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Effectiveness of Prescribed Fire to Re-Establish Sagebrush Steppe Vegetation and Ecohydrologic Function on Woodland-Encroached Sagebrush Rangelands, Great Basin, USA: Part I: Vegetation, Hydrology, and Erosion Responses

Christopher Jason Williams
USDA, Agricultural Research Service

Frederick B. Pierson
USDA, Agricultural Research Service

Sayjro K. Nouwakpo
University of Nevada, Reno

Osama Z. Al-Hamdan
Texas A&M University

Patrick R. Kormos
USDA, Agricultural Research Service

Mark A. Weltz
USDA, Agricultural Research Service

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Effectiveness of prescribed fire to re-establish sagebrush steppe vegetation and ecohydrologic function on woodland-encroached sagebrush rangelands, Great Basin, USA: Part I: Vegetation, hydrology, and erosion responses

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

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

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ABSTRACT

Pinyon (*Pinus* spp.) and juniper (*Juniperus* spp.) woodland encroachment has imperiled a broad ecological domain of the sagebrush steppe (*Artemisia* spp.) ecosystem in the Great Basin Region, USA. As these conifers increase in dominance on sagebrush rangelands, understory vegetation declines and ecohydrologic function can shift from biotic (vegetation) controlled retention of soil resources to abiotic (runoff) driven loss of soil resources and long-term site degradation. Scientists, public land management agencies, and private land owners are challenged with selecting and predicting outcomes to treatment alternatives to improve ecological structure and function on these rangelands. This study is the first of a two-part study to evaluate effectiveness of prescribed fire to re-establish sagebrush steppe vegetation and improve ecohydrologic function on mid- to late-succession pinyon-and juniper-encroached sagebrush sites in the Great Basin. We used a suite of vegetation and soil measures, small-plot (0.5 m^2) rainfall simulations, and overland flow experiments (9 m^2) to quantify the effects of tree removal by prescribed fire on vegetation, soils, and rainsplash, sheetflow, and concentrated flow hydrologic and erosion processes at two woodlands 9-yr after burning. For untreated conditions, extensive bare interspace (87% bare ground) throughout the degraded intercanopy (69–88% bare ground) between trees at both sites promoted high runoff and sediment yield from combined rainsplash and sheetflow (~45 mm, 59–381 g m^{-2}) and concentrated flow (371–501 L, 2343–3015 g) processes during high intensity rainfall simulation (102 mm h^{-1}, 45 min) and overland flow experiments (15, 30, and 45 L min^{-1}, 8 min each). Burning increased canopy cover of native perennial herbaceous vegetation by > 5-fold, on average, across both sites over nine growing seasons. Burning reduced low pre-fire sagebrush canopy cover (< 1% to 14% average) at both sites and sagebrush recovery is expected to take > 30 yr. Enhanced herbaceous cover in interspaces post-fire reduced runoff and sediment yield from high intensity rainfall simulations by > 2-fold at both sites. Fire-induced increases in herbaceous canopy cover (from 34% to 62%) and litter ground cover (from 15% to 36%) reduced total runoff (from 501 L to 180 L) and sediment yield (from 2343 g to 115 g) from concentrated flow experiments in the intercanopy at one site. Sparser herbaceous vegetation (49% cover) and litter cover (8%) in the intercanopy at the other, more degraded site post-fire resulted in no significant reduction of total runoff (371 L to 266 L) and sediment yield (3015 g to 1982 g) for concentrated flow experiments. Areas underneath unburned shrub and tree canopies were well covered by vegetation and ground cover and generated limited runoff and sediment. Fire impacts on vegetation, ground cover, and runoff and sediment delivery from tree and shrub plots were highly variable. Burning litter covered areas underneath trees reduced perennial herbaceous vegetation and increased invasibility to the fire-prone annual cheatgrass (*Bromus tectorum* L.). Cheatgrass cover increased from < 1% pre-fire to 16–30%, on average, post-fire across the sites and was primarily restricted to areas around burned trees.

* Corresponding author at: Southwest Watershed Research Center, Agricultural Research Service, United States Department of Agriculture, 2000 East Allen Road, Tucson, AZ 85719, USA.

E-mail address: jason.williams@ars.usda.gov (C.J.,. Williams).

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High herbaceous cover (73%) under burned trees at the less degraded site resulted in similar low total runoff and sediment from concentrated flow experiments as pre-fire (136–228 L, 204–423 g). In contrast, fire-reduction of litter (from 79% to 49%) resulted in increased total runoff (from 103 L to 333 L) and sediment yield (from 619 g to 2170 g) from concentrated flow experiments in burned tree areas at the more degraded site. The experimental results demonstrate pinyon and juniper removal by prescribed fire can effectively re-establish a successional trajectory towards sagebrush steppe vegetation structure and thereby improve ecohydrologic function. Responses to burning at the more degraded site suggest results should be interpreted with caution however. Although burning substantially increased perennial grass cover and reduced fine-scale runoff and erosion at the more degraded site, poor sagebrush recovery, delayed litter recruitment, and persistent high concentrated flow erosion at that site suggest not all sites are good candidates for prescribed fire treatments. Furthermore, high levels of cheatgrass in burned tree areas (~30% of area) at both sites increases wildfire risk, but cheatgrass is expected to decline over time in absence of fire. Our results in context with the literature suggest fire-surrogate tree-removal treatments (e.g., tree cutting or shredding) may be more appropriate on degraded sites with limited pre-treatment sagebrush and perennial herbaceous vegetation and that seeding may be necessary to improve post-fire establishment of sagebrush steppe vegetation structure and associated ecohydrologic function under these conditions. Lastly, vegetation, runoff, and erosion responses in this study are not directly applicable outside of the Great Basin, but similar responses in woodland studies from the southwestern US suggest potential application of results to woodlands in that region. The concept of re-establishing vegetation structure to improve ecohydrologic function is broadly applicable to sparsely vegetated lands around the World.

1. Introduction

Woody-plant encroachment on water-limited lands is a worldwide concern (Van Auken, 2000; Barger et al., 2011; Eldridge et al., 2011; Sala and Maestre, 2014). Encroachment of herbaceous- or shrub-dominated communities by woody species commonly initiates through multiple forces including climate variability, land use practices, altered fire frequency, or CO_2 fertilization (Archer et al., 1995; Miller et al., 2005; Van Auken, 2009; Eldridge et al., 2011; Archer et al., 2017). These forces can alter plant community physiognomy and associated biotic and abiotic processes and thereby propagate landscape degradation (Davenport et al., 1998; Peters et al., 2004; Allen, 2007; Turnbull et al., 2008, 2012; Williams et al., 2016b, 2016c). One of the most documented accounts is the degradation of black grama grasslands (*Bouteloua eriopoda* [Torr.] Torr.) in the southwestern United States (US) following encroachment by creosote shrubs (*Larrea tridentata* [DC.] Coville) and mesquite trees (*Prosopis* spp.) (Buffington and Herbel, 1965; Grover and Musick, 1990; Bahre and Shelton, 1993). Ecological transition of these grasslands, once initiated, is sustained by high infiltration rates, enhanced soil water storage, and capture and retention of soil nutrients underneath shrub and tree canopies (Abrahams et al., 1995; Parsons et al., 1996a, 1996b; Wainwright et al., 2000; Bhark and Small, 2003; Turnbull et al., 2010). Over time, bare ground becomes extensive between vegetation islands ("islands of fertility") and high rates of runoff and soil loss from well-connected bare areas perpetuate woody plant dominance and long-term site degradation (Schlesinger et al., 1990, 1996; Field et al., 2012; Turnbull et al., 2012; Puttock et al., 2013; Puttock et al., 2014). These pattern and process relationships are common on water-limited landscapes around the Globe (Ludwig and Tongway, 1995; Dunkerley and Brown, 1995; Cerdà, 1997; Ludwig et al., 1997; Wilcox et al., 2003a; Ludwig et al., 2005; Puigdefábregas, 2005; Bautista et al., 2007; Pierson and Williams, 2016) and have been reported following woody plant encroachment in Africa (Manjoro et al., 2012), increased shrub cover in South America (Chartier and Rostagno, 2006), coarsening of woodland community structure in Australia (Ludwig et al., 2007), and shrubland and woodland degradation in North America (Wilcox et al., 1996a, 1996b; Davenport et al., 1998; Miller et al., 2005; Pierson et al., 2007, 2010, 2013; Williams et al., 2014a, 2016b, 2016c).

Range expansion of pinyon (*Pinus* spp.) and juniper (*Juniperus* spp.) conifers into sagebrush steppe (*Artemisia* spp.) communities has imperiled a vast domain in the western US (Noss et al., 1995; Suring et al., 2005). The sagebrush ecosystem once extended over 620,000 km^2 of western North America by some accounts and currently occupies < 60% of its historical range (see Davies et al., 2011; Miller et al., 2011). Pinyon and juniper woodlands now occupy an estimated 190,000 km^2 in the Intermountain Western US and about 90% of that domain was historically occupied by sagebrush vegetation (see Miller et al., 2008; Davies et al., 2011; Miller et al., 2011). Woodland encroachment into sagebrush steppe has been attributed to a combination of intensive grazing, decreased fire frequency, climate variability, and increased atmospheric CO_2 (Miller and Wigand, 1994; Knapp and Soulé, 1996; Miller and Tausch, 2001; Miller et al., 2005; Romme et al., 2009). Woodland development has been characterized into three successive phases (Miller et al., 2000; Miller et al., 2005; Johnson and Miller, 2006; Miller et al., 2008). In Phase I (early-succession), tree cover increases for 0–3-m height class, sagebrush shrubs and perennial bunchgrasses and forbs (sagebrush steppe vegetation) are the dominant vegetation, and runoff and erosion are limited by high infiltration rates and litter protection of surface soil. In Phase II (mid-succession), trees approach 10–50% cover, shrub and herbaceous cover decline due to competition for limited water and soil resources, and runoff and erosion rates increase with increasing bare ground. Phase III (late-succession) is reached when tree cover becomes the dominant cover type (> 75% shrub mortality) and exerts the primary control on ecohydrologic processes. Extensive bare ground in late Phase II and Phase III connects runoff and/or erosion processes across spatial scales (Pierson et al., 2010, 2013; Williams et al., 2014a; Roundy et al., 2017), marking deviation from water and soil retention characteristics common for sagebrush steppe vegetation structure (Pierson et al., 1994, 2008a, 2009). Dense woody fuel loading in all phases increases susceptibility to high severity wildfire under dry and windy conditions (Miller and Tausch, 2001). Susceptibility to fire further increases with invasion by the non-native annual cheatgrass (*Bromus tectorum* L.). Cheatgrass invades bare patches on sagebrush and woodland-encroached sites and thereby increases the horizontal connectivity of fuels and the likelihood of fire ignition (Brooks et al., 2004; Link et al., 2006; Chambers et al., 2007; Littell et al., 2009; Condon et al., 2011; Balch et al., 2013; Reisner et al., 2013; Rau et al., 2014). Fire commonly increases runoff and erosion rates in the first few years post-fire (Pierson et al., 2008a, 2009, 2013, 2014; Williams et al., 2014a; Pierson et al., 2015; Pierson and Williams, 2016; Williams et al., 2016b, 2016c) and frequent re-burning associated with cheatgrass invasion increases erosion risk (Pierson et al., 2011; Wilcox et al., 2012; Williams et al., 2014b, 2016c, 2016d). The conversion of sagebrush rangeland to a woodland vegetation type also poses negative ramifications to key wildlife species and can limit delivery of ecosystem goods and services (Knick et al., 2003; Aldrich et al., 2005; Suring et al., 2005; Davies et al., 2011; Knick and Connelly, 2011; Miller et al., 2011; Coates et al., 2017; Kormos et al., 2017).

Land managers across the western US are challenged with selecting and applying effective sagebrush steppe restoration treatments on woodland-encroached sites (Davies et al., 2011; Chambers et al., 2014a, 2014b; McIver et al., 2014; Chambers et al., 2017; Miller et al., 2017). Land managers are confronted with: 1) predicting site-specific responses to treatment alternatives, 2) selecting sites most likely to benefit from treatment, 3) determining the appropriate time and conditions in which to apply treatment, and 4) selecting and implementing the correct treatment to meet desired outcomes. The most common treatments include either prescribed fire, tree cutting, whole-tree shredding (mastication), tree chaining, or some combination of tree felling and prescribed fire with or without seeding (Bates and Svejcar, 2009; Bates et al., 2011, 2014; Chambers et al., 2014b; Davies et al., 2014; Miller et al., 2014; Roundy et al., 2014a, 2014b; Bybee et al., 2016; Bates and Davies, 2016; Bates et al., 2017). The likelihood of re-establishing sagebrush and perennial bunchgrasses and forbs with either approach is generally considered greater when treatments are applied early in the encroachment gradient (Phase I to Phase II) (Miller et al., 2005; Davies et al., 2011; Bates et al., 2014; Roundy et al., 2014a). In Phase III, coverage and the seedbank of sagebrush and native perennial bunchgrasses and forbs are greatly reduced and increased cover of cheatgrass is likely following tree removal (Koniak and Everett, 1982; Koniak, 1985; Miller et al., 2000; Bates et al., 2011, 2014). However, responses to tree removal vary widely across the diverse ecophysiological domain in which pinyon and juniper encroachment occurs, with tree density at the time of treatment, and for different treatment methods (Romme et al., 2009; Davies et al., 2011; Miller et al., 2013; Chambers et al., 2014b; Miller et al., 2014; Roundy et al., 2014a; Bates and Davies, 2016). Prescribed fire kills sagebrush, but may have a positive or negative short-term effect on perennial bunchgrasses depending on fire severity (Bates et al., 2006; Ellsworth and Kauffman, 2010; Bates et al., 2011, 2014; Miller et al., 2014; Bates and Davies, 2016). Sagebrush does not re-sprout after fire and re-establishment of a sagebrush overstory can take 15 yr to > 50 yr (Harniss and Murray, 1973; Ziegenhagen and Miller, 2009; Miller et al., 2013; Moffet et al., 2015). Recruitment of perennial bunchgrasses is paramount to limiting invasion by cheatgrass post-treatment regardless of treatment method (Chambers et al., 2007; Condon et al., 2011; Bates et al., 2014; Chambers et al., 2014b; Miller et al., 2014; Roundy et al., 2014a; Bybee et al., 2016). Mechanical tree-removal treatments (cutting and shredding) have minimal to no negative initial impact on sagebrush and perennial grasses (Bates et al., 2000, 2005; Chambers et al., 2014b; Roundy et al., 2014a; Bybee et al., 2016). Mechanical treatments, however, leave numerous juvenile pinyon and juniper that may re-establish tree dominance with time (Tausch and Tueller, 1997; Bates et al., 2005; Miller et al., 2005, 2013; Bates et al., 2017). Guidance on site-specific responses to management practices for US rangelands is available through ecophysiological-based Ecological Site Descriptions (ESDs) and through descriptions of plant community dynamics in State-and-Transition Models (STMs) (Stringham et al., 2003; Briske et al., 2005, 2008; Bestelmeyer et al., 2009; Petersen et al., 2009; Caudle et al., 2013). However, the knowledge needed to populate ESDs and STMs is limited relative to the vast domain of woodland encroachment and varying responses to treatments (Chambers et al., 2014a, 2014b; Williams et al., 2016c; Chambers et al., 2017). Site-specific information is needed over short- and long-term time scales to address these limitations and to improve broader knowledge of sagebrush steppe vegetation dynamics and ecohydrologic responses to tree removal (Petersen et al., 2009; Miller et al., 2014; Roundy et al., 2014a, 2014b; Bates and Davies, 2016; Williams et al., 2016c; Bates et al., 2017; Chambers et al., 2017).

The ability to understand and predict ecohydrologic responses of woodlands to tree-removal treatments necessitates knowledge of treatment effects at multiple spatial scales (Ludwig and Tongway, 1995; Ludwig et al., 1997; Wilcox et al., 2003a; Ludwig et al., 2005, 2007; Pierson et al., 2011, 2013; Williams et al., 2014a, 2014b, 2016b). Pinyon and juniper woodlands have been described as “resource-conserving” when the vegetation structure limits runoff and erosion and “non-conserving” or “leaky” when the vegetation structure promotes increased runoff or soil loss across spatial scales (Wilcox et al., 2003a). Runoff and erosion for “resource-conserving” woodland-encroached rangelands are buffered by high infiltration rates in well-vegetated interspaces between trees and shrubs and in litter or woody debris covered areas adjacent to shrub and tree canopies (Pierson and Williams, 2016). Runoff and sediment generated from isolated bare interspaces travel only a short distance before infiltrating and being deposited in vegetated or litter covered patches (Ludwig et al., 1997; Reid et al., 1999; Wilcox et al., 2003a; Ludwig et al., 2005). The transition from “resource-conserving” to “non-conserving” conditions in later stages of woodland encroachment (late Phase II – Phase III) into sagebrush steppe occurs when herbaceous vegetation declines and bare ground becomes well-connected throughout the intercanopy (area between tree canopies) (Pierson et al., 2010, 2013; Williams et al., 2014a, 2016b, 2016c). High levels of runoff and soil loss associated with fine-scale rainsplash and sheetflow (splash-sheet) processes in interspaces accumulate throughout the bare intercanopy and form concentrated flow paths with high flow velocity and sediment detachment and transport capacity (Wainwright et al., 2000; Petersen and Stringham, 2008; Pierson et al., 2010; Al-Hamdan et al., 2013; Pierson et al., 2013; Williams et al., 2014a, 2014b; Nouwakpo et al., 2016). This structural (pattern) and functional (process) connectivity increases risk of downslope runoff and sediment delivery and long-term soil loss (Wilcox et al., 1996b; Ludwig et al., 1997; Davenport et al., 1998; Turnbull et al., 2008; Pierson et al., 2011; Wainwright et al., 2011; Turnbull et al., 2012; Bracken et al., 2013; Williams et al., 2016b). The effectiveness of tree removal to disrupt structural and functional connectivity is predicated on herbaceous cover recruitment in bare areas; improved infiltration, reduced runoff, and decreased sediment delivery in interspaces; and dissipation of runoff and flow velocity where overland flow does occur (Pierson et al., 2007, 2013, 2014; Williams et al., 2014a; Pierson et al., 2015; Roundy et al., 2017). Tree removal by fire may initially increase runoff and erosion from areas underneath burned shrubs and trees, temporarily increasing hillslope-scale hydrologic vulnerability (Pierson et al., 2013, 2014, 2015; Williams et al., 2016b, 2016c, 2016d). Runoff and erosion rates may remain high in interspaces the first few years following tree-removal treatments without ample ground cover by tree debris or rapid colonization by herbaceous plants (Pierson et al., 2013, 2015). Tree-removal on woodland-encroached sites commonly increases plant available soil water that stimulates herbaceous vegetation production over time (Bates et al., 2000; Young et al., 2013a, 2013b; Miller et al., 2014; Roundy et al., 2014b). However, knowledge remains limited for the vast woodland domain regarding the amount of cover necessary to reverse cross-scale process connectivity (Williams et al., 2014a) and the time period required to re-establish “resource-conserving” conditions.

