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Short-Term Impacts of Tree Removal on Runoff and Erosion From Pinyon- and Juniper-Dominated Sagebrush Hillslopes☆,☆☆

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ABSTRACT

Tree removal is often applied to woodland-encroached rangelands to restore vegetation and improve hydrologic function, but knowledge is limited regarding effects of tree removal on hydrologic response. This study used artificial rainfall and overland flow experiments (9–13 m^2) and measures of vegetation and ground cover to investigate short-term (1–2 yr) responses to tree removal at two woodland-encroached sites. Plots were located under trees (tree zone) and in the intercanopy (shrub-interspace zone, 75% of area). Before tree removal, vegetation and ground cover were degraded and intercanopy runoff and erosion rates were high. Cutting and placing trees into the intercanopy did not significantly affect vegetation, ground cover, runoff, or erosion 1 yr posttreatment. Whole-tree mastication as applied in this study did not redistribute tree mulch within the intercanopy, but the treatment did result in enhanced herbaceous cover and hydrologic function in the intercanopy. Fire removal of litter and herbaceous cover increased tree-zone runoff and erosion under high-intensity rainfall by 4- and 30-fold at one site but had minimal impact at the other site. Site response differences were attributed to variability in burn conditions and site-specific erodibility. Burning had minimal impact on shrub-interspace runoff and erosion from applied high-intensity rainfall. However, 1 yr postfire, erosion from concentrated overland flow experiments was 2- to 13-fold greater on burned than unburned tree-zone and shrub-interspace plots and erosion for burned tree zones was 3-fold greater for the more erodible site. Two yr postfire, overland flow erosion remained higher for burned versus unburned tree zones, but enhanced intercanopy herbaceous cover reduced erosion from shrub-interspace zones. The net impact of burning included an initial increase in erosion risk, particularly for tree zones, followed by enhanced herbaceous cover and improved hydrologic function within the intercanopy. The overall results suggest that erosion from late-succession woodlands is reduced primarily through recruitment of intercanopy herbaceous vegetation and ground cover.

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Introduction

The encroachment of pinyon (*Pinus* spp.) and juniper (*Juniperus* spp.) species into sagebrush-steppe (*Artemisia* spp.) in the western United States has been associated with decreased shrub and herbaceous vegetation, amplified runoff and soil erosion, degraded wildlife habitat, and a reduced capacity to deliver various ecosystem goods and services (Knick et al., 2003; Aldrich et al., 2005; Miller et al., 2005; Pierson et al., 2010; Davies et al., 2011; Miller et al., 2011; Bates et al., 2014; Williams et al., 2014a). This woodland encroachment has been attributed to multiple factors including climate variability, land-use practices, decreased fire frequency, and CO_2 fertilization (Miller and Wigand, 1994; Miller and Rose, 1995; Knapp and Soulé, 1996; Miller and Tausch, 2001; Romme et al., 2009). Woodland development forms a continuum of increasing tree cover but has been categorized into three phases (Miller et al., 2000; Johnson and Miller, 2006; Miller et al., 2008; Roundy et al., 2014a). In phase I, tree cover increases for the 0- to 3-m height class, but shrubs and herbaceous species remain dominant. Phase II occurs once trees approach 10–50% of potential tree cover and understory shrub and herbaceous plants decline due to competition for limited water and soil resources. Phase III is reached when tree cover stabilizes as the dominant cover type and exerts primary control on key ecological

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processes. Declining understory cover in the later stages of phase II creates extensive and contiguous patches of bare ground within the intercanopy, reduces rainfall interception and infiltration, increases water available for runoff, and promotes overland flow (Pierson et al., 2007; Petersen and Stringham, 2008; Petersen et al., 2009; Pierson et al., 2010; Williams et al., 2014a). This encroachment-induced shift from a water- and soil-conserving sagebrush-steppe structure (Pierson et al., 1994, 2009) to a community of tree islands and bare intercanopy facilitates high rates of intercanopy runoff and erosion (Wilcox, 1994; Wilcox et al., 2003; Pierson et al., 2007, 2010, 2013; Williams et al., 2014a) and long-term loss of soil resources critical for plant productivity (Schlesinger et al., 1990; Davenport et al., 1998; Belnap et al., 2005; Ludwig et al., 2005; Turnbull et al., 2008, 2010, 2012). Tree removal is a common practice to rehabilitate or restore ecological structure and function of sagebrush-steppe rangelands (Miller et al., 2005; Pierson et al., 2007; Sheley and Bates, 2008; Bates and Svejcar, 2009; Pierson et al., 2013; Bates et al., 2014; McIver et al., 2014; Williams et al., 2014a).

Sagebrush-steppe vegetation response to tree removal can vary widely depending on the pretreatment plant community and site conditions, removal method, soil temperature and moisture regimens, and posttreatment weather patterns (Bates et al., 1998, 2000; Miller et al., 2005, 2013; Bates et al., 2014; Chambers et al., 2014; Miller et al., 2014). Restoration of sagebrush communities in late successional (late phase II–III) woodlands can be particularly challenging due to limited propagules and seed sources for perennial grasses and sagebrush reestablishment (Koniak and Everett, 1982; Miller et al., 2000, 2005; Bates et al., 2014; Miller et al., 2014; Roundy et al., 2014a). Bates et al. (2005, 2007, 2011, 2014) suggested that recruitment of sagebrush-steppe understory vegetation in heavily encroached systems requires pretreatment perennial grass and forb densities of at least 1–2 and 5 plants per square meter, respectively. Tree removal by prescribed burning can remove limited native perennial species and facilitate invasion by annual weeds, particularly on sites with mesic-aridic soil temperature-moisture regimens (> 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; Bates et al., 2011, 2014; Miller et al., 2014; Roundy et al., 2014a). Prescribed fire can also reduce residual sagebrush cover (Chambers et al., 2014; Miller et al., 2014; Roundy et al., 2014a). Sagebrush does not resprout after fire and can require 15–50 yr for postfire recovery (Barney and Frischknecht, 1974; Miller and Heyerdahl, 2008; Ziegenliagen and Miller, 2009). Mechanical tree mastication and cutting can limit treatment-related shrub and herbaceous mortality (Chambers et al., 2014; Miller et al., 2014; Roundy et al., 2014a) but often leave residual juvenile pinyon and juniper seedlings to repopulate posttreatment (Bates et al., 2005; Miller et al., 2005, 2013; O'Connor et al., 2013). Vegetation response to any treatment is also influenced by precipitation in the years following treatment (Bates et al., 1998, 2000; West and Yorks, 2002; Bates et al., 2007). Successful restoration of woodland-encroached sagebrush-steppe is most likely on frigid-xeric sites and when tree removal is applied during phase I or early phase II encroachment (Miller et al., 2005, 2013; Chambers et al., 2014; Roundy et al., 2014a). Currently, much of the woodland-encroached domain across the western United States is approaching late succession (Miller and Tausch, 2001; Miller et al., 2008). The diverse conditions in which woodlands occur and the varying responses of vegetation to tree removal present major management challenges to land managers and agencies (Miller et al., 2005; McIver et al., 2010, 2014).

Knowledge is limited regarding linkages between sagebrush-steppe restoration and hydrologic function following tree removal treatments given the vast range of pinyon and juniper expansion. The generally accepted hypothesis is that favorable canopy and ground cover recovery following tree removal will improve infiltration, reduce runoff and soil loss, and enhance soil water recharge and vegetation productivity (Pierson et al., 2007, 2013; Young et al., 2013; Miller et al., 2014; Mollnau et al., 2014; Roundy et al., 2014a, 2014b; Williams et al., 2014a). Young et al. (2013) evaluated the effects of whole-tree mastication on soil water availability and found that tree removal by mastication created 44.5 additional wet-soil days (soil water potential at 13–30 cm soil depth > −1.5 MPa) during spring and summer growing seasons. Roundy et al. (2014b) evaluated the effects of prescribed fire, tree cutting, and tree mastication on available soil water (upper 30 cm of soil profile) at 13 pinyon- and/or juniper-encroached sagebrush sites in the Great Basin over a 4-yr period. That study found tree removal by each method increased the number of wet-soil days (soil water potential > −1.5 MPa) in the spring but also noted the additional number of wet-soil days declined as the understory plant cover increased. Four yr posttreatment, there were 8.6 and 18 more wet-soil days in treated areas than in untreated controls for mid and high tree dominance. Pierson et al. (2007) found that enhanced herbaceous cover significantly reduced intercanopy runoff and soil erosion from a simulated rainfall event 10 yr after tree cutting in a western juniper (*J. occidentalis* Hook.) woodland. Pierson et al. (2013) and Williams et al. (2014a) found that burning of a late-succession western juniper–dominated site enhanced intercanopy herbaceous cover and infiltration and reduced intercanopy erosion within two growing seasons postfire. At the small catchment scale (300–1100 m^2), Hastings et al. (2003) found that tree cut-and-slash treatments reduced soil erosion by nearly 100-fold relative to adjacent, untreated, pinyon-juniper–dominated sites. Hastings et al. (2003) attributed the reduced soil erosion to treatment-induced increases in surface cover of herbaceous plants and slash debris.

The goal of this study was to increase understanding of tree-removal effects on hillslope runoff and erosion processes on sagebrush rangelands dominated by single-leaf pinyon (*P. monophylla* Torr. and Frém) and Utah juniper (*J. osteosperma* [Torr.] Little). Specifically, we used rainfall simulation and overland flow experiments to evaluate the effects of tree cutting, mastication, and prescribed fire on runoff and erosion processes at the patch scale (10–40 m^2). The primary objectives were to quantify vegetation and ground cover characteristics and hillslope runoff and erosion underneath tree canopies (tree zones) and in the intercanopy (shrub-interspace zones) before tree removal, and 1 and 2 yr following tree removal. This research is part of the 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 of the Great Basin cold desert, United States, and the effects of various sagebrush-steppe restoration methods (McIver et al., 2010; McIver and Brunson, 2014). The study areas were the subject of previous companion SageSTEP hydrologic studies by Pierson et al. (2010, 2014) and Cline et al. (2010). Pierson et al. (2010) evaluated runoff and erosion across small-plot (0.5 m^2) and patch scales at the two woodland sites before the tree removal. Cline et al. (2010) evaluated the impacts of tree mastication on small-plot scale infiltration, runoff, and erosion at one of the sites 1 yr following tree removal. Pierson et al. (2014) evaluated the effects of the whole-tree mastication and prescribed-fire tree-removal treatments on small-plot scale vegetation, soils, infiltration, runoff, and erosion at the sites 1 and 2 yr after tree removal. This study expands on the previous studies by Pierson et al. (2010, 2014) and Cline et al. (2010) through quantification of tree removal effects on patch-scale vegetation, soils, runoff, and erosion for the first 2 yr following the treatments. The larger-scale experiments (paired 13 m^2 plots) in this study relative to the smaller plots (0.5 m^2) in our previous post-treatment studies (Cline et al., 2010; Pierson et al., 2014) allow us to quantify treatment effects on runoff and erosion from combined rainsplash, sheetflow, and concentrated flow processes (Williams et al., 2014a).

