% structural integrity lost within 5 s of initial immersion; 2) 50% structural integrity lost within 5–30 s of initial immersion; 3) 50% structural integrity lost within 30–300 s of initial immersion or less than 10% soil remaining after five 1-s immersions; 4) 10–25% soil remains after five 1-s immersions; 5) 25–75% soil remains after five 1-s immersions; 6) 75–100% soil remains after five 1-s immersions. A mean stability rating for each plot was assigned as the average of the six ped ratings for the respective plot.

Rainfall Simulation Experiments

Rainfall simulations were conducted using instrumentation and methods in Pierson et al. (2010). Rainfall was applied to each plot at target intensities 64 $mm \cdot h^{-1}$ and 102 $mm \cdot h^{-1}$ for 45 min each using a portable oscillating-arm rainfall simulator fitted with 80–100 Vee-jet nozzles. Standard deviations for applied rates across all simulations during the study were within 1 to 2 $mm \cdot h^{-1}$ of the target intensities. The dry run was conducted on uniform dry antecedent-soil moisture conditions, and the wet run was applied approximately 30 min following the dry run. The dry run intensity applied for 5-, 10-, and 15-min durations is equivalent to respective local storm return intervals of 7, 15, and 25 yr, and the wet run intensity over the same durations is equivalent to local storm return intervals of 25, 60, and 120 yr (Bonnin et al. 2006). Timed samples of plot runoff were collected over 1-min to 3-min intervals throughout each 45-min rainfall simulation and were analyzed in the laboratory for runoff volume and sediment concentration. Rainfall simulators, raindrop characteristics, simulator calibration procedures, and runoff sample processing are described in detail by Pierson et al. (2010).

A set of hydrologic response variables was derived for each rainfall simulation. The mean runoff rate ($mm \cdot h^{-1}$) was calculated for each runoff sample interval as the cumulative runoff divided by the interval time. Cumulative runoff (mm) from each 45 min simulation was calculated as the integration of runoff rates over the total time of runoff. The runoff-to-rainfall ratio was derived by dividing cumulative runoff by total rainfall applied. Mean infiltration and erosion variables were derived for plots that generated runoff. An average infiltration rate ($mm \cdot h^{-1}$) for each sample interval was calculated as the difference between applied rainfall and measured runoff divided by duration of the sample interval. Cumulative sediment yield ($g \cdot m^{-2}$) was calculated as the integrated sum of sediment collected during runoff and was extrapolated to plot unit area by dividing cumulative sediment by total plot area. The sediment-to-runoff ratio ($g \cdot m^{-2} \cdot mm^{-1}$), a variable closely related to soil erodibility, was obtained by dividing cumulative sediment yield by cumulative runoff.

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

Statistical Analyses

All statistical analyses were conducted using SAS software, version 9.1.3 (SAS Institute Inc. 2007). Temporal (between-years) variability in canopy and ground cover, soil, and all runoff, infiltration, and sediment variables for unburned and burned conditions at a given site was analyzed using a repeated measures split-plot mixed model with two whole-plot or treatment factors: unburned and burned. Microsite was the subplot factor and had three levels: tree coppice, shrub coppice, and interspace. A compound symmetry covariance structure was used given there were only two sample dates (year 1 and year 2) for burned and unburned treatments (Littell et al. 2006). Normality and homogeneity were tested using the Shapiro-Wilk test and Levene's test (SAS Institute 2007) and deviance from normality was addressed by data transformation. Where necessary, arcsine-square root transformations were used to normalize proportion data (e.g., canopy cover, percentage wet) and logarithmic transformations were used to normalize WDPT, infiltration, runoff, and erosion data. Mean separation was determined using the LSMEANS procedure with Tukey's adjustment. Significant effects were evaluated at the $P < 0.05$ level. Significant temporal variability was observed for some variables in the unburned treatment at both sites. Therefore, tabular and graphical statistical presentations were restricted to within-year analyses at a site, and all reported comparisons are between treatments within-year (year 1 or year 2) at a site unless otherwise specified. Within-year analyses of treatment effects at Marking Corral were conducted using a split-plot mixed model with two treatment levels, unburned and burned, and three microsite levels: tree coppice, shrub coppice, and interspace. Within-year analyses of treatment effects at Onaqui were conducted using a split-plot mixed model with four treatment levels: unburned, burned, mulch-free, and mulch. The subplot factor for within-year analyses at Onaqui had three levels: tree coppice, shrub coppice, and interspace. Normality and homogeneity for within-year analyses were addressed as specified above for the between-years analyses. Backtransformed data are reported. For all analyses, plot location was designated a random effect and treatment and microsite were considered fixed effects.

RESULTS

Vegetation

Canopy and ground cover in unburned areas were consistent with those reported pretreatment (Tables 2 and 3; Pierson et al. 2010), but herbaceous cover exhibited significant temporal variability. Unburned interspace areas were degraded of cover at both sites and averaged 70–90% bare ground (bare soil and rock). Interspace canopy cover averaged 10–35%. Shrub coppice microsites were well vegetated with an average of 50% shrub canopy cover and 5–20% grass canopy cover. Forb canopy cover on shrub microsites ranged 0–5%. Litter cover underneath shrubs averaged 60–80% across both sites, but litter depths were generally less than 3 mm (Tables 2 and 3). Canopy cover underneath trees was less than 25% and was predominantly grass (0–5% at Marking Corral, 5–20% at Onaqui). The ground surface on unburned tree coppices was well protected with 80–100% litter cover. Tree litter depths averaged 39 mm at Marking Corral and 16 mm at Onaqui. Herbaceous canopy cover in unburned areas declined at both sites from year 1 to year 2 (Tables 2 and 3). Grass canopy cover declined ($P < 0.05$) 4- to 8-fold across interspace and shrub coppice microsites, respectively, at Marking Corral and 5-fold across tree and shrub coppice microsites at Onaqui from year 1 to year 2. The reason for the declines is unknown, but may be related to the below average trends in precipitation at both sites in years 1 and 2. There were no significant differences in canopy cover on unburned mulch, and mulch-free interspace plots at Onaqui (Table 3) in year 1. However, the mulch treatment provided 5- to more than 10-fold greater litter cover on mulch interspaces than was measured on interspaces in the mulch-free and unburned treatments (Table 3). Litter depths on mulch-covered interspaces averaged 22 mm and were similar to litter depths on unburned tree coppices.

