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ARTICLE

Special Feature: Sagebrush Steppe Treatment Evaluation Project

Assessing runoff and erosion on woodland-encroached sagebrush steppe using the Rangeland Hydrology and Erosion Model

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Abstract

The transition of sagebrush-dominated (*Artemisia* spp.) shrublands to pinyon (*Pinus* spp.) and juniper (*Juniperus* spp.) woodlands markedly alters resource-conserving vegetation structure typical of these landscapes. Land managers and scientists in the western United States need knowledge and predictive tools for assessment and effective targeting of tree-removal treatments to conserve or restore sagebrush vegetation and associated hydrologic function. This study developed modeling approaches to quantify the hydrologic vulnerability and erosion potential of sagebrush rangelands in the later stages of woodland encroachment and in response to commonly applied tree-removal treatments. Using experimental data from multiple sites in the Great Basin Region, USA, and process-based knowledge from decade-long vegetation and rainfall simulation studies at those sites, we (1) assessed the capability of the Rangeland Hydrology and Erosion Model (RHEM) to accurately predict patch-scale (12 m^2) measured runoff and erosion from tree canopy and intercanopy hydrologic functional units in untreated and burned woodlands 9 years postfire, and (2) developed and evaluated multiple RHEM approaches/frameworks to model aggregated effects of tree canopy and intercanopy areas on patch- and hillslope-scale (50 m length) runoff and erosion processes in untreated and treated (burned, cut, and masticated) woodlands. The RHEM accurately predicted measured runoff and sediment yield from patch-scale rainfall simulations as partitioned on untreated and treated tree canopy and intercanopy areas and effectively parameterized the dominant controls on runoff and erosion process in woodlands. With few exceptions, evaluated hillslope-scale RHEM frameworks similarly predicted reduced hydrologic vulnerability and erosion potential for conditions 9 years following tree removal by burning, cutting, and mastication treatments. Regressions of RHEM-predicted hillslope runoff, sediment, and hydraulic/erosion parameters with bare ground and

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ground cover attributes indicate all RHEM frameworks effectively represented the dominant controls on hydrologic and erosion processes for rangelands and woodlands. The results provide RHEM frameworks and recommendations for assessing hydrologic vulnerability and erosion potential on woodland-encroached sites and predicting the effectiveness of tree removal to reestablish a water and soil resource-conserving vegetation structure on sagebrush rangelands. We anticipate our RHEM or similar modeling approaches may be applicable to analogous water-limited landscapes elsewhere subject to woody plant encroachment.

KEYWORDS
cutting, fire, Great Basin, hydrologic recovery, infiltration, islands of fertility, juniper, mastication, pinyon, prescribed fire, restoration, runoff, sagebrush, Special Feature: Sagebrush Steppe Treatment Evaluation Project