Methods

Research Sites

This study was conducted on a single-leaf pinyon-Utah juniper site (Marking Corral—Nevada, United States) and a Utah juniper site

(Onaqui—Utah, United States) within the SageSTEP study network (McIver and Brunson, 2014; McIver et al., 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. The study sites are managed by the Bureau of Land Management (BLM) for grazing use but have been excluded from grazing since autumn 2005. Both study sites are in late phase II to early phase III woodland encroachment and historically consisted of sagebrush-steppe vegetation (Pierson et al., 2010).

Detailed geographic, climate, soils, and vegetation characteristics for the sites are provided in Table 1. Soil temperature-moisture regimens for the sites are at the fringe of mesic-aridic and frigid-xeric classifications (McIver and Brunson, 2014). Annual precipitation during the study period, yr 2006–2008, averaged approximately 60–85% of normal at Marking Corral, and, at Onaqui, was approximately 13% above average in 2006 and 80% of normal in 2007 and 2008 (Thornton et al., 2012). Pierson et al. (2010) measured vegetation, ground cover, and soil properties at both sites in early summer 2006, before tree removal treatments. Tree canopy cover and intercanopy area averaged approximately 25% and 75%, respectively, at both sites (Table 1). Litter mounds (coppices) underneath trees extended, on average, 2.5 m and 2.2 m from tree bases at Marking Corral and Onaqui, respectively. Litter mass underneath tree canopies pretreatment averaged 17.4 kg · m^{-2} at Marking Corral and 14.3 kg · m^{-2} at Onaqui. Pretreatment litter depth under trees was approximately 80 mm at Marking Corral and 50 mm at Onaqui and was less than 10 mm in interspaces and under shrubs at both sites. Surface soils (0–5 cm depth) underneath tree canopies at both sites were water repellent before tree removal treatments, whereas soils in interspaces and underneath shrub canopies were wettable (Pierson et al., 2010, 2014). Bulk density at 0- to 5-cm soil depth in interspaces and on tree and shrub coppices averaged 1.35, 1.08, and 1.14 g · cm^{-3}, respectively, at Marking Corral and 1.07, 0.82, and 1.01 g · cm^{-3} at Onaqui (Pierson et al., 2010).

Tree-Removal Treatments

Tree-removal treatments at the sites were applied by the BLM in late summer of 2006 (Fig. 1). At each site, there was one application of each treatment within the respective study domain. The total study area at Marking Corral consists of approximately 6.2 ha. The burned and cut treatment areas at Marking Corral are 2.7 and 2.2 ha, respectively. The total study area at Onaqui consists of approximately 7.0 ha. The burn, cut, and mastication treatment areas at Onaqui are 2.0, 2.4, and 1.6 ha, respectively. Tree cutting by chainsaw (cut-and-drop) and prescribed-fire treatments were applied in separate areas at Marking Corral in August 2006 and at Onaqui in September–October 2006. A whole-tree mastication treatment was also applied in a separate treatment area at Onaqui in September of 2006. At each site, tree cover, understory canopy and ground cover, hillslope angle, and surface soil texture and bulk density were statistically similar ($P > 0.05$) across all treatment areas before tree removal treatments (Pierson et al., 2010). The tree-cutting treatment downed all mature trees (≥1 m height) but left a residual of 56 and 167 juvenile (<1 m height) trees per hectare in cut treatment areas at Marking Corral and Onaqui, respectively. Tree mastication at Onaqui was applied using a rubber-tired Tigercat M726E Mulcher (see Cline et al., 2010). The mastication treatment uniformly removed overstory tree cover in the mastication treatment area but left a density of 56 juvenile trees per hectare. The mastication method reduced live shrub density by ~40% (residual 5185 shrubs per hectare) and yielded a patchy ground cover of tree mulch. Posttreatment, bare ground (bare soil and rock) in the mastication area was approximately 30%, and mulch cover was 40% (Cline et al., 2010). Mulch depth was approximately 90 mm in areas previously dominated by a tree canopy and 20 mm in infrequent isolated debris patches within the intercanopy.

Table 1
Site descriptions for the Marking Corral and Onaqui study sites before tree removal treatments (Yr 0). Data from Pierson et al. (2010, 2014).

Site characteristic	Marking Corral, Nevada	Onaqui, Utah
Woodland community	Single-leaf pinyon[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 (%)[8]	15,[1] 10[2]	26[2]
Trees per hectare[8]	329,[1] 150[2]	476[2]
Mean tree height (m)[8]	2.3,[1] 2.4[2]	2.4[2]
Juvenile trees per hectare[9]	296,[1] 139[2]	154[2]
Dead shrubs per hectare	2 065	957
Intercanopy shrub canopy cover (%)	21	5
Intercanopy herbaceous canopy cover (%)	13[10]	11[10]
Intercanopy bare soil and rock cover (%)	64	79
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 yr 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 yr 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 Live trees ≥ 1.0-m height.
9 Live trees < 1.0-m height.
10 Intercanopy grass and forb canopy cover.

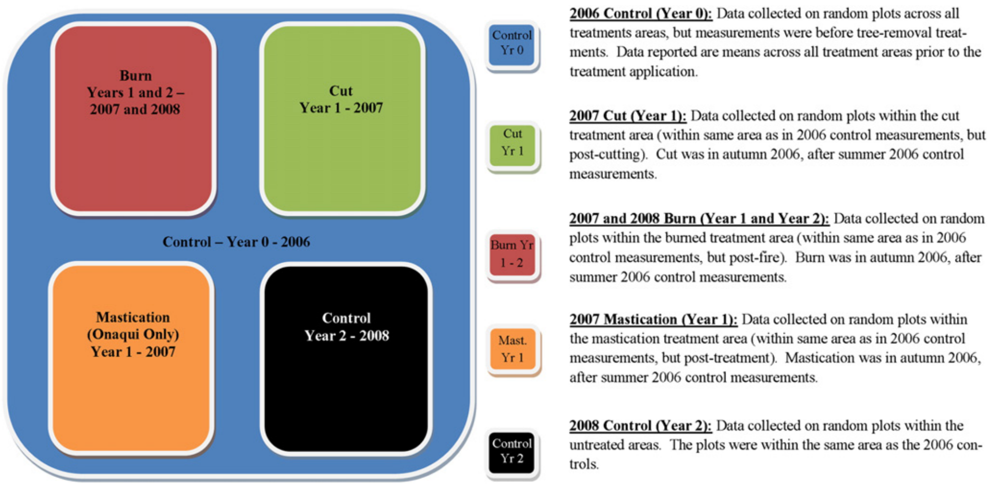

Fig. 1. Block diagram depicting treatment structure and chronology for treatments at the Marking Corral and Onaqui study sites.

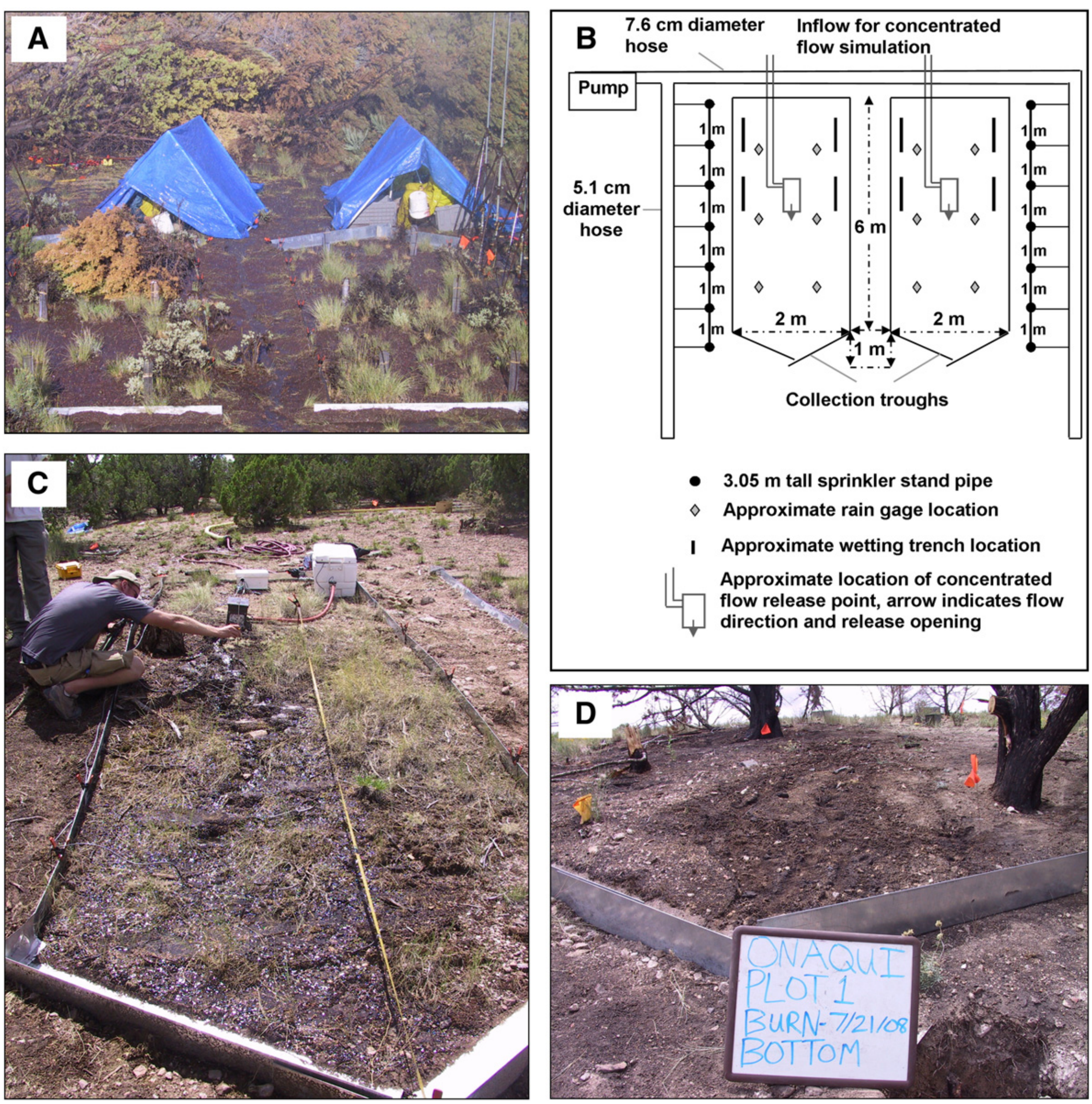

Fig. 2. Illustration showing large-plot rainfall simulation on shrub-interspaces with and without a cut-downed tree (A), paired large-plot layout and design (B), single large plot (13 m^2) on tree coppice with ongoing concentrated flow experiment (C), and an unbounded concentrated flow plot on a burned tree coppice (D). Figure modified from Pierson et al. (2010, 2013).