The net impact of the low/moderate severity burning on small-plot scale canopy and ground cover was creation of similar cover conditions across shrub and interspace microsites. The prescribed fires primarily removed shrub and grass cover and litter (Tables 2 and 3). Burning reduced shrub canopy to 1–10% on shrub coppice microsites across both sites. Grass canopies were reduced by 2- to 3-fold on interspace and shrub microsites at Marking Corral (Table 2) and more than 10-fold on tree coppices at Onaqui (Table 3). Differences in grass canopies across burned and unburned treatments at both sites were insignificant in year 2 due to the overall between-years decline in herbaceous cover in unburned areas. Burning reduced litter cover on shrub coppices at both sites by 2- to 3-fold (to < 35%) and significant differences between burned and unburned conditions persisted from year 1 to 2 (Tables 2 and 3). Fire consumed litter cover on tree coppices, but dead needlefall from burned tree canopies replenished litter on burned tree coppices to 75–80% in year 1. Litter consumption by burning at Marking Corral resulted in an approximately 20 mm difference in tree litter depth across treatments for years 1 and 2 (Table 2). No significant differences in tree litter depth were observed for burned versus unburned conditions at Onaqui (Table 3). Ash cover was minimal in interspaces, but was 12% and 6% on tree and shrub coppice plots at Marking Corral (Table 2) and 7% on tree coppices at Onaqui (Table 3).

Surface Soils

Measured surface soil properties at both sites were more influenced by microsite attributes than the tree-removal treatments. Surface soils were strongly hydrophobic on tree coppice plots for burned and unburned conditions at both sites, but were wettable in interspaces and on shrub coppices. Burning did not induce soil water repellency and did not significantly alter natural, background soil water repellency at either site (Fig. 1). Tree-coppice soil water repellency at Marking Corral was uniform over 0–5-cm depth for burned and unburned conditions in year 1, but, in year 2, was generally strongest near the mineral soil surface across both treatments (Fig. 1A). Water repellency on tree coppices at Onaqui was strong over 0–5-cm and 0–3-cm depths for burned and unburned conditions, respectively, in year 1, 0–4-cm depth on burned plots in year 2, and 0–1-cm depth on unburned plots in year 2 (Fig. 1B). Aggregate stability indices were highest for litter and ash covered tree coppices at both sites and suggest tree coppice plots maintained good surface soil stability postfire (Tables 2 and 3). Burned and unburned shrub coppice microsites generally had higher aggregate stability indices than interspace plots, but, in most cases, the differences were not significant. Plot-scale ground-surface roughness averaged 8–14 mm across both sites, and, with few exceptions, was unaffected by tree-removal treatments (Tables 2 and 3). Gravimetric soil moisture content at the time of sampling was uniformly low (< 10%) across all microsites and treatments at both sites each year.

Hydrologic and Erosion Responses

Infiltration and runoff generation for unburned conditions were strongly influenced by microsite cover and soil attributes. Canopy and litter cover on coppice plots at both sites intercepted and stored rainfall, mitigated effects of strong soil water repellency, and promoted infiltration into the soil profile. None of the unburned shrub coppice plots generated dry-run runoff and unburned tree coppices produced dry-run runoff at Onaqui solely (Tables 4 and 5). The dry-run simulations generated runoff on 50–70% of the untreated interspace plots (Tables 4 and 5). Dry-run sediment-to-runoff ratios were low ($< 1.00\ g \cdot m^{-2} \cdot mm^{-1}$) at Marking Corral for all unburned plots (Table 4), and were significantly higher ($P < 0.05$) at Onaqui (Table 5). The measured site differences in dry-run erosion rates are consistent with pretreatment dry-run rainfall simulations conducted at the sites one year prior to the treatments (Pierson et al. 2010).

Fire effects on dry-run runoff generation and erosion were limited to tree coppice microsites at Marking Corral (Table 4) and shrub coppice microsites at Onaqui (Table 5). On average, 70% of the burned tree coppice plots at Marking Corral generated dry-run runoff in contrast to 0% for unburned conditions. We attribute the differing tree coppice responses across the treatments at Marking Corral to fire-induced reductions in litter on strongly water-repellent soils (Table 2; Fig. 1A). Approximately 80–95% of the soil profile from 0–10-cm depth was wet following the dry run on unburned tree coppices at Marking Corral (Table 4). Only 65–70% of the soil profile was wet from 0–10-cm depth on burned tree coppices at the site (Table 4). Soil water repellency was strong on both

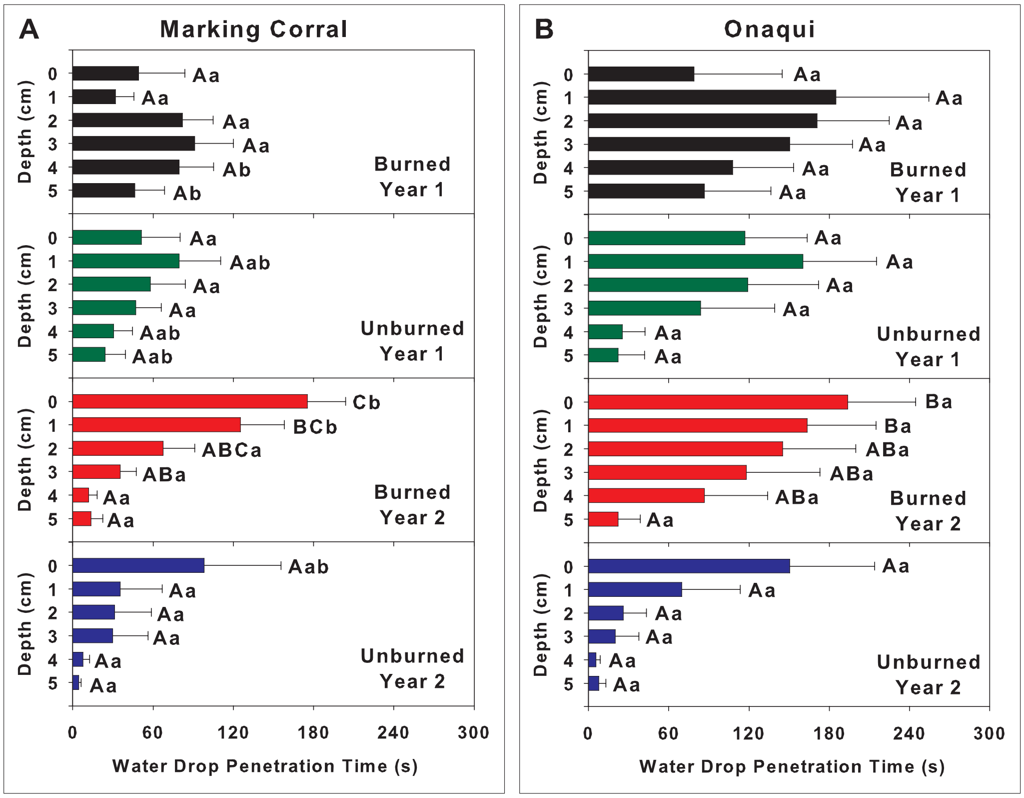

Figure 1. Water drop penetration times (WDPT, 300 s maximum) measured at 0–5 cm soil depths underneath tree canopies on burned and unburned small rainfall simulation plots (0.5 m^2) at the **A,** Marking Corral and **B,** Onaqui study sites 1 (year 1) and 2 (year 2) yr postfire. Soils were considered slightly water repellent if WDPT ranged from 5 to 60 s and strongly water repellent if WDPT exceeded 60 s (Bisdom et al. 1993). Error bars depict standard error. Site means across depths within a treatment and year combination followed by a different upper case letter are significantly different ($P < 0.05$). Site means for a specific soil depth across treatments and years followed by a lowercase letter are significantly different ($P < 0.05$).