This is the first of a two-part study to evaluate long-term effectiveness of prescribed fire to re-establish sagebrush steppe vegetation and improve ecohydrologic function on mid- to late-succession woodland-encroached sagebrush sites in the Great Basin, USA. Specifically, this study (Part I) used a suite of vegetation and soil measures, rainfall simulations, and overland flow experiments to quantify the effects of tree removal by prescribed fire on vegetation, soils, infiltration, and runoff and erosion by splash-sheet processes and by concentrated overland flow processes at two woodlands 9-yr after burning. The primary objectives for Part I were to quantify (1) vegetation and ground surface conditions at the hillslope (990 m^2 plots), small-plot (0.5 m^2), and patch (~10 m^2) spatial scales, (2) infiltration, runoff generation, and sediment delivery by splash-sheet processes for rainfall simulations at the small-plot scale, and (3) runoff and sediment delivery by concentrated overland flow processes at the patch scale for untreated and treated conditions 9 yr after burning. The spatial scale of the small-plots was designed to quantify treatment effects on interspace microsites between shrub and tree canopies and microsites underneath shrub and tree canopies. The patch scale experiments were designed to quantify treatment effects for areas representative of the intercanopy and areas immediately underneath/adjacent to tree canopies. Part II of the study (Nouwakpo et al., 2020) expands the inference space through use of large plot (12 m^2) rainfall simulation experiments at the same study sites to quantify prescribed-fire treatment effects on vegetation, soils, and combined splash-sheet and concentrated overland flow processes at the patch scale. The collective research is part of the Sagebrush Steppe Treatment Evaluation Project (SageSTEP, www.sagestep.org) aimed at investigating the ecological impacts of invasive species and woodland encroachment into sagebrush steppe ecosystems and the effects of various sagebrush steppe restoration approaches (McIver et al., 2010; McIver and Brunson, 2014; McIver et al., 2014). The study sites were the subject of multiple companion SageSTEP hydrology and erosion studies. Pierson et al. (2010) quantified vegetation and soil characteristics and runoff and erosion processes across small-plot and patch scales at the sites before prescribed fire treatments in 2006. Cline et al. (2010) evaluated the impacts of whole-tree shredding on small-plot scale infiltration, runoff, and erosion in unburned areas at one of the sites. Pierson et al. (2014, 2015) quantified short-term treatment effects on vegetation, soils, and hydrology and erosion processes at the small-plot and patch scales at both sites 1 yr and 2 yr after tree removal by burning, cutting, and shredding. Williams et al. (2016b) evaluated vegetation, hydrology, and erosion responses to burning at both sites across the small-plot to hillslope scales through a suite of field experiments and hydrologic modeling with the Rangeland Hydrology and Erosion Model (Nearing et al., 2011; Al-Hamdan et al., 2015). Parts I (this paper) and II (Nouwakpo et al., 2020) of the current study expand on the findings of the above noted studies by quantifying longer-term ecohydrologic responses of the study sites to prescribed-fire treatments across multiple spatial scales.

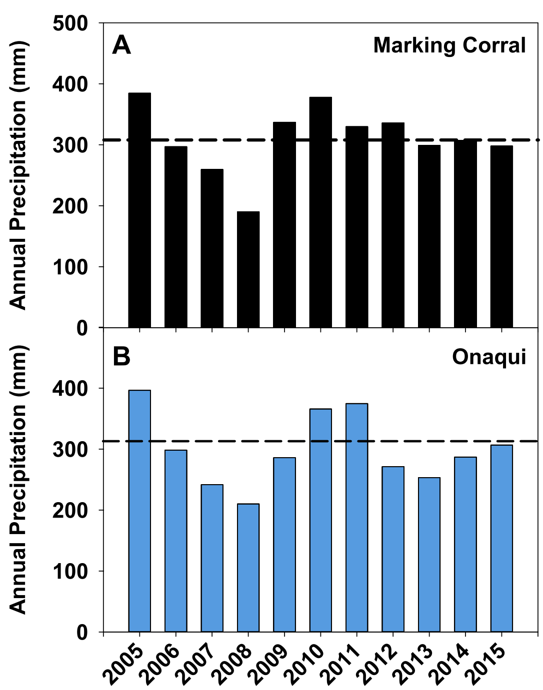

Fig. 1. Photographs of the Marking Corral study site in 2015 showing isolated tree islands and the degraded intercanopy area between trees in the control (A), the burned intercanopy (B), the unburned intercanopy with interspace rainfall simulation plots (0.5 m^2) (C), and the bunchgrass and bare ground structure of the burned intercanopy (D).

2. Materials and methods

2.1. 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) selected from the SageSTEP study network (McIver and Brunson, 2014; McIver et al., 2014). The Marking Corral site (Fig. 1; 39°27′17″N latitude, 115°06′51″W longitude) is located in the Egan Range, about 27 km northwest of Ely, Nevada, USA. The Onaqui site (Fig. 2; 40°12′42″N latitude, 112°28′24″W longitude) is located in the Onaqui Mountains, approximately 76 km southwest of Salt Lake City, Utah, USA. Both sites are managed by the US Department of the Interior, Bureau of Land Management (BLM) for grazing use, but have been excluded from grazing since autumn 2005. Detailed geographic, climate, soils, and vegetation characteristics for the sites are provided in Table 1. Precipitation at the sites is near or exceeds 300 mm most years and the soil temperature-moisture regimes at both locations are at the fringe of warm (mesic) - dry (aridic) and cool (frigid) - wet (xeric) classifications (McIver and Brunson, 2014). Estimated annual precipitation over the full study period (2006–2015) was near or exceeded the long-term average, with only 2 to 3 yr of > 15% below normal precipitation (Fig. 3). Across both sites, the year preceding prescribed burning received 125–130% of normal precipitation, the year of treatment received near normal precipitation, and the two years following burning were the driest (60–84% of normal) over the study period (Fig. 3).

The vegetation community structure at the sites prior to burning was typical of degraded sagebrush steppe in the later stages of woodland encroachment (late Phase II to early Phase III; Miller et al., 2000, 2005, 2008; Figs. 1 and 2). Pierson et al. (2010) and Williams et al. (2016b) reported vegetation, ground cover, and soil data measured at the sites in summer 2006 prior to the prescribed fires. Vegetation at both sites consisted of isolated tree islands surrounded by a degraded intercanopy (Figs. 1A and 2A). Pre-treatment intercanopy area and tree canopy cover in the burn treatment area at both sites were approximately 73% and 27%, respectively (Table 1; Williams et al., 2016b). The sites exhibited high shrub mortality attributable to woodland encroachment (Miller et al., 2000, 2005, 2008; Table 1). Shrub cover was the dominant intercanopy understory lifeform at Marking Corral (Fig. 1A) while the intercanopy understory at Onaqui was grass- and forb-dominated (Fig. 2A). The intercanopy ground surface was mostly bare before the fire at both sites (Figs. 1C and 2C), with combined bare soil and rock cover near 70% and > 80% in the intercanopy at Marking Corral and Onaqui, respectively (Williams et al., 2016b). Understory canopy cover underneath and adjacent to trees was near 20% across the sites before the fires and was mostly grasses (Williams et al., 2016b). The ground surface immediately underneath trees at the sites was approximately 80% to > 90% covered by a 20 mm to 40 mm thick litter layer, spanning approximately 2.2 m to 2.5 m in distance from tree bases (Williams et al., 2016b). The surface soil (0–5 cm depth) underneath tree canopies at both sites was water repellent before the fires, whereas surface soil in interspaces and underneath shrub canopies was wettable (Pierson et al., 2010, 2014). At Marking Corral, soil bulk density for 0–5 cm soil depth in the burn area prior to treatment averaged 1.26 g cm^{-3} in interspaces between shrubs and trees and 1.02 and 1.03 g cm^{-3} under shrub and tree canopies; the same measure at Onaqui averaged 1.08, 1.05, and 0.90 g cm^{-3} for interspaces and areas under shrub and tree canopies, respectively. At each site, tree cover, understory canopy and ground cover, hillslope angle, and surface soil texture and soil bulk density were statistically similar ($P > 0.05$) across

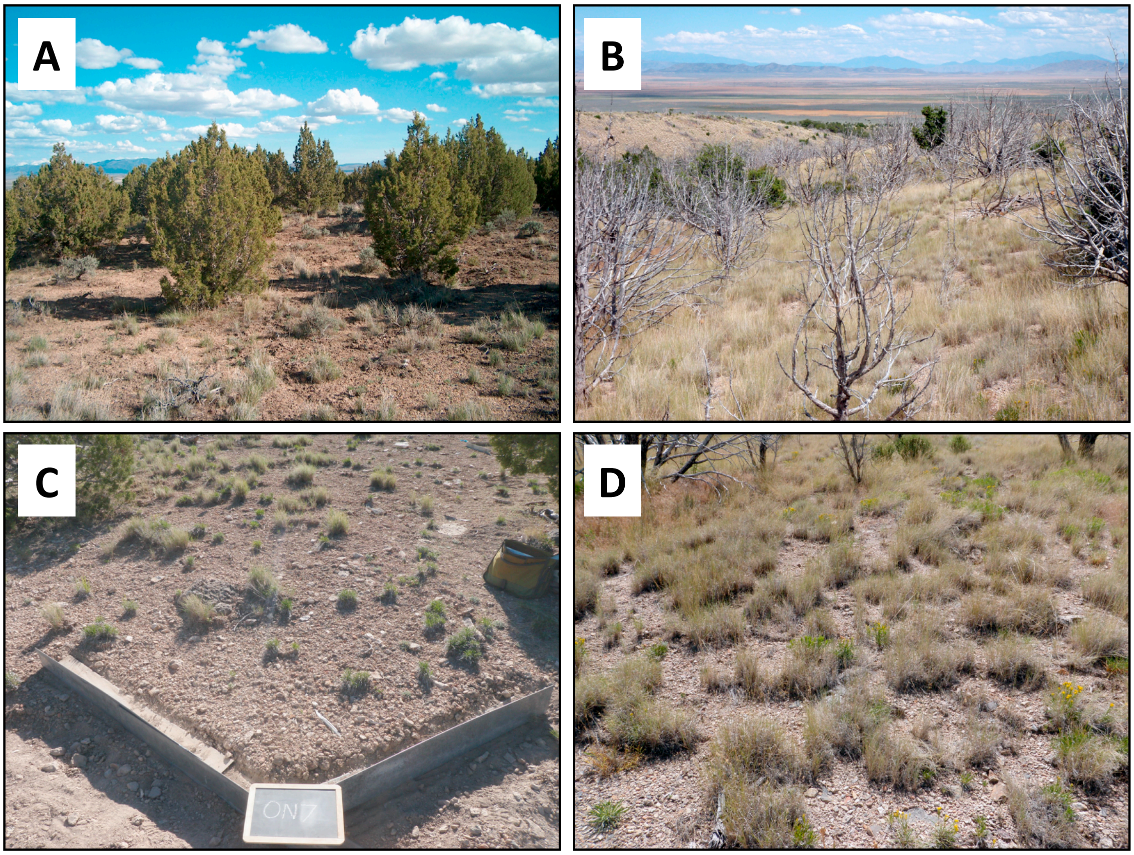

Fig. 2. Photographs of the Onaqui study site in 2015 showing isolated tree islands and the degraded intercanopy area between trees in the control (A), burned trees and intercanopy area (B), unburned intercanopy with shrub-interspace zone concentrated overland flow plot (C), and the bunchgrass and bare ground structure of the burned intercanopy (D).

control and burn treatment areas prior to prescribed burning (Pierson et al., 2010).

2.2. Prescribed fire treatments

Prescribed fire treatments were implemented by the BLM in late-summer - early autumn of 2006. The resulting control and burned areas were 1.3 ha and 2.7 ha at Marking Corral and 1.0 ha and 2.0 ha at Onaqui. Individual tree canopy scorch ranged from 50 to 75% at Marking Corral and 75–99% at Onaqui (Pierson et al., 2014). Standing burned tree and shrub skeletons were present at both sites post-fire. Burning reduced the density of juvenile trees and live shrubs by 80% each at Marking Corral and by 65% and 70% at Onaqui, respectively (Pierson et al., 2015). At the hillslope scale, burning did not significantly reduce understory total canopy cover at either site the 1st yr post-fire, but did reduce the denser shrub canopy and litter ground cover at Marking Corral (Table 1; Williams et al., 2016b). Hillslope-scale litter cover at Onaqui was low pre-fire (34%) and was not significantly reduced by burning (Table 1; Williams et al., 2016b). Burning reduced litter cover around trees from 88% to 67% at Marking Corral and from 79% to 30% at Onaqui as measured 1 yr post-fire (Williams et al., 2016b). At the patch scale, intercanopy litter cover 1 yr post-fire was reduced from near 30% to 10% at Marking Corral due to burning, but there was no significant reduction of the sparse litter cover (7–15%) in the intercanopy at Onaqui (Williams et al., 2016b). Burning had no effect on soil water repellency (0–5 cm depth) measured 1 yr post-fire on tree small plots, and surface soils on interspaces and underneath shrubs remained wettable after burning (Pierson et al., 2014). Bare ground (bare soil and rock cover) across both sites was near 70% 1 yr after the fires (Table 1). Burn severity was not quantified after the prescribed fires, but the presence of shrub skeletons, residual live and scorched tree needles, blackened litter, and downed-woody debris immediately post-fire at both sites is indicative of low- to moderate burn severities (Parsons et al., 2010).

2.3. Experimental design

Hillslope vegetation and ground cover in burned areas at each site were sampled on 30 m × 33 m site characterization plots established by Pierson et al. (2010) in 2006 prior to the fires. Pierson et al. (2010) randomly located and monumented three site characterization plots in the burn treatment area at each site for repeated sampling. The plots were sampled for tree cover, understory vegetation, and ground cover pre-fire (Pierson et al., 2010) and for understory vegetation and ground cover 1 and 2 yr post-fire (Williams et al., 2016b) (Table 1). This study re-sampled the three Pierson et al. (2010) site characterization plots in the burned area at each site as repeated measures to quantify hillslope-scale changes in vegetation and ground cover nine growing seasons post-fire.

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

Table 1
Topography, climate, soil, tree cover, and understory vegetation and ground cover at the Marking Corral and Onaqui sites before and after prescribed burning. Data from Pierson et al. (2010) and Williams et al. (2016b), except where indicated by footnote. Understory canopy and ground cover treatment means within a row followed by a different lowercase letter are significantly different ($P < 0.05$).

	Marking Corral, Nevada, USA	Onaqui, Utah, USA
Woodland community	Single-leaf pinyon[a]/Utah juniper[b]	Utah juniper[b]
Elevation (m) - Aspect	2250 – W to SW facing	1720 – N facing
Mean ann. precip. (mm)	307[c]	312[c]
Mean ann. air temp. (°C)	6.5[d]	8.9[e]
Slope (%)	10–15	10–15
Parent rock	Andesite and rhyolite[f]	Sandstone and limestone[g]
Soil association	Segura-Upatad-Cropper[f]	Borvant[g]
Depth to bedrock (m)	0.4–0.5[f]	1.0–1.5[g]
Soil surface texture	Sandy loam, 66% sand, 30% silt, 4% clay	Sandy loam, 56% sand, 37% silt, 7% clay
Tree canopy cover (%)[h,i]	21[a], 6[b]	28[b]
Trees per hectare[h,i]	465[a], 114[b]	532[b]
Mean tree height (m)[h,i]	2.3[a], 1.9[b]	2.3[b]
Juvenile trees per hectare[h,j]	444[a], 148[b]	167[b]
Live shrubs per hectare[h]	13,259	370
Dead shrubs per hectare[h]	1852	352
Intercanopy bare ground (%)[h,k]	68	84
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	

	Marking Corral			Onaqui		
	Pre-burn 2006	Burn 2007	Burn 2015	Pre-burn 2006	Burn 2007	Burn 2015
Understory Canopy Cover[m]						
Total canopy (%)	26.8 ab	40.0 b	76.9 c	19.8 a	17.6 a	65.4 c
Shrub (%)	17.7 c	6.2 b	8.7 bc	0.9 a	0.4 a	10.7 bc
Grass (%)	4.8 ab	10.0 b	63.1 d	6.2 ab	3.4 a	39.7 c
Forb (%)	0.1 a	10.6 de	0.9 ab	3.3 bc	6.0 cd	14.3 e
Ground Cover[m]						
Total ground (%)[n]	47.8 b	31.5 a	47.5 b	39.9 ab	32.5 a	48.8 b
Basal plant (%)	0.3 a	0.1 a	7.1 b	0.9 a	0.4 a	13.3 b
Litter (%)	47.4 c	31.4 ab	40.3 bc	34.4 ab	29.7 a	34.7 ab
Rock (%)[l]	25.4 cd	16.5 ab	12.8 a	29.0 cd	31.6 d	21.6 bc
Bare soil (%)	26.8 a	52.0 c	39.7 b	31.1 ab	35.9 ab	29.5 a
Bare ground (%)[k]	52.2 a	68.5 b	52.5 a	60.1 ab	67.5 b	51.1 a

[a] *Pinus monophylla* Torr. & Frém.
[b] *Juniperus osteosperma* [Torr.] Little.
[c] Estimated from 4 km grid for years 1971–2015 from Prism Climate Group (2017). Pierson et al. (2010) estimates (351 mm Marking Corral, 345 mm Onaqui) were from Prism Climate Group (2009) for years 1971–2000. Pierson et al. (2015) estimates (382 mm Marking Corral, and 468 mm Onaqui) were for years 1980–2011 based on Dayment (Thornton et al., 2012).
[d] Estimated from 4 km grid for years 1971–2015 from Prism Climate Group (2017). Pierson et al. (2010) estimate (7.2 °C) was for years 1928–1958 from Western Regional Climate Center (WRCC), Station 264–199–2, Kimberly, Nevada (WRCC, 2009).
[e] Estimated from 4 km grid for years 1971–2015 from Prism Climate Group (2017). Pierson et al. (2010) estimate (7.5 °C) was for years 1972–2005 from WRCC, Station 424–362–3, Johnson Pass, Utah (WRCC, 2009).
[f] Natural Resources Conservation Service (NRCS) (2007).
[g] NRCS (2006).
[h] Data from Pierson et al. (2010), but restricted to the area subsequently burned.
[i] Live tree data for trees ≥ 1 m height.
[j] Live trees < 1.0 m height.
[k] Bare soil and rock[l].
[l] Rock fragments > 5 mm diameter.
[m] Pre-burn 2006 and burn 2007 data from 990 m^2 vegetation plots reported in Williams et al. (2016b); burn 2015 data are new data from 990 m^2 vegetation plots in the current study. Canopy cover values exclude trees ≥ 1 m height.
[n] Cryptogam, litter, live and dead plant bases, and woody dead cover; excludes rock[l] cover.

and 2014) small plot vegetation, ground cover, soil, and rainfall simulation response measurements in summer 2015 on the previously established control and burned plots. The average slope gradient for small plots was similar across treatments at a site and was approximately 12% at Marking Corral and 18% at Onaqui (Williams et al., 2016b). The number of small plots sampled in 2015 for each site × treatment × microsite combination is shown in Table 2.