Burn severity was not quantified following the prescribed fires, but presence of residual and scorched tree needles, blackened litter, and downed-woody debris immediately postfire at both sites were indicative of low to moderate burn severity (Parson et al., 2010). Burning had no effect on soil water repellency under trees, and surface soils in interspaces and on shrub coppices remained wettable postfire (Pierson et al., 2014). Individual tree canopy scorch ranged from 50–75% at Marking Corral and 75–99% at Onaqui (Pierson et al., 2014). Burned tree and shrub skeletons were present at both sites. Burning reduced the density of juvenile trees by 80% and 65% and live shrub density by 80% and 70% at Marking Corral and Onaqui, respectively. Fire reduced litter cover underneath trees at both sites from nearly 100% to approximately 50% immediately postfire and reduced litter cover under shrubs from 65% to 10% at Marking Corral and from 35% to 20% immediately postfire at Onaqui (Pierson et al., 2014). Overall, burn characteristics at the Marking Corral site were consistent with those of a low- to moderate-severity burn, whereas burn characteristics at Onaqui were consistent with a moderate severity burn (Parson et al., 2010). Treatment areas were not seeded at either site.

Experimental Design

Rainfall simulation and overland flow experiments were performed at both sites immediately before tree removal (Yr 0—summer 2006) and 1 (Yr 1—summer 2007) and 2 (Yr 2—summer 2008) yr following tree removal (Fig. 1). Plot data for Yr 0 were reported in Pierson et al. (2010) but are included in this study as a control for evaluating tree removal effects. Rainfall simulations were conducted on ~2-m wide × 6.5-m long plots (Fig. 2A and B; long axis perpendicular to hillslope contour) in untreated areas in Yr 0 and in all tree-removed areas in Yr 1 to assess effects of tree removal on runoff and erosion by rainsplash, sheetflow, and concentrated-flow processes (Pierson et al., 2009, 2010, 2013; Williams et al., 2014a). Concentrated overland-flow simulations (Fig. 2C and D; 2-m wide × 4.5-m long) were conducted within rainfall simulation plots (after rainfall simulations) in untreated areas in Yr 0 and in all tree-removed areas in Yr 1 (Fig. 1). In Yr 2, concentrated overland flow simulations were conducted as independent experiments (no rainfall simulation plots in Yr 2) on untreated (controls) plots and burned plots (Fig. 1). Concentrated flow experiments were used to measure erosion solely from concentrated flow or rill processes (Pierson et al., 2008, 2009, 2010; Al-Hamdan et al., 2012a, 2012b; Pierson et al., 2013; Williams et al., 2014a).

Rainfall simulation and concentrated flow plots in Yr 0 and Yr 1 were installed in pairs, separated by a 1-m wide × 6.5-m long strip (Fig. 2A and B), as described by Pierson et al. (2010). In Yr 2, concentrated flow plots were installed without plot borders (Fig. 2D), but, as in Yr 0–1, contained runoff collection troughs in a “V” pattern at the plot base (Pierson et al., 2013; Williams et al., 2014a). Trees on control and burned plots were trimmed or removed 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.

Rainfall and concentrated-flow plot locations were selected randomly within shrub-interspace zones (varying amounts of shrub coppice and interspace area) and tree zones (tree coppice with minor interspace component) at each site (Pierson et al., 2010). Within the cut-tree areas, only the shrub-interspace zone was used to evaluate the effects of tree cutting. We anticipated substantial changes in understory vegetation due to tree cutting would require 2 or more yr (Bates et al., 1998, 2000, 2005, 2007, 2011; Miller et al., 2014) and therefore focused the cut-tree experiments on the potential for downed trees to reduce runoff and erosion in the shrub-interspace zone. The immediate hydrologic and erosional effects of cut-downed trees were examined by placing a randomly selected cut pinyon or juniper tree on individual shrub-interspace rainfall and concentrated-flow plots. Cut trees were placed on plots approximately 1 m upslope of the runoff collection trough, perpendicular to the predominant hillslope contour, with the long axis of the tree partially in contact with and parallel to the ground surface. The average height, trunk diameter, and crown diameter for cut trees placed on plots were respectively 1.9 m, 10.8 cm, and 1.4 m at Marking Corral and 2.1 m, 12.3 cm, and 1.5 m at Onaqui.

Table 2
Topography, canopy and ground cover, and cover gaps measured on rainfall simulation and concentrated-flow plots (13 m^2) 1 yr before treatments (Control—Yr 0) and in cut (with and without downed tree) and burned areas 1 yr post-treatment (Yr 1) at Marking Corral. Means within a row followed by a different lowercase letter are significantly different ($P < 0.05$).

Marking Corral study site	Control—Yr 0		Cut, no downed tree—Yr 1	Cut, downed tree[1]—Yr 1	Burned—Yr 1	
Plot characteristic	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Shrub-interspace zone	Shrub-interspace zone	Tree zone
Slope (%)	9.3 a	9.3 a	9.9 a	10.1 a	9.4 a	9.5 a
Surface roughness (mm)	17 ab	20 b	17 ab	17 ab	15 ab	13 a
Total canopy cover (%)[2]	39.0 bc	26.6 b	41.6 bc	43.6 c	23.0 ab	6.2 a
Total herbaceous canopy cover (%)[3]	13.1 bc	4.5 a	21.2 c	22.8 c	19.2 c	5.7 ab
Shrub canopy cover (%)	20.9 c	4.0 a	16.3 c	12.7 c	0.4 ab	0.0 a
Grass canopy cover (%)	12.3 b	4.4 a	16.0 b	17.4 b	5.1 a	1.5 a
Litter cover (%)	33.2 b	90.1 d	18.7 ab	21.4 ab	10.4 a	66.9 c
Rock cover (%)	37.6 c	3.1 a	9.0 ab	9.8 ab	15.6 b	3.6 a
Ash (%)	—	—	—	—	0.2 a	4.8 b
Bare ground (%)[4]	63.9 c	6.3 a	69.2 cd	69.3 cd	85.5 d	27.2 b
Canopy gaps 25–50 cm (%)[5]	9.9 b	3.4 a	14.2 b	13.4 b	15.2 b	2.2 a
Canopy gaps 51–100 cm (%)[5]	18.6 b	9.6 a	23.8 b	20.7 b	17.1 b	5.4 a
Canopy gaps 101–200 cm (%)[5]	29.8 c	14.6 ab	13.3 ab	16.8 abc	24.5 bc	4.5 a
Canopy gaps > 200 cm (%)[5]	5.5 a	54.5 b	1.1 a	7.6 a	15.2 a	76.8 b
Basal gaps 25–50 cm (%)	5.9 bc	1.4 a	9.7 bc	10.2 c	3.5 ab	0.7 a
Basal gaps 51–100 cm (%)	14.6 bc	4.1 a	25.7 c	20.7 bc	10.2 b	1.7 a
Basal gaps 101–200 cm (%)	33.6 b	5.4 a	27.5 b	34.0 b	24.3 b	7.1 a
Basal gaps > 200 cm (%)	37.2 ab	80.2 cd	23.3 a	21.2 a	58.0 bc	88.6 d
Average canopy gap (cm)[5]	75.1 a	201.8 b	50.8a	59.8 a	73.6 a	234.5 b
Average basal gap (cm)	125.0 ab	304.7 c	88.9 a	90.8 a	170.1 b	368.9 c
No. of plots	12	12	6	6	6	6

1 Plots with a single-leaf pinyon (*Pinus monophylla* Torr. & Frem.) or Utah juniper (*Juniperus osteosperma* [Torr.] Little) tree laying, partially in contact with and parallel to the ground surface, perpendicular to the long axis of the respective plot (across plot along hillslope contour). Tree placed on plot immediately before rainfall simulation.
2 Excludes tree canopy removed immediately before rainfall simulation.
3 Grass and forb canopy cover.
4 Rock, ash, and bare soil.
5 Canopy gaps measured after tree removal for rainfall simulation.

Table 3
Topography, canopy and ground cover, and cover gaps measured on rainfall simulation and concentrated-flow plots (13 m^2) 1 yr before treatments (Control—Yr 0) and in cut (with and without downed tree), mastication, and burned areas 1 yr posttreatment (Yr 1) at Onaqui. Means within a row followed by a different lowercase letter are significantly different ($P < 0.05$).

Onaqui study site	Control—Yr 0		Cut, no downed tree—Yr 1	Cut, downed tree[1]—Yr 1	Masticated—Yr 1		Burned—Yr 1	
Plot characteristic	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Shrub-interspace zone	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Tree zone
Slope (%)	14.0 a	14.6 a	13.1 a	13.5 a	13.6 a	14.1a	18.5 a	19.3 a
Surface roughness (mm)	26 ab	31 b	20 a	18 a	25 ab	42 c	26 ab	26 ab
Total canopy cover (%)[2]	18.6 b	21.0 b	30.8 bc	25.6 b	48.6 c	10.9 ab	17.2 b	3.3 a
Total herbaceous canopy cover (%)[3]	10.6 b	16.3 c	16.9 c	18.3 cd	25.4 d	5.8 ab	7.9 ab	1.1 a
Shrub canopy cover (%)	5.2 b	0.5 a	6.6 bc	4.1 ab	13.3 c	1.9 ab	0.2 a	0.1 a
Grass canopy cover (%)	7.5 b	15.0 c	8.8 b	9.0 bc	16.5 c	5.4 ab	6.1 b	0.7 a
Litter cover (%)	8.2 a	76.2 c	8.3 a	8.8 a	19.2 ab	83.4 c	15.1 a	30.1 b
Rock cover (%)	50.5 c	6.8 ab	45.6 c	41.5 c	16.6 b	2.1 a	38.1 c	9.7 ab
Ash (%)	—	—	—	—	—	—	0.2 a	17.4 b
Bare ground (%)[4]	79.3 b	14.6 a	85.3 b	85.1 b	70.1 b	15.4 a	81.0 b	68.1 b
Canopy gaps 25–50 cm (%)[5]	9.1 bc	9.9 bc	9.0 bc	11.5 bc	14.1 c	4.3 ab	7.1 bc	0.5 a
Canopy gaps 51–100 cm (%)[5]	19.5 b	14.7 ab	19.1 b	19.5 b	20.9 b	8.8 a	17.0 ab	2.5 a
Canopy gaps 101–200 cm (%)[5]	24.9 c	18.6 bc	25.4 c	23.4 c	19.4 bc	16.8 b	27.3 c	5.1 a
Canopy gaps > 200 cm (%)[5]	22.3 a	24.9 a	21.6 a	23.2 a	14.1 a	61.4 bc	38.9 ab	89.1 c
Basal gaps 25–50 cm (%)	6.8 bc	10.4 c	3.6 ab	7.6 bc	12.5 c	5.4 b	4.4 b	0.3 a
Basal gaps 51–100 cm (%)	15.1 bc	12.1 bc	22.9 c	17.1 c	27.1 c	9.5 abc	8.7 ab	3.0 a
Basal gaps 101–200 cm (%)	26.1 b	20.3 b	25.9 b	29.9 b	20.2 b	14.8 ab	21.3 b	4.1 a
Basal gaps > 200 cm (%)	42.2 ab	42.4 ab	38.6 ab	36.8 ab	21.8 a	64.4 bc	62.7 bc	90.5 c
Average canopy gap (cm)[5]	92.8 a	84.7 a	93.4 a	83.1 a	60.4 a	198.6 bc	110.1 ab	356.0 c
Average basal gap (cm)	117.9 a	115.3 a	119.4 a	108.1 a	79.3 a	224.2 b	167.6 b	367.0 c
No. of plots	18	18	6	6	4	4	6	6

1 Plots with a Utah juniper (*Juniperus osteosperma* [Torr.] Little) tree laying, partially in contact with and parallel to the ground surface, perpendicular to the long axis of the respective plot (across plot along hillslope contour). Tree placed on plot immediately before rainfall simulation.
2 Excludes tree canopy removed immediately before rainfall simulation.
3 Grass and forb canopy cover.
4 Rock, ash, and bare soil.
5 Canopy gaps measured after tree removal for rainfall simulation.