treatments (Fig. 1A), but deeper litter on unburned tree plots likely enhanced rainfall interception and storage, inhibited runoff generation, and facilitated infiltration into water-repellent soils. Dry-run sediment yield was amplified by burning tree coppices at Marking Corral, but sediment-to-runoff ratios ($\sim$1.35 $g \cdot m^{-2} \cdot mm^{-1}$) and cumulative sediment ($\sim$20 $g \cdot m^{-2}$) were low relative to the more erodible burned tree plots at Onaqui ($\sim$4.65 $g \cdot m^{-2} \cdot mm^{-1}$ and $\sim$45 $g \cdot m^{-2}$). Burning accentuated dry-run runoff and erosion from shrub coppices at Onaqui (Table 5). In contrast to unburned plots, most of the burned shrub coppices at the site generated runoff during the dry run. Dry-run sediment-to-runoff ratios ($\sim$6.60 $g \cdot m^{-2} \cdot mm^{-1}$) and cumulative sediment ($\sim$35 $g \cdot m^{-2}$) for burned shrub coppices were similar to the same measures for degraded interspaces and burned tree coppices at the site (Table 5). Dry-run simulations in burned interspaces yielded similar runoff and erosion values as in unburned interspaces at each site, but both measures were greater at Onaqui ($P < 0.05$; Tables 4 and 5). Burned interspaces at Onaqui produced, on average, 15 mm of runoff and 62 $g \cdot m^{-2}$ soil erosion from dry-run simulations. Burned interspaces at Marking Corral produced 4 mm of runoff and less than 10 $g \cdot m^{-2}$ soil erosion from the dry-run storm.

Infiltration of the wet-run rainfall was largely controlled by the distribution of litter and the strength of soil water repellency. For unburned plots, wet-run infiltration was well correlated with percent litter cover (Fig. 2A). As with the dry-run, canopy and litter cover on unburned tree and shrub coppices was capable of reducing the water available for runoff, resulting in runoff-to-rainfall ratios of less than 25% for the wet run (Tables 4 and 5). Average infiltration of the wet-run simulations on runoff-generating unburned tree and shrub coppice microsites averaged 70 to 95 $mm \cdot h^{-1}$ (Figs. 3A and 4A and 4B). In contrast, wet-run runoff-to-rainfall ratios were 40–65% for the mostly bare burned and unburned interspaces at both sites and were higher than for any other microsite $\times$ treatment combination. Infiltration for the wet-run on interspaces ranged from 50 to 60 $mm \cdot h^{-1}$ at Marking Corral (Figs. 3A and 3B) and 35 to 50 $mm \cdot h^{-1}$ at Onaqui (Figs. 4A and 4B). Burning had no significant effect on infiltration and runoff of the wet-run on interspace microsites. In year 1, wet-run infiltration on shrub coppices was unaffected by burning at Marking Corral (Table 4) likely due to persistence of at least 30% litter cover and 20–40% herbaceous canopy cover (Table 2). Wet-run infiltration on shrub coppices was reduced 30% by burning at Onaqui (Table 5) due to more limited litter cover and low herbaceous canopy cover (Table 3). Wet-run infiltration rates for burned tree coppices at Onaqui were similar to those of unburned tree coppices in year 1, but were 30% lower than on unburned tree coppices in year 2. The year 1 similarities across treatments on tree coppices at Onaqui are likely related to the similarities in litter cover and soil water repellency through the soil profile that year. In year 2, soil water repellency was strong on burned tree coppice plots to a

Table 4. Average runoff, infiltration, sediment, and wetting depth response variables measured on burned and unburned rainfall simulation plots (0.5 m^2) at Marking Corral 1 (year 1) and 2 (year 2) yr following burning. Means within a row by study year (year 1 or year 2) followed by a different lowercase letter are significantly different ($P < 0.05$).

Marking Corral	Year 1						Year 2					
	Burned			Unburned			Burned			Unburned		
Rainfall simulation variable	Interspace	Tree coppice	Shrub coppice	Interspace	Tree coppice	Shrub coppice	Interspace	Tree coppice	Shrub coppice	Interspace	Tree coppice	Shrub coppice
Dry-run simulation (64 $mm \cdot h^{-1}$, 45 min)												
Cumulative runoff (mm)	3 ab	7 b	0 a	6 b	0 a	0 a	4 ab	9 b	0 a	6 ab	0 a	0 a
Runoff-to-rainfall ($mm \cdot h^{-1}$) $\times 100\%$	5 ab	15 b	0 a	12 b	0 a	0 a	8 ab	18 b	0 a	12 ab	0 a	0 a
Mean infiltration rate ($mm \cdot h^{-1}$)[1]	54 a	47 a	-	50 a	-	-	51 a	48 a	-	48 a	-	-
Cumulative sediment ($g \cdot m^{-2}$)[1]	6 a	17 a	-	7 a	-	-	9 a	20 a	-	10 a	-	-
Sediment/runoff ($g \cdot m^{-2} \cdot mm^{-1}$)[1]	0.95 a	1.32 a	-	0.89 a	-	-	0.85 a	1.37 a	-	0.88 a	-	-
Percent wet at 0–6 cm depth	100 b	71 a	91 b	100 b	85 ab	99 b	100 b	66 a	100 b	100 b	93 b	100 b
Percent wet at 0–10 cm depth	99 c	72 a	94 bc	99 c	83 ab	92 bc	98 b	65 a	100 b	98 b	94 b	100 b
Percent wet at 0–20 cm depth	73 a	68 a	75 a	71 a	62 a	58 a	71 a	60 a	77 a	61 a	85 a	62 a
Percent of plots with runoff	38	63	0	57	0	0	38	75	0	50	0	0
Wet-run simulation (102 $mm \cdot h^{-1}$, 45 min)												
Cumulative runoff (mm)	35 c	21 b	8 a	31 bc	0 a	3 a	35 c	22 b	5 a	41 c	0 a	0 a
Runoff-to-rainfall ($mm \cdot h^{-1}$) $\times 100\%$	46 c	28 b	10 a	41 bc	0 a	4 a	45 c	29 b	6 a	53 c	0 a	0 a
Mean infiltration rate ($mm \cdot h^{-1}$)[1]	54 a	68 a	81 ab	60 a	-	93 b	56 a	68 ab	88 b	49 a	-	-
Cumulative sediment ($g \cdot m^{-2}$)[1]	41 a	46 a	48 a	23 a	-	6 a	35 a	75 a	27 a	42 a	-	-
Sediment/runoff ($g \cdot m^{-2} \cdot mm^{-1}$)[1]	1.10 a	1.96 a	2.07 a	0.66 a	-	1.01 a	0.92 a	2.14 a	2.49 a	0.90 a	-	-
Percent of plots with runoff	100	88	50	100	0	40	100	88	50	100	0	0
No. of plots	8	8	4	7	8	5	8	8	4	4	4	2

