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Original Research

Vegetation, Hydrologic, and Erosion Responses of Sagebrush Steppe 9 Yr Following Mechanical Tree Removal☆

C. Jason Williams [a,*], Frederick B. Pierson [b], Patrick R. Kormos [c,d], Osama Z. Al-Hamdan [e], Sayjro K. Nouwakpo [f], Mark A. Weltz [g]

[a] Research Hydrologist, Southwest Watershed Research Center, US Department of Agriculture (USDA)—Agricultural Research Service (ARS), Tucson, AZ 85719, USA
[b] Research Leader and Supervisory Research Hydrologist, Northwest Watershed Research Center, USDA-ARS, Boise, ID 83712, USA
[c] Currently, Research Hydrologist, Colorado Basin River Forecast Center, US Department of Commerce (USDC) —National Oceanic and Atmospheric Administration and National Weather Service, Salt Lake City, UT 84116, USA
[d] Formerly, Research Hydrologist, Northwest Watershed Research Center, USDA-ARS, Boise, ID 83712, USA
[e] Assistant Professor, Department of Civil and Architectural Engineering, Texas A&M University-Kingsville, Kingsville, TX 78363, USA
[f] Research Assistant Professor, Natural Resources and Environmental Science, University of Nevada-Reno, Reno, NV 89557, USA
[g] Research Leader, Great Basin Rangelands Research Unit, USDA-ARS, Reno, NV 89512, USA

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ABSTRACT

Land managers across the western United States are faced with selecting and applying tree-removal treatments on pinyon (*Pinus* spp.) and juniper (*Juniperus* spp.) woodland-encroached sagebrush (*Artemisia* spp.) rangelands, but current understanding of long-term vegetation and hydrological responses of sagebrush sites to tree removal is inadequate for guiding management. This study applied a suite of vegetation and soil measures (0.5 — 990 m^2), small-plot rainfall simulations (0.5 m^2), and overland flow experiments (9 m^2) to quantify the effects of mechanical tree removal (tree cutting and mastication) on vegetation, runoff, and erosion at two mid- to late-succession woodland-encroached sagebrush sites in the Great Basin, United States, 9 yr after treatment. Low amounts of hillslope-scale shrub (3 — 15%) and grass (7 — 12%) canopy cover and extensive intercanopy (area between tree canopies) bare ground (69 — 88% bare, 75% of area) in untreated areas at both sites facilitated high levels of runoff and sediment from high-intensity (102 mm • h^{-1}, 45 min) rainfall simulations in interspaces (~45 mm runoff, 59 — 381 g • m^{-2} sediment) between trees and shrubs and from concentrated overland flow experiments (15, 30, and 45 L • min^{-1}, 8 min each) in the intercanopy (371 — 501 L runoff, 2 342 — 3 015 g sediment). Tree cutting increased hillslope-scale density of sagebrush by 5% and perennial grass cover by twofold at one site while tree cutting and mastication increased hillslope-scale sagebrush density by 36% and 16%, respectively, and perennial grass cover by threefold at a second more-degraded (initially more sparsely vegetated) site over nine growing seasons. Cover of cheatgrass (*Bromus tectorum* L.) was < 1% at the sites pretreatment and 1 — 7% 9 yr after treatment. Bare ground remained high across both sites 9 yr after tree removal and was reduced by treatments solely at the more degraded site. Increases in hillslope-scale vegetation following tree removal had limited impact on runoff and erosion for rainfall simulations and concentrated flow experiments at both sites due to persistent high bare ground. The one exception was reduced runoff and erosion within the cut treatments for intercanopy plots with cut-downed-trees. The cut-downed-trees provided ample litter cover and tree debris at the ground surface to reduce the amount and erosive energy of concentrated overland flow. Trends in hillslope-scale vegetation responses to tree removal in this study demonstrate the effectiveness of mechanical treatments to reestablish sagebrush steppe vegetation without increasing cheatgrass for mid- to late-succession woodland-encroached sites along the warm-dry to cool-moist soil temperature — moisture threshold in the Great Basin. Our results indicate improved hydrologic function through sagebrush steppe vegetation recruitment after mechanical tree removal on mid- to late-succession woodlands can require more than 9 yr. We anticipate intercanopy runoff and erosion rates will decrease over time at both sites as shrub and grass cover continue to increase, but follow-up tree removal will be needed to prevent pinyon and juniper recolonization. The low intercanopy runoff and erosion measured underneath isolated cut-downed-trees in this study clearly demonstrate that tree debris following mechanical

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* Correspondence: C. Jason Williams, Southwest Watershed Research Center, USDA Agricultural Research Service, 2000 East Allen Rd, Tucson, AZ 85719, USA.
E-mail address: jason.williams@ars.usda.gov (C.J. Williams).

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treatments can effectively limit microsite-scale runoff and erosion over time where tree debris settles in good contact with the soil surface.

Introduction

Mitigation of pinyon (*Pinus* spp.) and juniper (*Juniperus* spp.) woodland encroachment is a primary tenet in the conservation of the sagebrush steppe (*Artemisia* spp.) ecosystem in the western United States (Davies et al., 2011; Knick and Connelly, 2011; Miller et al., 2011; McIver et al., 2014). Recent reviews indicate pinyon and juniper woodlands now occupy an estimated 190 000 km^2 in the Intermountain Western United States and that about 90% of that domain was historically sagebrush vegetation (Miller et al., 2008; Davies et al., 2011; Miller et al., 2011). Pinyon and juniper woodland encroachment and development on sagebrush rangelands have been characterized into three successive phases (Miller et al., 2000, 2005; Johnson and Miller, 2006; Miller et al., 2008). In phase I (early-succession), pinyon and juniper cover increases for 0- to 3-m height class, but sagebrush, perennial bunchgrasses, and forbs (sagebrush steppe vegetation) remain the dominant vegetation. In phase II (midsuccession), pinyon and juniper approach 10 − 50% of potential tree cover, trees begin influencing site-level ecological processes, and shrub and herbaceous cover decline due to competition with trees for limited water and soil resources. Phase III (late-succession) is reached when trees become the dominant cover type (> 75% shrub mortality) and exert the primary control on key site-level ecological processes. Roundy et al. (2014a) developed a tree dominance index (TDI [0 − 1], TDI = tree cover / (tree + shrub + tall perennial grass cover) to relate declines in shrub and tall perennial grass cover with increases in tree cover on sagebrush rangelands. Reduction of sagebrush and perennial grass cover and the associated vegetation structure following woodland encroachment can amplify runoff and erosion rates (Pierson et al., 2007; Petersen and Stringham, 2008; Petersen et al., 2009; Pierson et al., 2010, 2013; Williams et al., 2014a, 2016a, 2016b), degrade wildlife habitat (Knick et al., 2014; Coates et al., 2017; Prochazka et al., 2017), alter timing of water availability (Roundy et al., 2014b; Kormos et al., 2017), and limit delivery of ecosystem goods and services (Aldrich et al., 2005; Davies et al., 2011). Persistence of tree dominance over time can propagate site degradation through 1) loss of the native shrub and grass vegetation and the associated seed bank (Koniak and Everett, 1982; Miller et al., 2000; Bates et al., 2005; Miller et al., 2005; Bates et al., 2014; Chambers et al., 2014a; Williams et al., 2017), 2) alteration of the fuel structure and fire regime (Miller and Tausch, 2001; Miller et al., 2013), and 3) postfire increases in cover of exotic fire-prone annual grasses (Barney and Frischknecht, 1974; Koniak, 1985; Bates et al., 2014; Bates and Davies, 2016), and long-term soil loss (Pierson et al., 2011; Wilcox et al., 2012; Williams et al., 2016b, 2016c). Pinyon and juniper removal through prescribed fire and/or mechanical treatments (cutting, mastication/shredding, whole tree harvesting) is commonly employed to increase sagebrush steppe vegetation and thereby reverse negative ecological ramifications associated with woodland encroachment (Pierson et al., 2007; Bates and Svejcar, 2009; Miller et al., 2014a; Pierson et al., 2014; Roundy et al., 2014a, 2014b; Pierson et al., 2015; Bates and Davies, 2016; Bates et al., 2017; Holmes et al., 2017; Severson et al., 2017; Williams et al., 2017; Williams et al., 2018).

Sagebrush steppe vegetation response to tree removal varies with site biophysical attributes, treatment type and methodology, and weather in the years following treatment (Miller et al., 2013; Bates et al., 2014; Chambers et al., 2014a; Miller et al., 2014a; Roundy et al., 2014a, 2014b; Bates and Davies, 2016). Tree removal is most likely to elicit increases in sagebrush steppe vegetation where treatment is applied to sites in earlier phases (phases I − II) of woodland encroachment and development with an intact sagebrush and a native perennial herbaceous understory (Miller et al., 2005; Bates et al., 2011, 2014; Roundy et al., 2014a; Bates et al., 2017; Williams et al., 2017). Studies across the Great Basin have found that residual densities of 0.26 − 0.40 sagebrush shrubs, 1 − 3 perennial grass plants, and > 5 perennial forb plants per m^2 following burning or mechanical tree removal are generally sufficient to reestablish sagebrush steppe vegetation and reduce risk of invasion by the exotic annual cheatgrass (*Bromus tectorum* L.) (Bates et al., 2005; Ziegenhagen and Miller, 2009; Bates et al., 2011, 2014; Miller et al., 2014a). Cheatgrass invades bare patches on sagebrush and woodland-encroached sites and thereby increases the horizontal connectivity of fuels, as well as the likelihood of fire ignition (Brooks et al., 2004; Link et al., 2006; Chambers et al., 2007; Condon et al., 2011; Balch et al., 2013; Reisner et al., 2013; Rau et al., 2014). Cheatgrass is a prolific seed producer, readily establishes post fire, and can outcompete native perennial vegetation for soil nutrients and water, potentially forming a cheatgrass monoculture (Young et al., 1969; Melgoza et al., 1990; Knapp, 1996; Arredondo et al., 1998; Humphrey and Schupp, 2001; West and Yorks, 2002; Brooks et al., 2004; Chambers et al., 2007). The risk of cheatgrass invasion is greatest on sites with warm-dry (mesic-aridic) soil temperature-moisture regimes and is typically more limited on sites with cool-moist (frigid-xeric) soil temperature-moisture regimes (Chambers et al., 2007; Miller et al., 2013; Chambers et al., 2014a, 2014b; Miller et al., 2014a; Roundy et al., 2014a). The elevation thresholds for warm versus cool soil temperature regimes vary within the Great Basin Region but occur at 1 675 m to 1 980 m elevation in the central Great Basin (Miller et al., 2013). The annual precipitation threshold for dry versus moist soil moisture regimes is approximately 300 mm (Miller et al., 2013). Removal of pinyon and juniper trees on conifer-encroached sagebrush rangelands has been shown to increase seasonal soil water and nutrient availability important for cheatgrass establishment and dominance (Bates et al., 2000, 2002; Blank et al., 2007; Chambers et al., 2007; Rau et al., 2007; Vasquez et al., 2008; Young et al., 2013; Roundy et al., 2014b). Prescribed burning to reduce trees can be particularly risky where fire results in high mortality of perennial bunchgrasses and extensive bare conditions favorable to cheatgrass invasion (Bates et al., 2011, 2014; Bates and Davies, 2016). Burning also kills sagebrush and can therefore prolong recovery of the sagebrush understory (Harniss and Murray, 1973; Ziegenhagen and Miller, 2009; Miller et al., 2013, 2014a; Moffet et al., 2015). In contrast, mechanical tree removal treatments have minimal negative impact on sagebrush and perennial grass cover (Bates et al., 2000, 2005; Miller et al., 2005; Chambers et al., 2014a; Roundy et al., 2014a; Bybee et al., 2016; Williams et al., 2017) but often leave numerous pinyon and juniper seedlings that ultimately reestablish tree dominance over time (Tausch and Tueller, 1997; Bates et al., 2005; Miller et al., 2013; Bates et al., 2017). Hybrid treatments of tree cutting followed by cool- to cold-season prescribed burning can be used to reduce tree cover while limiting mortality of perennial bunchgrasses and forbs (Bates and Svejcar, 2009; Bates et al., 2014; Bates and Davies, 2016). Sites with limited sagebrush and native perennial grass and forb understories post treatment may require seeding to reestablish sagebrush steppe vegetation and prevent cheatgrass establishment (Bates et al., 2014; Davies et al., 2014; Roundy et al., 2014a; Bybee et al., 2016), but the effectiveness of seeding can be highly variable due to challenges with seeding methodologies and, as with natural revegetation, varying environmental conditions and precipitation in the immediate years post treatment (Hardegree et al., 2016a, 2016b, 2018).

Tree removal impacts on hydrologic function and erosion processes for a given site are governed by the degree in which the respective treatment alters the amount and distribution of vegetation and ground cover

(Pierson et al., 2007, 2013, 2014; Williams et al., 2014a; Pierson et al., 2015; Williams et al., 2016a; Roundy et al., 2017; Williams et al., 2018). Runoff and erosion are generally minimal on well-vegetated sagebrush steppe communities (Pierson et al., 1994, 2008, 2009; Williams et al., 2016d) and occur on these rangelands primarily as rainsplash and sheetflow (splash-sheet) processes in isolated bare interspaces between shrubs and bunchgrasses (Blackburn, 1975; Johnson and Gordon, 1988; Johnson and Blackburn, 1989; Blackburn et al., 1990; Pierson et al., 2002). Runoff generated in bare patches on well-vegetated sagebrush sites typically travels a short distance downslope before infiltrating and depositing sediment near plant bases or in litter-covered areas (Seyfried, 1991; Pierson et al., 1994; Pierson and Williams, 2016). Woodland encroachment affects runoff and erosion on sagebrush sites through reduction of vegetation and ground cover and alteration of the water- and soil-conserving vegetation structure characteristic to these rangelands (Pierson et al., 1994, 2007, 2009, 2010; Williams et al., 2014a; Pierson and Williams, 2016; Williams et al., 2016a, 2016b, 2016c). As pinyon and juniper trees outcompete shrubs and herbaceous vegetation in late phase II (near 0.5 TDI), bare ground increases and becomes well connected throughout the intercanopy between trees (Pierson et al., 2010, 2013; Williams et al., 2014a, 2016a, 2016b). High rates of runoff and sediment detachment in these bare patches accumulate and facilitate formation of high-velocity, concentrated overland flow with high sediment detachment and transport capacity (Pierson et al., 2007, 2010, 2011, 2013, 2014; Williams et al., 2014a, 2014b, 2016a). Pinyon and juniper removal treatments targeting improved ecohydrologic function aim to enhance interspace infiltration through increases in vegetation and ground cover and to reduce erosion through protection of the surface soil and limiting the amount, connectivity, and erosive energy of intercanopy, concentrated overland flow (Pierson et al., 2007; Cline et al., 2010; Pierson et al., 2013, 2014; Williams et al., 2014a, 2016b, 2016c; Roundy et al., 2017; Williams et al., 2018). Immediate and short-term (first few years post treatment) effects of tree removal on infiltration, runoff characteristics, and sediment delivery exhibit high spatial variability along a hillslope associated with variable vegetation, ground cover, and surface soil water repellency conditions and the connectivity of bare ground and runoff and erosion processes (Pierson et al., 2007; Cline et al., 2010; Pierson et al., 2013, 2014; Williams et al., 2014a; Pierson et al., 2015; Williams et al., 2016a). Tree removal by burning can increase the spatial connectivity of bare ground, sediment availability, runoff, and erosion in the short-term post treatment but may improve hydrologic function where burning stimulates perennial herbaceous cover (Pierson et al., 2013; Williams et al., 2014a; Pierson et al., 2015; Williams et al., 2016a, 2018). Mechanical tree removal may have limited immediate impact on runoff and erosion without substantial increases in vegetation and ground cover and good distribution of tree debris throughout the intercanopy (Pierson et al., 2013, 2014, 2015). The time required for vegetation and ground cover to enhance infiltration and reduce erosion through tree removal remains largely unknown across the vast domain of woodland-encroached sagebrush steppe. Short-term hydrologic and erosion responses reported in the literature are highly variable (Cline et al., 2010; Pierson et al., 2013, 2014; Williams et al., 2014a; Pierson et al., 2015; Williams et al., 2016a; Roundy et al., 2017). Literature is limited regarding long-term tree removal effects on hydrologic processes for sagebrush steppe, but Pierson et al. (2007) found increased intercanopy vegetation and ground cover over a 10-yr period following tree cutting was effective at limiting runoff and reducing the velocity of overland flow and sediment yield.