The number of rainfall simulation plots (reps) by yr × treatment × microsite (zone type) combination is shown in Table 2 for Marking Corral and Table 3 for Onaqui. The number of concentrated flow plots (reps) by yr × treatment × microsite (zone type) combination for Yr 0–1 is consistent with those for rainfall simulation plots at both sites. In Yr 2, six concentrated-flow plots were placed in shrub-interspace zones and in tree zones within the burned area at each site and two to three concentrated plots were placed in each zone type within untreated (control) areas at each site.

Vegetation Characterization

Canopy and ground cover by life form were measured using line-point intercept procedures on rainfall simulation and concentrated flow plots (Pierson et al., 2010). Cut trees placed on rainfall and concentrated flow plots were excluded from canopy and ground cover measurements. Canopy and ground cover on Yr 0 and Yr 1 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 (295 points per plot). In Yr 2, canopy and ground cover on concentrated flow 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 (216 points per plot). Percentage cover by cover type for each plot was determined from the number of point contacts or hits for each respective life form divided by the total number of points sampled within the plot. The ground surface roughness for each rainfall and concentrated flow plot was estimated as the average of the standard deviations of the ground surface heights across the line–point transects. The relative ground–surface height at each cover sample point was calculated as the distance between the ground surface and a survey transit level line above the respective sample point.

Gaps between plant canopies and basal cover on rainfall and concentrated flow plots were estimated using the gap-intercept method along the cover line–point transects (Pierson et al., 2010, 2013). Plant canopy and basal gaps exceeding 20 cm were recorded along each transect. Average canopy and basal gap sizes were determined as the mean of all respective gaps measured in excess of 20 cm. Percentages of canopy and basal gaps representing gap classes 25–50, 51–100, 101–200, and > 200 cm were determined for each transect and averaged across the transects on each plot to determine gap-class plot means (Herrick et al., 2005).

Rainfall Simulations

Rainfall was applied at rates of 64 mm · h^{-1} (dry run) and 102 mm · h^{-1} (wet run) for 45 min using methodology described by Pierson et al. (2010, 2013). The dry run was conducted with uniform dry antecedent soil moisture conditions (<10% gravimetric), and the wet run was applied within 30 min following the dry run. The dry-run intensity over 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). Each paired-rainfall simulation was conducted with a Colorado State University–type rainfall simulator (Fig. 2A and B; Holland, 1969; Pierson et al., 2010, 2013; Williams et al., 2014a). The applied simulator design produces rainfall drop diameters within approximately 1 mm of natural rainfall, and the rainfall energy is approximately 70% of that for a natural convective rainfall event (Holland, 1969; Neff, 1979). The 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 (Fig. 2B).

Timed samples of plot runoff and sediment were collected by direct bottle samples of the discharge at the plot outlet at 1-min to 3-min intervals throughout each 45-min rainfall simulation. Runoff from direct rainfall on the large-plot runoff collection troughs (trough catch, see Fig. 2B) was estimated by sampling collection trough runoff before plot-generated runoff occurred (Pierson et al., 2010, 2013). Runoff volume and sediment concentration were measured for each runoff sample by weighing the sample before and after drying at 105°C. Sample weights were adjusted to account for trough catch (Pierson et al., 2010, 2013).

A set of hydrologic response variables was derived for each rainfall simulation. A runoff rate ($mm \cdot h^{-1}$) was calculated for each runoff sample interval as the interval cumulative runoff divided by the interval time duration. The cumulative runoff (mm) from each 45-min simulation was calculated as the integration of runoff rates over the total time of runoff. Erosion variables were calculated solely for plots that generated runoff. Cumulative sediment yield ($g \cdot m^{-2}$) was determined as the integrated sum of sediment collected during runoff and was extrapolated to plot unit area by dividing cumulative sediment by total plot area. A sediment-to-runoff ratio ($g \cdot m^{-2} \cdot mm^{-1}$) was obtained by dividing cumulative sediment yield by cumulative runoff.

Concentrated Flow Simulations

Concentrated overland flow was applied using computer-controlled flow regulators and methodology described by Pierson et al. (2010, 2013) and Williams et al. (2014a). Flow regulators were used to apply release rates of 15, 30, and 45 $L \cdot min^{-1}$ to each concentrated flow plot within 1–2 hours after rainfall simulation in Yr 0 and Yr 1 and on each independent concentrated flow plot in Yr 2. Yr 2 concentrated flow plots were unconfined with respect to width given plot walls were not present (Fig. 2D). Concentrated flow plots in Yr 2 were prewet with a gently misting sprinkler to generate similar surface soil moisture conditions as on Yr 1 plots that received rainfall simulations before concentrated flow experiments. The concentrated flow release rate sequence was 12 min at 15 $L \cdot min^{-1}$, immediately followed by 12 min at 30 $L \cdot min^{-1}$, immediately followed by 12 min at 45 $L \cdot min^{-1}$. Each of the individual flow release rates (15, 30, and 45 $L \cdot min^{-1}$) was applied to each plot from a single location, 4 m upslope of the runoff collection point (Fig. 2C). Concentrated flow was routed through a metal box filled with Styrofoam pellets and released through a 10-cm-wide mesh-screened opening at the base of the box (Fig. 2C). Plot runoff samples were collected at 1- to 2-min intervals for each 12-min flow rate simulation and were processed in the laboratory for runoff and sediment as described for large plots. Runoff and erosion variables for each flow release rate were calculated for an 8-min time period beginning at runoff initiation. The 8-min runoff and sediment variables were derived consistent with those for 45-min rainfall simulations. Concentrated 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 using electrical conductivity probes to track the mean transit time of the salt over a 2-m flow path length (Pierson et al., 2007, 2008, 2009). Flow velocity ($m \cdot s^{-1}$) for each release rate on each plot was calculated by dividing the flow path length (2 m) by the mean of two to three sampled salt travel times ($n = 2$–3 per rate per plot) in seconds.

Statistical Analyses

Statistical analyses were conducted using SAS software, version 9.2 (SAS Institute Inc., 2008). Rainfall-simulation plot data at Marking Corral were analyzed using a mixed model with four treatment levels (control—Yr 0, burned—Yr 1, cut: no downed tree—Yr 1, and cut: downed tree—Yr 1) and two microsite levels (shrub-interspace zone and tree zone). Rainfall simulation data at Onaqui were analyzed in the same manner as those of Marking Corral but included one additional treatment level, mastication—Yr 1. Cover data for concentrated flow plots across Yr 0 and Yr 1 for a site were analyzed in the same manner as those of rainfall-simulation plot data. For Yr 2, cover data for concentrated flow plots by site were analyzed using a mixed model with two treatment levels, control—Yr 2 and burned—Yr 2, and two microsite levels, shrub-interspace zone and tree zone. Concentrated flow runoff and erosion data for each site were analyzed with a repeated measure mixed-model using the treatment and microsite levels specified above for concentrated flow-plot cover data. Flow release rate was the repeated measure for concentrated-flow runoff and erosion analyses, with three levels: 15, 30, and 45 $L \cdot min^{-1}$. Carryover effects of concentrated flow releases were modeled with an autoregressive order 1 covariance structure (Littell et al., 2006). Plot location was designated a random effect and treatment and microsite were considered fixed effects in all respective analyses. Normality and homogeneity were tested before ANOVA using the Shapiro-Wilk test and Levene's test, respectively (SAS Institute Inc., 2008), and deviance from normality was addressed by data transformation. Where necessary, arcsine-square root transformations were used to normalize proportion data (e.g., canopy and ground cover) and logarithmic transformations were used to normalize runoff and erosion data. Backtransformed results are reported. Mean separation was determined using the LSMEANS procedure with Tukey's adjustment (SAS Institute Inc. 2008). All reported significant effects (mean differences and correlations) were tested at the $P < 0.05$ level.

Results

Vegetation

Before tree removal, the intercanopy at both sites (~75% at each site) was relatively denuded of vegetation and ground cover, but the ground surface underneath tree canopies was well protected from erosion. Bare ground (bare soil and rock) within shrub-interspace zones exceeded 60% at Marking Corral (Table 2) and was nearly 80% at Onaqui (Table 3). Vegetation in shrub-interspace zones at Marking Corral was predominantly shrubs (21% cover) and grasses (12% cover) (Table 2) and, at Onaqui, consisted of equal (~5% each) amounts of shrubs, grasses, and forbs (Table 3). Canopy cover in tree zones at Marking Corral was mainly woody debris (13%) with minor amounts of grasses (4%) and shrubs (4%). Tree zone canopy cover at Onaqui consisted almost entirely of grasses (15%). Limited vegetation in the tree zones at Marking Corral resulted in larger average canopy and basal gaps for tree (202–305 cm) versus shrub-interspace (75–125 cm) zones, but the ground surface in tree zone gaps was well covered by litter (Table 2). Average gap lengths between plant canopies and bases at Onaqui were not significantly different for tree (85–115 cm) versus shrub-interspace zones (92–118 cm), but, as at Marking Corral, ground cover by litter protected the soil surface within tree zone gaps (Table 3).

The cut and mastication treatments yielded only minor changes in the understory canopy and ground cover one growing season posttreatment (Tables 2 and 3). With few exceptions, canopy and ground cover on shrub-interspace plots at both sites 1 yr following tree cutting were similar to values measured in Yr 0 controls (Tables 2 and 3). The mastication treatment at Onaqui resulted in limited transfer of tree debris to the intercanopy. Shrub-interspace litter cover (including tree mulch) within the mastication treatment in Yr 1 was not significantly different from litter cover in Yr 0 shrub-interspace plots (Table 3). Herbaceous and shrub canopy cover were twofold greater in shrub-interspace plots of the mastication treatment in Yr 1 relative to Yr 0 controls, but the values were similar with those measured in the cut treatment area (Table 3). Mastication reduced herbaceous cover in tree zones at Onaqui, but tree mulch and litter provided more than 80% ground cover (Table 3).