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

depth of 4 cm versus a depth of 1 cm for unburned conditions. We therefore attribute the year 2 differences in tree coppice response to persistence of strong soil water repellency throughout the upper 4 cm of the soil profile on burned plots. Wet-run infiltration on burned tree coppices at Marking Corral averaged 60–70 $mm \cdot h^{-1}$, similar to all other burned plots. However, the infiltration rate was substantially less than that of unburned tree coppice plots which exceeded the 102 $mm \cdot h^{-1}$ simulated storm. We attribute the differences in wet-run infiltration on burned and unburned tree coppices at Marking Corral to fire-reduced litter depths (50% reduction) and rainfall storage over strongly water-repellent surface soils.

The magnitude of soil erosion across all treatment×microsite combinations was dictated by protection of the ground surface, amount of runoff available for sediment transport, and site-specific erodibility. Wet-run infiltration rates for burned and unburned interspaces were similar across both sites, but sediment discharge was near an order of magnitude greater at Onaqui than Marking Corral (Figs. 3 and 4). The lower wet-run erosion rates at Marking Corral resulted in a poor correlation between litter cover and sediment yield (Fig. 2B). In contrast, litter cover exerted significant influence on wet-run sediment yield at the more erodible Onaqui site (Fig. 2B). Wet-run sediment yield at both sites was well correlated with runoff, but the strength of the correlation was stronger for Onaqui (Fig. 2C). Burning increased wet-run erosion on tree coppice microsites at Marking Corral, but had no significant effect on wet-run erosion from shrub coppices due to the low erodibility at the site (Table 4; Figs. 3C and 3D). The dependence of wet-run erosion on litter and runoff generation at Onaqui demonstrates the importance of maintaining surface cover for interception and storage and to buffer the detachment and transport of sediment by rainsplash and overland flow on the highly erodible site. Fire reductions of litter and grass cover and increased runoff at the site postfire resulted in 220 to 350 $g \cdot m^{-2}$ wet-run soil erosion across all microsites, and the high erosion rates persisted 2 yr posttreatment. Application of tree mulch to interspaces at Onaqui significantly enhanced wet-run infiltration and reduced sediment discharge and cumulative erosion (Fig. 5; Table 5), underscoring the effect of ground cover. Only 10% of applied wet-run rainfall was transferred to runoff on mulched interspaces, and cumulative erosion from the wet-run on mulched interspaces (50 $g \cdot m^{-2}$) was 5-fold less in comparison to mulch-free interspaces (233 $g \cdot m^{-2}$) in the mastication treatment and all burned microsites ($\sim$315 $g \cdot m^{-2}$).

DISCUSSION

Woodland Response and Treatment Effectiveness

Our results from experiments in the unburned treatments are consistent with earlier studies at both woodlands and indicate that they are capable of generating substantial long-term soil loss. Tree and shrub coppice plots at both sites were stable hydrologically and overall generated only minor amounts of soil loss. However, interspace plots produced significantly higher rates of runoff and erosion than the well protected coppice microsites (Tables 4 and 5). Interspace microsites occupy approximately 60% of the total area at Marking Corral

Table 5. Average runoff, infiltration, sediment, and wetting depth response variables measured on burned, unburned, mulch, and mulch-free rainfall simulation plots (0.5 m^2) at Onaqui 1 (year 1) and 2 (year 2) yr following treatments. Means within a row by study year (year 1 or year 2) followed by a different lowercase letter are significantly different ($P < 0.05$).

Onaqui	Year 1								Year 2					
	Burned			Unburned			Mulch	Mulch-free	Burned			Unburned		
Rainfall simulation variable	Interspace	Tree coppice	Shrub coppice	Interspace	Tree coppice	Shrub coppice	Interspace	Interspace	Interspace	Tree coppice	Shrub coppice	Interspace	Tree coppice	Shrub coppice
Dry-run simulation (64 $mm \cdot h^{-1}$, 45 min)														
Cumulative runoff (mm)	12 c	5 ab	2 a	10 bc	7 bc	0 a	1 a	8 bc	16 d	6 bc	3 ab	12 cd	2 ab	0 a
Runoff-to-rainfall ($mm \cdot h^{-1}$) $\times 100\%$	25 c	11 ab	3 a	20 bc	14 bc	0 a	2 a	16 bc	34 d	12 bc	7 ab	26 cd	5 ab	0 a
Mean infiltration rate ($mm \cdot h^{-1}$)[1]	47 a	51 ab	59 b	44 a	51 ab	-	-	51 ab	42 a	51 ab	57 b	39 a	55 b	-
Cumulative sediment ($g \cdot m^{-2}$)[1]	64 a	57 a	18 a	69 a	33 a	-	-	44 a	60 b	30 a	48 ab	70 b	8 a	-
Sediment/runoff ($g \cdot m^{-2} \cdot mm^{-1}$)[1]	5.15 a	6.84 a	6.79 a	4.80 a	4.02 a	-	-	4.66 a	3.51 ab	2.43 a	6.38 b	3.72 ab	1.72 a	-
Percent wet at 0–6 cm depth	100 b	67 a	100 b	100 b	97 b	100 b	97 b	100 b	100 b	67 a	100 b	100 b	75 a	99 b
Percent wet at 0–10 cm depth	97 b	65 a	100 b	99 b	93 b	96 b	98 b	98 b	97 b	63 a	97 b	89 ab	70 a	97 b
Percent wet at 0–20 cm depth	83 a	72 a	87 a	69 a	67 a	64 a	80 a	77 a	66 a	58 a	69 a	49 a	40 a	77 a
Percent of plots with runoff	100	60	60	67	75	0	10	80	100	60	60	67	50	0
Wet-run simulation (102 $mm \cdot h^{-1}$, 45 min)														
Cumulative runoff (mm)	49 c	13 a	22 ab	41 bc	16 a	6 a	10 a	37 bc	48 d	23 bc	30 cd	45 d	9 ab	2 a
Runoff-to-rainfall ($mm \cdot h^{-1}$) $\times 100\%$	64 c	18 a	29 ab	56 bc	22 a	8 a	13 a	49 bc	63 d	31 bc	39 cd	59 d	12 ab	3 a
Mean infiltration rate ($mm \cdot h^{-1}$)[1]	37 a	76 cd	63 bc	45 ab	70 bcd	91 d	82 cd	52 b	37 a	60 b	62 b	42 ab	83 c	-
Cumulative sediment ($g \cdot m^{-2}$)[1]	351 c	294 c	220 bc	233 bc	98 ab	33 a	50 a	233 bc	280 b	242 b	230 b	274 b	19 a	-
Sediment/runoff ($g \cdot m^{-2} \cdot mm^{-1}$)[1]	7.11 bc	10.40 c	7.90 c	5.53 bc	4.65 ab	4.99 ab	3.53 a	5.79 bc	5.85 b	6.28 b	6.85 b	6.11 b	1.52 a	-
Percent of plots with runoff	100	80	80	100	75	100	70	100	100	80	100	100	75	33
No. of plots	10	5	5	3	4	3	10	10	10	5	5	3	4	3