Recent studies have greatly advanced knowledge regarding mid- to long-term vegetation and ground cover responses to tree removal in sagebrush steppe (Bates and Svejcar, 2009; Bates et al., 2011, 2014; Miller et al., 2014a; Roundy et al., 2014a, 2014b; Bates and Davies, 2016; Bates et al., 2017; Williams et al., 2017), but current understanding regarding long-term effects of tree removal on vegetation and hydrologic and erosion processes remains inadequate for guiding management across the vast domain in which woodland-encroachment occurs. Furthermore, the lack of current understanding limits ecological process-based enhancement of management tools such as Ecological Site Descriptions and State-and-Transition Models (Briske et al., 2005; Herrick et al., 2006; Briske et al., 2008; Petersen et al., 2009; Chambers et al., 2014a; Stringham et al., 2016; Williams et al., 2016b, 2016c). Land managers throughout the western United States depend on these and other tools and the best available knowledge to 1) evaluate which sites are most

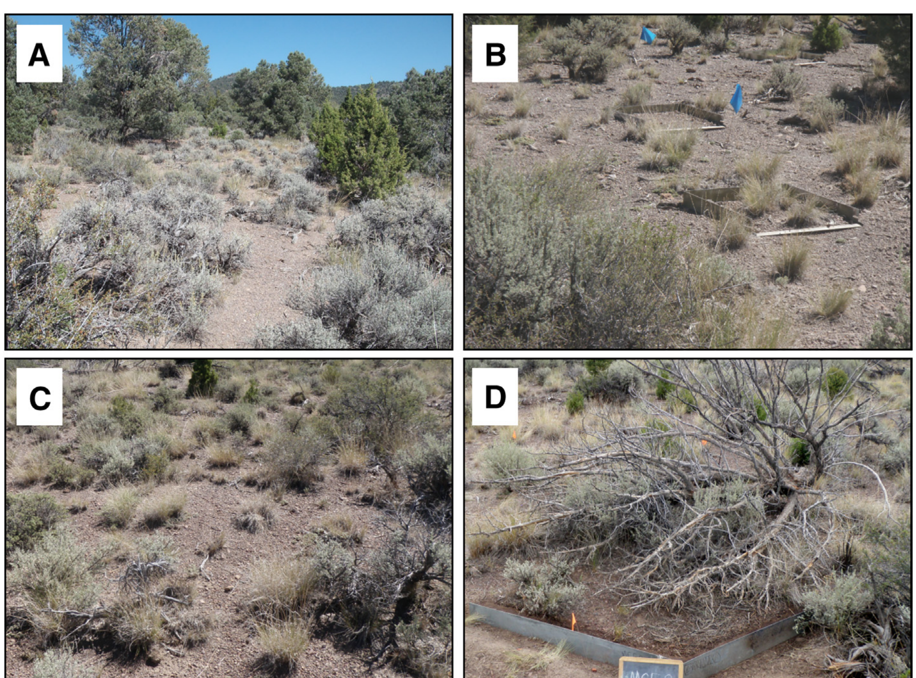

Figure 1. Photographs of the Marking Corral study site in 2015 showing isolated tree islands and degraded intercanopy area between trees in the control (untreated) area (**A**), untreated intercanopy with interspace rainfall simulation plots (0.5 m^2) (**B**), intercanopy within the cut treatment (**C**), and shrub-interspace zone concentrated overland flow plot (9 m^2) in the cut treatment area with a cut downed-tree (cut-downed-tree treatment, **D**). Figure modified from Williams et al., (2018).

appropriate for treatment, 2) predict responses to treatment alternatives, 3) select the correct treatment to meet desired outcomes, and 4) determine appropriate time and conditions in which to apply treatment (Miller et al., 2005, 2013; Chambers et al., 2014b; Miller et al., 2014b; Brown and Havstad, 2016; Chambers et al., 2017). This study evaluated the long-term effectiveness of mechanical tree removal treatments to reestablish sagebrush steppe vegetation and improve ecohydrologic function on two mid- to late-succession woodland-encroached sagebrush sites in the Great Basin. Specifically, we applied a suite of vegetation and soil measures, small-plot (0.5-m^2) rainfall simulations, and concentrated overland flow experiments (9 m^2) to quantify the effects of tree removal by cutting and mastication on vegetation, soils, and hydrology and erosion processes at two woodlands 9 yr after treatment. Small-plot measures were selected to quantify tree removal effects on infiltration, runoff generation, and splash-sheet erosion processes for interspace microsites between shrub and tree canopies and microsites underneath shrub and tree canopies. Overland flow experiments were designed to quantify treatment effects on patch scale (~10 m^2) concentrated overland flow processes in the intercanopy and in areas underneath and immediately adjacent to tree canopies. This study 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 and the effects of various sagebrush-steppe restoration approaches (McIver and Brunson, 2014; McIver et al., 2014). The current study expands on our previous pretreatment and short-term (1 – 2 yr post-treatment) tree removal studies at both sites as part of SageSTEP (Cline et al., 2010; Pierson et al., 2010, 2014, 2015; Williams et al., 2016a, 2018) by quantifying longer-term ecohydrologic responses to mechanical treatments.

Methods

Study Area

This study was conducted on a single-leaf pinyon – Utah juniper woodland (*P. monophylla* Torr. & Frém. – *J. osteosperma* [Torr.] Little) (Marking Corral) and a Utah juniper woodland (Onaqui) within the SageSTEP study network (McIver and Brunson, 2014; McIver et al., 2014). The Marking Corral site (Fig. 1; 39°27'17"N lat, 115°06'51"W long) is located in the Egan Range, ~27 km northwest of Ely, Nevada. The Onaqui site (Fig. 2; 40°12'42"N lat, 112°28'24"W long) is located in the Onaqui Mountains, ~ 76 km southwest of Salt Lake City, Utah. Both sites are managed by the US Department of the Interior, Bureau of Land Management for grazing but have been excluded from grazing since autumn 2005. Detailed physiographic, climate, soils, and vegetation attributes for the sites are provided in Table 1**.** Precipitation averages about 300 mm at the sites, and the soil temperature-moisture regimes at both locations are at the fringe of warm-dry and cool-wet classifications (McIver and Brunson, 2014). Estimated annual precipitation during the study period (2006 – 2015) was near or exceeded the long-term average for both sites, with only 2 – 3 yr of > 15% below normal precipitation (Fig. 3).

The vegetation community structure at both sites before tree removal was typical of degraded or declining sagebrush steppe in late phase II to phase III (TDI > 0.5) of woodland development (Miller et al., 2000, 2005, 2008; see Figs. 1 and 2). Pierson et al. (2010) quantified vegetation and ground cover attributes and soil features at the sites in summer 2006 before tree removal. Vegetation at both sites consisted of isolated tree islands surrounded by sparsely vegetated intercanopy (see Figs. 1A – B and 2A – B). Tree canopy cover and intercanopy area in the treatment areas at both sites were approximately 25% and 75%, respectively, before tree removal (see Table 1). The sites exhibited high shrub mortality associated with woodland encroachment (Miller et al., 2000, 2005, 2008; see Table 1). Approximately 80 – 85% of the shrub density and 75% to > 90% of total shrub canopy cover at the sites was sagebrush, but shrub canopy cover was ≤ 15% at the hillslope scale across the sites. Hillslope-scale tall perennial grass canopy cover averaged 9% at Marking Corral and 8% at Onaqui. Cheatgrass canopy cover was < 1% at the hillslope scale across both sites. Shrub and grass cover were the dominant intercanopy understory lifeforms at Marking Corral (see Fig. 1A) while the intercanopy understory at Onaqui was grass dominated (see Fig. 2A – B). The intercanopy ground surface was mostly bare before tree removal at both sites (see Figs. 1B and 2B). Combined intercanopy bare soil and rock cover was near 60% at

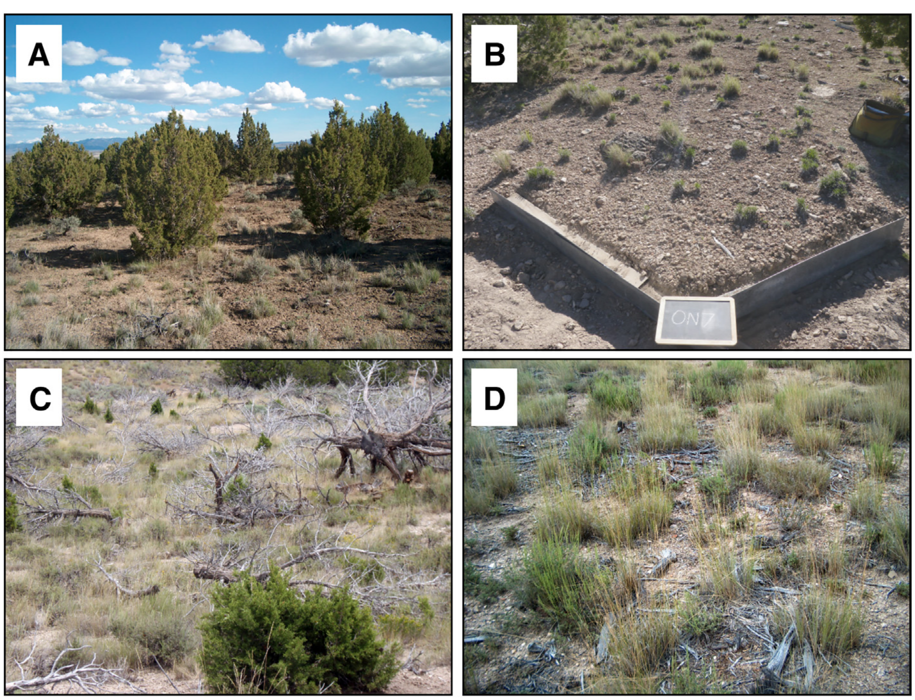

Figure 2. Photographs of the Onaqui study site in 2015 showing isolated tree islands and degraded intercanopy area between trees in the control (untreated) area (**A**), untreated intercanopy with shrub-interspace zone concentrated overland flow plot (9 m^2) (**B**), intercanopy in the cut treatment (**C**), and intercanopy in the mastication treatment (**D**). Figure modified from Williams et al., (2018).

Table 1
Woodland community characteristics, topography, climate, soil, and common understory plants at the Marking Corral and Onaqui study sites.[1]

Site characteristic	Marking Corral, Nevada	Onaqui, Utah
Woodland community	Single-leaf pinyon[2]/Utah juniper[3]	Utah juniper[3]
Encroachment phase[4]	Late phase II — early phase III	Late phase II — early phase III
Tree dominance index[4]	0.51	0.66
Elevation (m) and aspect	2 250—W to SW facing	1 720—N facing
Slope (%)	10-15	10-15
Mean ann. precip. (mm)	307[5]	312[5]
Mean ann. air temperature (°C)	6.5[6]	8.9[7]
Parent rock	Andesite and rhyolite[8]	Sandstone and limestone[9]
Soil association	Segura-Upatad-Cropper[8]	Borvant[9]
Depth to bedrock (m)	0.4-0.5[8]	1.0-1.5[9]
Depth to restrictive layer (m)	0.4-0.5[8]	0.3-0.5[9]
Soil surface texture	Sandy loam, 66% sand, 30% silt, 4% clay	Sandy loam, 56% sand, 37% silt, 7% clay
Tree canopy cover (%)[10]	10[2,11], 15[3,11]	27[3,11], 24[3,12]
Trees per ha[10]	195[2,11], 185[3,11]	441[3,11], 455[3,12]
Mean tree height (m)[10]	2.4[2,11], 2.8[3,11]	2.7[3,11], 2.3[3,12]
Juvenile trees per ha[13]	148[2,11], 130[3,11]	130[3,11], 167[3,12]
Live shrubs per m^2	1.09[11]	0.61[11], 0.82[12]
Dead shrubs per m^2	0.23[11]	0.06[11], 0.19[12]
Intercanopy shrub canopy cover (%)	21[11]	10[11], 5[12]
Intercanopy herbaceous canopy cover (%)[14]	18[11]	9[11], 13[12]
Intercanopy bare soil and rock (%)	59[11]	80[11], 74[12]
Common understory plants	*Artemisia tridentata* Nutt. ssp. *wyomingensis* Beetle & Young; *Artemisia nova* A. Nelson; *Artemisia tridentata* Nutt. ssp. *vaseyana* (Rydb.) Beetle; *Purshia* spp.; *Poa secunda* J. Presl; *Pseudoroegneria spicata* (Pursh) A. Löve; and various forbs	

[1] Data from Pierson et al. (2010) unless otherwise indicated by footnote.
[2] *Pinus monophylla* Torr. & Frém.
[3] *Juniperus osteosperma* [Torr.] Little.
[4] See Miller et al. (2005) for description of woodland encroachment phases (I-III) and Roundy et al. (2014a) for description of tree dominance index (TDI, 0-1), TDI = tree cover / (tree + shrub + tall perennial grass cover).
[5] Estimated from 4 km grid for yr 1971 — 2015 from Prism Climate Group (2017). Pierson et al. (2010) estimates (351 mm Marking Corral, 345 mm Onaqui) were from Prism Climate Group (2009) for yr 1971 — 2000. Pierson et al. (2015) estimates (382 mm Marking Corral, and 468 mm Onaqui) were for yr 1980 — 2011 based on Dayment (Thornton et al., 2012).
[6] Estimated from 4-km grid for yr 1971 — 2015 from Prism Climate Group (2017). Pierson et al. (2010) estimate (7.2°C) was for yr 1928 — 1958 from Western Regional Climate Center (WRCC), Station 264199-2, Kimberly, Nevada (WRCC, 2009).
[7] Estimated from 4 km grid for yr 1971 — 2015 from Prism Climate Group (2017). Pierson et al. (2010) estimate (7.5°C) was for yr 1972 — 2005 from WRCC, Station 424362-3, Johnson Pass, Utah (WRCC, 2009).
[8] Natural Resources Conservation Service (NRCS) (2007).
[9] NRCS (2006).
[10] Live trees ≥ 1 m height.
[11] Data from Pierson et al. (2010) but restricted to cut treatment area before treatment.
[12] Data from Pierson et al. (2010) but restricted to mastication treatment area before treatment.
[13] Live trees < 1.0 m height.
[14] Intercanopy grass and forb canopy cover.

Marking Corral and 80% at Onaqui, indicative of more degraded or sparser cover conditions at the latter site. Understory canopy cover directly under trees averaged 2 — 13% across treatment areas for the sites before tree removal and was mostly grasses. The ground surface immediately underneath tree canopies in treatment areas averaged near 100% litter cover before treatment, with litter depth averaging 42 — 82 mm. The soil surface (0- to 5-cm depth) underneath tree canopies at both sites was water repellent before the treatments, whereas surface soil in interspaces and underneath shrub canopies was wettable. At Marking Corral, pretreatment soil bulk density for 0- to 5-cm soil depth in the cut treatment area was 1.44 g cm^{-3} in interspaces between shrubs and trees and was 1.26 and 1.12 g cm^{-3} under shrub and tree canopies, respectively. At Onaqui, the same measure pretreatment in interspace, under shrub, and under tree canopy locations was 1.01, 0.87, and 0.86 g cm^{-3}, respectively, in the cut treatment area and 1.11, 1.13, and 0.73 g cm^{-3}, respectively, in the mastication treatment area. At each site, values of tree cover, understory canopy and ground cover, hillslope angle, and surface soil texture, water repellency, and bulk density by microsite were statistically similar ($P > 0.05$) across control and treatment areas before tree removal (Pierson et al., 2010).

Tree Removal Treatments

Tree removal treatments were applied in late-summer to early autumn of 2006. At each site, there was one application of each implemented treatment (see Pierson et al., 2015). The resulting control and cut areas were 1.3 ha and 2.2 ha, respectively, at Marking Corral. Control, cut, and mastication treatment areas at Onaqui were 1.0 ha, 2.4 ha, and 1.6 ha, respectively. Treatment areas were not seeded following tree removal at either site. Tree cutting at both sites removed all trees ≥ 1 m height. Trees were cut by chainsaw and allowed to fall naturally and remain in place (cut-and-drop treatment). Tree crowns commonly exhibited an oblong to round (ball-like) vertical form (see Figs. 1A — 2A) that caused cut trees (hereafter referred to as downed trees) to roll a short distance (< 2 m) upon falling. Downed trees had minimal direct contact with the ground surface immediately after cutting due to the crown shape (Pierson et al., 2015). The cutting treatment left a residual of 56 and 167 juvenile (< 1 m height) trees per hectare in cut treatment areas at Marking Corral and Onaqui, respectively (Pierson et al., 2015). Hillslope-scale canopy cover of downed trees (e.g., Fig. 2C) was 10% at Marking Corral and 19% at Onaqui 1 yr post treatment (Pierson et al., 2015). With few exceptions, hillslope-scale canopy cover of shrubs and herbaceous vegetation 1 yr after cutting were similar to pretreatment conditions for both sites (Table 2). The tree-cutting treatment had no impact on hillslope-scale ground cover and bare ground (bare soil and rock) the first yr post treatment at Marking Corral but contributed to a nearly twofold increase in hillslope-scale litter cover and reduction of bare ground at Onaqui (see Table 2). The cutting treatment had no impact on patch-scale ground cover measured in the intercanopy at both sites 1 yr after treatment (Pierson et al., 2015). Intercanopy bare ground was near 70% and 85% the first year after cutting at Marking Corral and Onaqui, respectively (Pierson et al., 2015).