INTRODUCTION

The retention and restoration of sagebrush-steppe (*Artemisia* spp.) vegetation are priority land management objectives in the western United States (Chambers et al., 2017; Davies et al., 2011; Suring et al., 2005). Sagebrush rangelands provide a host of ecosystem services, including critical wildlife habitat, cultural resources, forage for wild and domestic ungulates, and retention of water and soil (Connelly et al., 2011; Davies et al., 2011; Kormos et al., 2017; Pierson Jr et al., 1994). These services at a given site decline with reductions in density, productivity, and distribution of sagebrush and associated perennial bunchgrasses and forbs (Knick et al., 2003; Miller et al., 2011; Pierson et al., 2007; Williams, Pierson, Al-Hamdan, et al., 2014; Williams, Pierson, Nouwakpo, et al., 2019). Intact mid- to high-elevation sagebrush-steppe rangelands exposed to natural fire and disturbance regimes are generally resistant to plant community transitions and resilient to periodic perturbations (Chambers et al., 2017; Chambers, Bradley, et al., 2014; Chambers, Miller, et al., 2014; Miller et al., 2013). Inordinately prolonged fire-free periods on these rangelands allow encroaching native pinyon (*Pinus* spp.) and juniper (*Juniperus* spp.) conifers to persist and outcompete sagebrush and understory herbaceous vegetation for limited water and soil resources (Miller et al., 2000, 2008, 2019; Figures 1 and 2a). In the early stages of this woodland encroachment (Phase I), the amount and structure of sagebrush and perennial bunchgrass cover remain intact and reductions in ecosystem attributes and services are minimal. Pinyon and juniper increase in number and size in absence of fire and begin influencing water and soil resource availability (Phase II; Figure 1). Over time, pinyon and juniper dominate site resources (Phase III; Figure 1) and sustain a woodland vegetation structure with tree islands surrounded by extensive bare intercanopy (Figure 3a). In Phase III, residual intercanopy cover of sagebrush and perennial bunchgrasses is minimal and understory plants exhibit low vigor. Extensive bare ground between plants may increase invasibility to weeds, such as cheatgrass (*Bromus tectorum* L.), and propagate cross-scale runoff and soil loss. These significant alterations of sagebrush steppe associated with pinyon and juniper woodland encroachment are broadly occurring on western US rangelands, particularly in the Great Basin Region, USA (Davies et al., 2011; Miller et al., 2011). Land managers commonly apply pinyon and juniper removal treatments to reduce resource competition between trees and understory plants and re-establish sagebrush-steppe vegetation and associated ecosystem attributes and services, including water and soil retention (McIver & Brunson, 2014; Reinhardt et al., 2020; Williams et al., 2018). Although woodland encroachment into sagebrush steppe described here is a western US concern, similar woody plant encroachment occurs worldwide and its ecological ramifications and management are of global interest (Chartier & Rostagno, 2006; Eldridge et al., 2011; Ludwig et al., 2007; Turnbull et al., 2010a, 2010b; Van Auken, 2000, 2009).

Hydrologic vulnerability and erosion potential along rangeland and woodland hillslopes are a function of the susceptibility of the ground surface to runoff generation and sediment detachment and transport and the magnitude of water input to drive responses (Williams, Pierson, Al-Hamdan, et al., 2014; Williams, Pierson, Robichaud, & Boll, 2014; Figure 2). The amount and spatial connectivity of bare ground, soil infiltrability, and soil erodibility collectively exert the primary controls on surface susceptibility to runoff and erosion. Runoff and erosion rates on rangelands and woodlands are commonly greatest for well-connected bare areas in comparison with litter-covered and well-vegetated patches (Nouwakpo et al., 2016; Pierson et al., 2009, 2010, 2013; Williams, Pierson, Robichaud, Al-Hamdan, Boll, & Strand, 2016). Soil texture, structure, and moisture conditions influence the movement of water into and through the soil profile and affect soil erodibility (Pierson & Williams, 2016). Soil erodibility controls the amount of sediment potentially available for detachment and transport in bare areas and along hillslopes (Al-Hamdan, Pierson, Nearing, Williams, et al., 2012; Nyman et al., 2013). Susceptibility to runoff and erosion is also affected by the amount and distribution of vegetation, ground cover, and surface roughness elements, which capture and store rainfall/runoff and provide resistance to erosive energy of raindrops and overland flow (Al-Hamdan et al., 2013; Pierson & Williams, 2016; Williams, Pierson, Robichaud, & Boll, 2014). Topography may amplify susceptibility where it accentuates concentration of high-velocity overland flow or reduce susceptibility where it forces flow dispersal (Al-Hamdan et al., 2013; Nouwakpo et al., 2016). Rainfall intensity and duration determine the magnitude of water input at the event scale, whereas climate and weather patterns regulate seasonal to annual water input. The magnitude of water input dictates the degree to which susceptible conditions are subjected to various hydrologic and erosion processes by controlling the dynamic connectivity of runoff and sediment sources (Williams, Pierson, Robichaud, Al-Hamdan, Boll, & Strand, 2016; Williams, Pierson, Robichaud, & Boll, 2014). Given these fundamental ecohydrologic relationships, understanding and predicting hydrologic and erosion impacts of woody plant encroachment and associated management practices requires knowledge of the complexities in respective vegetation dynamics and shifts in community structure and ground surface conditions (Davenport et al., 1998; Ludwig et al., 2005; Roundy et al., 2017; Turnbull et al., 2008; Williams et al., 2018; Williams, Pierson, Al-Hamdan, et al., 2014; Williams, Pierson, Robichaud, Al-Hamdan, Boll, & Strand, 2016; Williams et al., 2016a).