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

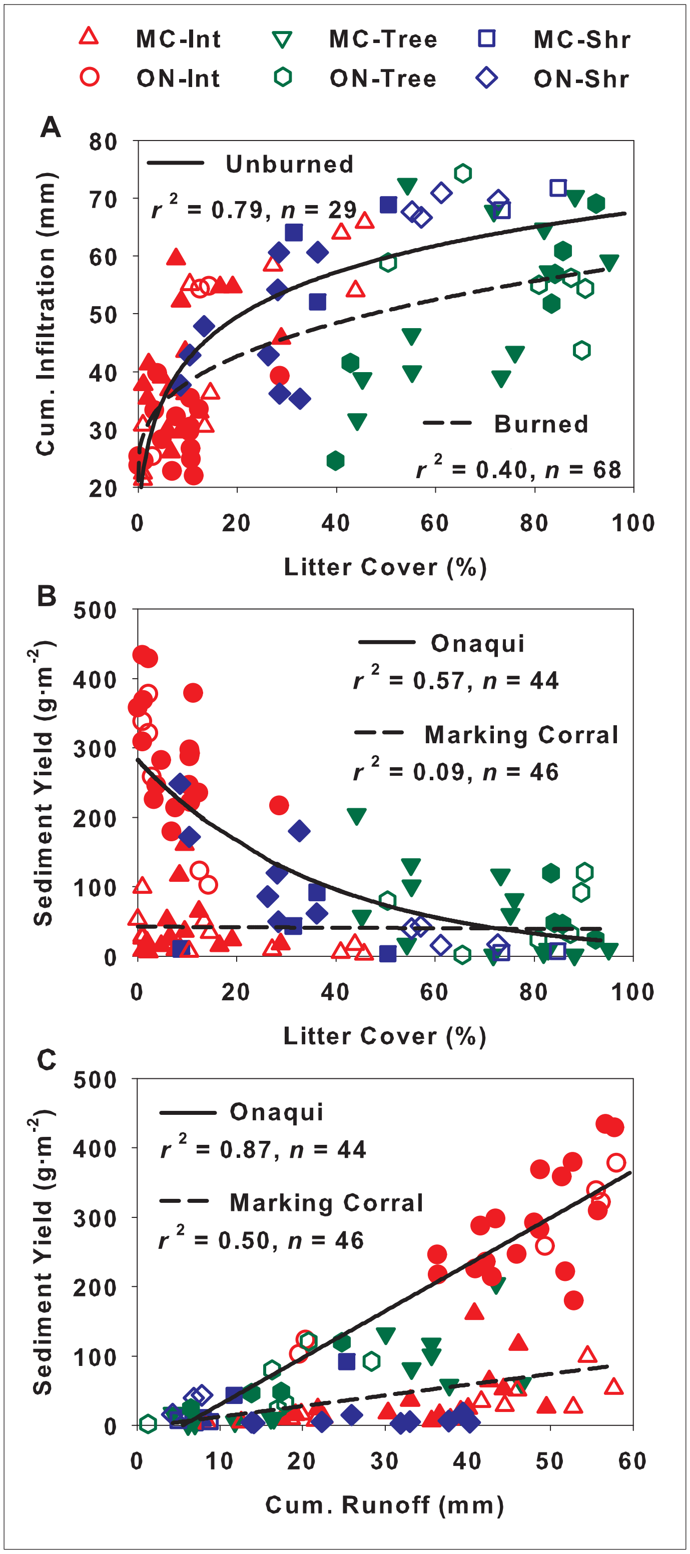

Figure 2. Cumulative infiltration versus **A,** litter cover and sediment yield versus **B,** litter cover and **C,** runoff as measured on year 1 and year 2 burned (shaded symbols) and unburned (un-shaded symbols) small rainfall simulation plots (0.5 m^2) during the wet-run (102 mm $\cdot$ h^{-1}, 45 min). Data points are shown for interspace (Int), tree coppice (Tree), and shrub coppice (Shr) microsites at the Marking Corral (MC) and Onaqui (ON) study sites.

and 70% of total area at Onaqui. At both sites, more than 70% of the interspace is bare soil and rock. Pierson et al. (2010) found intercanopy erosion during artificial rainfall experiments at the sites increased with increasing plot scale and that erosion increased exponentially where bare ground exceeded 50–60%. The amplified cross-scale erosion relationship and widespread bare ground at both sites are indicative of high erosion potential (Allen 2007; Turnbull et al. 2008) and poor ecohydrologic resilience (Williams et al. 2014). Persistence of the current vegetation and hydrologic/erosion conditions at the sites may result in an irreversible progression towards a woodland stable state without some reversing natural disturbance or management intervention (Scheffer et al. 2001; Briske et al. 2008; Petersen et al. 2009; Miller et al. 2013; Williams et al. 2014). The potential for this progression appears most likely for the Onaqui site given the limited intercanopy ground cover and high rates of erosion measured in this study.