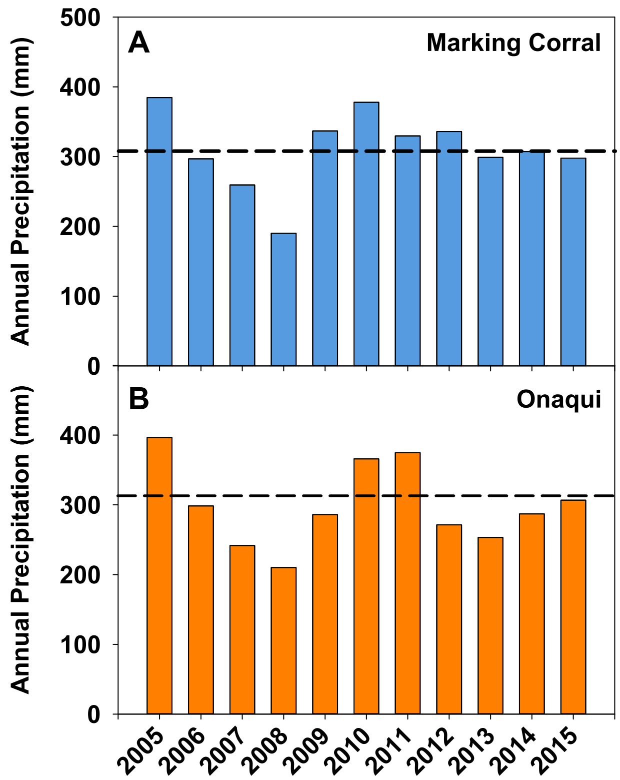

Figure 3. Estimated annual precipitation for the Marking Corral (**A**) and Onaqui (**B**) study sites 1 yr before study initiation (2005) and for the duration of the study period (2006 – 2015). *Bold dashed horizontal line* in each graph indicates mean annual precipitation for the respective site for yr 1971 – 2015. Data from Prism Climate Group (2017) as estimates from a 4-km spatial grid. Figure adapted from Williams et al., (2018).

Tree mastication at Onaqui was accomplished using a rubber-tired Tigercat M726E Mulcher (see Cline et al., 2010). The mastication treatment uniformly removed all trees ≥ 1 m in height through shredding of aboveground tree biomass but left a density of 56 juvenile trees per hectare (Pierson et al., 2015). Trees were masticated in place, and the resulting shredded tree debris (mulch) was allowed to naturally fall to the ground surface (minimal redistribution of mulch after shredding). Addition of mulch to the ground surface contributed to a nearly twofold increase in hillslope-scale litter cover and a reduction in bare ground (see Table 2). Mulch primarily accumulated within a radius of 2 m from shredded tree bases. Mulch ground cover the first year was 2% in the intercanopy and 79% in areas previously covered by tree canopy (Pierson et al., 2015). Mulch depth averaged near 20 mm in the isolated patches within the intercanopy and near 90 mm in areas near tree bases (Pierson et al., 2015). The retention of mulch near tree bases resulted in no significant changes in litter cover and bare ground in intercanopy areas and tree microsites 1 yr post treatment (Pierson et al., 2015). Intercanopy bare ground was approximately 70% 1 yr after the mastication treatment (Pierson et al., 2015).

Experimental Design

Hillslope-scale vegetation and ground cover in cut and mastication treatment areas at each site were sampled on 30 m × 33 m site characterization plots established by Pierson et al. (2010) in 2006 before tree removal. Pierson et al. (2010) randomly located and established three site characterization plots in the cut and mastication treatment areas for repeated sampling. The plots were sampled for tree cover, understory vegetation, and ground cover before tree removal (Pierson et al., 2010) and for understory vegetation and ground cover 1 yr post treatment (Pierson et al., 2015) (see Table 2). This study resampled the three Pierson et al. (2010) site characterization plots in treated areas at each site as repeated measures to quantify changes in vegetation and ground cover nine growing seasons after tree removal.

All small plots (0.7 m × 0.7 m, Fig. 1B) for rainfall simulation experiments in this study were installed before the tree-removal treatments in summer 2006 as described in Pierson et al. (2010) and were left in place for sampling in subsequent years. Plots were randomly selected and installed within control and cut and mastication treatments in the interspaces (Fig. 1B) between shrubs and trees and in areas immediately underneath shrub (shrub coppices) and tree (tree coppices) canopies to partition treatment effects by microsite (Pierson et al., 2008, 2009, 2014). Small-plot vegetation, ground cover, soil, and rainfall simulation response data were collected by Pierson et al. (2010) in control and treated areas in summer 2006 before treatments and, as part of the Pierson et al. (2014) study, as repeated measures in untreated control and mastication treatment areas in the summers of 2007 and 2008. This study repeated Pierson et al.'s (2010 and 2014) small-plot

Table 2
Understory canopy cover and ground cover characteristics measured on 30 × 33 m site characterization plots in cut and mastication treatment areas at the Marking Corral and Onaqui sites 1 yr before tree removal (2006) and 1 yr (2007) and 9 yr (2015) after tree removal treatments. Means within a row followed by different lowercase letters are significantly different ($P < 0.05$).

Site characteristic	Marking Corral Cut			Onaqui Cut			Onaqui Mastication		
	Untreated 2006[1,2]	Cut 2007[3]	Cut 2015	Untreated 2006[1,2]	Cut 2007[3]	Cut 2015	Untreated 2006[1,4]	Mastication 2007[3]	Mastication 2015
Canopy cover									
Total (%)	32.4 a	67.9 bc	73.1 c	21.2 a	55.2 b	59.9 bc	30.8 a	29.1 a	53.9 b
Shrub (%)	14.6 c	14.3 c	28.7 d	3.4 a	5.0 ab	16.9 c	6.1 ab	3.1 a	9.9 bc
Grass (%)	12.4 ab	21.4 bc	30.2 cd	7.3 a	13.7 ab	27.1 cd	11.0 ab	13.9 ab	38.5 d
Forb (%)	1.0 a	3.7 abc	1.4 ab	3.2 abc	12.1 d	7.4 cd	2.0 ab	5.3 bc	4.0 abc
Standing dead (%)	4.4 bc	7.3 cd	0.1 a	7.0 cd	5.8 cd	0.6 a	11.7 d	4.8 bc	1.1 ab
Ground cover									
Total (%)[5]	51.5 bc	48.2 bc	56.2 cd	32.6 a	48.6 bc	47.3 bc	42.5 ab	64.9 d	55.6 cd
Basal plant (%)	0.3 a	0.3 a	8.2 b	0.6 a	0.2 a	9.5 b	0.7 a	0.1 a	16.1 c
Cryptogam (%)	0.1 a	0.3 a	0.1 a	4.7 d	4.9 d	0.5 ab	3.2 cd	1.7 bc	2.5 cd
Woody dead (%)	5.0 c	1.6 b	0.3 a	1.1 ab	1.9 b	1.4 b	1.0 ab	1.0 ab	2.8 bc
Litter (%)	46.1 cd	46.0 cd	47.6 d	26.2 a	41.6 bcd	35.8 abc	37.6 bcd	62.1 e	34.2 ab
Rock (%)[6]	22.0 cd	11.3 b	1.3 a	29.8 d	22.3 cd	17.0 bc	20.1 bcd	11.2 b	14.4 bc
Bare soil (%)	26.4 a	40.5 c	42.5 c	37.7 bc	29.1 ab	35.7 bc	37.4 bc	23.9 a	30.0 ab
Bare ground (%)[7]	48.4 bc	51.8 bc	43.8 ab	67.5 d	51.4 bc	52.7 bc	57.5 cd	35.1 a	44.4 ab

[1] Data from Pierson et al. (2010) but restricted to plots in area subsequently cut[2] or masticated[4] at the respective site.
[3] Data obtained from measurements by Pierson et al. (2015).
[5] Cryptogam, litter, live and dead plant bases, and woody dead cover; excludes rock[6] cover.
[6] Rock fragments > 5 mm in diameter.
[7] Bare soil and rock cover.[6]

Table 3
Average surface roughness, aggregate stability, and cover variables measured on rainfall simulation plots (0.5 m^2) in control and cut treatments at Marking Corral 9 yr post treatment. Means within a row followed by different lowercase letters are significantly different ($P < 0.05$).

Marking Corral	Control			Cut		
Plot characteristic	Interspace	Shrub coppice	Tree coppice	Interspace	Shrub coppice	Tree coppice
Surface roughness (mm)	11 ab	15 b	12 ab	9 a	15 b	11 ab
Aggregate stability class (1-6)[1]	1.9 a	1.7 a	4.2 b	1.4 a	2.0 a	4.3 b
Total canopy cover (%)[2]	37.5 ab	94.8 c	13.2 a	41.3 b	118.3 c	44.4 b
Shrub canopy cover (%)	0.3 a	66.9 b	1.4 a	0.6 a	56.7 b	3.6 a
Grass canopy cover (%)	35.6 b	23.3 ab	5.4 a	37.8 b	38.3 b	35.5 b
Forb canopy cover (%)	0.5 ab	2.9 b	0.1 a	2.1 ab	0.0 a	0.0 a
Standing dead canopy cover (%)[2]	0.8 a	1.4 a	4.6 a	0.8 a	20.7 a	4.2 a
Total ground cover (%)[3]	13.0 a	69.8 b	82.2 bc	10.3 a	85.5 bc	89.6 c
Plant basal cover (%)	7.8 bc	13.5 c	1.9 a	6.5 ab	13.7 c	4.1 ab
Litter ground cover (%)	4.8 a	55.8 b	79.5 c	3.8 a	71.1 bc	84.8 c
Rock cover (%)[4]	7.8 ab	6.1 ab	2.2 a	18.2 b	1.6 a	2.8 a
Bare soil (%)	79.2 c	24.1 b	15.6 ab	71.5 c	12.9 ab	7.6 a
Bare ground (%)[5]	87.1 c	30.2 b	17.8 ab	89.7 c	14.5 ab	10.4 a
Litter depth (mm)	1 a	6 b	34 c	4 ab	8 b	45 c
No. of plots	6	4	8	6	4	8

[1] Stability classes: 1) < 10% stable aggregates, 50% structural integrity lost within 5 s; 2) < 10% stable aggregates, 50% structural integrity lost within 5-30 s; 3) < 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] Cryptogam, litter, live and dead basal plant, and woody dead cover; excludes rock[4] cover.
[4] Rock fragments > 5 mm in diameter.
[5] Bare soil and rock[4] cover.

vegetation, ground cover, soil, and rainfall simulation response measurements in summer 2015 on the previously established control, cut, and mastication plots. The average slope gradient for small plots was similar ($P > 0.05$) across treatments at a site and was approximately 12% at Marking Corral and 18% at Onaqui (Pierson et al., 2010). The number of small plots installed and sampled in 2015 for each site × treatment × microsite combination is shown in Tables 3 and 4.

Concentrated overland flow plots (2 m wide × 4.5 m long) at each site in this study were established as new plots in 2015 within the same control and cut and mastication treatment areas as the small plots. Concentrated flow plots were randomly selected and installed using methods described by Pierson et al. (2015). Plots were installed in shrub-interspace zones (variable amounts of shrub coppice and interspace area) and tree zones (tree coppice with minor interspace component) in each treatment at each site to separate treatment effects for intercanopy areas and areas underneath tree canopies (Pierson et al., 2013; Williams et al., 2014a; Pierson et al., 2015; Williams et al., 2016a). Plots were oriented with the long axis perpendicular to the hillslope contour and were installed borderless with a steel "V-shaped" runoff and sediment collection trough inserted 5 cm into the soil at the downslope plot base (Figs. 1D and 2B). Collection troughs spanned a 2-m plot width and were designed to direct runoff and sediment through a plot outlet. The effect of downed trees on intercanopy concentrated overland flow processes was assessed through additional shrub-interspace zone plots in cut treatment areas with an in-place downed tree situated on each plot within 1 m upslope of the plot outlet (cut-downed-tree treatment; Fig. 1D). Plot collection troughs on cut-downed-tree plots were installed consistent with all other

Table 4
Average surface roughness, aggregate stability, and cover variables measured on rainfall simulation plots (0.5 m^2) in control, cut, and mastication treatments at Onaqui 9 yr post treatment. Means within a row followed by different lowercase letters are significantly different ($P < 0.05$).

Onaqui	Control			Cut			Mastication		
Plot characteristic	Interspace	Shrub coppice	Tree coppice	Interspace	Shrub coppice	Tree coppice	Interspace	Shrub coppice	Tree coppice
Surface roughness (mm)	10 a	14 b	10 a	13 ab	20 c	14 b	11 a	16 bc	13 ab
Aggregate stability class (1-6)[1]	2.6 ab	3.8 bc	5.1 c	3.0 b	2.6 ab	5.0 c	1.8 a	2.9 ab	5.0 c
Total canopy cover (%)[2]	23.8 a	76.5 c	38.7 ab	48.6 ab	89.0 c	60.0 bc	42.3 ab	125.1 d	73.0 c
Shrub canopy cover (%)	0.0 a	47.0 b	0.5 a	2.9 a	57.0 bc	7.6 a	2.3 a	71.2 c	8.4 a
Grass canopy cover (%)	9.7 a	12.4 a	21.3 ab	24.6 ab	12.8 a	36.4 bc	20.6 ab	30.3 ab	57.0 c
Forb canopy cover (%)	7.3 a	6.3 a	4.7 a	14.1 a	6.1 a	5.7 a	12.5 a	6.5 a	4.2 a
Standing dead canopy cover (%)[2]	6.2 a	9.8 a	11.6 a	6.4 a	10.9 a	9.3 a	6.9 a	17.0 a	3.4 a
Total ground cover (%)[3]	13.8 a	46.5 c	72.4 d	25.2 a	78.5 d	70.9 d	16.3 a	66.5 cd	84.0 d
Plant basal cover (%)	3.6 ab	10.7 c	2.0 a	4.4 ab	5.8 abc	6.1 bc	6.1 bc	5.3 abc	8.9 bc
Litter ground cover (%)	7.8 a	33.9 b	69.6 c	18.8 ab	71.1 c	64.2 c	9.1 a	60.4 c	74.1 c
Rock cover (%)[4]	40.2 c	19.8 b	6.7 a	32.3 bc	4.1 a	3.9 a	34.6 bc	3.5 a	1.1 a
Bare soil (%)	46.0 b	33.7 ab	20.8 a	42.5 b	17.4 a	25.1 a	49.1 b	30.0 ab	14.9 a
Bare ground (%)[5]	86.2 c	53.5 b	27.6 a	74.8 c	21.5 a	29.1 a	83.7 c	33.5 ab	16.0 a
Litter depth (mm)	1 a	6 b	29 c	6 b	7 b	24 c	3 ab	1 ab	27 c
No. of plots	6	6	8	10	5	5	10	5	5

[1] Stability classes: 1) < 10% stable aggregates, 50% structural integrity lost within 5 s; 2) < 10% stable aggregates, 50% structural integrity lost within 5-30 s; 3) < 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] Cryptogam, litter, live and dead basal plant, and woody dead cover; excludes rock[4] cover.
[4] Rock fragments > 5 mm in diameter.
[5] Bare soil and rock[4] cover.

Table 5
Canopy and ground cover and cover gaps measured on concentrated flow plots (9 m^2) in control and cut treatment areas at Marking Corral 9 yr post treatment. Means within a row followed by different lowercase letters are significantly different ($P < 0.05$).

Marking Corral	Control		Cut		Cut-downed-tree[1]
Plot characteristic	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Tree zone	Shrub-interspace zone
Total canopy cover (%)[2]	56.0 ab	42.8 ab	58.8 b	35.6 a	102.8 c
Shrub canopy cover (%)	17.8 b	12.7 ab	14.2 ab	7.3 a	8.7 ab
Grass canopy cover (%)	33.1 bc	21.8 ab	40.5 c	17.6 a	42.2 c
Forb canopy cover (%)	1.1 a	0.1 a	1.6 a	0.0 a	0.1 a
Standing dead canopy cover (%)[2]	3.2 ab	5.9 b	1.7 a	0.8 a	1.4 a
Total ground cover (%)[3]	31.5 a	88.5 b	34.6 a	92.9 b	91.2 b
Plant basal cover (%)	11.8 b	4.1 a	11.8 b	2.3 a	3.7 a
Litter ground cover (%)	15.3 a	83.4 c	20.2 a	82.1 c	62.6 b
Rock cover (%)[4]	18.1 b	2.2 a	29.2 b	1.1 a	1.4 a
Bare soil (%)	50.4 c	9.3 a	36.2 b	6.0 a	7.5 a
Bare ground (%)[5]	68.5 b	11.5 a	65.4 b	7.1 a	8.8 a
Canopy gaps 25-50 cm (%)[2]	16.1 b	11.7 b	16.6 b	17.4 b	2.5 a
Canopy gaps 51-100 cm (%)[2]	12.3 b	10.0 b	15.3 b	19.4 b	0.4 a
Canopy gaps 101-200 cm (%)[2]	2.1 a	11.5 b	4.0 a	17.8 b	0.7 a
Canopy gaps 201-400 cm (%)[2]	0.0 a	8.2 b	0.0 a	0.0 a	0.0 a
Basal gaps 25-50 cm (%)	22.0 b	14.1 ab	16.9 b	15.2 ab	8.5 a
Basal gaps 51-100 cm (%)	31.9 c	18.0 ab	29.3 bc	23.2 bc	9.1 a
Basal gaps 101-200 cm (%)	11.4 ab	23.4 b	20.7 b	24.5 b	1.5 a
Basal gaps 201-400 cm (%)	2.9 ab	9.0 b	2.9 ab	4.2 ab	0.0 a
Average canopy gap (cm)[2]	45.1 ab	66.5 c	47.0 ab	60.7 bc	40.3 a
Average basal gap (cm)	60.2 ab	72.0 b	67.5 b	69.0 b	44.4 a
No. of plots	5	5	5	5	5

[1] Plots in the cut treatment area with a residual cut dead single-leaf pinyon (*Pinus monophylla* Torr. & Frem.) or Utah juniper (*Juniperus osteosperma* [Torr.] Little) tree laying parallel to the ground surface and perpendicular to the long axis of the respective plot (across plot along hillslope contour).
[2] Excludes tree canopy cover for trees ≥ 1 m in height.
[3] Cryptogam, litter, live and dead basal plant, and woody dead cover; excludes rock[4] cover.
[4] Rock fragments > 5 mm in diameter.
[5] Bare soil and rock[4] cover.

shrub-interspace zone plots and without disturbing downed trees. Average slope gradient for concentrated flow plots was consistent ($P > 0.05$) across treatments at a site and averaged 11% at Marking Corral and 15% at Onaqui. Five concentrated flow plots were installed and sampled in 2015 for each site × treatment × microsite (zone) combination (Tables 5 and 6).