Concentrated flow plots (approximately 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 burned treatment areas as the small plots. Concentrated flow plots at both sites were randomly selected and installed using procedures described in Pierson et al. (2015). Plots were installed in shrub-interspace zones (varying amounts of shrub coppice and interspace area) and tree zones (tree coppice with minor interspace component) in each treatment at each site to separate fire effects for intercanopy areas and areas underneath tree canopies (Pierson et al., 2010, 2013; Williams et al., 2014a; Pierson et al., 2015; Williams et al., 2016b). 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 base (Fig. 2C). Collection troughs spanned the 2-m plot width and were designed to route runoff and sediment directly through

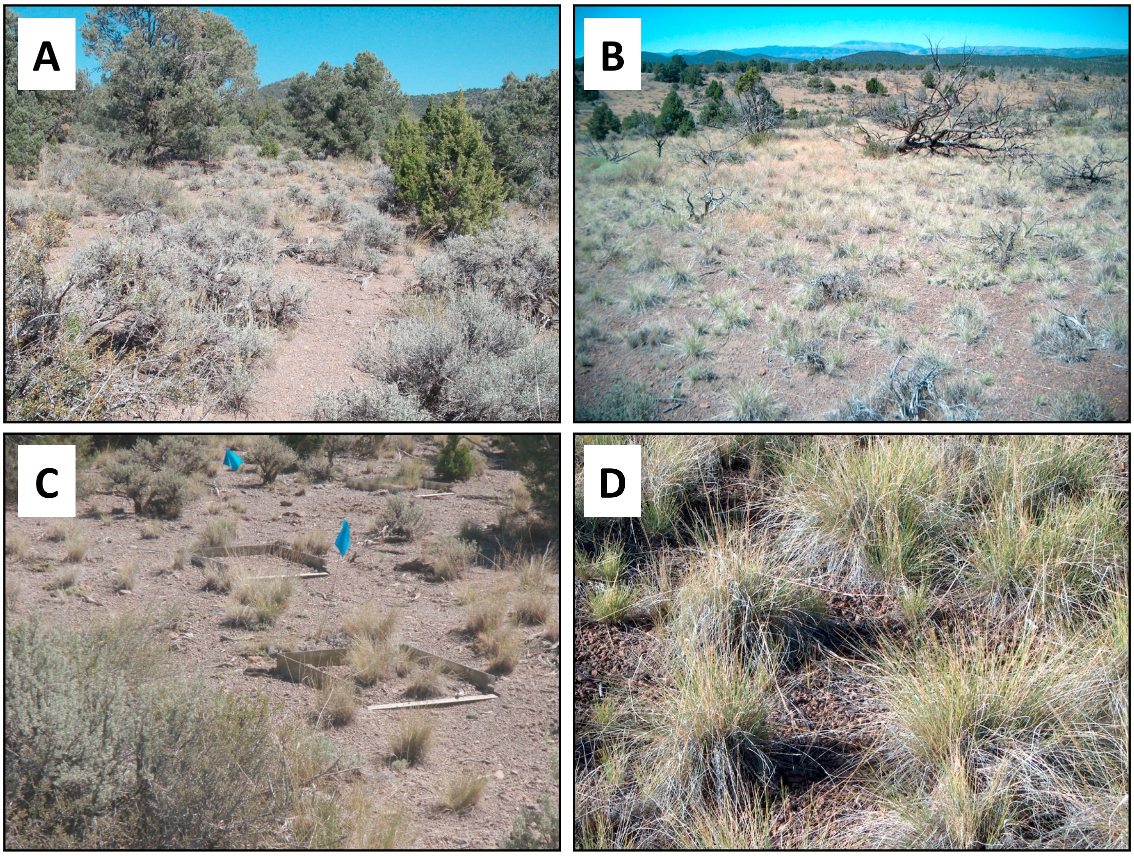

Fig. 3. Estimated annual precipitation for the Marking Corral (A) and Onaqui (B) study sites for the year prior to the study (2005) and the duration of the study period (2006–2015). The bold dashed horizontal line in each graph marks the mean annual precipitation for the respective site for years 1971–2015. Data from Prism Climate Group (2017) as estimates from a 4-km spatial grid.

Table 2
Average cover, surface roughness, and soil aggregate stability attributes measured on control and burned rainfall simulation plots (0.5 m^2) at Marking Corral and Onaqui study sites 9 yr following prescribed fire. Means within a row for a study site (Marking Corral or Onaqui) followed by a different lowercase letter are significantly different ($P < 0.05$).

	Marking Corral						Onaqui					
	Control			Burned			Control			Burned		
	Interspace	Shrub coppice	Tree coppice	Interspace	Shrub coppice	Tree coppice	Interspace	Shrub coppice	Tree coppice	Interspace	Shrub coppice	Tree coppice
Total canopy cover (%)[a]	37.5 ab	94.8 d	13.2 a	65.4 cd	56.0 bc	53.3 bc	23.8 a	76.5 c	38.7 ab	63.1 c	89.1 c	58.0 bc
Shrub canopy cover (%)	0.3 a	66.9 b	1.4 a	3.5 a	1.2 a	0.0 a	0.0 a	47.0 b	0.5 a	1.4 a	5.4 a	1.1 a
Grass canopy cover (%)	35.6 b	23.3 ab	5.4 a	61.6 c	50.0 bc	53.0 bc	9.7 a	12.4 a	21.3 ab	37.3 bc	48.6 c	34.9 bc
Forb canopy cover (%)	0.5 a	2.9 b	0.1 a	0.0 a	0.0 a	0.0 a	7.3 a	6.3 a	4.7 a	11.8 a	7.1 a	11.1 a
Standing dead canopy cover (%)[a]	0.8 ab	1.4 ab	4.6 b	0.3 a	1.9 ab	0.1 a	6.2 a	9.8 a	11.6 a	12.3 a	20.6 a	10.3 a
Total ground cover (%)[b]	13.0 a	69.8 b	82.2 b	21.8 a	39.4 a	77.0 b	13.8 a	46.5 c	72.4 d	24.9 ab	34.7 bc	42.3 c
Plant basal cover (%)	7.8 bc	13.5 c	1.9 a	6.5 ab	4.6 ab	3.2 ab	3.6 a	10.7 b	2.0 a	3.1 a	9.3 b	6.1 ab
Litter ground cover (%)	4.8 a	55.8 cd	79.5 d	14.9 ab	32.7 bc	70.2 d	7.8 a	33.9 b	69.6 c	19.7 b	20.1 b	33.8 b
Rock cover (%)[c]	7.8 ab	6.1 ab	2.2 a	13.7 b	5.6 ab	5.6 ab	40.2 b	19.8 ab	6.7 a	36.9 b	17.8 ab	5.3 a
Bare soil (%)	79.2 c	24.1 ab	15.6 a	64.5 c	55.0 bc	17.4 a	46.0 b	33.7 ab	20.8 a	38.2 b	47.5 b	52.4 b
Bare ground (%)[d]	87.1 b	30.2 a	17.8 a	78.2 b	60.6 b	23.0 a	86.2 b	53.5 b	27.6 a	75.1 b	65.3 b	57.7 b
Litter depth (mm)	1 a	6 a	34 c	3 a	12 ab	27 bc	1 a	6 a	29 b	4 a	4 a	9 a
Surface roughness (mm)	11 a	15 a	12 a	12 a	11 a	12 a	10 a	14 b	10 a	12 ab	14 b	14 b
Aggregate stability class (1–6)[e]	1.9 a	1.7 a	4.2 b	1.4 a	2.3 ab	3.8 b	2.6 a	3.8 ab	5.1 b	3.4 a	3.6 ab	3.8 ab
No. of plots	6	4	8	6	4	8	6	6	8	10	5	5

[a] Excludes tree canopy removed for rainfall simulation.
[b] Cryptogam, litter, live and dead basal plant, and woody dead cover; excludes rock[c] cover.
[c] Rock fragments > 5 mm in diameter.
[d] Bare soil and rock[c] cover.
[e] 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).

a plot outlet. Average slope gradient for concentrated flow plots was similar across treatments at a site and was approximately 11% at Marking Corral and 16% at Onaqui. Five concentrated flow plots per zone type were installed and sampled in control and burned treatment areas at each site.

2.4. Vegetation, ground cover, and soil sampling

2.4.1. Hillslope scale

Hillslope-scale understory vegetation and ground cover were measured on each 30 m × 33 m site characterization plot using line-point intercept and gap-intercept methods along five 30-m transects spaced 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 recorded at 60 points with 50-cm spacing along each of the five transects for a total of 300 sample points per plot. Percent cover for each cover type sampled was derived for each plot as the frequency of respective cover type hits divided by the total number of points sampled. Distances between plant bases (basal gaps) in excess of 20 cm were measured along each of the five 30 m transects on each plot. Average basal gap size was determined 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 cm, 51–100 cm, 101–200 cm, and 201–600 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 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 tallied tree. The crown radius for each tree was calculated as one-half the average of the minimum and maximum crown diameters. Individual tree crown area was assumed equivalent to the area of a circle, calculated with the respective crown radius. Total tree cover for each plot was calculated as the sum of measured tree cover values (crown areas) on the respective plot. The numbers of shrubs > 5-cm height and tree seedlings (5-50-cm height) were counted along three evenly spaced (6 m apart) belt transects (2 m wide × 30 m long) within each plot. Shrub and tree seedling densities for each plot were calculated as the sum of respective measures counted along each of the respective three belt transects divided by total belt transect area (180 m^2).

2.4.2. Small plot scale

Canopy (foliar) cover, ground cover, and ground surface roughness were measured on small plots using point frame methodologies (Mueller-Dombois and Ellenberg, 1974; Pierson et al., 2010). Canopy and ground cover for each plot were sampled at 15 points (spaced 5 cm apart) along each of seven evenly 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 calculated 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 ruler as the distance between a point frame level line and the ground surface. A ground surface roughness for each plot was derived as the arithmetic average 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) along the outside edge of each of the two plot borders located perpendicular to the hillslope contour. An average litter depth was derived for each plot as the mean of the eight litter depths measured.

Surface soil water repellency was quantified immediately adjacent (within ~50 cm) to each small plot prior to rainfall simulations using the water drop penetration time (WDPT) procedure (DeBano, 1981). Eight water drops (~3-cm spacing) were applied to 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 for an additional eight drops. This process was repeated 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 calculated as the arithmetic average of the means from each of the 1-cm depths sampled. Soils were classified as wettable when WDPT < 5 s, slightly water repellent when WDPT ranged 5 s to 60 s, and strongly water repellent when WDPT > 60 s (Bisdom et al., 1993).

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

2.4.3. 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 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, spaced 20 cm apart, and oriented 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 recorded along each line-point transect. Average canopy and basal gap sizes were determined for each plot 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 location was calculated as the distance between the ground surface and a survey transit level-line above the respective sample point. Ground surface roughness for each concentrated flow plot was derived as the arithmetic average of the standard deviations of the ground surface heights across the line-point transects (Pierson et al., 2010).

2.5. Hydrology and erosion measurements

2.5.1. Small plot rainfall simulations

Rainfall was applied on each small plot at target intensities of 64 mm h^{-1} (dry run) and 102 mm h^{-1} (wet-run) for 45 min each using a portable oscillating-arm rainfall simulator fitted with 80–100 Vee-jet nozzles. The rainfall simulator, raindrop characteristics, and simulator calibration procedures are described in detail in Pierson et al. (2010). The dry run was conducted on dry antecedent-soil moisture conditions (< 11% gravimetric), and the wet run was applied approximately 30 min following the dry run. The mean rainfall intensity and cumulative rainfall applied by run type were similar ($P > 0.05$) across control and burned conditions at both sites and the standard deviations by run type across all plots in the study were within 1–2 mm h^{-1} of the respective target intensities. For both study sites, the dry run intensity applied for 5-, 10-, and 15-min durations is equivalent to respective local storm return intervals of 7, 15, and 25 yr, and the wet run intensity over the same durations is equivalent to local storm return intervals of 25, 60, and 120 yr (Bonnin et al., 2006).

Timed samples of plot runoff were collected over 1-min to 3-min intervals throughout each 45-min rainfall simulation and were analyzed in the laboratory for runoff volume and sediment concentration. Runoff volume and sediment concentration for each runoff sample were determined by weighing the sample before and after oven drying at 105 °C. A suite of hydrologic and erosion response variables was 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 interval time. Cumulative runoff (mm) from each simulation was calculated as the integration of runoff rates over the total time of runoff. A runoff-to-rainfall ratio (mm mm^{-1}) was derived for each simulation by dividing cumulative runoff by total rainfall applied. A mean infiltration rate (mm h^{-1}) for each sample interval was calculated as the difference between applied rainfall and measured runoff divided by duration of the sample interval. Sediment discharge (g s^{-1}) for each sample interval was derived as the cumulative sediment for the sample interval divided by the interval time. Cumulative sediment yield (g m^{-2}) for each simulation was calculated as the integrated sum of sediment collected during runoff and was extrapolated to a unit area by dividing cumulative sediment by the 0.5 m^2 plot area. A sediment-to-runoff ratio (g m^{-2} mm^{-1}), a surrogate for erodibility, was calculated for each simulation by dividing cumulative sediment yield by cumulative runoff.

Soil profile wetting patterns were investigated over 0–20-cm depths immediately following dry-run rainfall simulations on each plot. 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 so as to not affect wet-run simulations. The percent wetted area of each exposed soil profile was measured using a 2 cm × 2 cm grid. Each grid area was determined to be dry or wet based on the dominant condition in the grid area. The area wet to 6-, 10-, and 20-cm soil depths for each 50-cm long trench was recorded as the percentage of wetted area from 0 to 6 cm, 0–10 cm, and 0–20 cm depths, respectively.

2.5.2. Concentrated overland flow simulations

Concentrated overland flow was applied using datalogger-controlled flow regulators and methodologies described by Pierson et al. (2010, 2015). Flow regulators were used to apply release rates of 15, 30, and 45 L min^{-1} to each concentrated flow plot. Each plot was prewet with a gently misting sprinkler to create wet soil conditions (~20% gravimetric) similar to those under which runoff occurs, but without detaching and eroding sediment (Pierson et al., 2015). The concentrated flow release rate sequence for each simulation was 12 min at 15 L min^{-1}, immediately followed by 12 min at 30 L min^{-1}, immediately followed by 12 min at 45 L min^{-1}. Each of the individual flow release rates was applied to each plot from a single location, ~4 m upslope of the plot outlet. Flow was routed from flow regulators 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). Plot 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 described for small plots.

Runoff and erosion response variables for each flow release rate were calculated for an 8-min time period beginning at runoff initiation (Pierson et al., 2010, 2013, 2015). A mean runoff rate (L min^{-1}) was calculated for each sample interval as the cumulative runoff divided by the interval time. Cumulative runoff (L) by release rate for each plot was calculated as the integration of runoff rates over the respective 8-min time of runoff. An averaged sediment concentration (g L^{-1}) was derived for each sample interval as the cumulative sediment divided by the interval time, and the mean sediment concentration for each flow release on each plot was determined as the average of all sediment concentrations for the respective rate on the plot. Cumulative sediment (g) by release rate for each plot was calculated as the integrated sum of sediment collected during the 8-min runoff period. Total runoff (L) and total sediment (g) for each plot was calculated as the sum of cumulative runoff and sediment, respectively, from all release rates.

Overland flow velocity and flowpath widths and depths were measured on each plot to characterize overland flow. Overland flow velocity was measured for each flow release rate on each plot by releasing a concentrated salt solution ($CaCl_2$, ~50 mL) into the flow and using electrical conductivity probes to track the mean transit time of the salt over a 2-m flowpath length (Pierson et al., 2008a, 2010, 2015). The flow velocity (m s^{-1}) was calculated by dividing the flowpath length (2 m) by the mean of multiple sampled salt travel times (n = 2 to 3 per rate per plot) in seconds. The width and depth of all flowpaths for each rate on each plot were measured at a cross-section located 3 m downslope of the flow release point. A mean flowpath width and depth for each simulation was derived as the arithmetic average of flowpath widths and depths measured at the 3-m cross-section.