The differing hydrologic and erosion responses of the two sites to burning underscore the need to consider site specific erodibility when considering tree-removal treatment alternatives. Runoff and erosion at the Marking Corral site were amplified by burning for the tree coppice microsites solely. Tree coppices occupy approximately 25% of the landscape at that site. We did not evaluate runoff and erosion beyond the microsite scale. However, Pierson et al. (2010) found tree coppice erosion at the site increased from less than 5 g $\cdot$ m^{-2} to approximately 40 g $\cdot$ m^{-2} with increasing plot area (0.5–13 m^2) for unburned conditions. Pierson et al. (2010) also reported intercanopy erosion increased 4-fold over the 0.5–13 m^2 scales. This suggests that burning likely increased erosion over larger scales at Marking Corral. Erosion from burned areas at the site is expected to decline over time with understory cover recruitment (Bates et al. 2005; Pierson et al. 2007; Bates and Svejcar 2009; Bates et al. 2011; Pierson et al. 2013; Williams et al. 2014). The strong linear correlation of sediment yield and runoff at Onaqui indicate that the site is capable of producing substantial erosion with increasing runoff (Fig. 2C). The prescribed fire at Onaqui essentially yielded shrub and tree coppice soil erosion rates similar to that of the degraded interspace plots (Table 5). Tenfold fire-induced increases in erosion persisted at the site in year 2. Longer-term studies are needed to assess the temporal effects of the prescribed fire on soil erosion at the site, but the initial impact is high rates of runoff and soil erosion across all microsites. In contrast, masticated tree material on degraded and rapidly eroding interspace plots at Onaqui significantly increased surface soil protection and decreased high rates of runoff and erosion by 4- to 5-fold (Table 5; Fig. 5). Cline et al. (2010) reported runoff and erosion rates from tree and shrub coppices in the mastication area were unaffected by the treatment.

Initial canopy and ground cover responses to the prescribed fires were typical for low/moderate fires in late-succession woodlands, but may have been affected by below average precipitation the second year after treatment. The prescribed fires in this study did not burn entire tree canopies due to the fuels structure and fire weather at the time of ignition. Residual dead needles on burned piñon and juniper trees continued to provide litter input to the ground surface through year 2. Williams et al. (2014) reported a more than 90% reduction in litter cover and more than 40 mm decrease in litter depth on

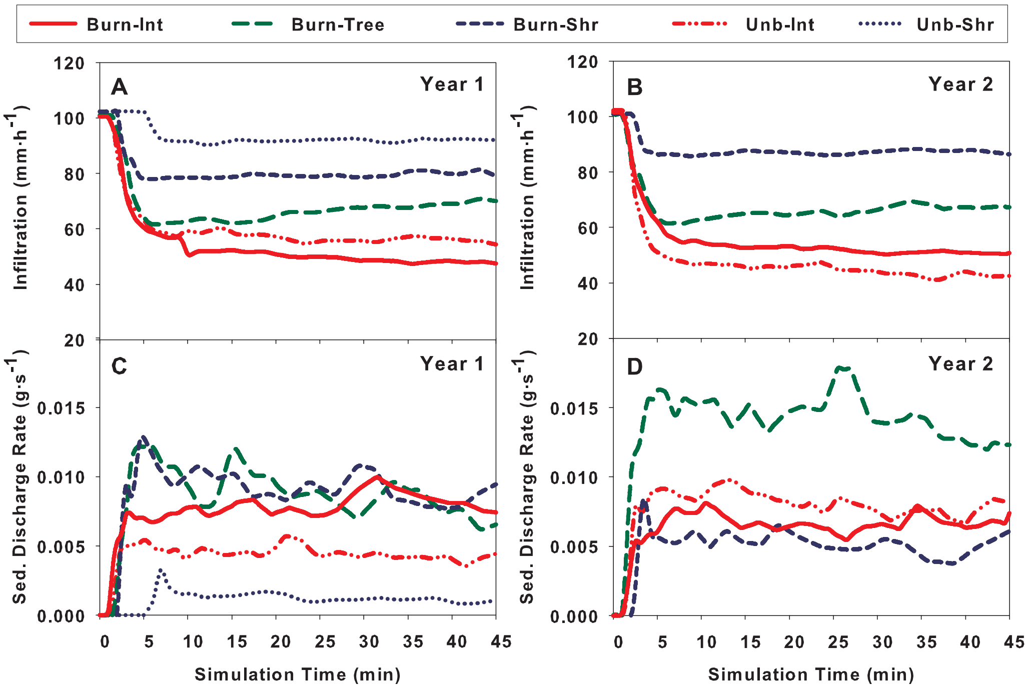

Figure 3. **A** and **B,** Infiltration and **C** and **D,** sediment discharge for wet-run (102 mm · h^{-1}, 45 min) rainfall simulations at Marking Corral that generated runoff on burned (Burn) and unburned (Unb) interspace (Int), tree coppice (Tree), and shrub coppice (Shr) microsites 1 (year 1) and 2 (year 2) yr postfire.

burned tree coppices one year following a high severity wildfire in a western juniper woodland. Tree canopies were nearly entirely consumed in the fire and litter cover on tree coppices 2 yr postfire was approximately 10%. Pierson et al. (2008a) reported nearly 100% consumption of litter on a productive sagebrush site during high severity wildfire that required 3 yr to return to near prefire levels. The limited litter consumption underneath trees in the low/moderate severity burns of this study likely provided greater surface protection against runoff and erosion (Pannkuk and Robichaud 2003) than would be expected for higher severity fires like those reported in the Pierson et al. (2008a) and Williams et al. (2014) studies. The levels of shrub and grass cover removal (Tables 2 and 3) were expected, but we anticipated a more rapid recovery of grass

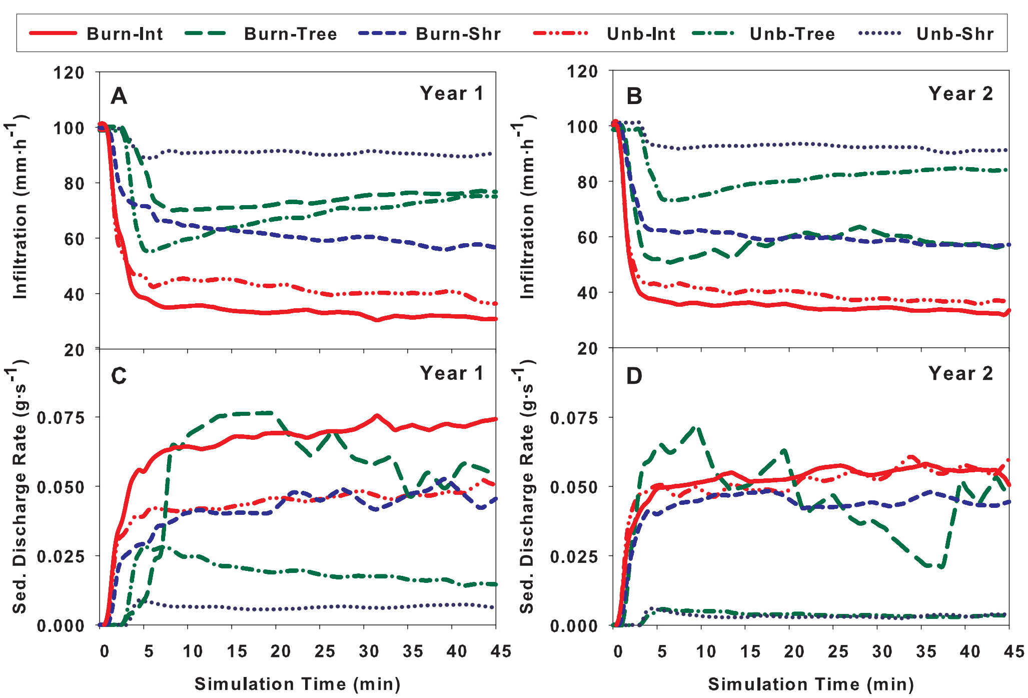

Figure 4. **A** and **B,** Infiltration and **C** and **D,** sediment discharge for wet-run (102 mm · h^{-1}, 45 min) rainfall simulations at Onaqui that generated runoff on burned (Burn) and unburned (Unb) interspace (Int), tree coppice (Tree), and shrub coppice (Shr) microsites 1 (year 1) and 2 (year 2) yr postfire.