Table 6
Canopy and ground cover and cover gaps measured on concentrated flow plots (9 m^2) in control, cut, and mastication treatment areas at Onaqui 9 yr post treatment. Means within a row followed by different lowercase letters are significantly different ($P < 0.05$).

Onaqui	Control		Cut		Cut-downed-tree[1]	Mastication	
Plot characteristic	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Shrub-interspace zone	Tree zone
Total canopy cover (%)[2]	19.2 a	51.6 bc	45.9 b	70.0 c	89.3 d	51.7 bc	54.0 bc
Shrub canopy cover (%)	0.1 a	0.2 a	13.6 bc	21.0 c	21.0 c	18.0 c	5.7 b
Grass canopy cover (%)	4.4 a	28.9 bc	14.2 ab	25.4 bc	33.0 c	25.2 bc	40.2 c
Forb canopy cover (%)	10.6 b	8.5 b	10.3 b	10.8 b	7.4 b	6.3 ab	2.9 a
Standing dead canopy cover (%)[2]	3.2 ab	12.9 c	5.2 ab	9.9 bc	3.3 ab	1.7 a	4.1 ab
Total ground cover (%)[3]	12.3 a	88.4 c	17.7 a	60.9 b	61.8 b	24.3 a	85.5 c
Plant basal cover (%)	3.2 a	5.7 a	6.2 a	10.8 bc	6.8 ab	7.6 abc	11.8 c
Litter ground cover (%)	8.2 a	79.3 c	9.3 a	42.6 b	43.4 b	14.2 a	68.6 c
Rock cover (%)[4]	51.8 d	4.4 a	17.7 bc	4.4 a	10.7 ab	26.7 cd	4.1 a
Bare soil (%)	35.9 bc	7.1 a	64.6 d	34.7 bc	27.5 b	49.0 cd	10.4 a
Bare ground (%)[5]	87.7 c	11.6 a	82.3 c	39.1 b	38.2 b	75.7 c	14.5 a
Canopy gaps 25-50 cm (%)[2]	15.6 b	14.6 b	14.3 b	6.5 a	6.2 a	14.8 b	11.1 ab
Canopy gaps 51-100 cm (%)[2]	22.4 d	10.3 abc	19.1 cd	3.6 a	6.4 ab	14.8 bcd	9.7 abc
Canopy gaps 101-200 cm (%)[2]	13.3 b	2.6 a	10.9 b	2.7 a	4.2 ab	6.2 ab	9.9 ab
Canopy gaps 201-400 cm (%)[2]	7.8 b	0.0 a	2.5 ab	1.9 ab	1.8 ab	0.0 a	4.3 ab
Basal gaps 25-50 cm (%)	16.7 bc	22.6 c	16.3 bc	12.5 ab	9.2 a	16.7 bc	12.3 ab
Basal gaps 51-100 cm (%)	24.7 b	21.0 b	20.8 b	14.2 ab	11.4 a	23.9 b	18.8 ab
Basal gaps 101-200 cm (%)	25.2 a	9.0 a	22.2 a	9.9 a	11.8 a	13.2 a	18.7 a
Basal gaps 201-400 cm (%)	9.0 ab	1.6 a	7.6 ab	2.7 a	0.0 a	1.3 a	12.4 b
Average canopy gap (cm)[2]	62.1 a	45.0 a	59.8 a	52.5 a	63.6 a	51.3 a	65.0 a
Average basal gap (cm)	71.6 a	54.6 a	69.6 a	62.6 a	60.4 a	58.2 a	78.1 a
No. of plots	5	5	5	5	5	5	5

[1] Plots in the cut treatment with a residual cut dead Utah juniper (*Juniperus osteosperma* [Torr.] Little) tree laying parallel to the ground surface and perpendicular to the long axis of the respective plot (across plot along hillslope contour).
[2] Excludes tree canopy cover for trees ≥ 1 m in height.
[3] Cryptogam, litter, live and dead basal plant, and woody dead cover; excludes rock[4] cover.
[4] Rock fragments > 5 mm in diameter.
[5] Bare soil and rock[4] cover.

Vegetation, Ground Cover, and Soil Sampling

Hillslope Scale

Hillslope-scale vegetation and ground cover were assessed on each 30 m × 33 m site characterization plot using line-point intercept and gap-intercept methods along five 30-m transects oriented 5 − 8 m apart and perpendicular to the hillslope contour (Herrick et al., 2005; Pierson et al., 2010). Canopy (foliar) and ground cover on each plot were measured at 60 points with 50-cm spacing along each of the five transects for a total of 300 sample points on each plot. Percent cover for each cover type sampled was calculated for each plot as the frequency of respective cover type hits divided by the total number of sample points. Distances in excess of 20 cm between plant bases (basal gaps) were measured along each of the five 30-m transects on each plot. Average basal gap size was calculated for each plot as the mean of all respective gaps measured in excess of 20 cm. Percentages of basal gaps representing gap classes 25 − 50, 51 − 100, 101 − 200, and 201 − 600 cm were derived for each transect and averaged across the transects on each plot to determine gap-class plot means (Herrick et al., 2005). The number of live trees > 0.5 m in height was tallied for each plot, and tree height and maximum and minimum crown diameters were measured for each tree. The crown radius for each tallied tree was calculated as one-half the average of the minimum and maximum crown diameters. Individual tree crown area was derived as equivalent to the area of a circle, calculated with the respective crown radius. Total tree cover for each plot was derived as the sum of measured tree cover values on the respective plot. The number of shrubs > 5-cm height and tree seedlings 5-50-cm height within each plot were counted along three evenly spaced (6 m apart) belt transects (2 m wide × 30 m long). Shrub and tree seedling densities for each plot were calculated as the respective sums tallied along each of the respective three belt transects divided by total belt transect area (180 m^2).

Small-Plot Scale

Small-plot canopy (foliar) cover, ground cover, and ground surface roughness were measured using point frame methods described by Pierson et al. (2010). Canopy and ground cover for each plot were sampled on 15 points (spaced 5 cm apart) along each of seven equally spaced transects (10 cm apart and parallel to hillslope contour) for a total of 105 sample points per plot. Percent cover for each cover type sampled on a plot was derived from the frequency of respective cover type hits divided by the total number of points sampled within the plot. The relative ground surface height at each sample point on each plot was measured by a metal ruler as the distance between a point-frame level-line and the ground surface. Ground surface roughness for each plot was calculated as the mean of the standard deviations of the ground surface heights for each of the seven transects sampled on the respective plot. Litter depth on each plot was measured to the nearest 1 mm at four evenly spaced points (~15-cm spacing) located along the outside edge of each of two plot borders located perpendicular to the hillslope contour. An average litter depth was calculated for each plot as the mean of the eight litter depths measured.

Surface soil water repellency was assessed immediately adjacent (within ~ 50 cm) to each small plot before rainfall simulations using the water drop penetration time (WDPT) method (DeBano, 1981). Eight water drops (~ 3-cm spacing) were placed on the mineral soil surface after litter was carefully removed, and the time required for infiltration of each drop was recorded up to a 300-s maximum time. Following this procedure, 1 cm of soil was excavated immediately underneath the previously sampled area and the WDPT method was repeated with eight additional drops. This process was continued until a full 5-cm soil depth was sampled. The mean WDPT at 0-, 1-, 2-, 3-, 4-, and 5-cm soil depths for each plot was recorded as the average of the eight WDPT (s) samples at the respective depth. A plot mean soil water repellency across all sample depths was derived as the arithmetic average of the means from each of the 1-cm depths sampled. Soils were classified as wettable if WDPT < 5 s, slightly water repellent if WDPT ranged 5 − 60 s, and strongly water repellent if WDPT > 60 s (Bisdom et al., 1993).

Surface soils for each plot were also sampled for soil moisture and aggregate stability before rainfall simulations. Soil samples were obtained for 0- to 5-cm depth immediately adjacent to each small plot and were analyzed gravimetrically in the laboratory for soil water content. Surface soil aggregate stability for each plot was determined using a modified sieve test described by Herrick et al. (2001, 2005). Six soil aggregates approximately 6 − 8 mm in diameter and 2- to 3-mm thick were excavated from the soil surface immediately adjacent to each plot and evaluated using the stability test. Each soil aggregate was assigned a stability class as suggested by Herrick et al. (2005) (see Tables 3 and 4). A mean aggregate stability for each plot was derived as the arithmetic average of the classes assigned to the six aggregate samples from the respective plot.

Patch Scale

Canopy and ground cover by cover type and distances between plant canopies (canopy gaps) and bases (basal gaps) were measured on each concentrated overland flow plot using line-point intercept and gap-intercept methodologies (Herrick et al., 2005). Canopy (foliar) and ground cover on each plot were sampled at 24 points (spaced 20 cm apart) along each of nine line-point intercept transects 4.6 m in length, oriented 20 cm apart and perpendicular to the hillslope contour (216 points per plot). Percent cover for each cover type sampled on each plot was calculated from the frequency of respective cover type hits divided by the total number of points sampled within the plot. Plant canopy and basal gaps exceeding 20 cm were measured along each line-point transect. Average canopy and basal gap sizes for each plot 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 201 − 400 cm were determined for each transect and averaged across the transects on each plot to determine gap-class plot means (Herrick et al., 2005). The relative ground-surface height at each line-point sample point was calculated as the distance between the ground surface and a survey transit level-line above the respective sample point. Ground surface roughness for each concentrated overland flow plot was derived as the arithmetic average of the standard deviations of the ground surface heights across the line-point transects.

Hydrology and Erosion Measurements

Small Plot Rainfall Simulations

Rainfall was applied to each small plot at target intensities 64 mm • h^{-1} (dry run) and 102 mm • h^{-1} (wet-run) for 45 min each using an oscillating-arm rainfall simulator fitted with 80 − 100 Vee-jet nozzles. The rainfall simulator, rainfall characteristics, and simulator calibration procedures are described in detail by Pierson et al. (2008, 2009, 2010). The dry run was conducted on dry antecedent − soil moisture conditions (~ 5 − 15% gravimetric), and the wet run was applied approximately 30 min after the dry run. The mean rainfall intensity and total rainfall applied by run type were similar across control, cut, and mastication conditions at both sites ($P > 0.05$), and the standard deviations of rainfall rates by run type across all plots in the study were within 3 mm • h^{-1} of the respective target intensities. For both study sites, the dry run intensity applied for 5-, 10-, and 15-min durations approximates respective local storm return intervals of 7, 15, and 25 yr, and the wet run intensity over the same durations approximates 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 during each 45-min rainfall simulation and were analyzed in the laboratory for runoff volume and sediment concentration. Runoff volume and sediment concentration for each runoff sample were obtained by weighing the sample before and after oven drying at 105°C. Hydrologic and erosion response variables were derived for each rainfall simulation using the timed runoff samples. The mean runoff rate (mm • h^{-1}) for each sample interval was calculated as the cumulative runoff over the sample interval divided by the respective interval time. A cumulative runoff (mm) from each simulation was derived as the integration of runoff rates over the total time of runoff. A runoff-to-rainfall ratio ($mm \cdot mm^{-1}$) was calculated for each simulation by dividing cumulative runoff by the total rainfall applied. The mean infiltration rate ($mm \cdot h^{-1}$) for each sample interval was derived as the difference between applied rainfall and measured runoff divided by duration of the sample interval. The sediment discharge ($g \cdot s^{-1}$) for each sampled interval was derived as the cumulative sediment for the sample interval divided by the interval time. The cumulative sediment yield ($g \cdot m^{-2}$) for each simulation was calculated as the integrated sum of sediment collected during runoff and was extrapolated to the total plot area by dividing cumulative sediment by the 0.5 m^2 plot area. A sediment-to-runoff ratio ($g \cdot m^{-2} \cdot mm^{-1}$), a surrogate for erodibility, was determined for each simulation by dividing cumulative sediment yield by cumulative runoff.

Soil wetting patterns were assessed over 0- to 20-cm depths immediately following dry-run rainfall simulations on each plot. Wetting patterns for each plot were measured by excavating a 50-cm-long trench to a depth of 20 cm. A single wetting trench was excavated immediately adjacent to each small plot to avoid impacting wet-run simulations. The percent wetted area of each exposed soil profile was measured using a 2 cm × 2 cm cell grid. Each grid area was determined to be dry or wet on the basis of the dominant moisture 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 percent of wetted area over 0- to 6-cm, 0- to 10-cm, and 0- to 20-cm depths, respectively.

Concentrated Overland Flow Simulations

Concentrated overland flow was applied using methods described by Pierson et al. (2010, 2015). Flow release rates of 15, 30, and 45 $L \cdot min^{-1}$ were applied to each concentrated flow plot by using datalogger-controlled flow regulators. Each plot was prewet with a gently misting sprinkler to create wet soil conditions (~ 20 − 25% gravimetric) similar to those under which overland flow occurs, but without detaching and eroding sediment (Pierson et al., 2015). The concentrated flow release rate sequence applied in each simulation 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 flow release rates was applied to each plot from a single location, approximately 4 m upslope of the plot outlet. Flow from flow regulators was routed through a metal box filled with Styrofoam pellets and was released through a 10-cm-wide mesh-screened opening at the base of the box (see Pierson et al., 2010). Runoff samples were collected at the plot outlet at 1- to 2-min intervals for each 12-min flow rate simulation and were processed in the laboratory for runoff and sediment concentration as explained for small plot rainfall simulations.

Runoff and erosion response variables for each flow release rate were derived for an 8-min time period beginning at runoff initiation. The mean runoff rate ($L \cdot min^{-1}$) was determined for each sample interval as the cumulative runoff divided by the interval time. The cumulative runoff (L) by release rate for each plot was derived as the integration of runoff rates over the respective 8-min time of runoff. The averaged sediment concentration ($g \cdot L^{-1}$) was derived for each sample interval as the cumulative sediment divided by the cumulative runoff, and a mean sediment concentration for each flow release rate on each plot was calculated as the average of all sediment concentrations for the respective rate on the plot. Cumulative sediment (g) by release rate for each plot was derived as the integrated sum of sediment collected during the 8-min runoff period. Total runoff (L) and total sediment (g) for each plot were calculated as the sum of cumulative runoff and sediment, respectively, across all flow release rates.

Overland flow velocity and the widths and depths of flowpaths were measured on each plot to characterize overland flow. Overland flow velocity was measured for flow paths during 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 flowpath length (Pierson et al., 2008, 2010, 2015). A mean flow velocity ($m \cdot s^{-1}$) for each flow rate on each plot was derived by dividing flowpath length (2 m) by the mean of multiple sampled salt travel times ($n = 2$ to 3 per rate per plot) in seconds. The width and depth of all flowpaths for each release rate on each plot were measured at a cross-section 3 m downslope of the flow release point. A mean flowpath width and depth for each flow rate on each plot was calculated as the arithmetic average of respective flowpath widths and depths measured at the 3-m cross-section.

Statistical Analyses

Statistical analyses were conducted with SAS software, version 9.4 (SAS Institute Inc., 2013). All statistical analyses were restricted to within-site comparisons except where explicitly stated. Hillslope-scale vegetation and ground cover data collected on 30 m × 33 m site characterization plots (this study with comparisons to previous yr [Pierson et al., 2010, 2015]) were analyzed across both sites using a repeated-measures mixed-model with multiple treatment levels (precut, yr-1 cut, yr-9 cut, premastication, yr-1 mastication, yr-9 mastication) and sample yr (2006, 2007, 2015) as the repeated measure. The covariance structure in all analyses of site characterization plot data was evaluated using fit statistics suggested by Littell et al. (2006), and the best fit model was applied. All data from rainfall simulation plots at Marking Corral were analyzed using a mixed model with two treatment factors (control and cut) and three microsite levels (interspace, shrub coppice, and tree coppice). All data from rainfall simulation plots at Onaqui were analyzed in the same manner as those of Marking Corral but included one additional treatment level, mastication. Vegetation, ground cover, flowpath dimension, and velocity data from concentrated flow plots at Marking Corral were analyzed using a mixed model with three treatment levels (control, cut, cut-downed-tree) and two microsite levels (shrub-interspace zone and tree zone). The same measures from concentrated flow plots at Onaqui were analyzed as described for Marking Corral but included the one additional treatment level, mastication. Concentrated flow runoff and erosion data for a site were analyzed with a repeated-measures mixed model using the site-specific treatment and microsite levels specified above for all other concentrated flow-plot 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 site, treatment, and microsite were considered fixed effects in all respective analyses. Normality and homogeneity were tested before analysis of variance using the Shapiro-Wilk test and Levene's test, respectively, and deviance from normality was addressed through data transformation. Where necessary, arcsine-square root transformations were used to normalize proportion data (e.g., canopy and ground cover) and logarithmic transformations were applied to normalize WDPT, infiltration, runoff, and erosion data. Backtransformed means are reported. Mean separation was determined using the LSMEANS procedure. All reported significant effects (mean differences and correlations) were determined at the $P < 0.05$ level.