The ecological impacts of pinyon and juniper removal on sagebrush sites have received substantial attention

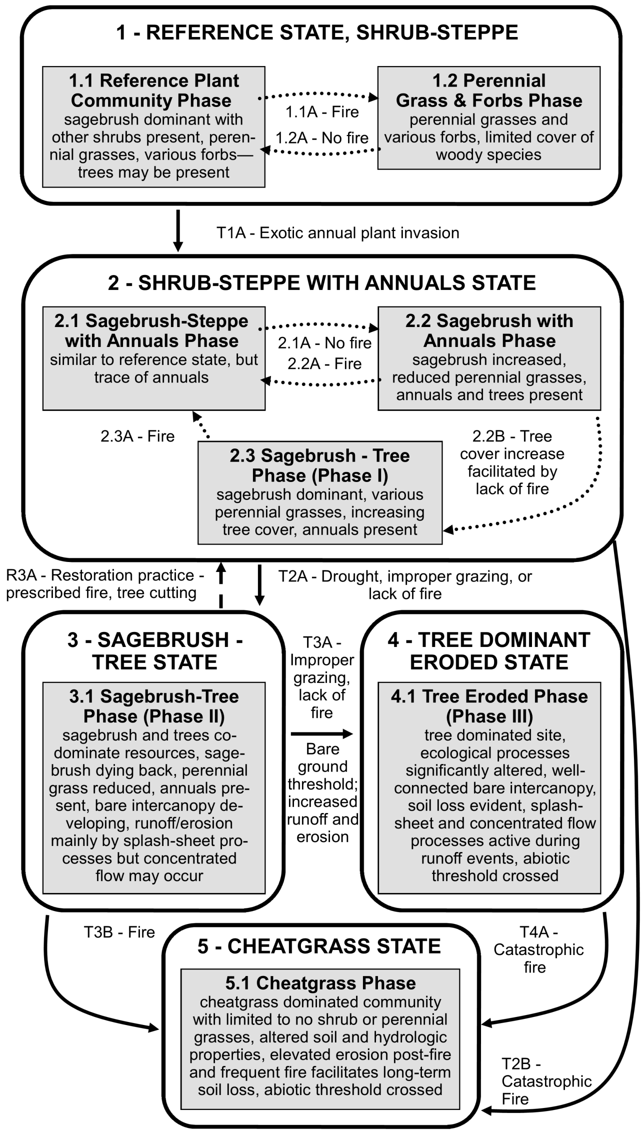

FIGURE 1 Generalized state-and-transition model showing ecological states and plant community phases for the Marking Corral and Onaqui study sites, as typical of pinyon (*Pinus* spp.) and juniper (*Juniperus* spp.) (trees noted in figure) encroached sagebrush (*Artemisia* spp.) rangelands in the Great Basin Region, USA. Individual ecological states are delineated by bold black boxes/rectangles, each with one or more within-state plant community phases (shaded box/rectangles). State transitions are indicated by solid black arrows. Within-state community pathways are indicated by dotted black arrows. Restoration pathways associated with management actions (such as tree removal) are indicated by dashed black arrows. Runoff and erosion for the Reference State and Shrub-Steppe with Annuals State are low and primarily occur by rainsplash and sheetflow (splash–sheet) processes in isolated bare patches. Bare area increases with tree cover in the Sagebrush-Tree State (Phase II pinyon/juniper woodland encroachment) and runoff and erosion risks are amplified. Further increases in pinyon and juniper cover are associated with transition to the Tree Dominant Eroded State (Phase III pinyon/juniper woodland encroachment). In that state, sagebrush and understory cover decline substantially, bare ground is well connected throughout the intercanopy between trees, and substantial soil loss occurs due to combined splash–sheet and concentrated overland flow processes during runoff events. The Cheatgrass State is marked by more frequent fire. Runoff and erosion rates are highest in the immediate postfire condition and frequent re-burning promotes long-term soil loss.