2.6. Statistical analyses

Statistical analyses were conducted using SAS software, version 9.4 (SAS Institute Inc., 2013). All statistical analyses were restricted to within-site comparisons except where explicitly stated in results. Hillslope-scale vegetation and ground cover data collected on 30 m × 33 m site characterization plots (this study with comparisons to previous years [Pierson et al., 2010; Williams et al., 2016b]) were analyzed using a repeated measures mixed-model with three treatment levels (preburn, burned year 1, and burned year 9) and sample year (2006, 2007, 2015) as the repeated measure. Covariance structure was evaluated using fit statistics suggested by Littell et al. (2006) and the best fit model was applied. Data from small rainfall simulation plots for each site were analyzed using a mixed model with two treatment levels (control and burned) and three microsite levels (interspace, shrub coppice, and tree coppice). Vegetation, ground cover, and flowpath dimension and velocity data from concentrated flow plots for each site were analyzed using a mixed model with two treatment levels (control and burned) and two microsite levels (shrub-interspace zone and tree zone). Concentrated flow runoff and erosion data for each site were analyzed with a repeated measures mixed-model using the 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 min^{-1}. Carryover effects of concentrated flow releases were modeled with an autoregressive order 1 covariance structure (Littell et al., 2006). Plot location was considered a random effect and site, treatment, and microsite were considered fixed effects in all respective analyses. Normality was tested prior to ANOVA using the Shapiro-Wilk test and deviance was addressed by data transformation. Where necessary, arcsine-square root transformations were used to normalize proportion data (e.g., canopy and ground cover) and logarithmic transformations were used to normalize runoff and erosion data. Back transformed means are reported. Mean separation was determined using the LSMEANS procedure (SAS Institute Inc., 2013). All reported significant effects (mean differences and correlations) were tested at the $P < 0.05$ level.

3. Results

3.1. Vegetation and surface conditions

3.1.1. Hillslope-scale vegetation

Burning stimulated hillslope-scale canopy cover at both sites, but had minimal impact on ground cover recruitment at the same spatial-scale after nine growing seasons (Table 1). The degraded thinning shrub understory at Marking Corral transitioned to a grass-dominated community post-fire, with > 60% grass canopy cover (Fig. 1B). Burning at Onaqui likewise promoted substantial grass cover (40%, Fig. 2B), but also facilitated increased forb canopy cover (14%). Although grass cover was well distributed at both sites post-fire, perennial bunchgrasses were more prevalent in intercanopy areas (Figs. 1, 2, and 4) and cheatgrass was more prevalent in areas previously covered by tree canopy and litter (Fig. 4). Perennial grass canopy cover increased from near 6% on average at the sites pre-fire to 33% at Marking Corral and 24% at Onaqui following burning. Bluebunch wheatgrass (*Pseudoroegneria spicata* [Pursh] Á. Löve) was the dominant perennial grass at both sites 9 yr post-fire, with 31% canopy cover at Marking Corral and 19% canopy cover at Onaqui. Other tall perennial grasses present in small amounts at both sites 9 yr post-fire included Indian rice grass (*Achnatherum hymenoides* [Roem. & Schult.] Barkworth, < 1% canopy cover) and squirreltail (*Elymus elymoides* [Raf.] Swezey, ~2% canopy cover). The short perennial grass Sandberg bluegrass (*Poa secunda* J. Presl) was also present at both sites post-fire with canopy coverage of 1–2% on average. Cheatgrass canopy cover was < 1% in the burn treatment area at both sites prior to the fire and was 30% at Marking Corral and 16% at Onaqui after burning. As expected, burning consumed sagebrush shrubs, but increased cover of root-sprouting yellow rabbitbrush shrubs (*Chrysothamnus viscidiflorus* [Hook.] Nutt.). Sagebrush comprised 83% and 10% of the total live shrub density (Table 1) pre-fire at Marking Corral and Onaqui, respectively. Combined canopy cover of all sagebrush species at Marking Corral decreased from 14% pre-fire to 6% 9 yr post-fire. Canopy cover of sagebrush species was < 1% at Onaqui before and after burning. The increase in rabbitbrush after burning and gradual sagebrush seedling recruitment over nine growing seasons resulted in no change in total shrub canopy cover at Marking Corral (9%) and an increase in shrub canopy cover at Onaqui (11%) (Table 1). Sagebrush species comprised 85% of the total live shrub density at Marking Corral (5315 shrubs per ha) and 3% of the total live shrub density at Onaqui (11,056 shrubs per ha) in the 9th yr. Live shrub density post-fire at Onaqui was dominated by yellow rabbitbrush (10,444 shrubs per ha, 95% of total). Basal plant cover was stimulated by the prescribed fire treatment at both sites (Table 1). Basal plant cover increased by factors of 24 and 15 at Marking Corral and Onaqui, respectively, over the nine growing seasons. The distance between plant bases tended to be greater at Onaqui relative to Marking

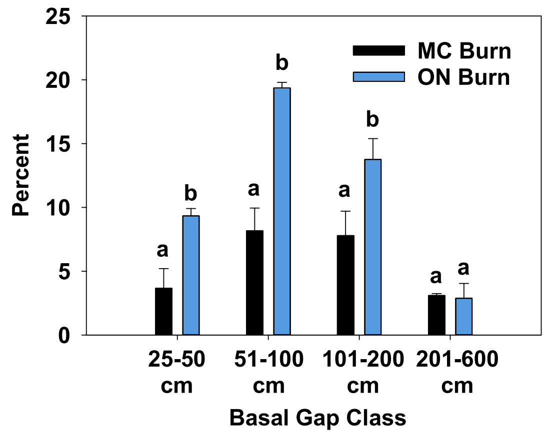

Fig. 5. Basal cover gaps (distance between plant bases) by gap class measured on site characterization plots (990 m^2) in burned areas at the Marking Corral (MC) and Onaqui (ON) study sites 9 yr after prescribed fire. Error bars depict standard error. Site means within a gap class followed by a different lowercase letter are significantly different ($P < 0.05$).

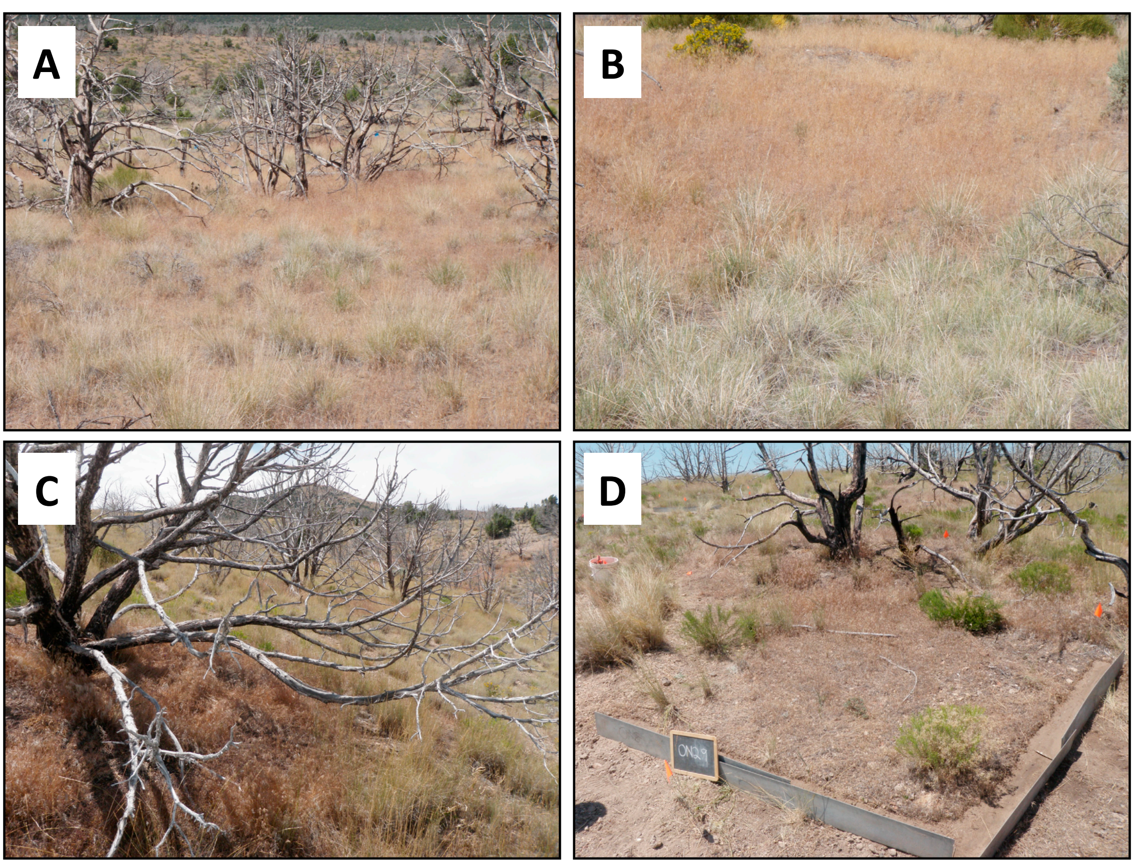

Fig. 4. Photographs showing islands of brown, senescing cheatgrass (*Bromus tectorum* L.) in areas previously covered by tree canopies, and bunchgrass-dominated intercanopy areas between burned trees at the Marking Corral (A and B) and Onaqui (C and D) sites 9 yr post-fire. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Corral (Fig. 5). Although burning facilitated understory plant growth, hillslope-scale total ground cover, litter cover, and bare ground 9 yr post-fire were similar to pre-fire levels for both sites (Table 1).

3.1.2. Small plot vegetation and surface conditions

The primary fire effect on vegetation at the small-plot scale was an increase in grass canopy cover (Table 2). This effect was consistent across all burned interspace plots, but varied by site for burned shrub and tree coppices. Interspaces in the controls at Marking Corral and Onaqui contained an average of 38% and 24% total canopy and 36% and 10% grass canopy cover, respectively. The burn treatment increased interspace total canopy and grass canopy covers by factors of 2 to 4 across the sites. All other canopy cover values were similar across control and burned interspace plots at each site (Table 2). Burning reduced shrub canopy cover by 56-fold and 9-fold on shrub coppices at Marking Corral and Onaqui, respectively. The shrub removal resulted in reduced total canopy cover (from 95% to 56%) on shrub coppices at Marking Corral, but substantial grass recruitment as live and standing dead cover resulted in similar total canopy cover for control (77%) and burned (89%) shrub plots at Onaqui (Table 2). Grass cover was substantially greater on burned (53%) than control (5%) tree plots at Marking Corral, increasing the total canopy cover on burned versus control tree plots at that site. Grass canopy cover was high on control tree plots (21%) at Onaqui and was not altered by the fire treatment at that site (Table 2). Overall, burning created a more uniform vegetation structure than in controls, with grass cover as the dominant vegetation type across all burned microsites at both sites (Table 2).

The fire treatment had limited impact on ground cover and ground surface conditions measured at both sites nine growing seasons post-fire (Table 2). Total ground cover (13% to 25%) and bare ground (75% to 87%) were comparable for control and burned interspace plots at a site. Total ground cover and basal cover were lower for burned (39% and 5%) than control (70% and 14%) shrub coppice plots at Marking Corral, but there were no significant differences in ground cover across treatments for shrub coppices at Onaqui. There were no significant fire impacts on ground cover for tree coppices at Marking Corral. In contrast, total ground and litter covers were lower for burned (42% and 34%) than control (72% and 70%) tree coppices at Onaqui. Litter depth was not significantly different for control versus burned conditions at Marking Corral, but was 3-fold lower for burned than control tree coppices at Onaqui (Table 2). With exception of tree plots at Onaqui, soil surface roughness by microsite was comparable for control and burned treatments at a site (Table 2). Surface soil aggregate stability by microsite was similar for control and burned plots at a site and was generally greater for tree than shrub and interspace plots (Table 2).

3.1.3. Patch scale vegetation and surface conditions

Prescribed fire stimulated herbaceous (grass and forb) vegetation in the intercanopy and underneath trees at both sites (Figs. 1, 2, and 4). Herbaceous canopy cover was 2- to 3-fold greater in burn versus control treatment areas nine growing seasons after burning. Grass canopy cover in the burn treatment at Marking Corral exceeded 60% on shrub-interspace and tree zone plots and was the primary canopy cover type at the site (Fig. 6A). The same measure averaged 33% and 22% on control shrub-interspace and tree zone plots at that site. At Onaqui, grass canopy cover averaged 29% and 61% on burned shrub-interspace and tree zones and 4% and 29% for the same microsites in the control (Fig. 6B). Forb canopy cover was minimal (< 1%) at Marking Corral and was not affected by the burn treatment. Forb canopy cover at Onaqui averaged 13% across all microsites and was not altered by the fire. Shrub canopy response to burning differed by site (Fig. 6). At Marking Corral, shrub canopy cover declined on both shrub-interspace and tree zones largely due to fire removal of mature sagebrush cover (Fig. 6A). Shrub cover was generally low at Onaqui and increased slightly in the burn area associated with root sprouting of rabbitbrush and minor sagebrush seedling recruitment (Fig. 6B).

The effects of burning on ground cover varied with cover type and were inconsistent across the study sites (Fig. 7). Ground cover was generally improved or unchanged by the burn treatment at Marking Corral and was unchanged or declined following burning at Onaqui. At Marking Corral, burning increased litter cover on shrub-interspace plots and had no significant effect on litter recruitment in tree zones (Fig. 7A). In contrast, burning reduced litter cover in tree zones at Onaqui, but had no effect on litter accumulation in shrub-interspace zones (Fig. 7B). Bare ground in shrub-interspace zones remained high at both sites nine growing seasons post-fire (51% at Marking Corral and 77% at Onaqui) and was comparable across treatments. Bare ground in tree zones averaged 12% to 33% across control and burned conditions at the sites and was significantly greater on burned (33%) than control (12%) tree zones at Onaqui solely. Plant basal cover by microsite was comparable for control and burned conditions at Marking Corral, but increased across all plots at Onaqui after the fire (Fig. 7). Overall, burning increased the ground surface protection in shrub-interspace zones at Marking Corral through litter recruitment over a 9-yr period while not substantially altering the existing litter layer in well-protected tree zones. In contrast, the burn at Onaqui had no effect on litter cover in shrub-interspace zones and reduced litter and increased bare ground in tree zones.

As with ground cover, fire effects on canopy and basal gaps varied by site. All vegetation gap measures at Marking Corral were similar for control and burned shrub-interspace zones. Burning of shrub-interspace zones at Onaqui reduced the percentage of canopy and basal gaps in excess of 50 cm by 2- to 6-fold and yielded gap values similar to those for the more vegetated shrub-interspace zones at Marking Corral (Table 3). The ground surface in unburned tree zone gaps was well-protected by an extensive litter layer. Burning reduced gap sizes for

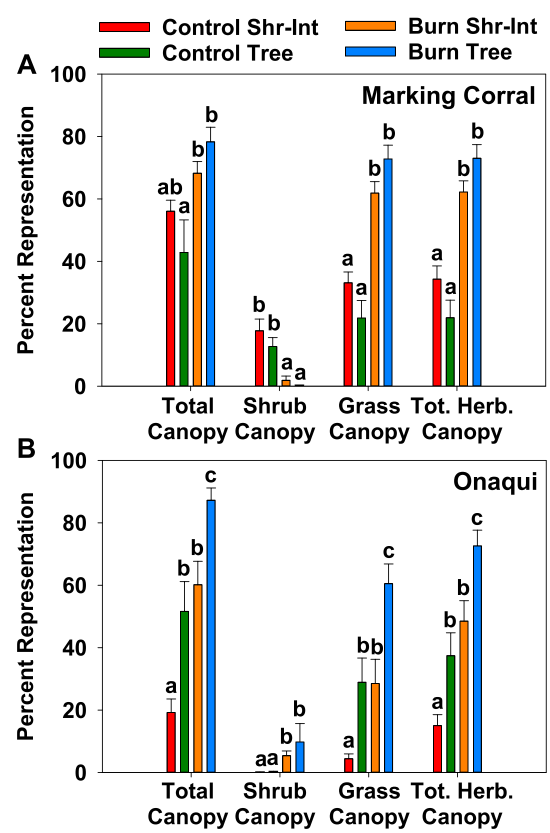

Fig. 6. Canopy (foliar) cover characteristics measured on shrub-interspace zone (Shr-Int) and tree zone (Tree) concentrated flow plots (9 m^2) in control and burn treatment areas at the Marking Corral (A) and Onaqui (B) sites 9 yr following prescribed fire. Error bars depict standard error. Means within a cover type (e.g., total canopy) followed by different lowercase letters are significantly different ($P < 0.05$).

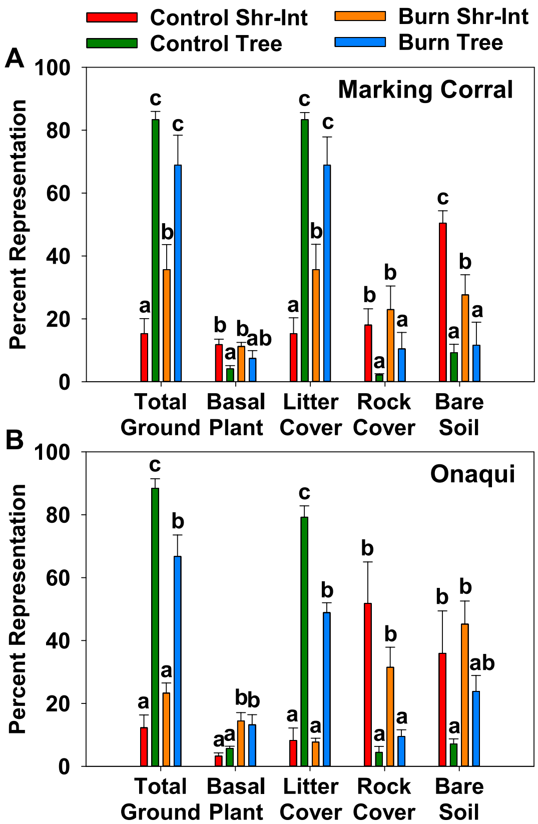

Fig. 7. Ground cover characteristics measured on shrub-interspace zone (Shr-Int) and tree zone (Tree) concentrated flow plots (9 m^2) in control and burn treatment areas at the Marking Corral (A) and Onaqui (B) sites 9 yr following prescribed fire. Error bars depict standard error. Means within a cover type (e.g., total ground) followed by different lowercase letters are significantly different ($P < 0.05$).

most gap classes in tree zones at Marking Corral and had less impact on gaps in tree zones at Onaqui (Table 3). The ground surface roughness was unchanged by the burn treatment at the sites and averaged 15 to 18 mm at Marking Corral and 17 to 20 mm at Onaqui.