Results

Vegetation, Ground Cover, and Soils

Hillslope Scale Vegetation and Ground Cover

Mechanical tree-removal treatments increased hillslope-scale understory canopy cover at both sites over nine growing seasons but had varied impact on ground cover (see Table 2). Total canopy cover 9 yr after tree removal was more than twofold greater, on average, across all treatments at the sites (54 − 73%) relative to pretreatment values (21 − 32%). Bare skeletons of downed trees (see Figs. 1D and 2C) made up 12% and 7% of total canopy cover in the cut treatments at

Marking Corral and Onaqui, respectively. The density of live trees ≥ 1 m in height was ≈239 trees per ha (94 single-leaf pinyon and 145 Utah juniper trees per ha) in the cut treatment at Marking Corral. The same measures in the cut and mastication treatments at Onaqui were 17 and 20 Utah juniper trees per ha, respectively. Canopy cover of live trees ≥ 1 m in height averaged < 1 − 3% across treatments at both sites and canopy cover of live trees < 1 m in height did not exceed 1% at either site. The low initial canopy cover of shrubs at both sites (3 − 15%, see Figs. 1A-B and 2A-B) was enhanced to 29% at Marking Corral and 17% at Onaqui by tree cutting but remained near 10% in the mastication treatment at Onaqui 9 yr after tree removal (see Table 2). Approximately 87% of the total shrub density and 82% of shrub canopy cover was sagebrush (0.90 shrubs per m^2, 24% canopy cover) in the cut treatment at Marking Corral. Sagebrush density was similar across cut (0.71 shrubs per m^2) and mastication (0.83 shrubs per m^2) treatments at Onaqui and made up approximately 82%, on average, of total live shrub density post treatment at that site. Canopy cover of sagebrush at Onaqui was 15% in the cut treatment and 9% in the mastication treatment, a fivefold increase from pretreatment values for the cut treatment area. Grass canopy cover was near 10%, on average, across the sites before tree removal and increased to an average of 32% across all treatments in the ninth yr (see Table 2). Tall perennial bunchgrasses made up 24% of the grass canopy cover at Marking Corral after cutting (Fig. 1C), a nearly threefold increase from pretreatment levels (Fig. 1B). At Onaqui, tall perennial grass cover (Fig. 2B-D) increased more than threefold in the cut (23%) and mastication (33%) treatment areas over the nine growing seasons. Bluebunch wheatgrass (*Pseudoroegneria spicata* [Pursh] A. Love) was the dominant perennial bunchgrass at both sites post treatment, representing 74 − 81% of the total grass cover across treatments after nine growing seasons. Cheatgrass cover remained low (1 − 7%) across both sites following tree removal. All treatments increased basal cover of plants, but total ground cover during the study only increased in the treatment areas at the initially more bare Onaqui site (see Table 2). Downed trees provided < 2% ground cover across the cut treatments, while mulch from shredded trees (Fig. 2D) provided 18% ground cover in the mastication area at Onaqui. In general, mechanical tree removal enhanced coverage of sagebrush steppe vegetation at both sites, but bare ground remained high (44 − 53%) in all treatment areas (see Table 2) 9 yr post treatment.

Small Plot Vegetation and Surface Conditions

Microsite influences on vegetation and ground surface conditions before treatment tended to persist 9 yr after tree removal regardless of treatment, with treatment effects on vegetation and ground cover mainly limited to shrub and tree coppices (see Tables 3 and 4). Cutting and mastication treatments elicited no significant changes in the percentages of canopy and ground cover or surface roughness across interspaces at the sites (see Tables 3 and 4). Therefore, interspaces 9 yr post treatment exhibited similar bare ground (75 − 90% average bare ground) conditions to those measured on control interspaces (~87% bare ground). Tree removal had minimal impact on canopy or ground cover attributes on the well-vegetated shrub coppices at Marking Corral, solely altering forb canopy cover from 3% to < 1%, on average, post treatment (see Table 3). At Onaqui, tree mastication increased total canopy and shrub canopy cover on shrub coppices by approximately twofold, on average, and both tree-removal treatments increased shrub-coppice litter cover by nearly twofold (see Table 4). Ground cover increases on shrub coppices in the cut treatment at Onaqui reduced bare ground and increased ground surface roughness (see Table 4). Tree cutting increased total canopy cover and grass canopy cover on tree coppices at Marking Corral but had no effect on canopy or ground cover by any other cover type for tree plots at that site (see Table 3). Likewise, tree cutting had a negligible impact on tree-coppice canopy and ground cover attributes at Onaqui but did increase plant basal cover slightly and surface roughness by 4 mm on tree plots at that site (see Table 4). The mastication treatment increased total canopy and grass canopy covers and plant basal ground cover on tree coppices at Onaqui but had minimal impact on other canopy and ground values on tree plots. Surface roughness tended to be highest for areas underneath shrubs and was unaltered by tree removal except for shrub and tree coppices within the cut at Onaqui (see Tables 3 and 4). Aggregate stability and litter depth tended to be greatest for areas under trees, followed by shrubs and then interspaces, and were minimally affected by the tree removal treatments (see Tables 3 and 4). Overall, the ground surface before and 9 yr after tree removal treatments was mainly bare in interspaces and was well covered by vegetation and a thin litter layer on shrub coppices and a dense well-distributed litter layer on tree coppices (see Tables 3 and 4).

Patch-Scale Vegetation and Surface Conditions

Tree-removal treatments increased patch-scale vegetation and ground cover at both sites, but treatment effects at the more vegetated Marking Corral site were mainly restricted to shrub-interspaces in the cut with a downed-tree (cut-downed-tree treatment; see Tables 5 and 6). Total canopy cover was substantially greater for shrub-interspace plots in the cut-downed-tree treatment relative to control shrub-interspace plots at both sites due largely to 49% and 24% canopy cover of downed trees at Marking Corral and Onaqui, respectively. All other canopy cover values were similar for treated and untreated shrub-interspace plots at Marking Corral, but the cut-downed-tree treatment enhanced litter cover (dead needle fall) by fourfold and reduced bare ground by eightfold at that site (Table 5). Substantial cover of dead tree needles on cut-downed-tree plots at Marking Corral may have contributed to a 3-fold reduction in plant basal cover (see Table 5). The cut-downed-tree treatment at Onaqui increased shrub canopy cover by more than 200-fold, grass cover by 8-fold, and litter cover by 5-fold (see Table 6). The substantial increase in litter cover under downed trees at Onaqui reduced bare ground from near 90% to 38% (see Table 6). Downed trees also effectively reduced shrub-interspace zone canopy and basal gaps for the 25- to 50-cm and 51- to 100-cm gap classes at both sites (see Tables 5 and 6). Cutting also increased shrub canopy cover on shrub-interspace zone plots without downed trees at Onaqui but had no favorable effect on enhancement of other vegetation and ground cover attributes on those plots (see Table 6). The mastication treatment improved shrub and grass canopy cover in shrub-interspaces at Onaqui but had no significant impact on ground cover levels. Canopy and ground cover in tree zones at Marking Corral were principally unaltered by tree removal (see Table 5). Both the cut and mastication treatments at Onaqui facilitated increased shrub canopy cover and plant basal cover in tree zones, but mastication decreased tree-zone forb cover (see Table 6). The cut treatment reduced litter cover and increased bare ground in tree zones at Onaqui by threefold, while mastication had no significant impact on litter and bare ground in tree zones. Mechanical treatments reduced basal gaps for the 25- to 50-cm gap class in tree zones at Onaqui, but the mastication treatment increased the proportion of gaps in the 201- to 400-cm size class for tree zones at that site. The ground surface within gaps between plants in masticated tree zones was well covered, however, with tree mulch (51% cover) and other litter (18% cover). Ground surface roughness at Marking Corral averaged 18 mm across control plots and cut shrub-interspaces without downed trees and was substantially increased in shrub-interspace zones by the cut-downed-tree treatment (42 mm) and by the cutting treatment in tree zones (30 mm). Ground surface roughness at Onaqui averaged 18 mm across control plots and cut shrub-interspaces without downed trees and was increased by the cut-downed-tree treatment (46 mm) and by cutting in tree zones (46 mm). Ground surface roughness in the mastication treatment at Onaqui was 26 mm and 29 mm on shrub-interspace and tree zones, respectively, and the microsite values were not significantly different from control plots. Collectively, the study sites exhibited contrasting patch-scale vegetation and ground cover responses to tree removal with exception of enhanced surface soil protection in the cut-downed-tree

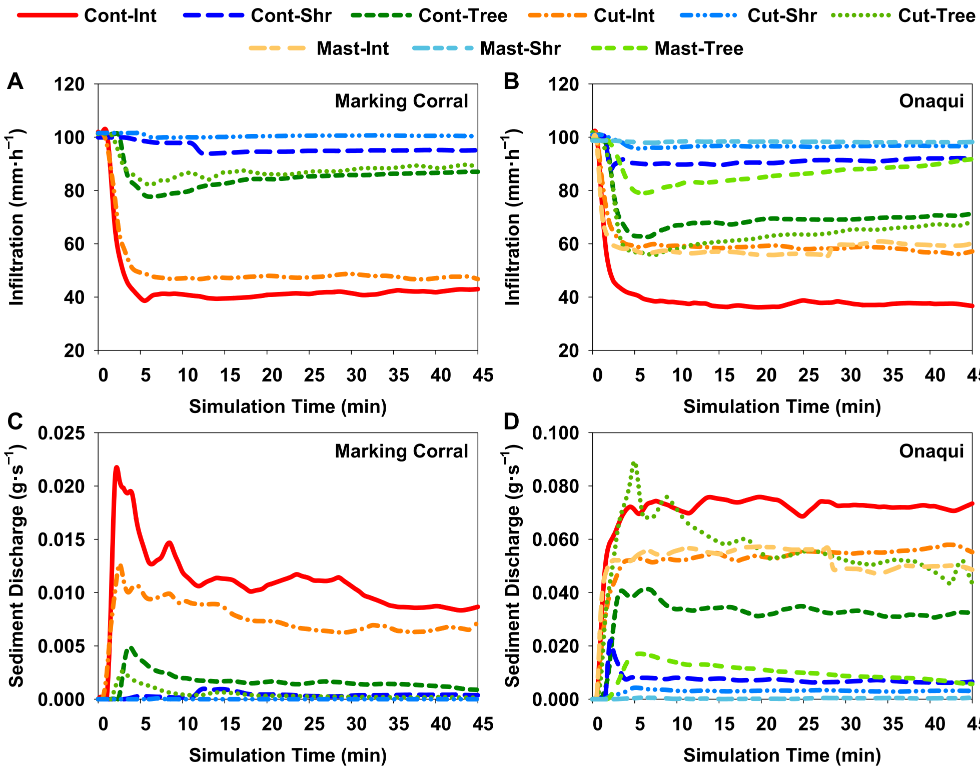

Figure 4. Infiltration (**A** and **B**) and sediment discharge (**C** and **D**) for wet-run (102 mm • h^{-1}, 45 min) rainfall simulations on interspace (Int), shrub coppice (Shr), and tree coppice (Tree) small plots (0.5 m^2) in control (Cont), cut (Cut), and mastication (Mast) treatments at Marking Corral and Onaqui 9 yr after treatment.

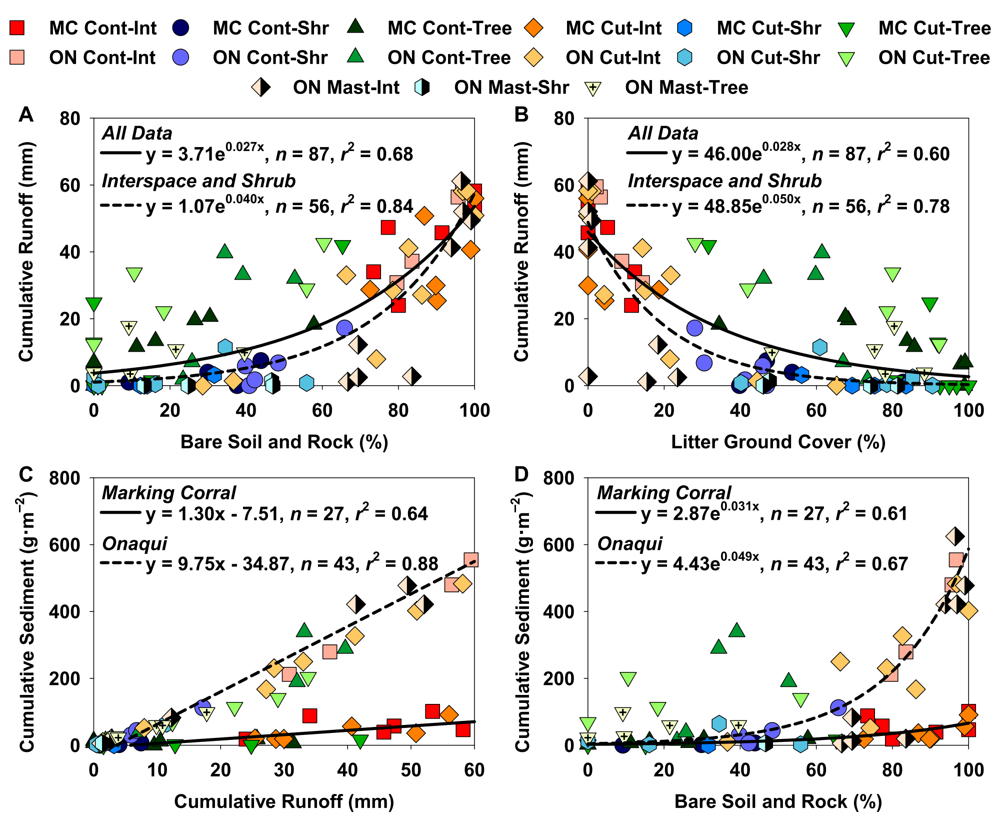

Figure 5. Correlations of cumulative runoff with bare soil and rock cover (**A**) and litter ground cover (**B**) and of cumulative sediment with cumulative runoff (**C**) and bare soil and rock cover (**D**) for wet-run (102 mm • h^{-1}, 45 min) rainfall simulations on interspace (Int), shrub coppice (Shr), and tree coppice (Tree) small plots (0.5 m^2) in control (Cont), cut (Cut), and mastication (Mast) treatments at Marking Corral (MC) and Onaqui (ON) 9 yr after treatment. All correlations shown were significant ($P < 0.05$).

Table 7
Average runoff, infiltration, sediment, wetting depth (percent wet), and soil water repellency response variables measured for dry- and wet-run rainfall simulations (0.5 m^2) in control and cut treatments at Marking Corral 9 yr post treatment. Means within a row followed by different lowercase letters are significantly different ($P < 0.05$).

Marking Corral	Control			Cut		
Rainfall simulation variable	Interspace	Shrub coppice	Tree coppice	Interspace	Shrub coppice	Tree coppice
Dry run simulation (64 $mm \cdot h^{-1}$, 45 min)						
Cumulative runoff (mm)	18 c	1 a	8 ab	10 bc	0 a	7 ab
Runoff-to-rainfall ($mm \cdot mm^{-1}$) × 100%	37 c	1 a	17 ab	21 bc	0 a	14 ab
Mean infiltration rate ($mm \cdot h^{-1}$)[1]	40 a	—	50 b	47 ab	—	41 ab
Cumulative sediment ($g \cdot m^{-2}$)[1]	18 b	—	9 ab	7 ab	—	6 a
Sediment/runoff ($g \cdot m^{-2} \cdot mm^{-1}$)[1]	1.02 b	—	0.83 b	0.44 ab	—	0.27 a
Percent wet at 0- to 6-cm depth	100 a	100 a	83 a	100 a	96 a	86 a
Percent wet at 0-10 cm depth	97 a	100 a	77 a	96 a	96 a	80 a
Percent wet at 0-20 cm depth	59 a	70 a	69 a	78 a	69 a	68 a
Mean soil water repellency (s)[2]	—	—	80 a	—	—	58 a
Depth of max water repellency (cm)[3]	—	—	1 a	—	—	1 a
Percent of plots with runoff	100	25	88	83	0	38
Wet run simulation (102 $mm \cdot h^{-1}$, 45 min)						
Cumulative runoff (mm)	44 c	3 ab	12 b	39 c	1 a	10 ab
Runoff-to-rainfall ($mm \cdot mm^{-1}$) × 100%	57 c	4 ab	16 b	51 c	1 a	13 ab
Mean infiltration rate ($mm \cdot h^{-1}$)[1]	44 a	94 b	83 b	51 a	—	75 b
Cumulative sediment ($g \cdot m^{-2}$)[1]	59 b	3 a	10 a	40 b	—	5 a
Sediment/runoff ($g \cdot m^{-2} \cdot mm^{-1}$)[1]	1.35 c	0.41 ab	0.67 bc	0.97 c	—	0.15 a
Percent of plots with runoff	100	75	88	100	25	50
No. of plots	6	4	8	6	4	8

[1] Means based solely on plots that generated runoff.
[2] Mean soil water repellency for 0- to 5-cm soil depth assessed as water drop penetration time (WDPT, 300 s maximum). Soils were classified slightly water repellent if WDPT ranged 5 to 60 s and strongly water repellent if WDPT exceeded 60 s (Bisdom et al., 1993).
[3] Soil depth (below mineral soil surface) with highest average WDPT.

treatment associated with increases in litter and reductions in bare ground and canopy and basal gaps (see Tables 5 and 6). All treatments were effective at improving sagebrush shrub cover at the more degraded Onaqui site (see Table 6).