over the last decade. The regional multidisciplinary Sagebrush Steppe Treatment Evaluation Project (SageSTEP, www.sagestep.org) was developed in 2005 in part to study ecological responses of woodland-encroached sagebrush steppe to various tree-removal treatments (McIver et al., 2014; McIver & Brunson, 2014). Over more than a decade and beginning in 2006, SageSTEP scientists evaluated the initial and midterm effects of pinyon and juniper removal across a network of 11 sagebrush sites within the Great Basin and in various phases of woodland encroachment (Chambers, Miller, et al., 2014; Freund et al., 2021; Miller et al., 2014; Roundy, Miller, et al., 2014; Roundy, Young, et al., 2014; Williams et al., 2017). Treatment effects in the study exhibited some variation with woodland type, initial woodland phase and site conditions, and treatment method, but were generally consistent with other short- and long-term studies of tree removal in the region (see Miller et al., 2013, 2019). Prescribed fire and mechanical tree-removal treatments effectively increased plant available soil water for understory vegetation, with some increases persistent over a 14-year period posttreatment (Roundy et al., 2020; Roundy, Young, et al., 2014). Prescribed fire initially reduced sagebrush shrubs and perennial bunchgrass cover, but cover of bunchgrasses was typically greater on burned hillslopes than on control hillslopes within 3 years and at 6 years postfire (Miller et al., 2014; Williams et al., 2017). Prescribed fire also promoted increases in rabbitbrush (*Chrysothamnus viscidiflorus* [Hook.] Nutt.), annual grass cover (including cheatgrass), and annual and perennial forb covers by the second and third years posttreatment (Miller et al., 2014; Roundy, Miller, et al., 2014). Sagebrush does not resprout after fire and can take 30 years to more than 50 years to return to predisturbance levels on some sites (Miller et al., 2013; Moffet et al., 2015; Ziegenliagen & Miller, 2009). Therefore, sagebrush cover remained lower on burned than control plots 3 years posttreatment, but burning did promote increases in sagebrush seedling density (Miller et al., 2014). By the sixth year posttreatment, sagebrush cover was similar across control and burned plots, but was generally low in both (1%–14%) (Williams et al., 2017). Ten years after fire, burned plots