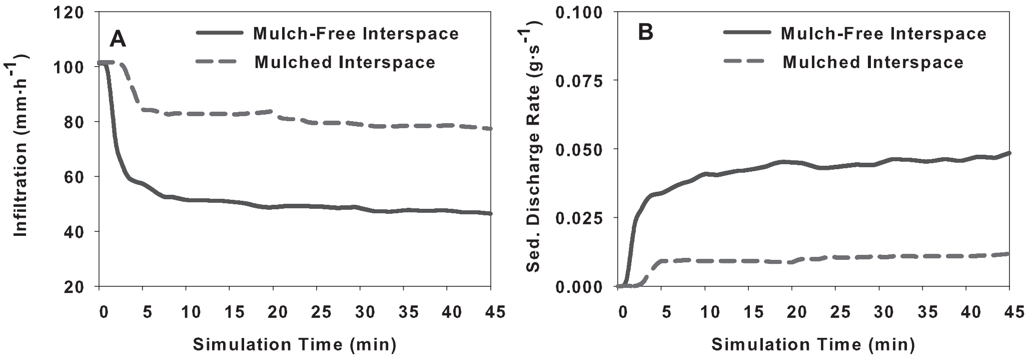

Figure 5. A, Infiltration and **B,** sediment discharge for wet-run (102 $mm \cdot h^{-1}$, 45 min) rainfall simulations at Onaqui that generated runoff on mulch-covered and mulch-free interspace microsites in year 1.

cover in interspaces and on shrub plots by year 2 (Bates et al. 2011; Miller et al. 2013; Williams et al. 2014). In year 1, unburned intercanopy perennial grass cover was approximately 25% at Marking Corral and 7% at Onaqui. Grass canopy cover was less than 10% in the unburned intercanopy at both sites in year 2. We suspect the delayed recruitment in the burn areas at the sites was associated with less than normal precipitation in year 2 that affected herbaceous cover across all treatments and possibly the hydrologic recovery of burned intercanopy plots. Williams et al. (2014) reported a favorable postfire herbaceous response within a burned western juniper intercanopy that improved interspace infiltration within 2 yr postfire.

Effects of Burning on Soil Water Repellency

Naturally occurring soil water repellency underneath piñon and juniper canopies persisted postfire, but its influence on infiltration was exacerbated by litter reductions postfire. We measured strong soil water repellency under tree litter to soil depths of 4 cm for unburned and burned treatments. The strength of soil water repellency for a particular depth was consistent for burned and unburned plots at a site, each year with few exceptions (Fig. 1). Soil water repellency did exhibit temporal variability at Marking Corral, increasing in strength at the soil surface across both treatments in year 2. The cause of the temporal variability is unknown, but temporal fluctuations in repellency strength are not uncommon (Doerr et al. 2000, 2009; Pierson et al. 2008a, 2008b, 2009, 2011). Infiltration hydrographs for burned and unburned tree coppice plots clearly indicate soil water repellency influenced infiltration (Figs. 3 and 4). Infiltration on water-repellent soils typically increases over the course of rainfall (minimum infiltration less than final infiltration) as the repellent layer is breached via macropores or gradually wets up (Meeuwig 1971; Doerr et al. 2000; Robichaud 2000; Pierson et al. 2008b). Thick litter mats underneath unburned trees mitigated the effects of repellency on infiltration rates as evident by the lack of runoff from unburned tree coppices at Marking Corral and relatively high infiltration rates on the same microsites at Onaqui (Tables 4 and 5). The impact of litter depth reductions on infiltration into the water-repellent soil is evident in the litter and infiltration relationship in Figure 2A. Infiltration was low for most burned tree coppice plots even though litter cover commonly exceeded 40%. Infiltration on unburned shrub and tree plots with more than 40% litter was nearly twice that of the burned tree plots (Fig. 2A). The difference in infiltration for burned versus unburned tree plots where litter exceeded 40% is explained only by the reduction in depth of the litter cover and concomitant repellency persistence. The effect of repellency on infiltration into burned tree coppices is further evident in the wetting trench data. The percent wetted area to 10-cm soil depth ranged 65–70% for burned tree coppice plots and 70–95% for unburned tree coppice plots. Furthermore, 60–90% of burned tree plots generated runoff from the dry- and wet-run simulations whereas none and 50–75% of unburned tree plots produced dry- and wet-run runoff at Marking Corral and Onaqui, respectively. Our results support previous work showing soil water repellency is a natural phenomenon in tree canopy areas on unburned and burned piñon and juniper woodlands in the Great Basin (Lebron et al. 2007; Madsen et al. 2008; Pierson et al. 2010; Robinson et al. 2010; Madsen et al. 2011, 2012; Pierson et al. 2013; Williams et al. 2014). Further, we find the presence of soil water repellency and its effect on infiltration in tree canopy areas are not necessarily fire-created, but rather are exacerbated by fire removal of litter. Burning did not create water-repellent conditions underneath shrubs or in the interspaces at either site in this study.

The persistence of soil water repellency following fire in this study may be related to the low to moderate soil burn severity at the study sites. Soil organic matter, the primary agent for soil water repellency development in sandy soils (Doerr et al. 2000), is combusted at soil burn temperatures $> 200°C$ and is completely consumed at 450–500°C (DeBano et al. 1998; Neary et al. 1999). Soil water repellency breaks down or is destroyed at soil burn temperatures between 270–400°C (Savage et al. 1972; DeBano et al. 1976; Giovannini and Lucchesi 1997; Doerr et al. 2000, 2004). Soil burn temperatures under trees were not measured in this study, but, based on residual woody debris and litter, likely did not exceed temperature thresholds for a long enough duration to substantially reduce soil organic matter and destroy soil water repellency (Parsons et al. 2010). Litter cover underneath tree canopies in this study was reduced by burning, but needle fall from burned trees returned litter cover to 75–80% on tree coppices within one year postfire (Tables 2 and 3). We opine the rapid litter reaccumulation had less effect on soil water repellency persistence than retention of soil organic matter associated with the low to moderate severity burns (Doerr et al. 2000, 2009). Williams et al. (2014) reported sustained strong soil water repellency under western juniper trees 2 yr following severe wildfire. Tree litter cover and litter depth in that study were reduced from near 100% and 43 mm to 12% and 2 mm, respectively, one year postfire. This study and others (Madsen et al. 2011; Pierson et al. 2013; Williams et al. 2014) indicate that burn temperature and duration required for complete destruction of soil water repellency in piñon and juniper woodlands may be uncommon except in cases with dense downed-woody fuel accumulations or tree slash. Soil temperatures under burning piñon and juniper slash piles commonly exceed 600°C for sustained periods (Sheley and Bates 2008; Bates et al. 2011).