Fire-induced decreases in canopy and ground cover the first growing season following burning were generally greater in tree than shrub-interspace zones and at Onaqui versus Marking Corral. Tree-zone understory canopy cover at both sites was reduced four- to sixfold by prescribed burning (Tables 2 and 3). Canopy losses from tree zones at Marking Corral were predominantly of tree debris (downed limbs, twigs, etc.) and, at Onaqui, were mostly associated with a 20-fold loss in grass cover. Tree litter cover 1 yr postfire was 1.5- and 2.5-fold less

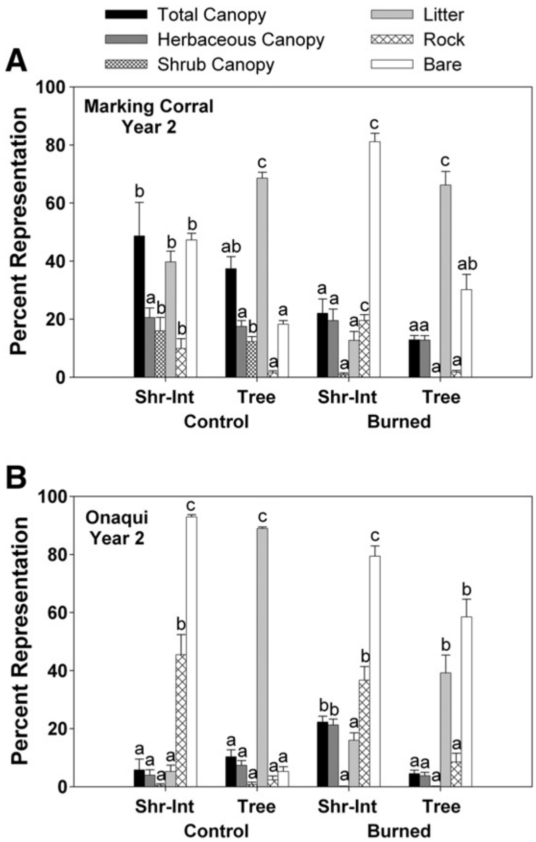

Fig. 3. Canopy, ground cover, and bare ground (bare soil, ash, and rock) representation on untreated (Control) and burned (Burned) shrub-interspace (Shr-Int) and tree (Tree) zone concentrated-flow plots (9 m^2) 2 yr posttreatment at Marking Corral **(A)** and Onaqui **(B)**. Error bars represent standard error. Means within a cover type (e.g., total canopy) followed by different lower case letters are significantly different ($P < 0.05$).

than pretreatment levels at Marking Corral and Onaqui, respectively (Tables 2 and 3). Burning did not increase canopy and basal gaps in tree zones at Marking Corral, although bare ground increased from 6–27% (Table 2). In contrast, fire-induced reductions of vegetation and litter in tree zones at Onaqui resulted in nearly 70% bare ground and a more than threefold increase in average canopy and basal gap lengths (Table 3). Shrub cover 1 yr postfire was nearly 0% across shrub-interspaces at both sites (Tables 2 and 3). Grass canopy cover in shrub-interspaces at Marking Corral was reduced from 12% prefire to 5% a year postfire. Shrub-interspace grass cover at Onaqui was low prefire and was unchanged 1 yr after the fire.

The sites exhibited differing vegetation and ground cover responses to fire by Yr 2 (Fig. 3). Bare ground within burned tree zones (30%) at Marking Corral in Yr 2 was similar to controls (~20%, Fig. 3A) due to retention of tree needle fall, but gaps in canopy and basal cover were greater for the burned (117 and 184 cm) than unburned (56 and 80 cm) plots. Shrub (1%) and grass (10%) canopy cover on burned shrub-interspace plots at Marking Corral remained below control levels (4–20%) in Yr 2. Bare ground in burned shrub-interspace zones (81%) at Marking Corral remained 1.5-fold greater than in the control (47%, Fig. 3A), and the average length of basal gaps was approximately twofold greater for burned (106 cm) versus unburned conditions (62 cm). In contrast to Marking Corral, burned tree-zone plots at Onaqui had approximately twofold less litter (39%) than unburned tree zone plots (90%, Fig. 3B). Average canopy and basal gaps in Yr 2 at Onaqui were not significantly different for burned versus unburned tree zones, but the ground surface within gaps was mostly bare (59%, Fig. 3B). Onaqui shrub-interspace-zone total canopy (22%) and herbaceous (21%) cover were greater 2 yr postfire than in controls (~5% each, Fig. 3B). Extensive bare ground (80%) remained on burned shrub-interspace plots in Yr 2, but the enhanced herbaceous cover on burned plots (Fig. 3B) reduced average canopy (56 cm) and basal gaps (87 cm) by twofold relative to controls (128 and 176 cm, respectively).

Rainfall Simulations

Runoff and erosion responses to rainfall before tree removal were greater from shrub-interspace zones than tree zones across both sites for the wet run (Tables 4 and 5), while dry-run microsite differences occurred solely at Marking Corral (Table 4). At Marking Corral, 24% and 47% of applied dry-run and wet-run rainfall, respectively, was converted to runoff on Yr 0 shrub-interspace plots, whereas less than 5% of applied rainfall was converted to runoff on Yr 0 tree zones for dry and wet runs (Table 4). At Onaqui, 10% and 45% of applied dry-run and wet-run rainfall was converted to runoff on the Yr-0 shrub-interspace plots, and 2–10% of applied dry- and wet-run rainfall exited the control tree-zone plots as runoff (Table 5). Dry-run erosion was generally low (<50 g · m^{-2}) on Yr 0 shrub-interspace and tree-zone plots at both sites (Tables 4 and 5). Wet-run erosion for Yr 0 at both sites was fivefold greater from shrub-interspace (222–296 g · m^{-2}) than tree zones

Table 4
Rainfall, runoff, and sediment response variables for rainfall simulations (13 m^2) 1 yr before treatments (Control—Yr 0) and in cut (with and without downed tree) and burned areas 1 yr posttreatment (Yr 1) at Marking Corral. Means within a row followed by a different lowercase letter are significantly different ($P < 0.05$).

Marking Corral study site	Control—Yr 0		Cut, no downed tree—Yr 1	Cut, downed tree[1]—Yr 1	Burned—Yr 1	
Rainfall simulation variable	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Shrub-interspace zone	Shrub-interspace zone	Tree zone
Dry Run Simulation (64 mm · h^{-1}, 45 min)						
Applied rain (mm)	45 a	46 a	44 a	47 a	46 a	45 a
Cumulative runoff (mm)	11 b	1 a	1 a	2 a	3 a	1 a
Cumulative sediment (g · m^{-2})[2]	45 c	18 b	8 ab	16 ab	25 b	6 a
Sediment/runoff (g · m^{-2} · mm^{-1})[2]	4.11 a	10.35 bc	8.55 abc	7.61 ab	7.54 ab	21.94 c
Percent of plots with runoff[3]	100	50	50	83	80	67
No. of plots	11	12	6	6	5	6
Wet Run Simulation (102 mm · h^{-1}, 45 min)						
Applied rain (mm)	81 a	81 a	83 a	86 a	80 a	80 a
Cumulative runoff (mm)	38 b	4 a	15 ab	24 b	34 b	11 a
Cumulative sediment (g · m^{-2})[2]	222 b	36 a	117 b	195 b	346 b	78 a
Sediment/runoff (g · m^{-2} · mm^{-1})[2]	5.75 a	6.13 a	7.13 a	8.58 a	9.56 a	7.15 a
Percent of plots with runoff[3]	100	67	100	100	100	100
No. of plots	10	9	6	6	6	6

[1] Plots with a single-leaf pinyon (*Pinus monophylla* Torr. & Frem.) or Utah juniper (*Juniperus osteosperma* [Torr.] Little) tree laying, partially in contact with and parallel to the ground surface, perpendicular to the long axis of the respective plot (across plot along hillslope contour). Tree placed on plot immediately before rainfall simulation.
[2] Means based solely on plots that generated runoff.
[3] Not included in statistical analysis.

Table 5
Rainfall, runoff, and sediment response variables for rainfall simulations (13 m^2) 1 yr before treatments (Control—Yr 0) and in cut (with and without downed tree), mastication, and burned areas 1 yr posttreatment (Yr 1) at the Onaqui site. Means within a row followed by a different lowercase letter are significantly different ($P < 0.05$).

Onaqui study site	Control—Yr 0		Cut, no downed tree—Yr 1	Cut, downed tree[1]—Yr 1	Masticated—Yr 1		Burned—Yr 1	
Rainfall simulation variable	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Shrub-interspace zone	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Tree zone
Dry Run Simulation (64 mm · h^{-1}, 45 min)								
Applied rain (mm)	49 a	50 a	53 a	51 a	55 a	56 a	45 a	48 a
Cumulative runoff (mm)	5 ab	1 a	3 ab	3 ab	2 ab	1 a	3 ab	11 b
Cumulative sediment (g · m^{-2})[2]	37 a	13 a	29 a	46 a	18 a	13 a	41 a	448 b
Sediment/runoff (g · m^{-2} · mm^{-1})[2]	12.73 a	14.18 a	15.56 a	16.32 a	11.72 a	9.14 a	10.52 a	35.70 b
Percent of plots with runoff[3]	94	78	100	67	100	75	60	100
No. of plots	17	18	6	6	4	4	5	5
Wet Run Simulation (102 mm · h^{-1}, 45 min)								
Applied rain (mm)	87 a	93 a	94 a	85 a	76 a	77 a	73 a	84 a
Cumulative runoff (mm)	39 c	9 a	37 c	29 bc	22 ab	9 ab	31 c	43 c
Cumulative sediment (g · m^{-2})[2]	296 b	66 a	310 b	332 b	158 a	69 a	491 b	1893 c
Sediment/runoff (g · m^{-2} · mm^{-1})[2]	7.64 a	7.22 a	8.64 ab	11.39 ab	9.37 ab	9.66 ab	16.01 b	44.67 c
Percent of plots with runoff[3]	100	100	100	100	100	100	100	100
No. of plots	17	16	6	6	4	4	5	5

[1] Plots with a Utah juniper (*Juniperus osteosperma* [Torr.] Little) tree laying, partially in contact with and parallel to the ground surface, perpendicular to the long axis of the respective plot (across plot along hillslope contour). Tree placed on plot immediately before rainfall simulation.
[2] Means based solely on plots that generated runoff.
[3] Not included in statistical analysis.