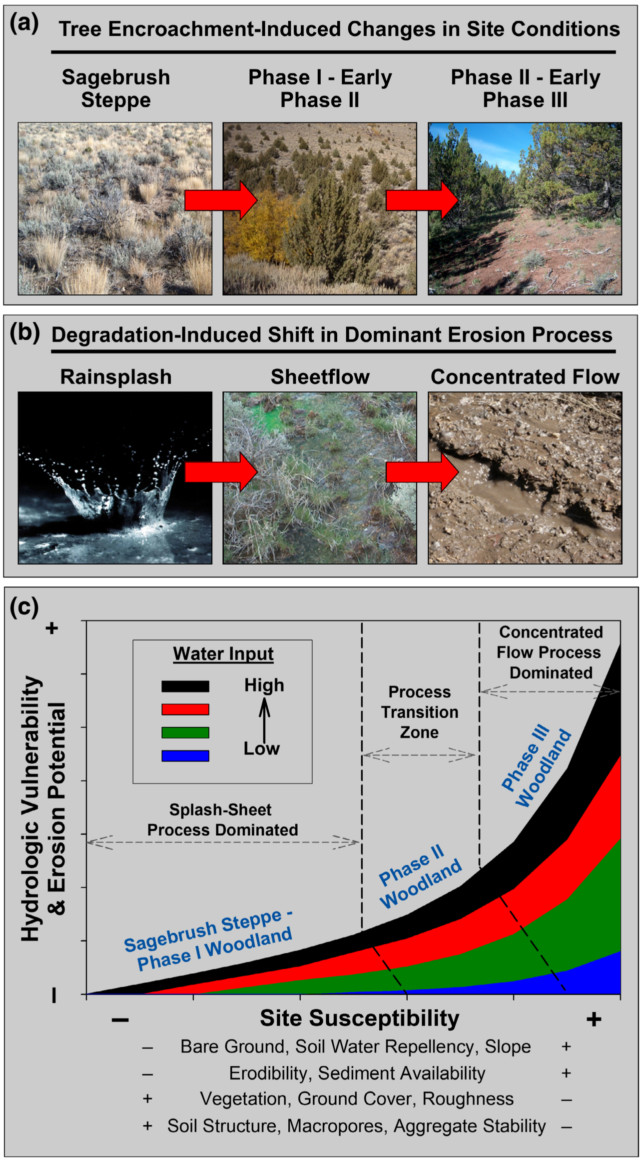

FIGURE 2 Illustrations showing (a) common vegetation and ground cover structural shifts on sagebrush (*Artemisia* spp.) steppe rangelands progressing through initial (Phase I), mid (Phase II), and later (Phase III) stages of pinyon (*Pinus* spp.) and juniper (*Juniperus* spp.) woodland encroachment, (b) associated degradation-induced shifts in dominant runoff and erosion processes with woodland encroachment progression, and (c) representative increases in hydrologic vulnerability and erosion potential (*y*-axis) associated with varying water input (see legend) and site susceptibility (*x*-axis) as attributed to respective dominant runoff/erosion processes (rainsplash, sheetflow, and/or concentrated flow). Site susceptibility is defined by the surface soil, ground cover, and topographic conditions that affect runoff and erosion responses. Symbols indicate directional increase (+) or decrease (−) in respective variable. Hydrologic vulnerability and erosion potential are low for intact sagebrush rangelands and are primarily associated with rainsplash and sheetflow processes. Concentrated flow occurs more often with progression to Phase II woodland conditions, becomes the dominate runoff and erosion process where bare soil approaches 50%–60%, and is the primary driver for long-term soil loss on highly susceptible surface conditions typical of Phase III woodlands. Concentrated flow has higher velocity than sheetflow and thereby exhibits greater energy for sediment detachment and transport than the combined effects of rainsplash and sheetflow. Figure modified from Williams, Pierson, Al-Hamdan, et al. (2014), Williams et al. (2016a), and Miller et al. (2013). Rainsplash photograph (b) courtesy of US Department of Agriculture, Natural Resources Conservation Service. All other photographs attributed to the authors.

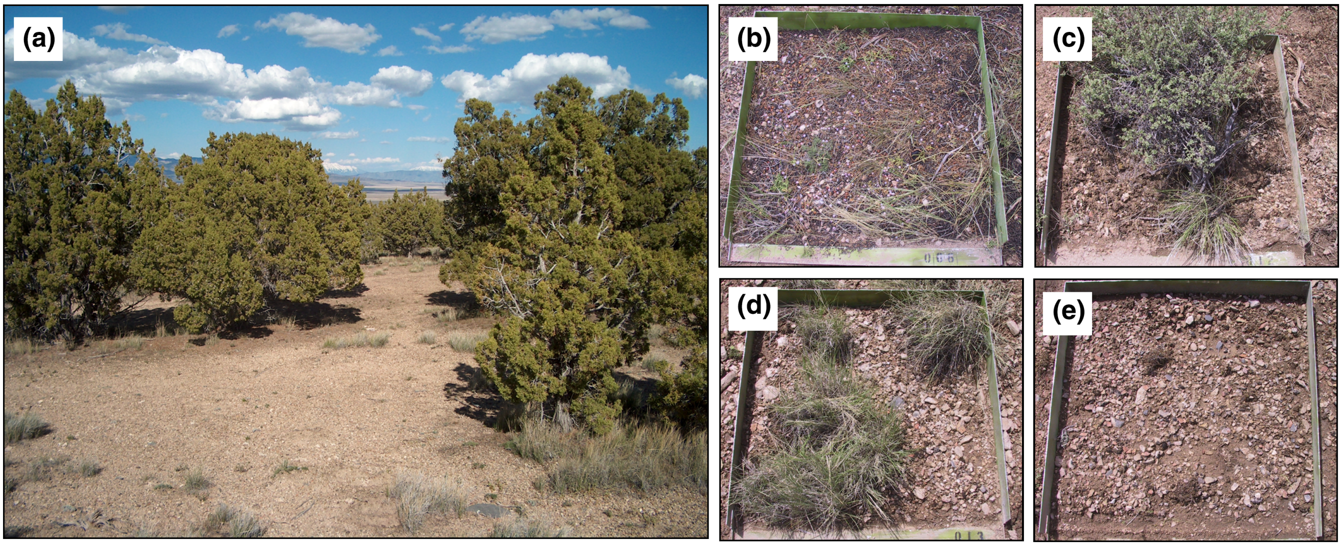

FIGURE 3 Photographs of a woodland-encroached sagebrush (*Artemisia* spp.) rangeland showing: (a) the common woodland vegetation structure in encroachment Phase III, with isolated litter-covered tree islands (tree canopy areas) and an extensive sparsely vegetated intercanopy area between trees; (b) the typical litter-covered ground surface underneath trees; and (c) shrub, (d) vegetated interspace (≥30% herbaceous cover), and (e) bare interspace (<30% herbaceous cover) microsites typical of areas within the intercanopy between trees. Figure modified from Williams, Johnson, Pierson, et al. (2020).