Aggregate Stability Measures and Soil Erosion Potential

Aggregate stability indices as measured in this study accurately depicted microsite differences in soil stability, but poorly depicted the measured differences in soil erosion potential associated with burning and site-specific erodibility. The aggregate stability indices clearly showed that the soil stability was greater underneath tree and shrub litter than in the more erodible and mostly-bare interspaces (Tables 2 and 3), as expected for rangelands (Blackburn and Pierson 1994; Bestelmeyer et al. 2006; Bird et al. 2007). In that regard, the index provided a good relative measure of where soils were most erodible. However, we found no correlation ($P > 0.05$) between aggregate stability indices and runoff and erosion as measured in this study. The aggregate stability index failed to capture the differences in soil erodibility between the two study sites. For example, aggregate stability indices ranged from 1 to 2 (less than 10% stable aggregates) for interspaces at both sites. Measured sediment-to-runoff ratios for interspace plots averaged 0.90 $g \cdot m^{-2} \cdot mm^{-1}$ at Marking Corral and 5.22 $g \cdot m^{-2} \cdot mm^{-1}$ at Onaqui. The index accurately depicted vulnerable conditions in interspaces for the two sites, but offered no indication of the significantly higher erodibility at Onaqui. Sediment-to-runoff ratios and sediment discharge both increased following burning of tree and shrub coppice at Onaqui (Table 5; Fig. 4). Aggregate stability indices for burned versus unburned tree and shrub coppices at Onaqui were not statistically different (Table 3). The index therefore also failed to capture the effects of burning on soil erosion potential. We conclude that the index approach provides a simple and quick relative indicator of soil stability for unburned conditions, but that it provides no predictive capability relative to quantifying actual erosion potential for burned and unburned woodlands.

Our results are contradictory to other recent studies using similar tests of surface soil aggregate stability on burned woodlands. Ross et al. (2012) and Owen et al. (2009) both reported decreased aggregate stability indices following pile-burning of tree debris in piñon-juniper woodlands, but did not quantify soil erosion. Burn temperatures were likely hotter on the pile burns in those studies than on the low- to moderate-severity burns in our study, and the hotter temperatures may have had a greater effect on surface soil aggregates. For sandy water-repellent soils with organic matter as the primary bonding agent, low to moderate fire severities commonly result in no change or only a slight increase or decrease in aggregate stability, whereas burn temperatures associated with high severity fires substantially reduce aggregate stability (Mataix-Solera et al. 2011). The persistence of soil water repellency 1 and 2 yr postfire in this study suggests retention of enough soil organic matter to sustain hydrophobic soil conditions (Doerr et al. 2000, 2009). Soil water repellency and organic matter likely promoted microaggregate stability under piñon and juniper canopies postfire (Jordán et al. 2011; Mataix-Solera et al. 2011). Additionally, the aggregate stability ped sampling method may have exaggerated surface soil stability given the method characterizes stability of extractable peds solely. Noncohesive, non-ped-forming soil particles are not characterized by the applied methodology. Regardless, any positive effects of soil particle bonding by organic matter or hydrophobic coatings on burned tree coppices were likely overwhelmed by amplified rainsplash effects associated with increased bare ground (ash, bare soil, and rock) (Terry and Shakesby 1993; Shakesby and Doerr 2006; Pierson et al. 2008a, 2009, 2011, 2013; Williams et al. 2014).

MANAGEMENT IMPLICATIONS

Results from small-plot scale measurements in this study clearly demonstrate that similarly degraded conditions in late-succession woodlands may result in highly variable hydrologic and erosion responses to tree-removal treatments. Our measurement scale allowed us to focus on microsite (tree, shrub, interspace) treatment effects and to evaluate site differences in soil erosion potential from mostly bare interspace areas. Erosion per unit of runoff from unburned interspaces in two phase II–III woodlands differed more than 6-fold and the site differences in soil erodibility produced significantly different erosion responses to tree removal by burning. Burning of the more erodible woodland generated 3- to more than 10-fold increases in erosion from areas underneath tree and shrub canopies and enhanced hydrologic and erosion vulnerability at the site. Postfire increases in erosion from tree and shrub plots at both sites persisted 2 yr following treatments. The persistence of fire effects, particularly for the less erodible site, was likely related to low precipitation and poor plant recruitment the second year posttreatment. High rates of runoff and soil erosion from unburned interspaces at the more erodible site were reduced 4- to 5-fold by application of masticated tree material (mulch). Overall results from the rainfall simulations suggest the following: 1) similarly degraded woodland sites with inherently different soils may require different treatments to reduce soil erosion, 2) amplified soil loss following woodland burning may persist more than 2 yr, particularly in periods of low precipitation, and 3) mastication or mulch treatments may be more appropriate than burning on highly erodible sites if more immediate soil loss reductions are desired. Our findings address short-term responses solely and do not address the potential for burning or mastication treatments to reduce long-term soil erosion. Furthermore, vegetation, hydrologic, and erosion responses to the low/moderate severe fire in this study may differ from those of sites that experience high-severity burning.

Our experimental design allowed us to assess the influence of burning on naturally occurring soil water repellency and the effectiveness of rapidly acquired aggregate stability measures to detect soil erosion potential. Soil water repellency was restricted to tree coppice microsites and was unaffected by burning at both woodland sites. The effects of repellency on infiltration were mitigated by litter on unburned tree plots, and were exacerbated by litter removal on burned tree plots. Burning reduced litter depths underneath trees and resulted in decreased infiltration and soil wetting and amplified runoff and erosion. We attribute the fire-induced hydrologic and erosion responses on tree coppices primarily to exacerbation of repellency effects on infiltration. Indices of aggregate stability accurately depicted microsite relative differences in soil stability, but failed to track the large differences in soil erodibility between sites and between burned versus unburned conditions. The results suggest that the aggregate stability index method is a good tool for identifying areas of relative soil surface instability but provides no indication of actual soil erosion potential. It is paramount that users of the index consider the method simply as a tool to identify areas of soil erosion susceptibility and that determination of soil erosion potential may require more detailed soil analysis.

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