(36–66 g · m^{-2}; Tables 4 and 5). Wet-run erosion across all control plots in Yr 0 was linearly correlated with runoff and generally increased where bare ground exceeded 50–60% in the intercanopy (Figs. 4 and 5).

Runoff and erosion were minimally affected by tree cutting at the sites but were lower for the wet-run shrub-interspaces in the mastication treatment relative to controls. Placing cut trees across shrub-interspace zone plots did not reduce runoff and erosion at either site (Tables 4 and 5). Dry-run runoff and erosion in the cut treatment at Marking Corral in Yr 1 were significantly less than measured on control plots but were not significantly different for the cut—no downed tree versus the cut—downed tree plots (Table 4). Runoff and erosion for the wet-run simulations were similar across all shrub-interspace plots in Yr 0 control and Yr 1 cut treatments at a site (Tables 4 and 5). In contrast, wet-run runoff (22 mm) and erosion (158 g · m^{-2}) on shrub-interspace plots in the mastication treatment at Onaqui were twofold less relative to shrub-interspace plots in the Yr 0 control and the Yr 1 cut—no downed tree treatment (~40 mm and 300–310 g · m^{-2}, Table 5). The mastication treatment did not affect runoff and erosion rates from tree zone plots at Onaqui.

Runoff and erosion responses in the burn treatments differed for the two sites. Dry-run runoff was fourfold less for burned versus unburned shrub-interspaces at Marking Corral and was negligible for burned and unburned tree zones at the site (Table 4). Dry-run erosion from burned shrub-interspace and tree (6–25 g · m^{-2}) zones at Marking Corral was twofold to threefold less than for respective unburned plots (18–45 g · m^{-2}; Table 4), but, overall, the dry-run erosion rates at Marking Corral were low. Wet-run runoff and erosion by microsite for Yr 1 burned and Yr 0 control plots were not significantly different at Marking Corral due in part to the variability associated with each treatment × microsite combination (Table 4, Fig. 4). At Onaqui, dry-run runoff and erosion were similar for burned and unburned shrub-interspace plots (Table 5) but were 11-fold and more than 30-fold greater for burned (11 mm and 448 g · m^{-2}) versus unburned (1-mm and 13 g · m^{-2}) tree zones, respectively. Cumulative runoff and sediment yield from wet-run simulations in shrub interspaces at Onaqui were not significantly different for burned versus unburned conditions, but the amount of sediment per unit of runoff on burned shrub-interspace plots was twice that of the unburned shrub-interspaces (Table 5). Wet-run runoff and erosion from burned tree zones at Onaqui were fourfold and nearly 30-fold greater, respectively, than in unburned tree zones, and the relationships of sediment yield with runoff and bare ground for tree zones were substantially altered by burning (Fig. 5). Burning at Onaqui

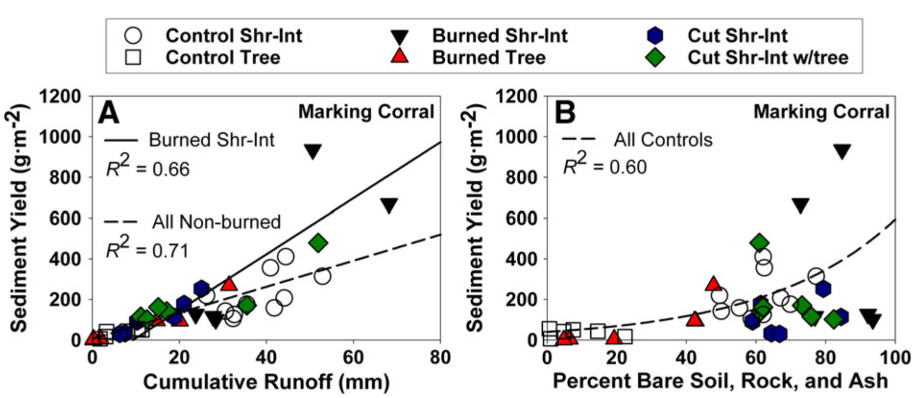

Fig. 4. Wet-run rainfall simulation (102 mm · h^{-1}, 45 min, 13 m^2) cumulative sediment yield versus runoff **(A)** and bare ground (bare soil, rock, and ash, **B**) measured on shrub-interspace (Shr-Int) and tree (Tree) zone plots 1 yr before treatments (Control) and in burned and cut areas 1 yr posttreatment at Marking Corral. Plots in cut treatment with a downed tree (w/tree) contain an unburned pinyon (*Pinus monophylla* Torr. & Frem.) or Utah juniper (*Juniperus osteosperma* [Torr.] Little) tree lying, partially in contact with and parallel to the ground surface, perpendicular to the long axis of the respective plot (across plot along hillslope contour).

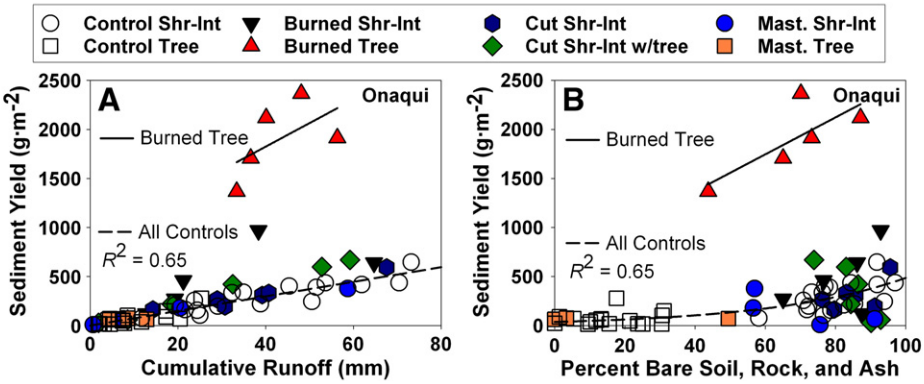

Fig. 5. Wet-run rainfall simulation (102 mm · h^{-1}, 45 min, 13 m^2) cumulative sediment yield versus runoff **(A)** and bare ground (bare soil, rock, and ash, **B**) measured on shrub-interspace (Shr-Int) and tree (Tree) zone plots 1 yr before treatments (Control) and in burned, cut, and mastication (Mast.) areas 1 yr posttreatment at Onaqui. Plots in cut treatment with a downed tree (w/tree) contain an unburned Utah juniper (*Juniperus osteosperma* [Torr.] Little) tree lying, partially in contact with and parallel to the ground surface, perpendicular to the long axis of the respective plot (across plot along hillslope contour).

essentially rendered the hydrologic and erosion response of tree zone plots elevated relative to those measured in the degraded shrub-interspace zone.

Concentrated Flow Simulations

Concentrated-flow runoff and erosion for each flow release rate in Yr 0 controls were higher for shrub-interspace than tree zones at both sites and were not reduced by placement of cut trees within shrub-interspace zones (Tables 6 and 7). Runoff and erosion from the combined 15–45 L · m^{-1} flow releases in Yr 0 controls at Marking Corral were approximately twofold and fivefold greater, respectively, for shrub-interspace (514 L and 1 317 g) than tree (303 L and 241 g) zones. The velocity of concentrated flow by flow release rate was also higher for shrub-interspace than tree zones in Yr 0 at Marking Corral (Table 6). At Onaqui, runoff and erosion from the combined 15–45 L · m^{-1} flow releases in Yr 0 controls were both twofold greater for shrub-interspace (537 L and 2 236 g) than tree (280 L and 1 086 g) zones. Concentrated flow velocity on Yr 0 control shrub-interspace zones at Onaqui was not significantly different from that measured on the control tree zones (Table 7). Runoff from Yr 0 control plots by microsite was similar for the two study sites, but erosion by microsite was generally two- to fivefold greater at Onaqui than Marking Corral (Tables 6 and 7). Microsite differences in concentrated flow runoff and erosion for controls plots persisted in Yr 2 at Marking Corral but were not evident in Yr 2 at Onaqui. The cutting treatment at Marking Corral had no effect on runoff from concentrated flow processes for shrub-interspace zones with or without a downed tree, but plots within the cut treatment at Marking Corral generated six- to ninefold more erosion and had higher flow velocities for the 45 L · m^{-1} release than control shrub-interspace zone plots (Table 6). The cut treatment at Onaqui had no significant effect on runoff or erosion from shrub-interspace zones (Table 7). Tree mastication had no effect on cumulative runoff and erosion from shrub-interspace or tree zones, but the velocity of concentrated flow in tree zones was reduced by the mulch cover.

Table 6
Runoff and sediment variables by flow release rate for concentrated flow experiments (9 m^2) in control plots 1 yr before treatment (Yr 0) and 2 yr posttreatment (Yr 2), cut plots 1 yr posttreatment (Yr 1), and burned plots 1 (Yr 1) and 2 yr (Yr 2) posttreatment at Marking Corral. Means within a row followed by a different lowercase letter are significantly different ($P < 0.05$).

Marking Corral study site	Release rate (L · min^{-1})	Control		Cut, no downed tree	Cut, downed tree[1]	Burned	
Concentrated flow variable		Shrub-interspace zone	Tree zone	Shrub-interspace zone	Shrub-interspace zone	Shrub-interspace zone	Tree zone
Yr 0 (Control) and Yr 1 (Burned and Cut)							
Cumulative runoff	15	51 b	4 a	31 b	38 b	57 b	28 ab
(L)	30	173 b	78 a	141 b	148 b	177 b	97 a
	45	290 b	221 a	284 b	300 b	300 b	234 a
Cumulative sediment	15	26 a	–	40 a	216 bc	142 b	499 c
(g)[2]	30	334 b	121 a	1 134 c	2 178 c	2 240 c	972 b
	45	957 b	120 a	6 514 c	8 942 c	4 845 c	1 875 b
Flow velocity	15	0.06 a	–	0.06 a	0.08 a	0.07 a	0.08 a
(m · s^{-1})[2]	30	0.08 b	0.04 a	0.13 b	0.12 b	0.12 b	0.12 b
	45	0.12 b	0.04 a	0.21 c	0.16 bc	0.17 bc	0.12 b
Yr 2 (Control and Burned)							
Cumulative runoff	15	11 a	0 a	–	–	20 a	28 a
(L)	30	109 b	9 a	–	–	94 b	90 b
	45	217 b	69 a	–	–	197 b	176 b
Cumulative sediment	15	34 a	–	–	–	45 a	1 227 a
(g)[2]	30	1 970 b	19 a	–	–	254 b	1 496 b
	45	2 410 b	375 a	–	–	911 ab	1 158 ab
Flow velocity	15	0.08 a	–	–	–	0.07 a	0.11 a
(m · s^{-1})[2]	30	0.14 b	0.05 a	–	–	0.11 ab	0.15 b
	45	0.18 b	0.07 a	–	–	0.16 ab	0.15 ab

[1] Plots with a single-leaf pinyon (*Pinus monophylla* Torr. & Frem.) or Utah juniper (*Juniperus osteosperma* [Torr.] Little) tree laying, partially in contact with and parallel to the ground surface, perpendicular to the long axis of the respective plot (across plot along hillslope contour). Tree placed on plot immediately before rainfall simulation.
[2] Mean based solely on plots that generated runoff.