maintained greater perennial grass cover than control plots, but sagebrush cover remained low, particularly for sites with warm/dry soil temperature/moisture regimes (Chambers et al., 2021; Freund et al., 2021). Cheatgrass cover remained higher for burned than control plots 10 years postfire (Chambers et al., 2021; Freund et al., 2021). For mechanical treatments, sagebrush cover was largely unaffected initially and cover of perennial bunchgrasses in treated areas exceeded that of controls in the first 2–3 years after tree removal (Miller et al., 2014) and in the sixth year posttreatment (Williams et al., 2017). Mechanical treatments also increased perennial forb cover by the second year after tree removal and annual grasses and forbs by the third year posttreatment (Miller et al., 2014). Ten years posttreatment, tree cover remained low, while sagebrush and perennial grass cover both increased in mechanical treatment plots relative to untreated plots (Chambers et al., 2021; Freund et al., 2021; Wozniak et al., 2020). Cheatgrass cover was greater in mechanical treatment than control areas 10 years after tree removal, but increases were less than in burned areas (Freund et al., 2021).

Some key implications emerged from the short- to midterm SageSTEP experiments. First, increases in nonnative annual grasses, such as cheatgrass, are most common with burning and with treatment applications on warmer and drier sites (Chambers et al., 2021; Chambers, Miller, et al., 2014; Miller et al., 2014; Roundy, Miller, et al., 2014). Second, tree-removal treatments applied in earlier phases of woodland encroachment are most effective for reestablishing sagebrush-steppe vegetation (Freund et al., 2021; Roundy, Young, et al., 2014; Williams et al., 2017). Third, seeding may be required to promote desired understory responses to tree removal on Phase III woodlands with depleted sagebrush and perennial bunchgrass covers, particularly for warmer and drier sites (Freund et al., 2021; Roundy, Young, et al., 2014). Lastly, significant trade-offs exist among treatments (see Roundy, Miller, et al., 2014; Roundy, Young, et al., 2014; Williams et al., 2017). Fire can effectively kill trees of all size classes and promote perennial bunchgrass productivity. However, burning also reduces existing sagebrush cover and increases rabbitbrush and risk of cheatgrass invasion on some sites (Freund et al., 2021; Miller et al., 2013). Mechanical treatments have less initial impact on sagebrush cover but create ample surface fuels, can increase cheatgrass cover, and may leave numerous residual tree seedlings to reestablish tree dominance over time. These implications are consistent with earlier and recent single and multisite studies of pinyon and juniper removal on sagebrush sites throughout the Great Basin (Bates et al., 2000, 2005, 2011, 2014, 2017; Bates & Davies, 2016; Bates & Svejcar, 2009; Bybee et al., 2016; Davies et al., 2014; Davies & Bates, 2019; Davies, Bates, & Boyd, 2019; Davies, Rios, et al., 2019).

By comparison with vegetation research, few well-replicated field studies exist regarding partitioning of runoff and erosion processes in woodland-encroached sagebrush steppe. Interception studies from the Great Basin suggest precipitation interception by individual pinyon and juniper trees ranges about 40% to near 70% at event to annual time scales (Eddleman, 1986; Eddleman & Miller, 1992;