3.2. Small plot infiltration, runoff, and erosion

The primary impacts of prescribed burning on hydrology and erosion at the small plot scale were increased infiltration and decreased sediment discharge for interspaces (Fig. 8, Table 4). Unburned interspaces exhibited the lowest infiltration and highest sediment discharge at each site (Fig. 8). Interspace runoff-to-rainfall ratios averaged near 40% and 60% for dry- and wet-run simulations in unburned areas at both sites. The burn treatment improved dry- and wet-run infiltration in the interspaces by > 25% and > 70% relative to controls (Fig. 8A and B; Table 4). Runoff-to-rainfall ratios for burned interspaces averaged 15 to 29% for the dry- and wet-runs. Cumulative sediment yield from burned interspaces was about 60% less than measured on controls for both simulation rates (Table 4). The decreased sediment delivery from interspace plots after burning at both sites was primarily controlled by the fire-induced reduction in runoff (improved infiltration), as the sediment-to-runoff ratio at a site was comparable for control and burned interspaces with exception of the wet run at Onaqui (Table 4). Runoff and sediment discharge were generally low for the litter-protected control shrub and tree coppice plots and fire impacts on runoff and erosion from these microsites varied across the two sites (Table 4, Fig. 8C and D). Runoff and sediment measures for shrub coppices at Marking Corral were 7- to nearly 20-fold greater for burned versus control treatments (Table 4). Runoff and erosion were similar for control and burned shrub plots at Onaqui (Table 4). Burning increased runoff from tree coppices at Marking Corral by 2- to 3-fold, but there were no significant differences in tree coppice runoff for burned versus control treatments at Onaqui (Table 4). Burning did not significantly alter sediment yield from tree coppices at either site, but the sediment-to-runoff ratio and sediment concentration measures were lower for burned than unburned tree plots at Onaqui (Table 4). Soil water repellency was strong on control and burned tree coppice plots at both sites, and repellency effects on wetting depth were observed over 0–10-cm soil depth in the control and burn treatments at Marking Corral and 0–6-cm soil depth in the control at Onaqui (Table 4). Mean infiltration for a treatment × microsite combination was similar across sites ($P > 0.05$; Fig. 8A and B). Sediment-to-runoff ratios, sediment concentrations, and sediment discharge rates across all microsites and treatments were greater for Onaqui than Marking Corral ($P < 0.05$; Table 4, Fig. 8C and D), clearly demonstrating that site as having the more erodible soil for control and burned treatments.

3.3. Concentrated flow runoff and erosion

The burn treatment reduced runoff and erosion from concentrated flow experiments in shrub-interspace zones at Marking Corral (Table 5), but had no effect on the same measures in shrub-interspace zones at Onaqui (Table 6). Across both sites, runoff from concentrated flow releases was primarily controlled by litter cover, and cumulative sediment was controlled by cumulative runoff and runoff velocity (Fig. 9). Runoff velocity from concentrated flow releases was regulated by litter

Table 3
Canopy and basal cover gaps by gap class for concentrated flow plots (9 m^2) in control and burned areas at the Marking Corral and Onaqui study sites 9 yr after prescribed fire. Means within a row for a study site (Marking Corral or Onaqui) followed by a different lowercase letter are significantly different ($P < 0.05$).

	Marking Corral				Onaqui			
	Control		Burned		Control		Burned	
Canopy and basal gap classes	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Tree zone
Canopy gaps 25–50 cm (%)[a]	16 b	12 b	11 b	2 a	16 b	15 b	15 b	4 a
Canopy gaps 51–100 cm (%)[a]	12 b	10 b	6 ab	1 a	22 c	10 bc	7 ab	2 a
Canopy gaps 101–200 cm (%)[a]	2 a	12 b	1 a	0 a	13 b	3 a	2 a	0 a
Canopy gaps 201–400 cm (%)[a]	0 a	8 b	0 a	0 a	8 b	0 a	0 a	0 a
Basal gaps 25–50 cm (%)	22 a	14 a	16 a	20 a	17 a	23 a	23 a	17 a
Basal gaps 51–100 cm (%)	32 b	18 ab	19 ab	11 a	25 b	21 b	24 b	9 a
Basal gaps 101–200 cm (%)	11 ab	23 b	12 ab	2 a	25 b	9 a	9 a	3 a
Basal gaps 201–400 cm (%)	3 ab	9 b	0 a	0 a	9 a	2 a	2 a	0 a
Average canopy gap (cm)[a]	45 a	66 b	43 a	34 a	62 b	45 a	42 a	36 a
Average basal gap (cm)	60 b	72 b	51 ab	40 a	72 b	55 a	55 a	44 a
No. of plots	5	5	5	5	5	5	5	5

[a] Excludes tree canopy cover for trees ≥ 1 m in height.

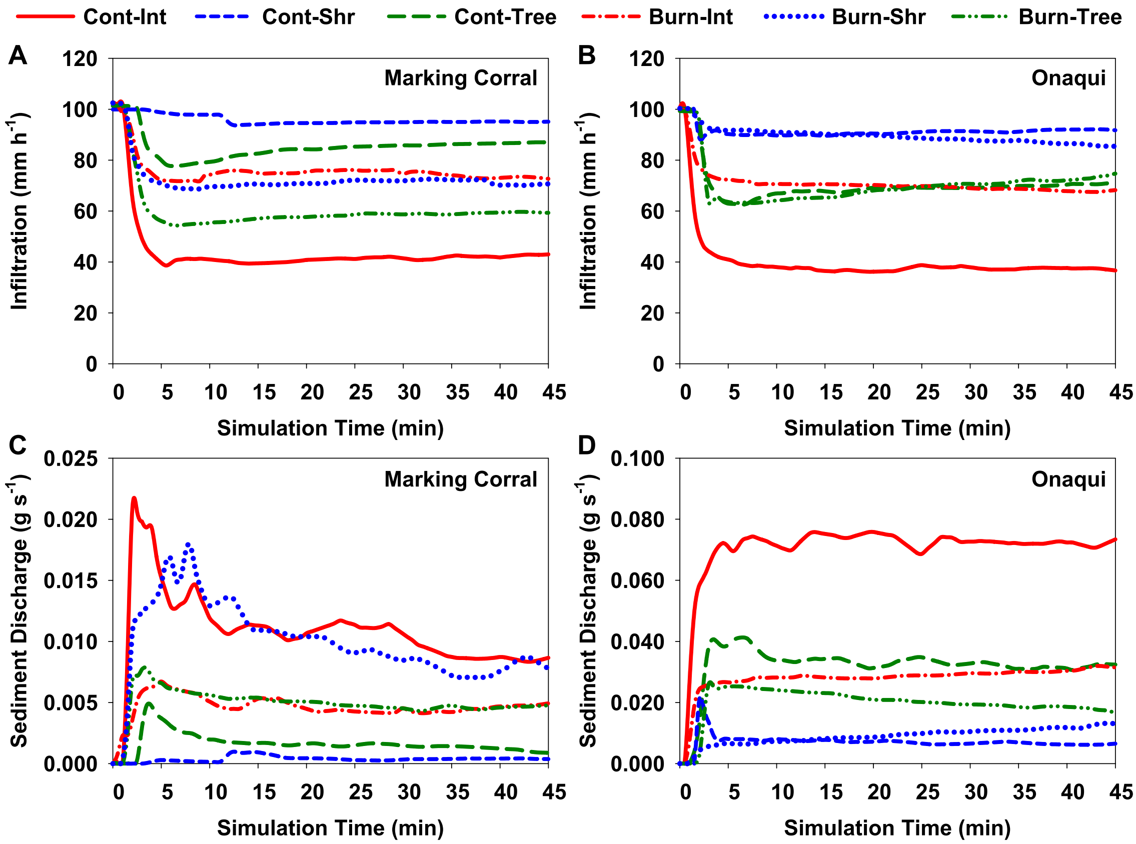

Fig. 8. Infiltration (A and B) and sediment discharge (C and D) for wet-run (102 mm h^{-1}, 45 min) rainfall simulations on interspace (Int), shrub coppice (Shr), and tree coppice (Tree) small plots (0.5 m^2) in control (Cont) and burned (Burn) areas at the Marking Corral and Onaqui study sites 9 yr after prescribed fire.

cover and cumulative runoff (Fig. 10). For unburned conditions, high levels of runoff from the degraded and bare shrub-interspace zones formed concentrated flow paths with high flow velocity, sediment detachment, and sediment transport capacity (Tables 5 and 6). Burning reduced shrub-interspace zone total runoff by 64% at Marking Corral (Table 5). The reduced flow spread out into wider flow paths than in the control and travelled downslope at ~50–60% slower flow velocity than measured on unburned shrub-interspace plots (Table 5). The reduced runoff and flow velocity limited sediment detachment and transport on burned shrub-interspaces at Marking Corral, resulting in 20-fold less total sediment than measured for the control shrub-interspace plots at the site (Table 5). With few exceptions, all runoff and sediment measures for burned shrub-interspaces at Onaqui were comparable to those for control shrub-interspace plots (Table 6). Overland flow released into burned shrub-interspace zones at Onaqui tended to concentrate into narrower flow paths than in the control, but the flow velocities and sediment concentration were similar across both treatments.

Burning had no significant effect on total runoff and sediment delivered from the well-protected tree zone plots at Marking Corral (Table 5), but increased runoff and sediment delivered from tree zones at Onaqui (Table 6). Total runoff and sediment from all tree zone plots at Marking Corral were low and were ~50–70% and ~80–90% less than measured on control shrub-interspace zones at the site. Overland flow in burned tree zones at Marking Corral formed wider flow paths than in control tree zones, but the flow velocity by release rate, total runoff, and total sediment were comparable across control and burned tree zone plots at the site (Table 5). In contrast, burned tree zones at Onaqui generated cumulative runoff and sediment values and flowpath characteristics similar to those measured for control and burned shrub-interspace zones at the site. Total runoff and sediment generated from burned tree zones at Onaqui were > 3-fold greater than measured on control tree zones at the site. Velocity for the 15 and 30 L min^{-1} release rates was greater for burned versus control tree zones at Onaqui and the greater total runoff and overall high flow velocities relative to control tree zones generated higher sediment yield (Table 6). Similar to erodibility at the small-plot scale, sediment concentration of runoff by flow release rate was higher for Onaqui than Marking Corral for both treatments ($P < 0.05$; Tables 5 and 6).

4. Discussion

4.1. Fire as a mechanism to re-establish sagebrush steppe vegetation on Great Basin woodlands

Post-fire perennial herbaceous and sagebrush vegetation succession in this study are consistent with the literature for tree removal on woodland-encroached sagebrush rangelands in the Great Basin (Barney and Frischknecht, 1974; Koniak, 1985; West and Yorks, 2002; Bates et al., 2011; Miller et al., 2013; Bates et al., 2014; Chambers et al., 2014b; Miller et al., 2014; Roundy et al., 2014a, 2014b; Bates and Davies, 2016). In a 4-yr study of 11 Great Basin sites, Miller et al. (2014) found low to moderate severity burning of pinyon- and juniper-encroached sagebrush rangelands initially reduced tall perennial grass cover (from 10% to 7%) and sagebrush cover (from 8% to < 1%), but cover of tall perennial grasses (~13%) in burned areas exceeded that in control areas (10%) 3 yr after burning. Sagebrush seedling density increased in burned areas throughout the study, but sagebrush cover remained low (1%) for burned areas 3 yr post-fire. Perennial forb cover was similar for burned and control treatments 1 yr post-fire, but was greater for burned (7%) than control (4%) treatments 3 yr post-fire (Miller et al., 2014). Roundy et al. (2014a, 2014b) investigated effects of pinyon and juniper removal on soil water availability and vegetation at 15 Great Basin sites, inclusive of the 11 sites from Miller et al. (2014). Roundy et al. (2014b) reported burning in later phases of woodland

Table 4

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 burned areas at Marking Corral and Onaqui study sites 9 yr following prescribed fire. Means within a row for a study site (Marking Corral or Onaqui) followed by a different lowercase letter are significantly different ($P < 0.05$).

	Marking Corral						Onaqui					
	Control			Burned			Control			Burned		
Rainfall simulation variable	Interspace	Shrub coppice	Tree coppice	Interspace	Shrub coppice	Tree coppice	Interspace	Shrub coppice	Tree coppice	Interspace	Shrub coppice	Tree coppice
Dry Run Simulation (64 mm h^{-1}, 45 min)												
Cumulative runoff (mm)	18 b	–	8 a	7 a	8 a	15 b	18 c	1 a	8 ab	8 ab	1 a	15 bc
Runoff-to-rainfall (mm mm^{-1}) × 100%	37 b	–	17 a	15 a	17 a	33 b	38 c	2 a	16 ab	17 ab	3 a	31 bc
Mean infiltration rate (mm h^{-1})[a]	40 a	–	50 b	51 b	49 b	39 a	38 a	61 c	54 bc	52 bc	58 bc	43 ab
Cumulative sediment (g m^{-2})[a]	18 bc	–	9 ab	8 a	23 c	18 bc	126 c	9 a	53 ab	52 ab	9 a	71 bc
Sediment/runoff (g m^{-2} mm^{-1})[a]	1.02 ab	–	0.83 a	0.76 a	2.30 b	1.06 ab	6.76 bc	5.55 abc	7.39 c	5.51 ab	4.10 a	4.64 a
Sediment concentration (g L^{-1})[a]	0.96 ab	–	0.67 a	0.67 a	2.08 b	1.07 ab	6.14 bc	3.86 abc	6.86 c	4.77 ab	3.43 a	3.78 a
Percent wet at 0–6 cm depth	100 a	100 a	83 a	98 a	99 a	87 a	88 b	90 b	58 a	96 b	100 b	87 b
Percent wet at 0–10 cm depth	97 b	100 b	77 a	98 b	98 b	81 a	82 ab	84 ab	54 a	92 b	100 b	80 ab
Percent wet at 0–20 cm depth	59 a	70 a	69 a	70 a	76 a	65 a	68 ab	54 ab	41 a	79 b	78 b	69 ab
Mean soil water repellency (s)[b]	–	–	80 a	–	–	62 a	–	–	98 a	–	–	88 a
Depth of max water repellency (cm)[c]	–	–	1	–	–	3	–	–	0	–	–	2
Percent of plots with runoff	100	25	88	83	75	88	100	60	100	100	60	100
Wet Run Simulation (102 mm h^{-1}, 45 min)												
Cumulative runoff (mm)	44 d	3 a	12 ab	20 bc	23 bc	31 cd	46 b	6 a	23 a	22 a	8 a	22 a
Runoff-to-rainfall (mm mm^{-1}) × 100%	57 d	4 a	16 ab	26 bc	29 bc	41 cd	61 b	9 a	30 a	29 a	11 a	29 a
Mean infiltration rate (mm h^{-1})[a]	44 a	94 c	83 c	76 bc	73 bc	60 ab	40 a	88 b	70 b	71 b	87 b	70 b
Cumulative sediment (g m^{-2})[a]	59 c	3 a	10 a	26 ab	54 bc	27 ab	381 b	48 a	174 a	152 a	61 a	108 a
Sediment/runoff (g m^{-2} mm^{-1})[a]	1.35 ab	0.41 a	0.67 a	1.05 ab	1.93 b	0.72 a	8.05 d	5.44 ab	7.30 cd	6.29 bc	6.06 abc	4.90 a
Sediment concentration (g L^{-1})[a]	1.32 ab	0.32 a	0.61 a	0.87 ab	1.79 b	0.69 a	7.83 d	5.01 ab	6.94 cd	6.00 bc	4.88 ab	4.54 a
Percent of plots with runoff	100	75	88	100	100	100	100	80	100	100	80	100
No. of plots	6	4	8	6	4	8	4	5	5	10	5	5

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

[b] Mean soil water repellency for 0–5 cm soil depth assessed as water drop penetration time (WDPT, 300 s maximum). Soils were classified slightly water repellent if WDPT ranged 5 to 60 s and strongly water repellent if WDPT exceeded 60 s (Bisdom et al., 1993).

[c] Soil depth (below mineral soil surface) with highest average WDPT.

Table 5
Runoff, sediment, and flowpath variables by flow release rate for concentrated flow experiments (9 m^2) in control and burned areas at the Marking Corral study site 9 yr after prescribed fire. Means within a row followed by a different lowercase letter are significantly different ($P < 0.05$).