Table 7
Runoff and sediment variables by flow release rate for concentrated flow experiments (9 m^2) in control plots 1 yr before treatment (Yr 0) and 2 yr posttreatment (Yr 2), cut and masticated plots 1 yr posttreatment (Yr 1), and burned plots 1 (Yr 1) and 2 yr (Yr 2) posttreatment at Onaqui. Means within a row followed by a different lower case letter are significantly different ($P < 0.05$).

Onaqui study site	Release rate ($L \cdot min^{-1}$)	Control		Cut, no downed tree	Cut, downed tree[1]	Masticated		Burned	
Concentrated flow variable		Shrub-interspace zone	Tree zone	Shrub-interspace zone	Shrub-interspace zone	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Tree zone
Yr 0 (Control) and Yr 1 (Burned, Cut, and Mastication)									
Cumulative runoff (L)	15	70 b	11 a	58 b	49 b	41 b	2 a	55 b	70 b
	30	175 b	66 a	185 b	177 b	142 b	52 a	161 b	176 b
	45	292 b	203 a	301 b	288 b	256 ab	182 a	279 b	287 b
Cumulative sediment (g)[2]	15	200 b	60 a	238 b	189 b	193 b	—	243 b	2 800 c
	30	605 b	225 a	800 bc	1007 bc	715 bc	228 a	1 372 c	3 422 d
	45	1 431 b	801 a	1 324 b	1 794 b	1 049 ab	776 a	2 398 bc	3 840 c
Flow velocity ($m \cdot s^{-1}$)[2]	15	0.07 ab	0.05 a	0.10 ab	0.06 ab	0.05 a	—	0.07 ab	0.11 b
	30	0.11 ab	0.07 a	0.14 bc	0.09 ab	0.08 ab	—	0.13 bc	0.19 c
	45	0.17 bc	0.14 b	0.17 bc	0.09 ab	0.10 b	0.04 a	0.17 bc	0.26 c
Yr 2 (Control and Burned)									
Cumulative runoff (L)	15	5 a	0 a	—	—	—	—	7 a	68 b
	30	85 a	29 a	—	—	—	—	59 a	163 b
	45	224 ab	145 a	—	—	—	—	208 ab	255 b
Cumulative sediment (g)[2]	15	34 a	—	—	—	—	—	102 a	3 411 b
	30	1 109 ab	1 237 ab	—	—	—	—	702 a	3 904 b
	45	4 093 a	2 426 a	—	—	—	—	1 920 a	3 571 a
Flow velocity ($m \cdot s^{-1}$)[2]	15	0.10 ab	—	—	—	—	—	0.07 a	0.16 b
	30	0.12 ab	0.10 a	—	—	—	—	0.11 a	0.19 b
	45	0.20 b	0.14 a	—	—	—	—	0.13 a	0.21 b

[1] Plots with a Utah juniper (*Juniperus osteosperma* [Torr.] Little) tree laying, partially in contact with and parallel to the ground surface, perpendicular to the long axis of the respective plot (across plot along hillslope contour). Tree placed on plot immediately before rainfall simulation.
[2] Mean based solely on plots that generated runoff.

Concentrated-flow runoff and erosion were greater for burned than unburned conditions on tree zones at both sites, but runoff and erosion responses in burned shrub-interspaces varied by site (Tables 6 and 7). Cumulative runoff from burned concentrated flow plots was similar to that of unburned plots at Marking Corral in Yr 1 (Table 6). However, runoff from 30 and 45 $L \cdot m^{-1}$ flow releases at Marking Corral on Yr 2 burned tree zones were 10-fold and twofold more than on Yr 2 control tree zones. Concentrated flow velocity by flow release rate and cumulative erosion from all release rates at Marking Corral were, on average, threefold and more than 10-fold greater for burned than control tree zones across all years and were similar to values measured on shrub-interspace controls (Table 6). Erosion from combined 15–45 $L \cdot m^{-1}$ flow releases in burned shrub-interspaces (7 227 g) at Marking Corral exceeded that of controls (1 317 g) for Yr 1 only (Table 6). At Onaqui, concentrated flow runoff, erosion, and velocity were greater for burned versus unburned tree zones but were generally similar across burned and unburned conditions in shrub-interspace zones (Table 7). Runoff from the combined 15–45 $L \cdot m^{-1}$ flow releases at Onaqui was twofold more for Yr 1 burned (533 L) than Yr 0 control (280 L) tree zones and was consistent with that measured for Yr 1 burned (495 L) and Yr 0 control shrub-interspaces (537 L, Table 7). The combined 15–45 $L \cdot m^{-1}$ flow releases at Onaqui generated ninefold more erosion from Yr 1 burned (10 062 g) than Yr 0 control (1 086 g) tree-zone plots, and flow velocity by flow release rate for Yr 1 burned tree plots at the site was twice that measured on Yr 0 control tree zones (Table 7). Erosion from the combined 15–45 $L \cdot m^{-1}$ flow releases on shrub-interspace plots at Onaqui was slightly elevated for Yr 1 burned (4 013 g) versus Yr 0 controls (2 236 g). The velocity of concentrated flow was similar across the three flow release rates for Yr 1 burned and Yr 0 control shrub-interspace plots (Table 7). Higher total runoff and erosion and flow velocity for burned versus unburned tree zones at Onaqui persisted through Yr 2, but the magnitude of erosion differences between treatments was reduced to twofold (Table 7). Total runoff and erosion and the velocity of concentrated flow at Onaqui in Yr 2 were all two- to fourfold greater for burned tree zones than burned shrub-interspaces. No significant differences in runoff and erosion were detected for burned versus unburned shrub-interspaces in Yr 2 at Onaqui (Table 7).

Discussion

Pretreatment Conditions and Hydrologic Responses

Before tree removal, both sites exhibited hydrologically stable tree islands surrounded by unstable shrub-interspace zones. Extensive bare interspace within shrub-interspace zones allowed overland flow during high-intensity rainfall events to accumulate in concentrated flow paths with high sediment detachment and transport capacity (Pierson et al., 2008, 2009, 2010; Al-Hamdan et al., 2012b, 2013; Pierson et al., 2013; Williams et al., 2014a). The linear relationship of runoff with soil erosion for the simulated high-intensity storms at both sites (Figs. 4A and 5A) clearly indicates that the degraded shrub-interspace zones were capable of generating substantial erosion during intense rainfall events. Erosion under simulated rainfall increased at both sites mainly where intercanopy bare ground exceeded 50–60% (Figs. 4B and 5B). The high rates of erosion measured within the intercanopies suggest both sites represented rapidly eroding conditions and that continued woodland succession at the sites would likely perpetuate substantial soil loss and further degradation over decadal or longer time scales (Wilcox, 1994; Wilcox et al., 1996; Davenport et al., 1998; Wilcox et al., 2003; Williams et al., 2014a).

Effects of Mechanical Treatments

Results from experiments in the cut treatments indicate cut-downed pinyon and juniper trees afford minimal surface erosion reduction within the first year following cutting. No substantial changes in understory vegetation and ground cover were expected over the first growing season following cutting (Bates et al., 1998, 2000, 2005, 2007, 2011; Miller et al., 2014). We anticipated that the only short-term hydrologic benefit to the cut treatment would be interception of rainfall and overland flow by the downed trees. We hypothesized that downed trees would reduce erosion through intercepting rainfall, trapping overland flow, and reducing the erosive energy of runoff (Hastings et al., 2003). Shrub-interspace runoff and erosion under simulated rainfall were reduced for the cut versus control treatments solely for the lower intensity dry run and only at Marking Corral (Tables 4 and 5). The cut treatment effect was consistent at that site regardless of whether the shrub-interspace plot had a downed tree or not (Table 4). Rock cover was the only significant cover difference between the cut (~10% rock) and control (38% rock) areas at Marking Corral (Table 2). The higher rock cover within the degraded control shrub-interspace plots may have reduced infiltration relative to the shrub-interspace plots in the cut treatment (Wilcox et al., 1988; Abrahams and Parsons, 1991; Pierson et al., 2010), although both treatment areas had nearly equal amounts of bare ground (bare soil and rock, Table 3).

We also observed overland flow on plots in the cut treatment at both sites tended to concentrate through narrow breaks in contact of the soil surface with downed trees. At Marking Corral, these concentrated flow paths typically incised and were enriched with flow-detached sediment, as evident by higher concentrated flow erosion relative to controls (Table 6). Concentrated flow erosion at Marking Corral was also higher for shrub-interspace zones in the cut—no downed tree treatment (Table 6). We therefore attribute the overall higher shrub-interspace zone concentrated-flow erosion rates in the cut treatment at Marking Corral to the greater bare soil (60% bare) in cut versus control treatment (26% bare) (Wilcox, 1994; Wilcox et al., 1996; Reid et al., 1999; Pierson et al., 2010, 2013; Williams et al., 2014a). Concentrated overland flow in the cut-downed tree treatment at Onaqui tended to route through downed trees similarly to that at Marking Corral, but concentrated flow erosion was not significantly different for the cut versus control treatments (Table 7). Bare soil and ground cover were similar across the control and cut shrub-interspace plots at that site (Table 3). Overall, overland flow under rainfall and from released concentrated flow formed relatively contiguous flow paths through extensive bare areas within shrub-interspaces regardless of the treatment.

Our results are consistent with a similar recent study at a western juniper–encroached sagebrush site in Idaho, United States (Pierson et al., 2013). Pierson et al. (2013) found overland flow routed through voids in contact of downed juniper trees with the soil surface and that the downed trees provided minimal ground surface protection from concentrated overland flow processes within the immediately postcut period (Pierson et al., 2013). Studies of postfire mitigation have found that incorrectly installed contoured-log treatments can concentrate overland flow and promote soil erosion (Wagenbrenner et al., 2006; Robichaud et al., 2008, 2010). We observed a similar effect of downed trees in the cut treatment at Marking Corral but anticipate a reduction in this effect as trees and debris settle over time and ground cover increases (Bates et al., 1998, 2000; Hastings et al., 2003; Bates et al., 2005, 2007; Pierson et al., 2007; Bates et al., 2011; Miller et al., 2014; Roundy et al., 2014a).