Small Plot Rainfall Simulations

Hydrologic and erosion responses to rainfall simulations reflect the limited effects of tree removal on vegetation and ground cover at the small-plot scale and the persistence of microsite controls on runoff generation and erosion (see Figs. 4 and 5; Tables 7 and 8). Runoff was controlled by bare ground and the percentage of litter cover (see Fig. 5A and B). Sediment yield was primarily controlled by the amount of runoff and percent bare ground (see Fig. 5C and D), with sediment delivery generally greater at Onaqui than Marking Corral regardless of treatment (see Tables 7 and 8, Figs. 4C-D and 5C-D). Infiltration was lowest, and runoff and erosion were highest for interspaces in control areas at both sites, and hydrologic and erosion responses to rainfall on interspaces were not altered by tree removal (see Tables 7 and 8). Runoff and erosion levels were low (0 − 6 mm, 0 − 48 $g \cdot m^{-2}$) for dry- and wet-run

Table 8
Average runoff, infiltration, sediment, wetting depth (percent wet), and soil water repellency response variables measured for dry- and wet-run rainfall simulations (0.5 m^2) in control, cut, and mastication treatments at Onaqui 9 yr post treatment. Means within a row followed by different lowercase letters are significantly different ($P < 0.05$).

Onaqui	Control			Cut			Mastication		
Rainfall simulation variable	Interspace	Shrub coppice	Tree coppice	Interspace	Shrub coppice	Tree coppice	Interspace	Shrub coppice	Tree coppice
Dry run simulation (64 $mm \cdot h^{-1}$, 45 min)									
Cumulative runoff (mm)	18 b	1 a	8 ab	13 b	1 a	12 b	12 b	0 a	2 a
Runoff-to-rainfall ($mm \cdot mm^{-1}$) × 100%	38 b	2 a	16 ab	27 b	1 a	27 b	27 b	0 a	5 a
Mean infiltration rate ($mm \cdot h^{-1}$)[1]	38 a	61 b	54 b	41 a	60 b	45 ab	40 a	—	57 b
Cumulative sediment ($g \cdot m^{-2}$)[1]	126 b	9 a	53 ab	133 b	8 a	109 b	130 b	—	14 a
Sediment/runoff ($g \cdot m^{-2} \cdot mm^{-1}$)[1]	6.76 a	5.55 a	7.39 a	8.21 a	4.97 a	7.58 a	7.23 a	—	5.92 a
Percent wet at 0-6 cm depth	88 bc	90 bc	58 ab	92 c	93 c	48 a	97 c	100 c	44 a
Percent wet at 0-10 cm depth	82 bc	84 bc	54 ab	84 bc	83 bc	42 a	92 c	99 c	44 a
Percent wet at 0-20 cm depth	68 ab	54 ab	41 a	57 ab	42 a	30 a	65 ab	82 b	41 a
Mean soil water repellency (s)[2]	—	—	98 a	—	—	82 a	—	—	88 a
Depth of max water repellency (cm)[3]	—	—	0 a	—	—	1 a	—	—	1 a
Percent of plots with runoff	100	60	100	80	40	100	78	0	100
Wet run simulation (102 $mm \cdot h^{-1}$, 45 min)									
Cumulative runoff (mm)	46 d	6 ab	23 bc	31 cd	3 a	28 cd	28 cd	0 a	9 ab
Runoff-to-rainfall ($mm \cdot mm^{-1}$) × 100%	61 d	9 ab	30 bc	41 cd	4 a	37 cd	37 cd	0 a	12 ab
Mean infiltration rate ($mm \cdot h^{-1}$)[1]	40 a	88 c	70 bc	56 ab	96 c	65 bc	62 ab	—	87 c
Cumulative sediment ($g \cdot m^{-2}$)[1]	381 c	48 ab	174 bc	316 c	20 a	300 c	258 c	—	53 ab
Sediment/runoff ($g \cdot m^{-2} \cdot mm^{-1}$)[1]	8.05 b	5.44 ab	7.30 ab	8.33 b	4.81 a	8.88 b	7.75 b	—	6.08 ab
Percent of plots with runoff	100	80	100	90	80	100	100	25	100
No. of plots	4	5	5	10	5	5	9	4	5

[1] Means based solely on plots that generated runoff.
[2] Mean soil water repellency for 0- to 5-cm soil depth assessed as water drop penetration time (WDPT, 300 s maximum). Soils were classified slightly water repellent if WDPT ranged 5 − 60 s and strongly water repellent if WDPT exceeded 60 s (Bisdom et al., 1993).
[3] Soil depth (below mineral soil surface) with highest average WDPT.

Table 9
Runoff, sediment, and flowpath variables by flow release rate for concentrated flow experiments (9 m^2) in control and cut treatment areas at Marking Corral 9 yr post treatment. Means within a row followed by different lowercase letters are significantly different ($P < 0.05$).

Marking Corral	Release rate ($L \cdot min^{-1}$)	Control		Cut		Cut-Downed-Tree[1]
Concentrated flow variable		Shrub-interspace zone	Tree zone	Shrub-interspace zone	Tree zone	Shrub-interspace zone
Cumulative runoff (L)	15	43 b	0 a	35 b	2 a	0 a
	30	171 b	30 a	143 b	40 a	6 a
	45	287 b	106 ab	257 b	67 a	59 a
	Total	501 b	136 a	435 b	109 a	65 a
Cumulative sediment (g)[2]	15	82 a	—	99 a	—	—
	30	718 c	203 ab	693 bc	161 a	—
	45	1542 b	220 a	1108 b	35 a	27 a
	Total	2342 b	423 a	1900 b	198 a	35 a
Sediment Concentration ($g \cdot L^{-1}$)[2]	15	2.3 a	—	3.9 a	—	—
	30	3.8 a	4.7 a	3.9 a	2.0 a	-
	45	5.2 b	1.9 ab	3.6 b	1.3 ab	0.8 a
Flow velocity ($m \cdot s^{-1}$)[2]	15	0.10 a	—	0.08 a	—	—
	30	0.15 b	0.04 a	0.16 b	0.06 a	—
	45	0.22 b	0.05 a	0.29 b	0.08 a	0.05 a
Flow path width (cm)[2]	15	30 a	—	63 b	—	—
	30	28 ab	13 a	74 c	63 bc	—
	45	28 a	30 a	62 b	62 b	46 ab
Flow path depth (cm)[2]	15	0.70 a	—	0.70 a	—	—
	30	0.91 a	1.04 a	0.91 a	0.95 a	—
	45	1.07 b	0.83 ab	1.13 b	0.67 ab	0.65 a
Percent of plots with runoff	15	100	0	100	40	0
	30	100	80	100	60	50
	45	100	100	100	100	100
No. of plots		5	5	5	5	4

[1] Plots in the cut treatment with a residual cut dead single-leaf pinyon (*Pinus monophylla* Torr. & Frem.) or Utah juniper (*Juniperus osteosperma* [Torr.] Little) tree laying parallel to the ground surface and perpendicular to the long axis of the respective plot (across plot along hillslope contour).
[2] Means based solely on plots that generated runoff.

simulations on shrub coppices in control areas at both sites, and, as with interspaces, shrub-coppice infiltration rates and cumulative runoff and erosion were not significantly affected by tree removal treatments at either site (see Fig. 4, Tables 7 and 8). However, the percentage of shrub coppices generating runoff and sediment declined following tree removal at both sites (see Tables 7 and 8). Tree coppices generated low levels of runoff (7 — 12 mm) and sediment (5 — 10 $g \cdot m^{-2}$) during rainfall simulations at Marking Corral regardless of treatment. Tree coppices at Onaqui generated low to moderate levels of runoff (8 — 28 mm) and moderate to high sediment yield (53 — 300 $g \cdot m^{-2}$) across control and cut conditions. Tree removal did not significantly affect runoff and sediment delivered from tree coppices at either site with respect to controls (see Tables 7

Table 10
Runoff, sediment, and flowpath variables by flow release rate for concentrated flow experiments (9 m^2) in control, cut, and mastication treatment areas at Onaqui 9 yr post treatment. Means within a row followed by different lowercase letters are significantly different ($P < 0.05$).

Onaqui	Release rate ($L \cdot min^{-1}$)	Control		Cut		Cut-Downed Tree[1]	Mastication	
Concentrated flow variable		Shrub-interspace zone	Tree zone	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Shrub-interspace zone	Tree zone
Cumulative runoff (L)	15	20 b	1 a	38 b	17 b	0 a	16 b	0 a
	30	106 b	13 a	141 b	74 b	21 a	81 b	5 a
	45	245 c	89 ab	262 c	175 bc	94 ab	186 bc	40 a
	Total	371 c	103 b	441 c	266 c	115 b	283 c	45 a
Cumulative sediment (g)[2]	15	135 a	—	251 a	199 a	—	126 a	—
	30	694 b	111 a	976 b	892 b	237 a	598 b	—
	45	2186 c	500 ab	2144 c	1400 bc	586 ab	1556 c	230 a
	Total	3015 c	619 b	3371 c	2491 c	823 b	2280 c	392 a
Sediment Concentration ($g \cdot L^{-1}$)[2]	15	6.6 a	—	6.6 a	8.1 a	—	6.3 a	—
	30	6.5 a	5.1 a	6.9 a	8.4 a	6.3 a	7.5 a	—
	45	8.3 a	5.5 a	8.3 a	7.6 a	6.1 a	8.2 a	6.1 a
Flow velocity ($m \cdot s^{-1}$)[2]	15	0.07 a	—	0.08 a	0.05 a	—	0.06 a	—
	30	0.10 b	0.04 a	0.11 b	0.06 a	0.06 a	0.10 b	—
	45	0.12 c	0.07 ab	0.18 d	0.10 bc	0.07 ab	0.16 cd	0.04 a
Flow path width (cm)[2]	15	93 b	—	90 b	40 a	—	57 a	—
	30	164 b	98 ab	88 ab	60 a	104 ab	52 a	—
	45	184 c	125 bc	97 abc	86 ab	88 abc	61 a	69 ab
Flow path depth (cm)[2]	15	0.58 a	—	0.72 a	0.63 a	—	0.58 a	—
	30	0.68 a	0.71 a	0.96 a	0.83 a	0.74 a	0.74 a	—
	45	0.99 b	0.79 b	1.16 b	1.02 b	0.78 b	0.85 b	0.40 a
Percent of plots with runoff	15	100	40	100	80	0	80	20
	30	100	60	100	80	60	100	20
	45	100	100	100	100	100	100	100
No. of plots		5	5	5	5	5	5	5

[1] Plots in the cut treatment with a residual cut dead Utah juniper (*Juniperus osteosperma* [Torr.] Little) tree laying parallel to the ground surface and perpendicular to the long axis of the respective plot (across plot along hillslope contour).
[2] Means based solely on plots that generated runoff.

and 8), but cumulative runoff and sediment yield were lower for tree coppices in the mastication (2 − 9 mm, 14 − 53 g • m^{-2}) than those in the cut treatment (12 − 28 mm, 109 − 300 g • m^{-2}) at Onaqui. Water-repellent conditions were persistent across tree coppice plots in control and treated areas at both sites (see Tables 7 and 8) but only affected the depth of wetting at Onaqui. Dense litter cover on tree coppices at both sites buffered the effects of repellency on runoff generation. However, the influence of repellency on tree coppice runoff is reflected in the rapid initial dip in infiltration (to a minimum) followed by continuous increasing infiltration throughout rainfall simulation on control and treated tree plots at both sites (Fig. 4A and B). Soil water repellency on tree plots increased variability in runoff for low levels of bare ground (Fig. 5A) and high levels of litter cover (Fig. 5B), but overall trends in primary controls on runoff and erosion were consistent across treatments at both sites and clearly demonstrate post-treatment persistence of microsite effects (see Figs. 4 and 5).

Concentrated Flow Experiments

Downed trees were effective at reducing runoff and sediment delivery from concentrated flow releases in the intercanopy at both sites, but reduction of litter following cutting increased concentrated flow runoff and erosion in tree zones at Onaqui (Tables 9 and 10). Runoff from concentrated flow experiments across both sites and all treatments was controlled by the amount of bare ground and litter cover (see Fig. 6A and B). Sediment delivery from concentrated flow experiments was controlled primarily by the amount of runoff but was also well correlated with the amount of bare ground and flow velocity (see Figs. 6C − E). Flow velocity increased with increasing runoff (see Fig. 6F) and bare ground ($r^2 = 0.58$ for 30 L • min^{-1}, $r^2 = 52$ for 45 L • min^{-1}) and, along with runoff and sediment delivery, was higher for shrub-interspace than tree zones at both sites (see Tables 9 and 10). Decreased bare ground and increased litter cover in cut-downed-tree treatments (see Tables 5 and 6) reduced total runoff from shrub-interspaces by more than sevenfold at Marking Corral and more than threefold at Onaqui (see Tables 9 and 10). The low levels of runoff and rougher surface conditions in the cut-downed-tree treatments buffered flow velocity, and the reduced runoff, flow energy, and bare ground resulted in lower total sediment delivery from shrub-interspace plots with downed trees relative to those in controls (see Tables 9 and 10). Shrub-interspace plots in the cut treatment at a site without a downed tree produced similar runoff and sediment amounts as shrub-interspace zones in the control (Tables 9 and 10), although flow paths were wider for all flow rates in the cut treatment relative to the control at Marking Corral (see Table 9). Tree cutting had no significant effect on the low runoff and sediment amounts delivered from tree zones at Marking Corral (see Table 9). In contrast, reduction of litter in tree zone plots within the cut at Onaqui (see Table 6) promoted 2- to nearly 20-fold higher runoff and sediment delivery relative to tree zone plots in the control at that site (see Table 10). Flow velocity for tree zones was similar across control and cut treatments at Onaqui (see Table 10), and, therefore, the increased erosion is attributed to the elevated runoff and increased availability of highly erodible soil (greater bare ground). The mastication treatment at Onaqui had no significant effect on amounts of runoff and sediment delivered from concentrated flow releases in shrub-interspace zones but did result in lower total runoff and sediment delivered from treated tree zone plots relative to control tree zone plots (see Table 10).

Discussion

Vegetation and Ground Cover Responses to Tree Removal

Trends in hillslope-scale shrub and perennial herbaceous cover responses to tree removal in this study (see Table 2) are consistent with those reported in other mid- to long-term studies of mechanical tree removal in Great Basin sagebrush steppe (Chambers et al., 2014a; Miller et al., 2014a; Roundy et al., 2014a; Bates and Davies, 2016; Bates et al., 2017; Williams et al., 2017). Chambers et al. (2014a) evaluated the effects of a cut-and-drop tree-removal treatment on vegetation over a 4-yr period at nine sagebrush sites in the Great Basin Region. The sites spanned a range of warm-dry to cool-moist soil moisture − temperature regimes, respectively, represented by Wyoming sagebrush (*A. tridentata* Nutt. ssp. *wyomingensis* Beetle & Young) and mountain big sagebrush (*A. tridentata* Nutt. ssp. *vaseyana* [Rydb.] Beetle) as the dominant shrub cover. The sites were in various stages of woodland encroachment by pinyon and juniper but had sagebrush and native perennial bunchgrasses and forbs in the understory at the time of treatment. Chambers et al. (2014a) found that the cut-and-drop treatment increased hillslope-scale shrub cover relative to untreated areas by the fourth yr after tree removal and increased perennial herbaceous cover in the second, third, and fourth yrs after tree removal. Shrub cover was ~10 − 14% in control areas and ~13 − 19% in the treated areas the fourth yr post treatment. Perennial herbaceous cover was near 25% in control areas and ~30 − 35% in cut treatment areas 4 yr after tree removal. Chambers et al. (2014a) found that perennial grass responses reflected pretreatment conditions and that post-treatment perennial grass cover was greater on sites treated with low initial tree cover (0 − 20%) than on sites treated with high initial tree cover (40 − 75%). Miller et al. (2014a), a companion study to the Chambers et al. (2014a), evaluated the impacts of tree removal on sagebrush steppe vegetation over a 3-yr period after treatment at 11 sites in the Great Basin spanning warm-dry to cool-moist soil temperature-moisture regimes. Total shrub and sagebrush cover 3 yr post treatment were similar for cut (12% and 8%) and control areas (10% and 6%) across all sites, but sagebrush seedling density was greater for the cut than control plots. Tall perennial grass cover was nearly twofold greater for cut (17%) than control (10%) areas 3 yr after treatment. Short perennial grass cover averaged 5% across treated and untreated areas in the third yr. Miller et al. (2014a) attributed increases in perennial grass cover over the 3 yr after treatment to enhanced productivity of residual plants associated with increases in soil water, as demonstrated in a companion study by Roundy et al. (2014b). The authors further reported that increases in perennial grass cover during the study occurred for sites spanning low to high tree dominance. Most sites in the Miller et al. (2014a) study had > 5% cover of perennial grasses before treatment. Roundy et al. (2014a) evaluated effects of cut-and-drop tree removal on vegetation at the same sites in Miller et al. (2014a) and the effects of tree mastication at four additional sagebrush sites over a 3-yr period post treatment. The study found tree cutting and mastication increased shrub and perennial herbaceous cover for all phases (I − III, low to high TDI) of pinyon and juniper encroachment. The authors reported that increases in perennial herbaceous cover occurred more rapidly following mechanical tree removal at low TDI but increased most following tree removal at mid to high TDI due to lower initial understory cover and greater increases in soil water post treatment under these conditions (Roundy et al., 2014a, 2014b). Williams et al. (2017) conducted follow-up vegetation measurements in control and cut treatments at the same sites as Miller et al. (2014a) and Roundy et al. (2014a) 6 yr after tree cutting. The study found that tree cutting increased shrub density over the 6-yr period after cutting, maintained sagebrush cover across low to high TDI, and resulted in nearly 3% more sagebrush cover in the sixth yr post treatment. Cutting also increased tall perennial grass cover at mid- to high-TDI by the third and sixth yr post-treatment. Bates et al. (2000, 2005, 2017) assessed the impacts of cutting western juniper (*J. occidentalis* Hook.) on sagebrush steppe vegetation at a single site over a 25-yr period. The site was in phase III of woodland encroachment at the time of treatment, with 26% cover (250 trees per ha) of western juniper and near 70% bare ground (Bates et al., 2000, 2005, 2017). The authors reported initial sagebrush cover at the site was < 1% and that 15 − 20% is common for similar sites without tree encroachment. Density of perennial grasses and basal cover of understory