Marking Corral study site		Control		Burned	
Concentrated flow variable	Release rate ($L\,min^{-1}$)	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Tree zone
Cumulative runoff (L)	15	43 c	0 a	0 a	9 b
	30	171 b	30 a	26 a	52 a
	45	287 a	106 a	153 a	167 a
	Total	501 b	136 a	180 a	228 a
Cumulative sediment (g)[a]	15	82 b	–	–	9 a
	30	718 b	203 a	31 a	29 a
	45	1542 b	220 a	84 a	166 a
	Total	2343 b	423 a	115 a	204 a
Sediment concentration ($g\,L^{-1}$)[a]	15	2.3 b	–	–	0.3 a
	30	3.8 b	4.7 b	0.9 a	0.5 a
	45	5.2 b	1.9 ab	0.5 a	0.9 a
Flow velocity ($m\,s^{-1}$)[a]	15	0.10 b	–	–	0.03 a
	30	0.15 b	0.04 a	0.07 a	0.05 a
	45	0.22 b	0.05 a	0.09 a	0.07 a
Flow path width (cm)[a]	15	30 a	–	–	75 b
	30	28 a	13 a	85 b	131 c
	45	28 a	30 a	160 b	158 b
Flow path depth (cm)[a]	15	0.70 a	–	–	0.38 a
	30	0.91 a	1.04 a	0.84 a	0.69 a
	45	1.07 a	0.83 a	1.04 a	0.93 a
Percent of plots with runoff (n = 5 per treatment × microsite combination)	15	100	0	0	60
	30	100	80	100	100
	45	100	100	100	100

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

encroachment increased days of plant available soil water in the spring-season resource growth pool in each of 4 yr post-fire. Roundy et al. (2014a, 2014b) and Miller et al. (2014) attributed increased perennial herbaceous cover post-fire to the longer period of seasonal soil water availability and associated enhanced growth of residual perennial plants. Bates et al. (2014) found burning of late-succession western juniper (*J. occidentalis* Hook.) woodlands reduced tall perennial grass cover by 75% to 85% 1 yr post-fire. Tall perennial grass cover returned to pre-fire levels (~3% to 10%) after three growing seasons, exceeded pre-fire levels 6 yr post-fire, and was > 2-fold greater than pre-fire levels 9 yr after burning. Burning had minimal effect on perennial forb cover (~1% to 5%) after nine growing seasons (Bates et al., 2014). Prescribed burning fully consumed sagebrush (~5% to 15% cover pre-fire), and sagebrush cover was < 2% 9 yr post-fire. In an extensive literature synthesis, Miller et al. (2013) reported that tall perennial grass cover returns to pre-treatment levels within 2 to 3 yr post-fire and that sagebrush recovery on treated woodlands commonly requires 20–35 yr (but as much as 50 yr) due to moisture requirements for germination, limited seed sources, and a short seed dispersal distance (~9 m to 31 m) for sagebrush. Prescribed burning in this study increased hillslope-scale perennial grass cover by factors of 4 to 7 over 9 yr. The primary tall grass constituent was bluebunch wheatgrass (~20% to 30% cover). Burning had minimal impact on forb cover over 9 yr at Marking Corral, but perennial forb cover increased by a factor of 3 at Onaqui (from 3% to 9%). Such responses of perennial vegetation to fire are strongly influenced by pre-fire vegetation and survival of existing plants during burning (Bates et al., 2006, 2009, 2011; Miller et al., 2013; Bates et al., 2014; Miller et al., 2014; Bates and Davies, 2016). Pre-fire perennial grass cover amounts (~6% on average) at our sites were lower than, but approached pre-fire amounts (~10–15%) for sites with favorable post-fire perennial grass recruitment in studies by Bates et al. (2014) and Miller et al. (2014). Hillslope-scale sagebrush cover decreased from 14% pre-fire to 6% (4537 plants per ha) 9 yr post-fire at Marking Corral and remained < 1% (333 plants per ha) in the burn at Onaqui. Fire consumption of sagebrush and its delayed post-fire recovery were expected (Barney and Frischknecht, 1974; Koniak, 1985; Ziegenhagen and Miller, 2009; Bates et al., 2011; Miller et al., 2011, 2014).

Favorable recruitment of perennial grasses in this study did not preclude ample increases of cheatgrass. The 15- to 30-fold increases in cheatgrass cover post-fire at our sites was unexpected given minimal pre-fire cover (< 1%) of the species and abundant perennial herbaceous vegetation post-fire, but is not atypical for the temperature-soil moisture regimes at the sites (Miller et al., 2013; Bates et al., 2014; Miller et al., 2014; Chambers et al., 2014b; Roundy et al., 2014a). The primary determinants of cheatgrass responses to burning are: 1) soil temperature and moisture regimes, 2) pre-fire presence and post-fire survival of perennial grasses and forbs, and 3) a cheatgrass seed source (Condon et al., 2011; Miller et al., 2013; Bates et al., 2014; Chambers et al., 2014a, 2017). Cheatgrass cover commonly increases post-fire on sites with warm-dry soil temperature-moisture regimes and is more limited on sites with cool-moist soil temperature-moisture regimes (Chambers et al., 2007; Miller et al., 2013; Chambers et al., 2014b; Miller et al., 2014; Roundy et al., 2014a). The elevation thresholds for warm versus cool soil temperature regimes occurs at 1675 to 1980 m msl in the central Great Basin, and the annual precipitation threshold for dry versus moist soil moisture regimes is approximately 300 mm (Miller et al., 2013). The classification places Marking Corral (2250 m) at the warmer/lower elevation end of the cool temperature regime and Onaqui (1720 m) at the boundary of warm and cool temperature regimes. The west- to southwest-facing orientation of slopes at Marking Corral may further render that site as warm and susceptible to cheatgrass recruitment post-fire (Koniak, 1985). The co-presence of black sagebrush (*A. nova* A. Nelson), Wyoming sagebrush (*A. tridentata* Nutt. ssp. *wyomingensis* Beetle & Young), and mountain big sagebrush

Table 6
Runoff, sediment, and flowpath variables by flow release rate for concentrated flow experiments ($9 m^2$) in control and burned areas at the Onaqui study site 9 yr after prescribed fire. Means within a row followed by a different lowercase letter are significantly different ($P < 0.05$).

Onaqui study site		Control		Burned	
Concentrated flow variable	Release rate ($L\,min^{-1}$)	Shrub-interspace zone	Tree zone	Shrub-interspace zone	Tree zone
Cumulative runoff (L)	15	20 a	1 a	17 a	21 a
	30	106 b	13 a	73 b	93 b
	45	245 c	89 a	175 b	219 bc
	Total	371 b	103 a	266 b	333 b
Cumulative sediment (g)[a]	15	135 b	8 a	150 b	224 b
	30	694 b	111 a	533 b	599 b
	45	2186 b	500 a	1300 b	1348 b
	Total	3015 b	619 a	1982 b	2170 b
Sediment concentration ($g\,L^{-1}$)[a]	15	6.6 b	3.4 a	7.2 b	6.2 b
	30	6.5 a	5.1 a	7.0 a	6.5 a
	45	8.3 a	5.5 a	7.4 a	6.2 a
Flow velocity ($m\,s^{-1}$)[a]	15	0.07 a	–	0.06 a	0.05 a
	30	0.10 bc	0.04 a	0.14 c	0.08 b
	45	0.12 b	0.07 a	0.15 b	0.10 ab
Flow path width (cm)[a]	15	93 a	62 a	56 a	95 a
	30	164 b	98 a	72 a	108 ab
	45	184 b	125 a	113 a	150 ab
Flow path depth (cm)[a]	15	0.58 a	0.80 a	0.62 a	0.52 a
	30	0.68 a	0.71 a	0.58 a	0.68 a
	45	0.99 a	0.79 a	0.88 a	0.80 a
Percent of plots with runoff ($n = 5$ per treatment × microsite combination)	15	100	40	80	60
	30	100	60	100	100
	45	100	100	100	100

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

(*A. tridentata* Nutt. ssp. *vaseyana* (Rydb.) Beetle) and the dominance of bluebunch wheatgrass post-fire 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 (Table 1). Chambers et al. (2014b) found that warm-dry to warm-moist sites dominated by Wyoming sagebrush with or without pinyon and juniper were less resistant to increases in cheatgrass post-fire than cool-moist sites with pinyon and juniper and mountain big sagebrush. Miller et al. (2014) reported 7-fold increases in non-native herbaceous cover, including cheatgrass, 3 yr after burning pinyon- and juniper-encroached sagebrush sites. The warmest sites exhibited the highest cover of cheatgrass before and after burning (Miller et al., 2014) and increases in cheatgrass and annual forb cover primarily occurred on woodlands burned in mid- to late-succession (mid to high tree dominance; Roundy et al., 2014a). Most sites had at least 5% perennial grass cover prior to treatment and tall perennial grass cover on burned treatments exceeded 13%, on average, across all sites after three growing seasons. Sites in the Miller et al. (2014) study span woodland encroachment Phases I-III and the cooler and wetter end of the warm-dry to warmer and drier end of the cool-moist soil temperature-moisture regimes. Bates et al. (2014) reported that prescribed fire increased cheatgrass cover over a 3–9 yr period on cool-moist Phase II and III juniper woodlands, but that cheatgrass cover was about 10 times greater for the Phase III (~30%) than Phase II (~3%) sites 9 yr after burning. Perennial grass cover was approximately 4-fold greater on Phase II sites pre-fire and 9 yr post-fire. Bates et al. (2014) attributed cheatgrass dominance on Phase III sites to limited pre-fire perennial herbaceous vegetation and potentially a poor perennial herbaceous seed bank. Phase II-III woodland-encroached sites in this study exhibited sufficient perennial grass cover (~6%) pre-fire to sustain perennial grasses post-fire (24–33% canopy cover), but those levels were clearly not enough to inhibit substantial increases in cheatgrass (Figs. 1, 2, and 4). Cheatgrass cover on a given site can increase dramatically during wet periods (West and Yorks, 2002; Bates et al., 2005, 2007), but precipitation was near normal at our sites in each of the 4 yr preceding the 2015 measurements in this study (Fig. 3). Therefore, precipitation alone does not explain substantial increases in cheatgrass measured 9 yr post-fire. Our results and those from the studies cited above suggest burning of Great Basin pinyon and juniper woodlands near the warm-dry and cool-moist soil temperature-moisture boundary can substantially increase cheatgrass cover with and without favorable perennial grass recruitment and that increases in cheatgrass along this boundary are most likely in late Phase II to Phase III of woodland encroachment.

Cheatgrass cover increases at both sites in this study are attributed to fire reduction of limited perennial vegetation and persistent sparse vegetation in tree zones the first few years post-fire. Total canopy cover was ≤6% in burned tree zones at both sites 1 yr post-fire and consisted of ~1% perennial grass and forb canopy cover (Pierson et al., 2015; Williams et al., 2016b). By year 2 post-fire, total canopy cover on burned tree zones was 13% at Marking Corral and 5% at Onaqui and was mostly annual forbs (Pierson et al., 2015). Vegetation in burned tree zones was dominated by cheatgrass by the 9th yr post-fire (Fig. 4). Other studies have documented delayed cheatgrass invasion of tree zones and areas under downed trees following tree removal due to limited herbaceous vegetation and/or creation of favorable conditions for cheatgrass establishment (Bates et al., 2005, 2007, 2011; Bates and Svejcar, 2009; Bates and Davies, 2016). Fire removal of perennial vegetation in tree zones at our study sites sustained large gaps between plant bases (> 300 m) 1 yr post-fire that were reduced, but still substantial (> 180 m) 2 yr post-fire (Pierson et al., 2015). Cheatgrass readily invades large gaps between vegetation on sagebrush rangelands and can capitalize on and outcompete native perennial herbaceous vegetation for seasonally available soil water and nutrients (Melgoza

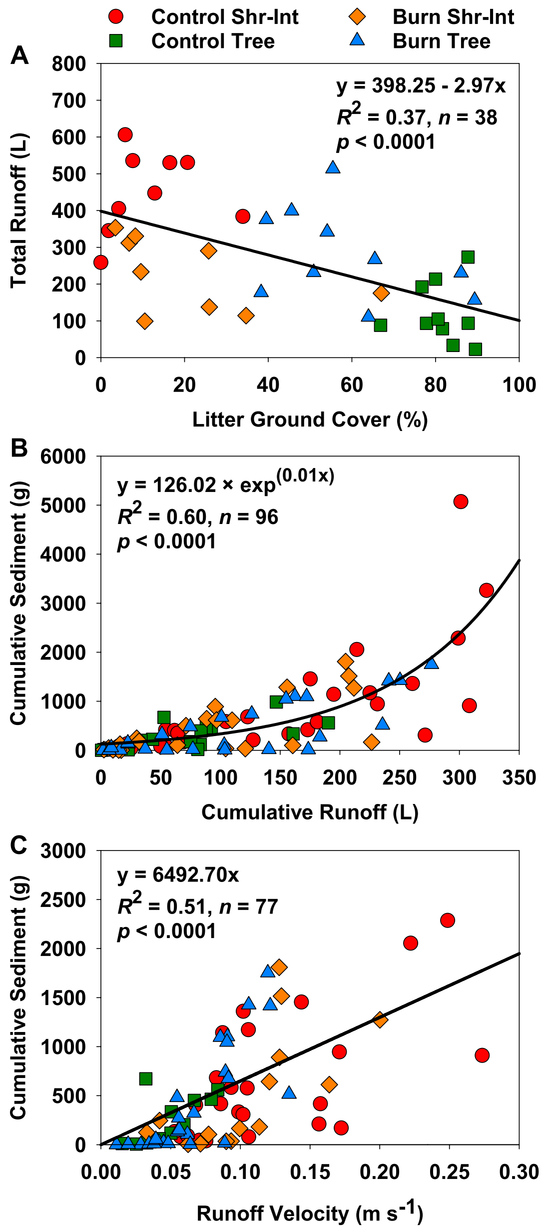

Fig. 9. Total runoff versus litter ground cover (A), cumulative sediment versus cumulative runoff (B), and cumulative sediment versus runoff velocity (C) as measured on shrub-interspace (Shr-Int) and tree (Tree) zone concentrated overland flow plots (9 m^2) in the control and burned areas at the Marking Corral and Onaqui study sites 9 yr after prescribed fire. Total runoff (A) data points represent the sum of cumulative runoff from 15, 30, and 45 L min^{-1} flow releases for a plot. Cumulative runoff and sediment data points (B and C) represent respective cumulative values for individual flow release rates for a plot.

et al., 1990; Arredondo et al., 1998; West and Yorks, 2002; Brooks et al., 2004; Chambers et al., 2007; Reisner et al., 2013; Rayburn et al., 2014; Rau et al., 2014). 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., 2013b; Roundy et al., 2014b). Seasonal soil water repellency of surface soils in tree zones may favor formation of cheatgrass islands through limiting germination and establishment of native perennial species (Madsen et al., 2011, 2012; Williams et al., 2014a). Soil water repellency was strong in surface soils of burned and unburned tree zones at sites in this study (Table 4; Pierson et al., 2010, 2014, 2015; Williams et al., 2016b), as commonly reported for pinyon and juniper woodlands in the Great Basin (Madsen et al., 2008, 2011; Pierson et al., 2013; Williams et al., 2014a; Fernelius et al., 2017; Zvirzdin et al., 2017). Tree zone soils can develop an isolated moist or wet soil layer near the soil surface that overlays a water repellent layer and subsequently dries out (Meeuwig, 1971; Doerr et al., 2000; Ritsema and Dekker, 2000; Lebron et al., 2007; Pierson et al., 2008b; Robinson et al., 2010; Madsen et al., 2011). Seedlings that establish in temporary wet conditions can be cut off from soil water subsequently stored beneath the water repellent layer and thereby undergo mortality (Madsen et al., 2011, 2012). Cheatgrass has competitive advantages over native perennial grass and forb seedlings in establishing and surviving on burned water-repellent tree zones. Cheatgrass germinates in the autumn, winter, and spring; has high germination rates; develops greater root strength over the winter wet season; and can more effectively establish with available soil moisture and nutrients during prolonged wet periods through earlier germination relative to native perennial species (Aguirre and Johnson, 1991, 1998; Arredondo et al., 1998; Chambers et al., 2007; Roundy et al., 2007; Hardegree et al., 2010, 2013; Mummey et al., 2016; Fernelius et al., 2017). We opine, given the line of evidence above, that the co-occurrence of favorable perennial vegetation and substantial cheatgrass coverage in this study (Fig. 4) was facilitated by: 1) enhanced production of perennial forbs and grasses within the intercanopy as soil water availability increased, and 2) cheatgrass invasion and infilling in burned tree zones void of perennial vegetation and with favorable soil climate and resources for cheatgrass establishment and survival.