Runoff and erosion were minimally affected by mulch in the mastication treatment as applied in this study (Tables 5 and 7). The mastication treatment did not substantially distribute the shredded tree debris (mulch) within the intercanopy (<5% cover by mulch). Mulch typically resided within the area immediately adjacent to a shredded tree, resulting in nearly 80% ground cover of mulch in tree zones. Runoff and erosion from the high-intensity, wet-run storm were both nearly twofold lower for shrub-interspace zone plots in the mastication treatment relative to those in the control (Table 5). However, we attribute the different runoff and erosion responses between the control and mastication treatments to differences in herbaceous cover rather than mulch deposition within the shrub-interspace zones. Herbaceous cover on shrub-interspace zone plots at Onaqui was 25% within the mastication treatment and 11% in the control (Table 3). This cover response was unexpected 1 yr posttreatment (Roundy et al., 2014a) but suggests the treatment may improve hydrologic function without the mulch effect pending a favorable herbaceous cover response. Pierson et al. (2014) and Cline et al. (2010) found manual application of tree mulch to interspace small plots (0.5 m^2) at Onaqui increased infiltration and decreased erosion by four- to fivefold from the same simulated rainfall events as used in this study. Those studies suggest that woodland mastication treatments aimed at reducing runoff and erosion should focus on distributing masticated tree debris and mulch throughout the intercanopy. However, the potential for tree debris to negatively impact intercanopy herbaceous recruitment should also be considered with regards to slash and debris management (Bates et al., 1998; Brockway et al., 2002; Bates et al., 2005, 2007; Bates and Svejcar, 2009). Recent multisite studies in the Great Basin have reported favorable herbaceous response to tree mastication within 2 to 3 yr post-treatment, with limited negative impacts on shrub and herbaceous vegetation in the immediate post-treatment period (Bybee, 2013; Roundy et al., 2014a). The large decline in shrub density after mastication in this study may represent an extreme case and did not result in a significant reduction in total shrub canopy cover (Table 3).

Effects of Prescribed Burning

Differences in tree-zone burn conditions for the two sites elicited differing runoff and erosion responses in the year following fire. The fire at each site substantially reduced tree litter cover and depth (Pierson et al., 2014), but burned trees retained 25–50% and 0–25% of canopy needles at Marking Corral and Onaqui, respectively. Subsequent needle cast from Yr 0 through Yr 2 at Marking Corral limited bare ground within tree zones to less than 30%. Bare ground in tree zones at Onaqui was approximately 70% in Yr 1 and 60% in Yr 2. Surface soils (0- to 5-cm depth) in tree zones were strongly water repellent before and after fire (Pierson et al., 2014). Litter retention and accumulation in tree zones after 1 yr at Marking Corral protected the soil surface and likely limited fire effects on runoff and erosion under intense rainfall (Table 4; Pannkuk and Robichaud, 2003; Cerdà and Doerr, 2008; Robichaud et al., 2013). In most cases, bare ground on tree zone plots at Marking Corral in Yr 1 was less than 20%, and the relationships of sediment yield with runoff and bare ground for burned tree zone plots were consistent with those for control tree zone plots at the site (Fig. 4). The persistence of extensive bare ground, strong soil water repellency, and readily entrainable sediment in burned tree zones at Onaqui resulted in extreme increases in tree zone runoff and erosion relative to all unburned plots (Fig. 5, Table 5; Pierson et al., 2009; Al-Hamdan et al., 2012b; Williams et al., 2014b; Al-Hamdan et al., 2015). Well-incised flow paths were easily discernible on burned tree zone plots at Onaqui following the wet-run rainfall simulations. Formation of well-defined concentrated flow paths during the wet-run promoted a sixfold increase in the sediment-to-runoff ratio and a nearly 30-fold increase sediment yield from tree zones (Table 5).

Differences in burned conditions for the two sites also influenced runoff and erosion from concentrated flow simulations (Tables 6 and 7). The subtle differences in litter cover for burned versus unburned conditions in tree zones at Marking Corral did not affect runoff in Yr 1 concentrated flow simulations. However, Yr 1 erosion from the combined 15–45 L · m^{-1} flow releases was nearly 15-fold higher for burned than control tree zones (Table 6). Litter depth at Marking Corral was reduced from 40 mm prefire to 23 mm in Yr 1 (Pierson et al., 2014). The reduction in both the vertical and horizontal components of litter in the tree zones at Marking Corral limited resistance to erosion associated with high rates of concentrated runoff. This suggests that although burned tree zones were reasonably protected against high-intensity rainfall, they were vulnerable to high rates of soil erosion from rare, extreme runoff events 1 yr postfire (Benavides-Solorio and MacDonald, 2005; Pierson et al., 2008, 2009, 2011; Robichaud et al., 2013; Williams et al., 2014b). At Onaqui, Yr 1 total runoff and erosion from concentrated flow experiments were two- and ninefold greater for the burned versus unburned tree zones and two- to threefold greater than burned tree zones at Marking Corral (Tables 6 and 7). Two yr postfire, litter depth was more than twofold lower on burned than unburned tree coppices at Marking Corral (Pierson et al., 2014) and total runoff and erosion from concentrated flow experiments in burned, litter-depleted tree zones at the site were both elevated relative to unburned controls (Table 6). As at Marking

Corral, total runoff and erosion and concentrated flow velocity in burned tree zones at Onaqui remained amplified relative to controls, but the magnitude of erosion responses was much higher for burned and control tree zones at Onaqui (Tables 6 and 7). These results suggest differences in burn conditions and inherent differences in soil erodibility are important in predicting woodland microsite-scale erosion responses to burning (Williams et al., 2014b). Furthermore, fire-induced soil erosion vulnerability at the microsite scale may be masked in the first few years postfire without extreme runoff events (Robichaud et al., 2013).

Burning increased short-term vulnerability to soil erosion within shrub-interspace zones, but the effects were buffered after two growing seasons. Bare ground was high within shrub-interspace zones at both sites prefire and was slightly greater postfire at Marking Corral (Tables 3 and 4). Shrub-interspace runoff and erosion from wet-run rainfall simulations were similar for burned and unburned treatments across both sites in Yr 1, but erodibility, as indicated by the sediment-to-runoff ratio, was highest for burned shrub-interspace plots at Onaqui (Tables 4 and 5). Erodibility of shrub-interspaces in the burn Yr 1 at Marking Corral was approximately twofold higher than measured for Yr 0 controls, but the difference was not significant due to variability in measured erosion from burned plots (Fig. 4). The overall effects of burning on soil erodibility within the intercanopy are evident in results from the concentrated flow experiments (Tables 6 and 7). Concentrated flow runoff and flow velocity on shrub-interspaces were not significantly altered by burning at either site. Shrub-interspace zone erosion from concentrated flow increased 1 yr following the burn at both sites (Tables 6 and 7). The increase in erosion without increases in cumulative runoff or flow velocity suggests that the extensive bare ground within burned shrub-interspace zones at each site was more erodible than before the fire. The fire effect on erodibility in shrub-interspace plots was dampened by Yr 2. In Yr 2, shrub-interspace runoff, flow velocity, and erosion for concentrated flow experiments were similar across burned and control treatments and total erosion from burned shrub-interspaces were similar to or less than those of control tree zones (Tables 6 and 7). We attribute the improved overall hydrologic function of burned shrub-interspaces in Yr 2 to the recruitment of herbaceous cover at Onaqui and forb cover at Marking Corral (Fig. 3; Pierson et al., 2009, 2013; Williams et al., 2014a). Williams et al. (2014a) also reported favorable herbaceous recruitment and improved intercanopy hydrologic function 2 yr following fire in a western juniper–encroached sagebrush site. The short-term results from both sites suggest that tree removal by burning has potential to improve long-term hydrologic function on degraded, pinyon- and/or juniper-encroached sites through recruitment of intercanopy herbaceous vegetation (Pierson et al., 2007; Bates and Svejcar, 2009; Bates et al., 2011, 2014; Williams et al., 2014a; Miller et al., 2014; Roundy et al., 2014a, 2014b).

The prolonged bare conditions and enhanced hydrologic vulnerability at Onaqui are not uncommon for woodlands burned at moderate to high severity (Madsen et al., 2011; Pierson et al., 2013; Williams et al., 2014a). Several studies have reported persistence of strong soil water repellency following burning on pinyon and juniper woodlands (Roundy et al., 1978; Madsen et al., 2011; Pierson et al., 2013; Williams et al., 2014a). In a laboratory experiment, Madsen et al. (2012) found soil water repellency in soils acquired from underneath burned Utah juniper trees reduced herbaceous seedling emergence and survival by decreasing soil moisture availability. Madsen et al. (2011) found that infiltration, soil water content, and understory cover following burning of a Great Basin pinyon and juniper woodland were strongly correlated with presence of strong soil water repellency. The study further reported herbaceous plant productivity under burned trees was limited by repellency-induced moisture deficits in surface soils. Williams et al. (2014a) also reported limited herbaceous recovery and prolonged hydrologic vulnerability in burned tree zones at a western juniper woodland burned by wildfire. As in our study, they did not measure soil moisture over time but did report strong soil water repellency in surface soils under burned trees 1 and 2 yr postfire. Pierson et al. (2014) reported persistence of strong water repellent conditions underneath burned Utah juniper at the Onaqui site within the same study area and period of this study. Herbaceous recovery on burned tree zones at Onaqui in this study was likely related to persistence of strong soil water repellency postfire, minimal survival of herbaceous plants during burning, and fire consumption of the herbaceous seed bank (Allen et al., 2008; Sheley and Bates, 2008; Bates et al., 2011; Miller et al., 2014). Regardless, the delayed plant recruitment in burned tree zones at Onaqui and persistence of strong water repellency and hydrologic vulnerability are consistent with other studies of moderate- to high-severity woodland burns and suggest that land managers should expect similar conditions following high-severity burns in woodland tree zones.

Management Implications

This study suggests that woodland tree removal treatments targeting erosion reduction should focus on establishment and retention of intercanopy vegetation and ground cover. We measured high rates of runoff and erosion from simulated rainfall and overland flow in degraded intercanopies at two late-succession woodlands in the Great Basin. The results demonstrate that woodland encroachment can facilitate conditions of rapid runoff during high-intensity rain events and promote organization of overland flow into concentrated flow paths with high sediment detachment and transport capacity. Cyclical high-erosion events associated with these processes promote long-term loss of critical soil resources when perpetuated over decades or longer time periods. Our results indicate that cross-felled pinyon and Utah juniper trees do not necessarily reduce runoff and erosion in the immediate postcut period, but cutting and mastication treatments may reduce runoff and erosion where the treatments elicit increased intercanopy herbaceous cover. Results further indicate that short-term hydrologic and erosional impacts of tree removal by burning are strongly related to the degree in which fire alters prefire vegetation and soils and to site-specific soil erodibility. Burning poses increased short-term erosion risk, particularly for tree islands, but erosion may be reduced over the long-term where burning enhances intercanopy vegetation and ground cover. Amplified runoff and erosion in tree islands after burning may persist for 3 or more yr postfire, but tree islands often occupy much less of the woodland total area than the intercanopy. Finally, our study in context with the literature suggests that mechanical tree removal or low-severity burns should be considered when more rapid (i.e., within several years) reduction in soil erosion is desired on highly erodible woodland sites.

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