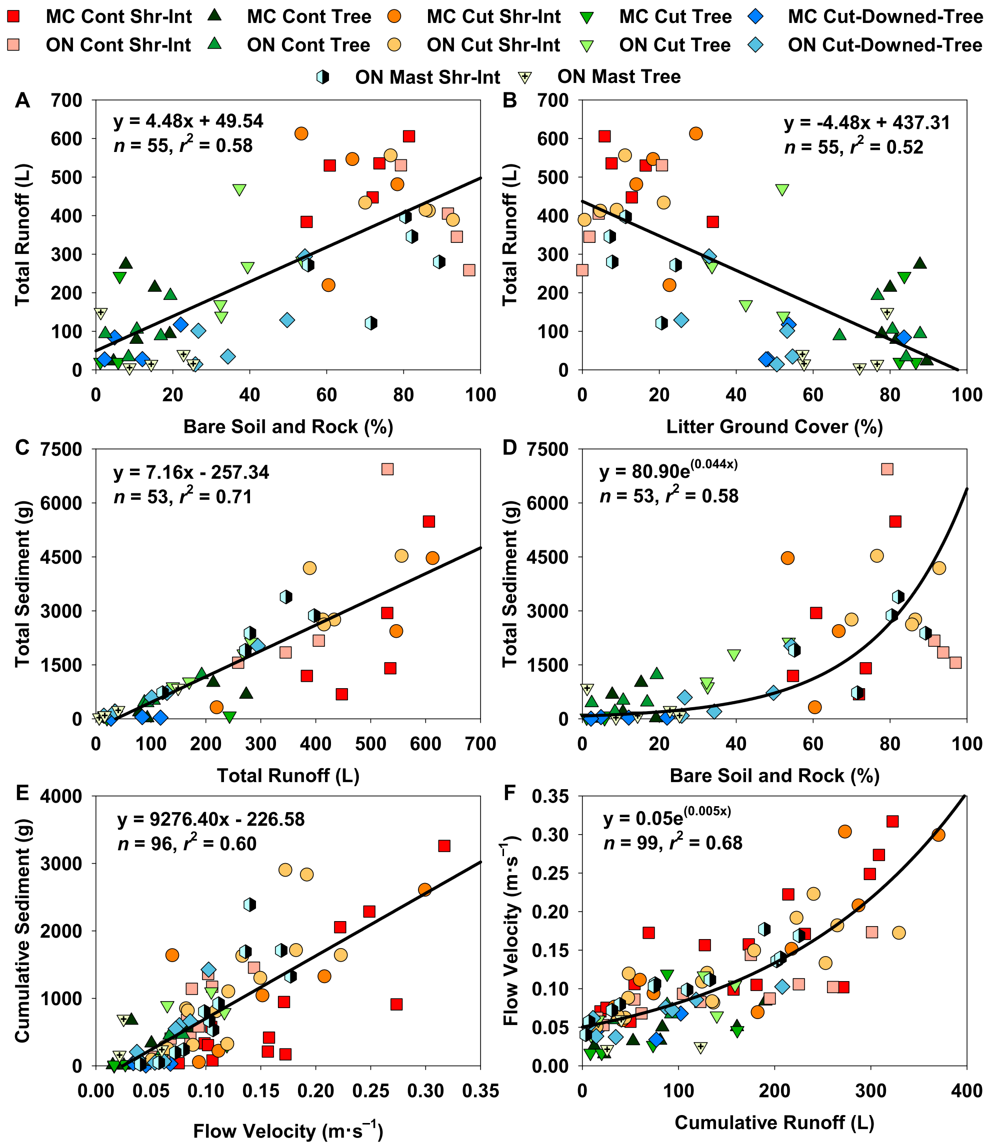

Figure 6. Correlations of total runoff with bare soil and rock cover (**A**) and litter ground cover (**B**); of total sediment with total runoff (**C**) and bare soil and rock cover (**D**); of cumulative sediment with flow velocity (**E**); and of flow velocity with cumulative runoff (**F**) for concentrated flow experiments (15, 30, 45 L • min^{-1}, 12 min each) on shrub-interspace zone (Shr-Int) and tree zone (Tree) plots (9 m^2) in control (Cont), cut (Cut), and mastication (Mast) treatments and in shrub-interspace zones within the cut treatment area with a cut-downed-tree (Cut-Downed-Tree) at Marking Corral (MC) and Onaqui (ON) Onaqui 9 yr after treatment. Total runoff and total sediment are the sums of cumulative runoff and sediment, respectively, for all three flow release rates on a given plot. All correlations shown were significant ($P < 0.05$).

perennial vegetation were 2.9 plants per m^2 and < 3%, respectively before tree removal (Bates et al., 2000). Shrub density was near 0 plants per m^2 before treatment and was similar for cut and control conditions until the sixth yr after cutting (Bates et al., 2005). Low initial sagebrush cover delayed recruitment following tree removal, and sagebrush cover remained statistically similar for treated and untreated conditions until 18 yr after tree cutting (Bates et al., 2017). Sagebrush cover was 4.4% in the cut treatment area and remained < 1% in the control 25 yr post treatment. Perennial bunchgrass density increased in the cut treatment by the second growing season, was five- to sixfold greater for the cut than control areas in the 6th-13th yr after treatment, and peaked the 14th yr after treatment (Bates et al., 2005, 2017). Perennial bunchgrass density declined in the cut treatment thereafter, but remained ~ threefold greater for cut than control conditions 25 yr post treatment (Bates et al., 2017). Experimental plots at both sites in this study were adjacent to, but separate from, treatment areas in the Chambers et al. (2014a), Miller et al. (2014a), and Roundy et al. (2014a, 2014b) studies cited earlier. Similar to those shorter studies and the Williams et al. (2017) study, generally low initial canopy coverage of shrubs (3 − 15%) and perennial herbaceous vegetation (9 − 12%) on our hillslope plots was stimulated over nine growing seasons (increased to 17 − 29% and 25 − 27%) through tree removal by cutting (see Table 2). Shrub cover on hillslope plots in the mastication was unchanged after nine growing seasons (6 − 10%) (see Table 2), but the density of sagebrush increased from 0.71 plants per m^2 pretreatment to 0.83 plants per m^2 over nine growing seasons after mastication. Tree shredding also increased perennial herbaceous cover from 13% pretreatment to 40% post treatment. Sagebrush species and tall perennial bunch grasses made up > 80% of total shrub and perennial herbaceous cover, respectively, across all treatments at our sites 9 yr after tree removal. Hillslope-scale forb cover was unaltered by tree removal treatments in our study (see Table 2), but forb cover response to tree removal can be highly variable in sagebrush steppe (Miller et al., 2013, 2014a; Bates et al., 2017). Sagebrush recovery in our study relative to that in the long-term study by Bates et al. (2017) reflects the greater initial sagebrush cover at our sites pretreatment. Perennial herbaceous cover responses are similar for our study and the longer-term Bates et al. (2017) study. Our findings are supported by the mid- to long-term studies mentioned earlier in demonstrating effectiveness of mechanical tree removal in increasing sagebrush steppe vegetation across low to high TDI (TDI of 0.51 − 0.66 in our study) and with at least 5% coverage, on average, of residual sagebrush and of native perennial grasses.

Results in this study corroborate that mechanical tree removal in phase II − III woodlands can increase sagebrush steppe vegetation on sites near the warm-dry and cool-moist soil temperature-moisture threshold without substantially increasing cheatgrass cover (Chambers et al., 2014a; Miller et al., 2014a; Roundy et al., 2014a). The primary determinants of cheatgrass responses to tree removal are 1) site soil temperature and moisture regimes, 2) pretreatment cheatgrass presence or a nearby seed source, 3) pretreatment presence and post-treatment survival and recruitment of native herbaceous vegetation, and 4) the expanse of well-connected bare ground (canopy and basal gap) post treatment (Chambers et al., 2007; Bates et al., 2011; Condon et al., 2011; Miller et al., 2013; Reisner et al., 2013; Bates et al., 2014; Chambers et al., 2014a, 2014b; Rau et al., 2014; Chambers et al., 2017). Chambers et al. (2014a) found that warm-dry to warm-moist sites dominated by Wyoming sagebrush were less resistant to increases in cheatgrass after mechanical tree removal than cool-moist sites with mountain big sagebrush. The Marking Corral site in our study is positioned at the warmer/lower elevation end of the cool soil temperature regime for the central Great Basin, and Onaqui is at the boundary of warm and cool temperature regimes (Miller et al., 2013). The west to southwest orientation of slopes at Marking Corral may further render that site as warm and susceptible to cheatgrass recruitment (Koniak, 1985). The copresence of black sagebrush (*A. nova* A. Nelson), Wyoming sagebrush, and mountain big sagebrush and the dominance of bluebunch wheatgrass post treatment at both sites are indicative of conditions near the warm-dry and cool-moist soil temperature − moisture thresholds (Miller et al., 2013). Mean annual precipitation for both sites is near the dry versus moist threshold (see Table 1). Although both sites in this study are potentially susceptible to substantial increases in cheatgrass following tree removal, cheatgrass cover was low (< 1%) before and 9 yr after (≤ 7%) treatment. We attribute limited recruitment of cheatgrass after tree removal primarily to 1) low initial cheatgrass cover, 2) > 10% canopy cover, on average, of native perennial vegetation pre-treatment, and 3) effectiveness of mechanical tree removal methods in preventing increases in bare ground. Chambers et al. (2014a) suggested 20% cover of native perennial herbaceous cover may be necessary to prevent large increases in cheatgrass after tree removal on warmer Wyoming and mountain big sagebrush sites. However, regional companion studies by Miller et al. (2014a) and Roundy et al. (2014a) reported low values of non-native herbaceous cover before and after mechanical tree removal across sites with an average of < 20% perennial herbaceous cover at the time of treatment. Sites in the Miller et al. (2014a) study averaged ~1 − 5% cover of non-native herbaceous vegetation, inclusive of cheatgrass, across treated and untreated areas 3 yr following tree cutting. Two sites in the Miller et al. (2014a) study exhibited high (~20 − 50%) cover of non-native annual herbaceous species before and after treatment. Cheatgrass cover in the Roundy et al. (2014a) study was < 6%, on average, across untreated and treated areas 3 yr post treatment. The long-term study by Bates et al. (2017) reported high variability in cheatgrass cover response to western juniper cutting over a 25-yr period. Cheatgrass cover was low (< 1%) before treatment in that study (Bates et al., 2000, 2005), but annual grass yield represented 20% of total herbaceous yield 25 yr after tree removal, partially due to temporarily favorable conditions for annuals. Our results are more consistent with those of the regional studies by Chambers et al. (2014a), Miller et al. (2014a), and Roundy et al. (2014a) indicating mechanical treatments can be effective tree removal practices to reestablish sagebrush vegetation while limiting cheatgrass. The more limited cheatgrass recruitment in our study relative to that of Bates et al. (2017) may have been facilitated by higher cover of sagebrush and perennial herbaceous vegetation pretreatment and post treatment at our sites (Chambers et al., 2007; Condon et al., 2011; Reisner et al., 2013; Bates et al., 2014; Chambers et al., 2014a; Miller et al., 2014a; Roundy et al., 2014a, 2014b; Chambers et al., 2017). Our results following mechanical tree removal contrast with those of tree removal with prescribed fire. Fire consumption of litter and perennial vegetation can leave large bare patches in the intercanopy and around tree bases that are favorable for cheatgrass establishment and growth (Melgoza et al., 1990; Blank et al., 2007; Chambers et al., 2007; Reisner et al., 2013; Chambers et al., 2014a, 2014b; Roundy et al., 2014b; Williams et al., 2014a; Chambers et al., 2017). Numerous studies have reported increased cheatgrass cover in bare patches, adjacent to burned trees, or underneath downed trees associated with fire consumption of litter and perennial herbaceous vegetation (Bates and Svejcar, 2009; Miller et al., 2013; Bates et al., 2014; Bates and Davies, 2016; Havrilla et al., 2017; Williams et al., 2018).

Minimal changes in canopy and ground cover values at the small-plot scale over nine growing seasons post treatment likely reflect the scale of measurement (see Tables 3 and 4) but also demonstrate that alteration of microsite-scale ground cover attributes through mechanical tree removal takes time. Contrasting canopy cover values measured across small-plot to hillslope-scales are partially attributed to inherent differences in measurement scales by plot type. That is, measures at the small-plot scale do not capture treatment effects on plant density that may occur over larger spatial scales (e.g., intercanopy, hillslope). In particular, interspace small plots exhibited a high variability in vegetation and ground cover before and after treatment (Cline et al., 2010; Pierson et al., 2010, 2014). In the current study, interspace plots commonly contained either a single bunchgrass plant or minor forb cover or were nearly 100% bare. Shrub coppice plots always contained a single shrub and, therefore, do not reflect treatment effects on shrub density over larger spatial scales. In the intercanopy, average values for grass cover were generally greater for shrub-interspace plots in treated areas than in control areas at both sites, but, with exception of the mastication treatment and cut-downed-tree plots, high variability in intercanopy grass cover resulted in no differences in grass canopy cover means for shrub-interspaces at a site (see Tables 5 and 6). Total canopy and shrub canopy cover were high on the 0.5-m^2 shrub coppice plots for untreated and treated conditions at both sites as expected (see Tables 3 and 4), were similar across untreated and cut shrub-interspace plots at Marking Corral (see Table 5), and were greater for shrub-interspaces in treated areas than control areas at Onaqui (see Table 6). The lack of increase in intercanopy shrub cover at Marking Corral (see Table 5) reflects the greater initial shrub cover at that site (see Table 2), as expected with a lower TDI (see Table 1; Roundy et al., 2014a, 2014b). Hillslope-scale measures of shrub density found tree removal maintained the number of shrubs at > 1 (1.04) plant per m^2 at Marking Corral and increased the number of shrubs from 0.61 to 0.82 plants per m^2 to 0.92 − 0.97 plants per m^2 at Onaqui over nine growing seasons. Hillslope-scale shrub cover increased at both sites following cutting but was unchanged by mastication at Onaqui (see Table 2). Although canopy cover trends varied with spatial scale, ground cover measures at all spatial scales reflect persistent (but reduced in some cases) high levels of intercanopy bare ground post treatment (see Tables 2 − 6). Nine growing seasons post treatment, bare ground remained > 70% in interspaces at both sites and bare conditions at the interspace-scale propagated well-connected bare intercanopy area at the hillslope scale (see Table 2). The persistence of extensive bare ground and low intercanopy litter cover at both sites clearly indicates substantial recruitment of hillslope ground cover on phase II − III woodlands with mid to high TDI can take more than a decade. Hillslope-scale litter cover 9 yr post treatment averaged 34 − 48% across sites and treatments and was similar for treated versus pretreatment conditions at both sites (see Table 2). Litter cover at the small-plot scale increased only for shrub coppices at Onaqui (see Table 4). The sole microsite-scale increases in intercanopy litter cover occurred in areas underneath cut-downed-trees (see Tables 5 and 6). Dead needles falling from cut-downed-trees were effective in reducing bare ground in shrub-interspaces at both sites, but these microsites represent near 10%, on average, of total area at the sites. The mastication treatment in this study did not target distribution of tree mulch into the intercanopy and therefore had limited impact on intercanopy litter cover. Consistent with our study, Miller et al. (2014a) found bare ground, on average, was similar for treated and untreated areas across 11 woodland sites 3 yr following tree cutting. In that study, bare ground and litter cover averaged 25% and 62% across cut treatment plots and 30% and 54% across control plots 3 yr after treatment. Bates et al. (2005, 2017) found that cutting western juniper reduced hillslope-scale bare ground after a single growing season at a sagebrush site in phase III of woodland encroachment, but bare ground still exceeded 40% in the cut treatment 13 yr after tree removal. Litter cover was similar across cut (35%) and uncut (32%) areas the 13th yr post treatment but was more evenly distributed in the cut treatment (Bates et al., 2005, 2017). The changes in canopy and ground cover across spatial scales in this study are typical for mechanical tree removal and contrast with more immediate changes common with tree removal by prescribed fire (Bates and Svejcar, 2009; Pierson et al., 2013; Bates et al., 2014; Miller et al., 2014a; Pierson et al., 2014; Roundy et al., 2014a; Williams et al., 2014a; Pierson et al., 2015; Bates and Davies, 2016; Williams et al., 2018).