Our results and those from early- to long-term post-treatment studies illustrate the complexities of evaluating and predicting vegetation outcomes of pinyon and juniper removal on Great Basin sagebrush steppe (Bates et al., 2005; Miller et al., 2013; Bates et al., 2014; Chambers et al., 2014a, 2014b; Miller et al., 2014; Roundy et al., 2014a, 2014b; Bates and Davies, 2016; Bates et al., 2017; Chambers et al., 2017). The prescribed fires at Marking Corral and Onaqui were successful in re-establishing perennial grasses important in the restoration of sagebrush steppe communities. However, the fires also substantially reduced cover of sagebrush species that are likewise key ecological components for these ecosystems. The 4537 plants per ha and 6% canopy coverage of sagebrush at Marking Corral 9 yr post-treatment suggest sagebrush, over the next 30–35 years, will potentially return to levels common for sagebrush communities near the boundary for cool-moist versus warm-moist soil temperature-moisture regimes (Harniss and Murray, 1973; Barney and Frischknecht, 1974; Ziegenhagen and Miller, 2009; Davies and Bates, 2010; Miller et al., 2013, 2014; Moffet et al., 2015). The low density of sagebrush (333 shrubs per ha) after 9 yr at Onaqui is concerning and suggests that the low pre-fire coverage (< 1%) of sagebrush at that site is limiting post-fire recruitment. Longer-term studies are needed to truly assess shrub recovery on both sites in this study, but slow sagebrush recruitment at Onaqui demonstrates the potential challenge of restoring sagebrush shrubs on sites in the later stages of woodland encroachment, with low initial sagebrush coverage and a limited seed bank (Miller et al., 2014; Roundy et al., 2014a; Bates et al., 2017). The fate of cheatgrass at Marking Corral and Onaqui is difficult to predict (Bates et al., 2011; Roundy et al., 2014a; Bates and Davies, 2016; Bates et al., 2017). Long-term successional studies generally indicate cheatgrass persistence on warm-dry sites and perennial dominance on cooler and wetter sites over time in the absence of fire (Barney and Frischknecht, 1974; Koniak and Everett, 1982; West and Yorks, 2002; Rew and Johnson, 2010; Miller et al., 2013; Bates et al., 2017). In an extensive review of literature, Miller et al. (2013) concluded current knowledge of cheatgrass long-term successional trends is limited, but that cheatgrass increases usually occur 5 to 12 yr post-fire followed by a gradual decline. Miller et al. (2013) noted, however, that cheatgrass can increase dramatically during wet years even on sites with high perennial grass cover. Overall, our results demonstrate that fire can effectively re-establish a

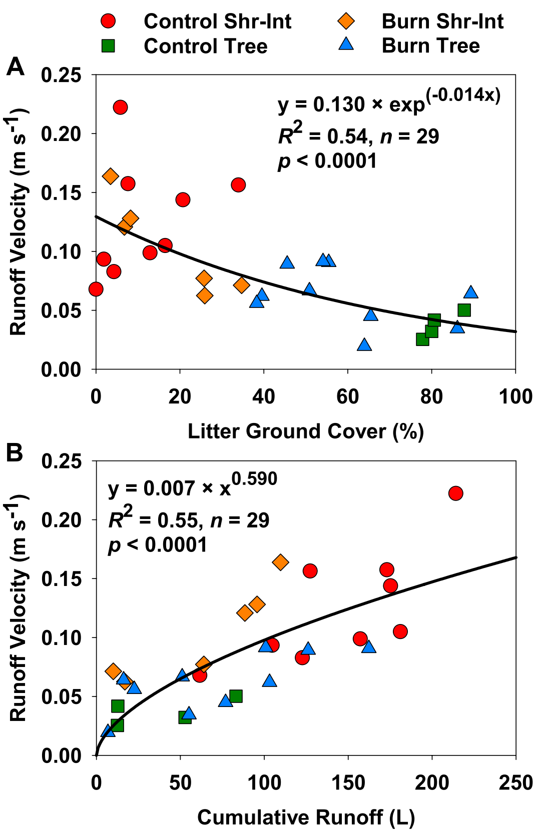

Fig. 10. Runoff velocity versus litter ground cover (A) and cumulative runoff (B) as measured on control and burned shrub-interspace (Shr-Int) and tree (Tree) zone concentrated overland flow plots (9 m^2) for the 30 L min^{-1} flow release rate at the Marking Corral and Onaqui study sites 9 yr after prescribed fire.

successional trajectory towards native sagebrush steppe vegetation dominance on late-succession woodlands near the warm-dry to cool-moist soil temperature-moisture threshold. Full re-establishment of a sagebrush steppe community structure at our study sites clearly requires more time. Given the high amounts of cheatgrass, both sites remain at risk of conversion to a cheatgrass-dominated ecological state following wildfire (Brooks et al., 2004; Chambers et al., 2007; Davies et al., 2012; Balch et al., 2013; Williams et al., 2016c, 2016d). However, resistance to cheatgrass dominance has likely increased at both sites through post-fire perennial grass recruitment and reduced likelihood of high intensity wildfire (reduced woody fuels) (Chambers et al., 2007; Condon et al., 2011; Reisner et al., 2013; Bates et al., 2014; Chambers et al., 2014a, 2014b; Miller et al., 2014; Roundy et al., 2014a, 2014b). Successional trajectories for sagebrush sites along the warm-dry to cool-moist soil temperature-moisture threshold suggest substantial pinyon and juniper infilling can begin within 10 to 20 yr after tree removal, rapidly increase within 50 to 70 yr after tree removal, and achieve tree dominance within approximately 70 to 125 yr in the absence of fire (Barney and Frischknecht, 1974; Koniak, 1985; Miller et al., 2005; Miller and Heyerdahl, 2008; Miller et al., 2013; Bristow et al., 2014; Bates et al., 2017). At 9 yr post-fire in this study, tree density for the 5–50 cm size class was 56 trees per ha at Marking Corral and 19 trees per ha Onaqui, and the density of trees > 0.5 m height was 158 trees per ha at Marking Corral and 17 trees per ha at Onaqui. These density values are approaching those reported for 30 to 40 yr post-fire in a multi-site study by Barney and Frischknecht (1974). Maintenance of the re-establishing sagebrush steppe vegetation structure will require follow up tree-removal as tree cover increases and affects understory vegetation production in the years ahead (Barney and Frischknecht, 1974; Tausch and Tueller, 1997; Bates et al., 2005; Miller et al., 2005; Roundy et al., 2014a; Bates et al., 2017).

4.2. Fire as a mechanism to improve ecohydrologic function on Great Basin woodlands

The primary driver of high runoff and erosion at both sites in this study for unburned conditions was extensive connectivity of runoff and erosion processes across intercanopy bare interspaces (Williams et al., 2014a, 2016b). Runoff and erosion on Great Basin rangelands increase with increasing bare ground and commonly increase exponentially where bare ground exceeds 50–60% (Pierson et al., 2008a, 2009, 2010; Al-Hamdan et al., 2013; Pierson et al., 2013; Williams et al., 2014a, 2014b; Pierson and Williams, 2016; Williams et al., 2016b). The exponential relation of runoff and erosion with bare ground for these ecosystems suggests that subtle changes in ground cover or bare ground near the 50% bare threshold can trigger conditions susceptible to high levels of runoff and erosion (Davenport et al., 1998). This threshold then, represents a transition from vegetation or ground cover (biotic) to runoff (abiotic) control of long-term soil loss. Bare ground at our study sites prior to burning was 50% to 60% (Table 1). Approximately 70% of the area at the woodlands was intercanopy and approximately 70 to 90% of the intercanopy area was bare interspace (bare soil and rock covered) (Fig. 7; Pierson et al., 2010). High amounts of runoff and sediment from rainsplash and sheetflow were delivered from bare interspaces (Table 4) within control shrub-interspace zones at both sites in this study and provided ample runoff and sediment sources for delivery across spatial scales (Pierson et al., 2010, 2013; Williams et al., 2014a, 2014b, 2016b). Pierson et al. (2010) observed formation of concentrated flow paths with high flow velocity during large plot (13 m^2) rainfall simulation experiments in shrub-interspace zones at Marking Corral and Onaqui before the prescribed fires. Pierson et al. (2010) attributed increasing erosion across small plot to large plot scales during rainfall simulations at the study sites to the observed high velocity concentrated overland flow within the bare intercanopy. Nouwakpo et al. (2020) measured comparable shrub-interspace zone runoff and erosion rates as Pierson et al. (2010) for unburned conditions at the sites using similar methodologies in conjunction with this study. We measured high overland flow velocity within unburned shrub-interspaces at both sites 9 yr post-fire (Tables 5 and 6), consistent with those reported by Pierson et al. (2010, 2015) for unburned shrub-interspaces immediately prior to and 1 yr after the fires. The amount and velocity of overland flow releases was strongly influenced by the percentage cover of litter at the ground surface (inverse of bare ground) (Figs., 9A and 10A), and the highest amounts of concentrated flow runoff and erosion were measured on the mostly bare unburned shrub-interspace plots (Fig. 9). Sediment delivery from concentrated flow plots was primarily driven by the amount of runoff and the velocity of the flow (Fig. 9B and C), implicating reduction of overland runoff generation as a key factor in reducing erosion from these systems. The amount and velocity of runoff, and therefore erosion, were low for the well-protected ground surface underneath unburned trees at both sites (Tables 4–6). The extensive bare ground with high rock content at the soil surface within the intercanopy at our study sites is indicative of substantial ongoing and long-term soil loss. The higher intercanopy rock cover at Onaqui (52%) relative to Marking Corral (18%) and generally greater sediment-to-runoff ratios, sediment concentrations, and erosion (Tables 4–6) for Onaqui indicate that site may be more vulnerable to long-term loss of critical surface soil. Results from this and our companion studies (Pierson et al., 2010, 2013; Williams et al., 2014a; Pierson et al., 2015; Nouwakpo et al., 2020) affirm conceptual models that suggest Great Basin sagebrush rangelands near the warm-dry and cool-moist soil temperature-moisture boundary can become highly erodible in the later stages of woodland encroachment and that these ecosystems are susceptible to transitioning to a degraded eroded ecological state without alteration of bare conditions within the intercanopy (Miller et al., 2005; Petersen et al., 2009; Chambers et al., 2014b; Williams et al., 2016c, 2016d).

Nine years post-fire plot-scale hydrologic and erosion responses at the Marking Corral and Onaqui sites reflect the combination of degraded pre-fire vegetation conditions, the associated initial fire effects, delayed litter accumulation, and inherent site attributes (Pierson et al., 2010, 2014, 2015; Williams et al., 2016b). Site physical (e.g., topography, slope angle, soil type) and biological (e.g, vegetation, ground cover, soil organic material) characteristics and the degree to which fire modifies these attributes dictate storm event and annual runoff and erosion rates the first few years post-fire (Pierson et al., 2011; Williams et al., 2014b; Pierson and Williams, 2016). Event runoff and erosion rates commonly increase by factors of 2 to 40 at the small plot scale and can increase by a factor of 100 over the patch scale following burning on Great Basin rangelands (Pierson and Williams, 2016). Fire-induced increases in runoff and erosion are typically greater for areas that were well-vegetated or litter covered than areas that were bare pre-fire (Pierson and Williams, 2016). Post-fire runoff rates on Great Basin rangelands often return to pre-fire levels within several years, but erosion may remain elevated for a longer period (Pierson et al., 2002, 2008a, 2009, 2011, 2013, 2014; Williams et al., 2014b, 2016a). Burning at the sites in this study initially increased bare ground by at least factors of 2 to 4 underneath individual trees and shrubs at the small-plot scale (Williams et al., 2016b) and across tree zones at the patch scale (Pierson et al., 2015; Williams et al., 2016b). Burning had more limited initial impact on bare ground on shrub-interspace zones due to the 64% to 84% average bare ground on these plots pre-fire (Pierson et al., 2015; Williams et al., 2016b). Initial reductions in infiltration and increases in runoff from small-plot rainfall simulations were restricted to tree and shrub coppices at the sites (Pierson et al., 2014). Patch-scale runoff from large-plot (13 m^2) rainfall simulations and concentrated flow experiments increased in tree zones at Onaqui solely due to a 3-fold litter reduction in tree zones at that site (Pierson et al., 2015; Williams et al., 2016b). Increases in small plot erosion 1 yr post-fire varied across sites, but were always confined to tree coppice or shrub coppice microsites with increased bare ground (Pierson et al., 2014). Initial post-fire erosion increases for large-plot rainfall simulations were recorded on shrub-interspace zones at Marking Corral and tree zones at Onaqui (Williams et al., 2016b). The elevated erosion on shrub-interspace plots at Marking Corral was attributed to substantial runoff and an increase in bare soil, and therefore sediment availability, with reduced litter cover on those plots (Williams et al., 2016b). Erosion from large-plot rainfall simulations in shrub-interspace zones at Onaqui did not increase even with elevated bare soil due to the already high erosion rates for unburned conditions at that site (Pierson et al., 2010; Williams et al., 2016b). Low runoff limited erosion the 1st yr post-fire on tree zones at Marking Corral (Williams et al., 2016b). The increased sediment delivery in tree zones at Onaqui was attributed to reduced litter, ample ash and sediment availability, and increased runoff post-fire (Williams et al., 2016b). Burning increased concentrated flow erosion in tree zones at both sites 1 yr post-fire, but first year increases in concentrated flow erosion on shrub-interspaces were mainly restricted to the more productive Marking Corral site (Pierson et al., 2015). Overall, the first year hydrologic and erosion responses reflect the initial low cover on interspaces at both sites, typical runoff/erosion increases for vegetated microsites, and the initial more limited cover and higher soil erodibility at Onaqui (Pierson and Williams, 2016). Fire effects on runoff and erosion for tree coppice, shrub coppice, and tree zones persisted the 2nd yr post-fire, reflecting the initial greater fire impact on locations that were vegetated or litter covered pre-fire. Nine years post-fire, the more degraded initial intercanopy conditions (lower shrub and grass cover) at Onaqui are evident by the greater percentages of basal gaps in 25–50 cm to 101–200 cm size classes for that site relative to Marking Corral (Fig. 5). The persistence of low litter cover and prolonged bare ground > 30% in tree zones at Onaqui may reflect initial hotter fire conditions (as evident by a 60% reduction in litter and high post-fire ash cover (Williams et al., 2016b)) or a general lower vegetative productivity relative to Marking Corral (Figs. 6 and 7). Post-fire recruitment of perennial herbaceous cover at both sites has improved infiltration and reduced erosion for interspaces (Table 4), the most dominant microsite pre-fire, but the improved hydrologic function is partially masked at Onaqui for concentrated flow runoff and erosion processes in shrub-interspace zones due to the slow litter recruitment, persistent high bare ground, and high soil erodibility (Fig. 7B, Table 6; Pierson et al., 2010). Runoff and erosion from concentrated flow experiments in tree zones 9 yr post-fire were similar for burned and unburned conditions at Marking Corral (Table 5) due to litter recovery (Fig. 7A) and low inherent erodibility at that site (Pierson et al., 2010). In contrast, slow litter recovery (Fig. 7B) and inherent high erodibility of soils at Onaqui (Pierson et al., 2010) resulted in an increase in tree zone concentrated flow runoff and erosion 9 yr post-fire (Table 6).

The results spanning early- to mid-succession post-fire support pervious assertions by the authors (Pierson et al., 2013; Williams et al., 2014a) that fire may act to reverse the abiotic threshold for soil loss on late-succession woodland-encroached sagebrush rangelands. Pierson et al. (2013) and Williams et al. (2014a) used the same rainfall simulation and concentrated flow methodologies to evaluate short-term (1 to 2 yr) impacts of wildfire on vegetation and runoff and erosion processes at a mountain big sagebrush site with cool-moist soil temperature-moisture regimes. Prior the fire, the site was progressing from Phase II to Phase III encroachment by western juniper and had crossed the ecohydrologic threshold of biotic-controlled soil retention to abiotic-controlled soil loss (Williams et al., 2014a). Bare ground measured in unburned areas at the hillslope scale was approximately 42%, the intercanopy (~75% of area) was mostly bare interspace (75 to 90% bare ground), and the ground surface underneath and adjacent to unburned trees was well protected with > 70% coverage of tree litter (Williams et al., 2014a). General pre-fire trends of low runoff and erosion from unburned areas underneath trees and shrubs and high runoff and sediment yield from unburned interspaces and shrub-interspace zones in the study are consistent with this study (Tables 4–6). Burning increased bare ground to > 75% at the hillslope scale and to near 90% across all plots at small plot scale (Pierson et al., 2013). Burning increased runoff of applied rainfall and overland flow in tree coppices and tree zones by 2- to 8-fold and increased erosion by approximately 7- to > 30-fold on these microsites. The fire had no initial effect on runoff from shrub coppices, interspaces, and shrub-interspace zone plots, but increased erosion on shrub coppice and interspace plots by 4- to > 20-fold (Williams et al., 2014a). Fire-induced increases in runoff and erosion on tree coppices and tree zones persisted 2 yr post-fire, but runoff in the 2nd yr was lower than pre-fire levels on interspace plots (Pierson et al., 2013). Improved infiltration in burned interspaces had no significant effect on concentrated flow runoff in burned shrub-interspace zones in the 2nd yr, but a reduction of flow path incision on burned shrub-interspaces in the 2nd yr indicated a fire-induced effect on flow path energy (Williams et al., 2014a). Williams et al. (2014a) and Pierson et al. (2013) reported that fire-induced increases in herbaceous vegetation in interspaces reduced the distance between plant bases. The authors suggested that the more uniform distribution of vegetation on burned shrub-interspace zones in the 2nd yr reduced the shear stress applied to soil by concentrated flow and thereby limited flow path incision and erosion (Williams et al., 2014a). The short-term nature of the Pierson et al. (2013) and Williams et al. (2014a) studies precludes determination of whether fire can reverse a site from abiotic-driven soil loss to biotic-controlled soil retention, as acknowledged by the authors. Results from the Marking Corral site 9 yr post-fire in this study indicate burning of late-succession encroachment woodlands can reverse the abiotic control on long-term soil loss. Fire-induced enhancement of herbaceous cover in interspaces (Table 2; ~60% of site area) and litter recruitment in the intercanopy (Fig. 7A; > 70% of site area) improved infiltration (Table 4, Fig. 8A) and caused overland flow from concentrated flow releases to spread out into wide flow paths with reduced runoff, flow velocity, and sediment delivery relative to unburned conditions (Table 5). Although burning reduced small-plot scale infiltration of tree coppices at the site (Table 4), any negative hydrologic impact of burning tree coppices (< 30% of site area) at the site was masked in tree zones by ample and evenly distributed herbaceous vegetation and litter ground cover 9 yr post-fire (Figs. 4A–B, 6A and 7A, Table 5). Persistent high sediment concentrations from rainfall simulations and concentrated flow experiments in burned areas at Onaqui (Tables 4 and 6) underscore the influence of pre-fire site conditions and inherent site attributes on treatment responses. Burning enhanced herbaceous cover in interspaces (> 70% of site area) at Onaqui, but similar runoff rates for burned interspaces as at Marking Corral delivered approximately 6-fold more sediment (Table 4). As previously noted, we primarily attribute the site differences in erosion for burned interspaces to the inherently higher soil erodibility at the Onaqui site (Pierson et al., 2010). Poor litter recruitment post-fire in the initially bare shrub-interspaces (Figs. 2C and 7B) at Onaqui limited infiltration of concentrated overland flow releases and resulted in no change in runoff and erosion from simulated concentrated flow in shrub-interspaces at that site 9 yr post-fire (Table 6). Delayed litter cover recruitment in tree zones post-fire at Onaqui (Fig. 7B) offered limited resistance to released overland flow, resulting in elevated runoff and erosion from concentrated flow releases (Table 6). The concentrated flow experiments are meant to quantify concentrated flow runoff and erosion rates if concentrated flow paths were to form under rainfall. Concentration of overland flow for these systems is dependent on runoff generation in bare areas at fine spatial scales (i.e., small-plot scale) and the accumulation and connectivity of that runoff over the patch scale (Pierson et al., 2010, 2013; Williams et al., 2014a; Pierson et al., 2015; Williams et al., 2016b, 2016c). Improved infiltration rates in interspaces 9 yr post-fire at Onaqui (Table 4; Fig. 8B) likely limit the amount of water available for broad-scale runoff concentration during natural rainfall in burned shrub-interspaces at the patch scale relative to unburned conditions (see Nouwakpo et al., 2020; Williams et al., 2014a, 2016b). Likewise, infiltration rates from small-plot rainfall simulations on burned tree coppices at Onaqui (Table 4; Fig. 8B) suggest limited runoff availability for overland flow delivery to and concentration in burned tree zones at that site given the near 50% litter ground cover (see Nouwakpo et al., 2020; Pierson et al., 2010; Williams et al., 2016b). Nevertheless, our results indicate that, where runoff does accumulate at Onaqui, erosion rates are likely to be high due to persistent bare conditions and inherent high soil erodibility at that site (Pierson et al., 2010). Overall, burning re-established "resource-conserving" vegetation and hydrologic and erosion function at Marking Corral and improved hydrologic and erosion function at Onaqui, but the persistence of high runoff and erosion for simulated concentrated flow processes at Onaqui demonstrates the time period required to fully re-establish "resource conserving" conditions is strongly influenced by pre-fire conditions and inherent site characteristics.