Ecohydrologic and Erosion Responses to Mechanical Tree Removal

The primary driver of high runoff and erosion at both sites in this study for untreated conditions was extensive connectivity of runoff and erosion processes throughout bare intercanopy areas (Williams et al., 2014a, 2016a). Runoff and erosion on Great Basin rangelands increase with increasing bare ground and commonly increase exponentially where bare ground surpasses 50 − 60% (Pierson et al., 2008, 2009; Al-Hamdan et al., 2013; Pierson et al., 2013; Williams et al., 2014a, 2014b; Pierson and Williams, 2016; Williams et al., 2016a). Bare ground measured at the hillslope scale was 48 − 68% at our study sites before tree removal (see Table 2). High levels of runoff and sediment from rainsplash and sheetflow were delivered from bare interspaces (see Tables 7 and 8) within control intercanopy areas at both sites in this study and, as demonstrated by Pierson et al. (2010), provided ample runoff and sediment for delivery across spatial scales (Pierson et al., 2013; Williams et al., 2014a, 2014b, 2016a). Pierson et al. (2010) observed formation of concentrated overland flow with high-flow velocity during large plot (13 m^2) rainfall simulation experiments conducted in shrub-interspace zones at Marking Corral and Onaqui before tree removal. That study measured increasing sediment yield with increasing spatial scale at Marking Corral and Onaqui before tree removal and attributed the cross-scale responses to the observed high-velocity concentrated overland flow within the bare intercanopy during rainfall simulations. In the current study, runoff, sediment delivery, and overland flow velocity were all well correlated with the percentage of bare ground (see Figs. 5 and 6). Sediment delivery was also strongly correlated with runoff (see Figs. 5C and 6C) and flow velocity (see Fig. 6E), implicating reduction of overland flow as a key determinant in reducing erosion from the sites. The highest levels of runoff and sediment were delivered from the mostly bare interspaces and shrub-interspace zones at both sites (see Tables 7 − 10). In contrast, litter cover in tree zones trees limited concentrated flow runoff and sediment and buffered flow velocities (see Tables 9 − 10). The extensive bare ground with high surface cover of rock within the intercanopy at both study sites is indicative of substantial ongoing and long-term soil erosion. The higher intercanopy rock cover at Onaqui (52%) relative to Marking Corral (18%) and greater sediment-to-runoff ratios and erosion (see Tables 7 − 10, Fig. 5C and D) for that site suggest Onaqui is more vulnerable to long-term soil loss than Marking Corral. Results from the current study and our companion studies of pinyon and juniper woodlands (Pierson et al., 2007, 2010, 2013, 2014; Williams et al., 2014a; Pierson et al., 2015; Williams et al., 2016a, 2018) affirm conceptual models that suggest Great Basin sagebrush rangelands can become highly erodible in the later stages of woodland encroachment with increases in bare ground and that these ecosystems are susceptible to transitioning to a degraded eroded state without reversal of bare intercanopy conditions (Petersen et al., 2009; Chambers et al., 2014a; Williams et al., 2016b, 2016c).

Increases in ground cover associated with downed trees reduced the amount and energy of runoff and sediment delivery from shrub-interspace zones over time at both sites, but the limited mulch distribution in the mastication treatment was ineffective at reducing the high levels of intercanopy runoff and sediment yield (see Table 10). Pierson et al. (2013) found that placing downed trees in the intercanopy 1 and 2 yr after tree cutting had no effect on runoff and erosion from rainfall simulations and concentrated overland flow experiments at a western juniper woodland in Idaho, United States. The sagebrush site was in late phase II to phase III of juniper encroachment with 9% shrub canopy cover and > 80% bare ground in the intercanopy. Cut-downed-trees with residual dead needle cover were placed perpendicular to the hillslope contour on 9 − 13 m^2 experimental plots subjected to the same rainfall simulation and concentrated overland flow release rate and duration sequences as applied in this study. Pierson et al. (2013) found that runoff from rainfall simulations and concentrated flow releases tended to route underneath downed trees where the tree material was not in contact with the soil surface. Bare ground conditions were similar for plots with and without cut trees in the Pierson et al. (2013) study, as the downed trees had not deposited dead needles at the time of the experiments. Our companion study, Pierson et al. (2015), similarly found cutting and placing downed-trees with residual needle cover into bare intercanopy areas was ineffective at reducing bare ground, runoff, and erosion on rainfall simulation and concentrated flow experimental plots at Marking and Onaqui 1 yr after tree cutting. Runoff in the Pierson et al. (2015) experiments tended to form flow paths of high-velocity flow through breaks in contact of tree debris with the bare ground surface and delivered high levels of sediment yield through the plot outlet. In the current study, cut-downed-trees at Marking Corral and Onaqui had settled in place over a 9 yr period since cutting and had deposited tree needles underneath the tree skeletons (Figs. 1D and 2C). The ground surface under tree skeletons had, on average, 63% and 43% litter cover at Marking Corral and Onaqui, respectively, and the tree limbs at the ground surface were integrated into the litter mat. We observed runoff from concentrated flow releases on the cut-downed-tree plots tended to pool up behind tree debris and slowly disperse around/under the debris and through litter mat. The effect of tree debris and litter on runoff and soil erosion processes is evident in the 3- to 8-fold reductions in total runoff, 4- to 67-fold reductions in total sediment, and 2- to 4-fold reductions in overland flow velocity for concentrated flow releases in shrub-interspace zones with downed trees relative to those in the controls (see Tables 9 and 10). Our results from the cut-downed-tree treatment are similar to those of Hastings et al. (2003), who found that cutting pinyon and juniper and evenly distributing tree debris within the intercanopy reduced erosion from high-intensity rain events on a degraded and rapidly eroding woodland. Erosion from natural rainfall events over two rainy seasons was one to three orders of magnitude more for untreated than treated microwatersheds (300 − 1 100 m^2 area). Hastings et al. (2003) attributed the reduced erosion following tree cutting to enhanced infiltration and soil water retention afforded by slash, herbaceous cover recruitment, and reduced interconnectivity of runoff and sediment source areas. Stoddard et al. (2008) found that scattering pinyon and juniper slash in interspaces reduced soil movement at two woodlands in Arizona, United States, 1 − 2 yr post treatment. In our mastication treatment, mulch from shredded trees was not distributed into the intercanopy. As result, microsite-specific litter and bare ground were generally similar for mastication and control areas across the small plot and patch scales (see Tables 4 and 6), and the mastication treatment had limited impact on runoff and erosion responses from rainfall simulations and concentrated flow experiments 9 yr post treatment (see Tables 8 and 10). Pierson et al. (2014) and Cline et al. (2010)

found that directly placing tree mulch on interspace plots in the mastication treatment at Onaqui reduced runoff and erosion by fourfold to fivefold for rainfall simulation experiments as applied in this study. Pierson et al. (2015) evaluated vegetation, hydrology, and erosion impacts of the mastication treatment on 13-m^2 rainfall simulation and concentrated flow plots at Onaqui the first year after treatment. Experimental methods in that study were consistent with those in this study. The mastication treatment initially increased herbaceous vegetation cover from 11% to 25% in shrub-interspace zones, reduced the same measure for tree zones from 16% to 6%, and had no impact on bare ground and litter cover across shrub-interspace and tree zone plots (Pierson et al., 2015). The treatment reduced runoff and erosion from shrub-interspaces by twofold for the wet-run rainfall simulations relative to control plots but had negligible impact on concentrated flow runoff and erosion and runoff from rainfall simulations in tree zones (Pierson et al., 2015). Results from the current study in context with those from our companion studies (Cline et al., 2010; Pierson et al., 2014, 2015) and the literature (Hastings et al., 2003; Stoddard et al., 2008) suggest distributing tree debris and mulch into the intercanopy can be an effective way to reduce microsite-scale runoff and erosion where tree debris is placed in good contact with the bare ground surface. However, the potential for tree debris, litter, and mulch to negatively impact intercanopy perennial herbaceous recruitment or favor cheatgrass establishment should also be considered with regards to slash and debris management (Bates et al., 1998, 2005, 2007; Bates and Svejcar, 2009).

Persistence of high interspace and intercanopy runoff and erosion levels at both sites (see Tables 7 — 10) indicate that the measured increases in vegetation over 9 growing seasons after tree removal (see Table 2) are not substantial enough to reestablish the soil- and water-conserving attributes of sagebrush steppe (see Pierson et al., 1994, 2009; Pierson and Williams, 2016; Williams et al., 2016b, 2016c), but both sites are on a trajectory toward improved hydrologic function. Tree cutting had minimal impact on understory vegetation, ground cover, runoff, and erosion within the intercanopy at both sites the first yr post treatment (Pierson et al., 2015). The tree mastication treatment at Onaqui increased intercanopy shrub and grass cover but had no effect on bare ground the first yr. Intercanopy runoff and erosion from rainfall simulations were reduced by the mastication treatment the first yr, but runoff and erosion from concentrated flow experiments were similar for the mastication and control shrub-interspace plots (Pierson et al., 2015). Nine yr after tree removal, intercanopy runoff and erosion from rainfall simulation and concentrated flow experiments were similar across all treatments at a site except for the plots in the cut treatment areas with downed trees (cut-downed-tree treatment; see Tables 7 — 10). Although these results indicate there has been no significant reduction in runoff and erosion for the applied experiments, hillslope- and patch-scale vegetation attributes that mitigate runoff and erosion are slowly improving across both sites (see Tables 2, 5, and 6). The cut treatment increased hillslope-scale shrub and grass canopy cover at both sites, tree mastication substantially increased hillslope-scale grass cover at Onaqui, and both treatments increased the density of sagebrush plants. Shrub coppices generated substantially less runoff and sediment yield than interspace plots at both sites before and after treatment (Pierson et al., 2010, 2014; see Tables 7 and 8). Cline et al. (2010) found that runoff and erosion from rainfall simulations at Onaqui were similar for shrub coppices and grass-dominated interspaces (34% perennial grass cover, 38% bare soil) but were greater for bare interspaces (< 1% perennial grass cover, 47% bare soil) than the grass-dominated interspaces. Similar findings have been reported for other sagebrush and woodland sites throughout the Great Basin and elsewhere in the western United States (Blackburn, 1975; Pierson et al., 1994; Pierson and Williams, 2016). In this study, runoff and erosion from wet-run rainfall simulations were 21 mm and 46 $g \cdot m^{-2}$ for grass-dominated interspaces (> 30% grass cover) and 46 mm and 321 $g \cdot m^{-2}$ for bare interspaces (< 30% grass cover) across all treatments and sites. We anticipate intercanopy runoff and erosion rates will decrease over time at both sites as shrub coppice and grass interspace areas increase and begin to reduce intercanopy bare ground (Pierson et al., 2007, 2013; Williams et al., 2014a). Pierson et al. (2013) and Williams et al. (2014a) found that increases in the amount and spatial distribution of intercanopy herbaceous vegetation 2 yr following wildfire at phase II — III western juniper woodland improved interspace infiltration of artificially applied rainfall and reduced intercanopy concentrated flow erosion. Pierson et al. (2007) reported increased intercanopy perennial herbaceous cover (from 2% to 14%) and litter cover (9% to 27%) 10 yr following cutting of western juniper on a sagebrush site in later stages of woodland encroachment. Runoff and erosion from rainfall simulations in cut treatment areas were negligible in that study, and bare intercanopy (84% bare ground) in untreated areas generated runoff and erosion rates 10- to more than 100-fold greater than in treated, than in treated intercanopy areas. Continued increases in intercanopy sagebrush and perennial grass cover are, of course, not a foregone conclusion for the sites in this study, but the likelihood of continued increases is supported by other long-term studies (Bates and Davies, 2016; Bates et al., 2017). Bates et al. (2017) found increased density of shrubs 6 yr after juniper cutting did not translate to significant increases in shrub canopy cover relative to untreated areas until the 18th yr after cutting, due in part to low initial shrub cover. In that study sagebrush density continued to increase until the 18th yr after tree removal and perennial grass cover and density peaked and stabilized the 6th — 14th yr post treatment. In a multisite study, Bates and Davies (2016) found cutting of western juniper increased intercanopy perennial bunchgrass grass cover in cut areas relative to control areas of a sagebrush site after three growing seasons and that perennial grass cover continued to increase in one of three sites six growing seasons post treatment. Cutting reduced intercanopy bare ground at two of the three sites by the sixth yr of the study. Results from our study indicate improved hydrologic function through vegetation and ground cover recruitment after mechanical tree removal on late phase II — III (TDI > 0.5) woodlands can require more than 9 yr and, in context with the literature, suggest that the period required depends on the time needed to establish well-distributed grass and shrub cover and reduce intercanopy bare ground to at least < 50% (Pierson et al., 2007, 2010, 2013; Williams et al., 2014a, 2016a). Seeding can be an effective tool at reducing the time required to increase sagebrush steppe vegetation and improve hydrologic function following tree removal (Davies et al., 2014; Roundy et al., 2014a; Bybee et al., 2016; Roundy et al., 2017) but was not applied in this study. Residual tree cover (< 4%, ~20 — 239 trees per ha) at both sites in this study needs to be addressed through follow-up treatments to limit competition for resources as the understory continues to reestablish. Follow-up tree removal is commonly necessary to prevent pinyon and juniper recolonization within the first 30 — 50 yr post treatment (Tausch and Tueller, 1997; Bates et al., 2005; Miller et al., 2005; O'Connor et al., 2013; Bristow et al., 2014; Roundy et al., 2014a; Bates et al., 2017).

Summary and Implications

Our results demonstrate that pinyon and juniper removal by tree cutting and mastication can effectively increase sagebrush steppe vegetation on mid- to late-succession woodlands in the central Great Basin and thereby establish a trajectory toward improved hydrologic function. We measured depauperate coverage of sagebrush and perennial herbaceous vegetation, extensive bare ground, and high rates of intercanopy runoff and erosion from rainsplash, sheetflow, and concentrated flow processes in untreated areas at two woodland-encroached sagebrush sites. Tree cutting and mastication effectively recruited sagebrush and native tall perennial grass cover at both sites over nine growing seasons but failed to substantially reduce high amounts of intercanopy bare ground. Increases in hillslope-scale vegetation were greater for the more degraded (initially more sparsely vegetated) site in this study.

Cheatgrass was low before tree removal, and cover of the species was not substantially increased by tree-removal treatments. Tree removal had limited impact on vegetation, ground cover, and hydrologic and erosion processes at the small-plot scale due to persistent microsite attributes at that spatial scale. Interspaces between plants remained mostly bare 9 yr after tree removal and generated runoff and sediment at similar levels to those pretreatment for high-intensity rainfall simulations. Areas underneath shrubs and trees were well vegetated and litter covered, respectively, before and after treatment and generally produced low and low-to-high runoff and sediment during high-intensity rainfall simulations at the less degraded and more degraded sites, respectively across all treatments. Intercanopy vegetation and ground cover responses to tree cutting were mixed across the two sites. Tree cutting had negligible impact on intercanopy ground cover on concentrated flow plots, and total runoff and sediment from concentrated flow simulations were similar for control and cut treatments within the intercanopy at both sites. The exception was that intercanopy plots with downed trees in the cut treatment areas had greater litter cover and lower bare ground than those without downed trees and generated low levels of total runoff and total sediment from concentrated flow experiments. The ground surface on concentrated flow plots underneath trees was well protected with litter before and after tree removal, and tree cutting did not reduce concentrated flow total runoff and total sediment from tree microsites. Mastication increased intercanopy shrub and grass cover but had negligible impact on ground cover and total runoff and sediment for intercanopy concentrated flow simulations. The mastication treatment reduced total runoff and sediment delivered from concentrated flow simulations in areas underneath trees where tree mulch accumulated. The trends in hillslope-scale vegetation responses to tree removal in this study demonstrate the effectiveness of mechanical treatments to recruit sagebrush steppe vegetation without increasing cheatgrass for mid- to late-succession woodland-encroached sites along the warm-dry to cool-moist soil temperature — moisture threshold in the Great Basin. The vegetation responses reflect initially low levels of cheatgrass and low but sufficient levels of sagebrush and native perennial bunchgrasses for reestablishment of sagebrush steppe over time. We attribute the greater increases in hillslope vegetation at the more degraded site to initially low cover levels at the site pretreatment and potentially to lesser increases in soil water at the less degraded site due to higher residual shrub cover. Our results are supported by numerous mid- to long-term studies of vegetation responses to tree removal in sagebrush steppe. Persistence of high runoff and erosion rates in interspaces and the intercanopy 9 yr post-treatment indicate that the measured increases in hillslope-scale shrub and grass cover were not sufficient to re-establish the soil and water conserving attributes of intact sagebrush steppe communities. Our results indicate improved hydrologic function through sagebrush steppe vegetation recruitment after mechanical tree removal on mid- to late-succession woodlands can require more than 9 yr. We anticipate intercanopy runoff, and erosion rates will decrease over time at both sites as shrub and grass cover continue to increase, but follow-up tree removal will be necessary to prevent pinyon and juniper recolonization and competition between trees and understory vegetation for limited soil and water resources. The low intercanopy runoff and erosion measured underneath isolated downed trees in this study clearly demonstrates that tree debris following mechanical treatments can effectively limit microsite-scale runoff and erosion over time. Mechanical tree-removal treatments targeting reduced runoff and erosion may be aided by distributing tree debris into bare intercanopy patches in good contact with the soil surface.

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