The diverging site ecohydrologic responses in this study underscore the importance of considering mechanical fire surrogate treatments (e.g., tree cutting, shredding/mastication, or chaining) for re-establishment of sagebrush steppe vegetation structure and hydrologic function on woodland encroached sites. Tree removal by prescribed fires in this study effectively increased herbaceous vegetation (Tables 1 and 2, Fig. 6) and improved hydrologic function (Table 4, Fig. 8; see also Nouwakpo et al., 2020) at both sites over a 9-yr period. However, the fires also initially consumed limited sagebrush and ground cover and increased soil erosion rates (Pierson et al., 2014, 2015; Williams et al., 2016b). Sagebrush recovery has been slow at Onaqui due to low initial sagebrush cover, and interspace erosion rates are generally reduced (Tables 4 and 6, Fig. 8D), but remain high for that site 9 yr post-fire. As with prescribed burning, mechanical treatments can effectively increase plant available soil water (Bates et al., 2000, 2002; Young et al., 2013a, 2013b; Roundy et al., 2014b) and thereby improve sagebrush and perennial herbaceous cover (Bates et al., 2000; Miller et al., 2005; Bates et al., 2005, 2007; Bybee et al., 2016; Bates et al., 2017). Most mechanical treatments have limited negative impact on existing understory vegetation, ground cover, soils, and runoff and erosion relative to that of fire (Miller et al., 2013; Chambers et al., 2014b; Miller et al., 2014; Roundy et al., 2014a), and recruitment of cheatgrass is typically lower following mechanical treatments than prescribed fire on sites with a cool soil temperature regime (Miller et al., 2013, 2014). Pierson et al. (2013, 2014, 2015) suggested that hydrologic function following tree cutting and shredding treatments improves over the first few years post-treatment if intercanopy vegetation cover increases and that initial (1st yr) reductions in runoff and erosion are most likely when tree debris is in good contact with the ground surface throughout the intercanopy. Pierson et al. (2013, 2015) reported minimal initial change in runoff and erosion through the placement of downed trees in intercanopy areas. The authors found that runoff generated in residual bare interspaces on recently-cut woodlands tends to route through downed trees as concentrated flow paths along narrow breaks in contact of tree debris with the soil surface. Cline et al. (2010) and Pierson et al. (2014) reported that application of shredded tree debris (mulch) to bare interspaces (0.5 m^2) improved infiltration and reduced erosion for those microsites. In a companion study, Pierson et al. (2015) found that tree shredding reduced runoff and erosion from shrub-interspace zones 1 yr post-treatment due to herbaceous cover recruitment rather than increased litter cover in the form of mulch. Distribution of tree debris in the study was restricted to tree zones and therefore had minimal impact on runoff and erosion from shrub-interspace zones. 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 juniper on a sagebrush site in later stages of woodland encroachment. Runoff and erosion from rainfall simulations in cut treatment areas were negligible. Rainfall simulations in the primarily bare intercanopy (84% bare ground) in untreated areas generated runoff and erosion levels 14- to > 85-fold greater, respectively, than in treated areas. Roundy et al. (2017) reported that pinyon and juniper removal by chaining combined with a seeding treatment increased intercanopy vegetation cover from 5% to 24% 1 yr post-treatment and to > 40% 3 yr post-treatment. Litter cover in the study was near 20% in untreated areas and exceeded 50% in treated areas within 3 yr. Intercanopy runoff and sediment delivery under natural rainfall on 10 m^2 plots were reduced from 44 L to 9 L and from 558 g to 54 g (averaged over 5 yr) following the tree removal and seeding. Collectively, these studies demonstrate that mechanical treatments can re-establish sagebrush steppe vegetation and improve ecohydrologic function on woodland encroached sagebrush sites over time without initial increases in runoff and erosion associated with burning. Mechanical alternatives may be preferred over prescribed fire on sites like Onaqui near the warm to cool soil temperature threshold and with high inherent soil erodibility, limited sagebrush cover, and a degraded intercanopy (Bates et al., 2005; Miller et al., 2013; Chambers et al., 2014b; Miller et al., 2014; Roundy et al., 2014a). Seeding may be necessary to limit cheatgrass cover and recruit perennial vegetation following any treatment where herbaceous cover density is below 1–3 perennial grass and 5 perennial forb plants per m^2 (Bates et al., 2005; Bates and Svejcar, 2009; Bates et al., 2014; Davies et al., 2014; Roundy et al., 2014a; Bybee et al., 2016). Mechanical treatments often leave numerous pinyon and juniper seedlings and juvenile trees and have minimal impact on the soil seed bank. Therefore, follow-up tree removal is usually necessary to prevent pinyon and juniper re-colonization 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). Some studies have reported favorable perennial vegetation responses and tree mortality with combined cutting and cool- to cold-season burning (Bates and Svejcar, 2009; Bates et al., 2011, 2014; Bates and Davies, 2016). Lastly, fire surrogate treatments can also leave substantial downed woody fuels that contribute to wildfire risk (Young et al., 2015).

4.3. Comparable responses of expansion woodlands elsewhere in North America

Pinyon and juniper woodlands in North American span beyond the Great Basin across a broad ecological domain of the southwestern US including the Colorado Plateau, the southern Rocky Mountain, and the Desert Southwest regions (Romme et al., 2009; Jacobs, 2011; Floyd and Romme, 2012; McAuliffe et al., 2014). Tree removal treatments in woodlands of the southwestern US have focused primarily on fuel reduction and re-establishment of shrub- or herbaceous-dominated vegetation (Huffman et al., 2009; Ross et al., 2012; Huffman et al., 2013; Redmond et al., 2013; Meddens et al., 2016; Coop et al., 2017; Havrilla et al., 2017; Huffman et al., 2017). Redmond et al. (2013) examined the long-term (20–40 yr) effects of seeding with tree removal by chaining at 17 sites on the Colorado Plateau dominated by two-needle piñon (*P. edulis* Engel.) and Utah juniper conifers. The results across all sites indicate the treatments effectively reduced tree cover and increased sagebrush and perennial herbaceous cover (mainly of seeded crested wheatgrass (*Agropyron cristatum* [L.] Gaertn.)) by factors of 3 and 4, respectively. Redmond et al. (2013) found the treatments had no effect on cheatgrass cover (≤3%). The authors attributed limited cheatgrass cover post-treatment to low pre-treatment cheatgrass cover, successful establishment of seeded crested wheatgrass, and the generally low invasibility for cheatgrass in the study domain. Chaining increased bare soil from ~35% to ~50% and amplified the potential for soil loss by wind and water erosion processes (Redmond et al., 2013). The authors attributed persistent low tree cover (~16%) > 40 yr post-treatment to the slow regeneration rate of two-needle piñon and Utah juniper for the region. Havrilla et al. (2017) found that removal of two-needle piñon and Utah juniper by shredding, cutting with broadcast burning, and cutting with pile burning treatments and seeding increased perennial grass and sedge cover by 7- to 14- fold 2 yr post-treatment and by 15- to 18-fold 6 yr post-treatment relative to seeded controls. Cheatgrass cover was minimal (< 1%) prior to tree removal, but was 2% to 26% across all tree-removal treatments 6 yr post-treatment. The authors found bare patches created by pile burning were highly susceptible to cheatgrass invasion. An additional treatment involving tree shredding without seeding produced nearly 2- to 16-fold more cheatgrass cover than all other treatments 6 yr after tree removal, indicating that seeding may have limited cheatgrass potential in the other treatments (Havrilla et al., 2017). Coop et al. (2017) evaluated 1–11 yr treatment effects of shredding pinyon and juniper at 192 sites spanning warm-wet to cool-dry climate conditions on the Colorado Plateau. Treatments generally increased perennial herbaceous vegetation and cheatgrass cover and increases in bare ground and cheatgrass (although low) were greater on warmer study sites. The studies noted above and others from the southwestern US (Jacobs and Gatewood, 1999; Ross et al., 2012; Jacobs, 2015; Meddens et al., 2016) are generally consistent with woodland literature from the Great Basin (Bates et al., 2014; Chambers et al., 2014b; Miller et al., 2014; Roundy et al., 2014a) indicating pinyon and juniper removal can increase perennial herbaceous vegetation. As also in the Great Basin, tree removal can stimulate cover of cheatgrass on pinyon and juniper woodlands in the Colorado Plateau, southern Rocky Mountain, and Desert Southwest regions (Owen et al., 2009; Ross et al., 2012; Huffman et al., 2013; Coop et al., 2017; Havrilla et al., 2017) and seeding may be necessary to increase perennial vegetation and limit cheatgrass recruitment on more degraded sites or after burning (Stoddard et al., 2008; Redmond et al., 2013; Havrilla et al., 2017; Huffman et al., 2017).

Hydrology and erosion studies indicate runoff and erosion on pinyon and juniper woodlands in the southwestern US are driven mainly by high intensity convective storms and that the magnitude of runoff and erosion events from these storms is strongly regulated by rainfall intensity, the distribution of vegetation and ground cover, and inherent soil properties (Wilcox, 1994; Wilcox et al., 1966a, 1996b; Davenport et al., 1998; Reid et al., 1999; Hastings et al., 2003; Wilcox et al., 2003a). As with Great Basin woodlands, hydrologic function and erosion rates on woodlands in the southwestern US differ for areas near or underneath tree canopies and with surface cover in the intercanopy (Wilcox and Breshears, 1995; Reid et al., 1999; Wilcox et al., 2003a, 2003b). Runoff and erosion rates are low for "resource conserving" woodlands in the southwestern US due to dissipation of isolated runoff and deposition of sediment in well distributed vegetative or litter covered patches (Wilcox et al., 2003a; Ludwig et al., 2005). Degraded woodlands develop well-connected intercanopy bare patches with high runoff and erosion rates, potentially resulting in increased soil loss with increasing spatial scale on sites with highly erodible soils (Wilcox et al., 1996a, 1996b; Davenport et al., 1998; Wilcox et al., 2003a). Watershed and hillslope restoration treatments on woodlands throughout the Colorado Plateau, southern Rocky Mountains, and Desert Southwest regions primarily target intercanopy vegetation and litter recruitment, bare ground reduction, improved infiltration, and soil retention/stability (Brockway et al., 2002; Hastings et al., 2003; Stoddard et al., 2008; Owen et al., 2009; Jacobs, 2015; Meddens et al., 2016). Brockway et al. (2002) found cutting of two-needle piñon and one-seed juniper (*J. monosperma* [Engelm.] Sarg.) on a Desert Southwest woodland increased herbaceous and litter cover and reduced bare soil regardless of retention or removal of slash, but that the increased cover over a nearly 2 yr period post-treatment did not affect the inherently low soil loss rate for the site. Hastings et al. (2003) found that cutting < 20-cm diameter 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 over two rainy seasons was one to three orders of magnitude more for untreated than treated micro-watersheds (300–1100 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. The study further reported that erosion was 2- to 100-fold greater for micro-watersheds with non-pumice soils versus those with pumice soils. Stoddard et al. (2008) investigated the effects of slash and seeding treatments on interspace vegetation and soil movement over a 2 yr period at two degraded two-needle piñon and Utah juniper woodlands in the Desert Southwest. Applying slash and seeding treatments to 1 m^2 plots reduced interspace bare ground (from ~99% to ~10%) and increased grass cover, although grass cover remained low (< 5%) 2 yr post-treatment (Stoddard et al., 2008). Scattering slash in interspaces reduced soil movement by factors of 2 to 3 across the two sites 2 yr post-treatment regardless of whether the plots were initially seeded or not (Stoddard et al., 2008). Owen et al. (2009) evaluated the effects of tree shredding and tree cutting with pile burning on vegetation and soil stability over a 3.5 yr period at a two-needle piñon and Utah juniper woodland in the southwestern US. Tree shredding increased grass and litter cover by 4- and 2-fold and reduced bare ground by nearly 30-fold (from 42% to 1.5%) over the study period. Pile burning of cut trees reduced total plant cover from 26% to 4% and increased bare ground to near 80% over the study period. Cheatgrass cover increased in both tree removal treatments during the study, and, similar to the study by Havrilla et al. (2017), formed dense cheatgrass rings in areas around burn piles (20% of site area). Surface soil stability was unaltered by the shredding treatment, but was substantially reduced by the cut and pile burn treatment 0.5 yr and 2.5 yr post-treatment. Overall, experimental results from the studies above for pinyon and juniper treatments in the southwestern US exhibit similar trends as in the Great Basin: 1) mid- to long-term vegetation recruitment is strongly associated with pre-treatment conditions (Miller et al., 2013; Bates et al., 2014; Chambers et al., 2014b; Miller et al., 2014; Roundy et al., 2014a; Bates et al., 2017), 2) seeding may be necessary to combat susceptibility to invasive weeds, particularly on warmer and drier sites (Miller et al., 2013; Chambers et al., 2014a; Davies et al., 2014; Roundy et al., 2014a; Bybee et al., 2016; Chambers et al., 2017), 3) reduction of runoff and erosion occurs mainly through increased vegetation and ground cover in bare intercanopy areas (Pierson et al., 2007; Cline et al., 2010; Pierson et al., 2013, 2014, 2015; Williams et al., 2014a, 2016b, 2016c), and 4) the magnitude of erosion reduction is influenced by inherent soil properties (e.g., erodibility) (Pierson et al., 2014, 2015). The ecological intricacies of woodlands and shrublands worldwide is outside the scope of this study, but the concept of re-establishing vegetation structure to improve hydrologic function and reduce erosion is broadly applicable to patchy vegetated landscapes around the World (Bergkamp et al., 1996; Van de Koppel et al., 1997; Ludwig and Tongway, 1995; Ludwig et al., 1997; Cammeraat and Imeson, 1999; Scanlan, 2002; Calvo-Cases et al., 2003; Ludwig et al., 2005, 2007; Mayor et al., 2009; Puigdefábregas, 2005; Eldridge et al., 2015; Martínez-Valderrama et al., 2016; Vandandorj et al., 2017).

5. Summary and conclusions

Experimental results in this study suggest pinyon and juniper removal by prescribed fire can effectively re-establish a successional trajectory towards sagebrush steppe vegetation structure and thereby improve ecohydrologic function on woodland-encroached sagebrush rangelands in the Great Basin. 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 degraded woodland-encroached sagebrush sites. Prescribed burning substantially increased native perennial herbaceous vegetation in the interspaces throughout the intercanopy (> 70% of area) at both sites over nine growing seasons post-fire. Enhanced perennial herbaceous vegetation and litter recruitment post-fire improved intercanopy ecohydrologic function and reduced erosion at one site. Sparser pre-fire vegetation and ground cover conditions and inherently high soil erodibility limited improvements in ground cover and ecohydrologic function at a second, more degraded site. Fire consumed sagebrush at both sites, but sagebrush density at the less degraded site was consistent with favorable trajectories towards long-term sagebrush recovery. Low pre-fire sagebrush cover at the more degraded site limited post-fire sagebrush recovery. Burning of trees and the perennial herbaceous vegetation underneath them increased site invasibility to fire-prone cheatgrass. Cheatgrass increased substantially at both sites in tree zones even though coverage of the species was < 1% pre-fire. Although tree zones represent < 30% of the area at both sites, substantial increases in cheatgrass have enhanced the risk of wildfire and the potential for cheatgrass dominance following burning. Cheatgrass is likely to decline at both sites over time with improved perennial grass cover in the absence of fire. However, cheatgrass increases following burning in tree zones further substantiates that fire can be a risky endeavor for tree removal on pinyon- and juniper-encroached sagebrush sites on the warmer end of the cool soil-temperature regime in the Great Basin. Diverging vegetation, ground cover, and ecohydrologic function across our study sites post-fire illustrate the complexity of predicting ecological responses to tree removal and demonstrate that vegetation and hydrologic responses to tree removal in the Great Basin are strongly affected by pre-fire site physical (geography, soils) and biological (vegetation) attributes at the time of treatment. Overall, our results show that prescribed burning can re-establish perennial herbaceous vegetation, sagebrush recruitment, and biotic regulation of hydrologic and erosion processes on woodland-encroached sagebrush steppe. However, the results should be interpreted with caution, as both sites remain at risk to cheatgrass dominance if subjected to wildfire. Furthermore, poor sagebrush recovery, delayed litter recruitment, and persistent high erosion at the more degraded site suggest not all sites are good candidates for prescribed fire treatments. Our results in context with the literature indicate mechanical tree removal may be more appropriate on sites with limited sagebrush and perennial herbaceous vegetation and that seeding may be necessary to improve post-fire establishment of sagebrush steppe vegetation structure and ecohydrologic function under these conditions. Lastly, vegetation trajectories and runoff and erosion responses in this study are not directly applicable outside of the Great Basin, but the concept of re-establishing ecosystem vegetation structure to improve ecohydrologic function following woody-plant encroachment is broadly applicable to sparsely vegetated landscapes around the Globe. In particular, our results from the Great Basin are strikingly similar to those for woodland tree-removal studies in the southwestern US.

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