Stringham et al., 2018; Young et al., 1984). This suggests canopy interception by pinyon and juniper can greatly influence the amount of water available for infiltration and runoff beneath trees. Soils underneath pinyon and juniper canopies, and litter are commonly water repellent (Madsen et al., 2008; Pierson et al., 2010; Williams, Johnson, Pierson, et al., 2020; Williams, Pierson, Al-Hamdan, et al., 2014). However, the thick litter layers beneath these trees intercept and store rainfall passing through the canopy layers. The prolonged storage delays runoff and allows water to infiltrate by progressive wetting of water-repellent layers and/or preferential flow through macropores and isolated wettable patches (Lebron et al., 2007; Madsen et al., 2008; Pierson et al., 2010; Robinson et al., 2010; Roundy et al., 1978; Williams, Pierson, Al-Hamdan, et al., 2014). Rainfall simulation studies have reported high infiltration rates for litter-covered water-repellent soils underneath pinyon and juniper (Cline et al., 2010; Pierson et al., 2010, 2013, 2014; Roundy et al., 1978; Williams, Pierson, Al-Hamdan, et al., 2014; Williams, Pierson, Kormos, et al., 2020; Williams, Pierson, Nouwakpo, et al., 2019, 2020). In comparison with tree canopy areas, the same studies reported relatively similar infiltration rates for shrub canopy areas and about 25% to 70% lower infiltration rates for bare- and grass-covered interspaces between shrub and tree canopies. Tree islands on sagebrush-steppe sites in woodland Phases II and III are a minor component (often <30%) of the community structure, which is mostly intercanopy (~70%) with limited shrub cover and extensive interspace area (Figure 3). Rainfall simulation experiments have shown intercanopies comprised of interspace and sagebrush microsites have infiltration rates about 30% less than that of tree islands and are the primary sources of hillslope runoff and sediment in encroachment woodlands (Nouwakpo et al., 2020; Pierson et al., 2010; Williams, Pierson, Al-Hamdan, et al., 2014). Sediment yields from intercanopy areas during low- and high-intensity rainfall events can be 150% to more than 600% (~400% on average) higher than from tree islands (Nouwakpo et al., 2020; Pierson et al., 2010; Williams, Pierson, Al-Hamdan, et al., 2014).

The hydrologic studies noted above and others (Noelle et al., 2017; Petersen & Stringham, 2008; Pierson et al., 2007; Roundy et al., 2017) from woodland-encroached sagebrush sites clearly indicate that effectiveness of tree removal to limit or reduce hillslope hydrologic vulnerability and erosion potential hinges on treatment-induced re-establishment of intercanopy vegetation and ground cover (see Williams et al., 2018). Pierson et al. (2007) found plot-scale (32.5 m^2) intercanopy runoff and erosion from rainfall simulation experiments were 14- and more than 85-fold greater for a sagebrush site dominated by western juniper (*Juniperus occidentalis* [Hook.]—Phase III) relative to an adjacent sagebrush site where juniper were mechanically removed (cut) 10 years earlier. The authors attributed the enhanced hydrologic function on the cut site to improved intercanopy perennial herbaceous cover and litter recruitment over the 10-year period after tree removal. At a sagebrush site dominated by Colorado pinyon (*Pinus edulis* Engelm.) and Utah juniper (*Juniperus osteosperma* [Torr.] Little), Roundy et al. (2017) found that mechanical tree removal and subsequent seeding enhanced intercanopy vegetation the first year and litter cover the third year posttreatment. The authors further found that measured (10-m^2 plots) intercanopy runoff and erosion from natural rainfall events were less in treated than untreated plots within 2 years, and, that, in the fifth year posttreatment, the same measures were 5- and 10-fold less, respectively, for treated versus untreated areas (5-year average). Paired experiments by Williams, Pierson, Nouwakpo, et al. (2019, 2020) and Nouwakpo et al. (2020) as part of the SageSTEP study found that prescribed burning at two woodland-encroached sagebrush sites (Phases II and III) increased intercanopy perennial bunchgrass cover over a 9-year period postfire and that the enhanced vegetation improved interspace infiltration and reduced intercanopy runoff and erosion rates. Working at the same sites, Williams, Pierson, Kormos, et al. (2019) found mechanical tree-removal treatments enhanced intercanopy cover over a 9-year period posttreatment, but that cover improvements were insufficient to significantly improve infiltration and reduce erosion by overland flow within intercanopies except in areas immediately adjacent to downed trees.

The recent substantive advances in understanding of pinyon and juniper encroachment and tree-removal impacts on vegetation and hydrologic/erosion processes provide the underpinning to populate and evaluate predictive tools for guiding management of woodland-encroached sagebrush steppe (Bates et al., 2017; Miller et al., 2014, 2019; Nouwakpo et al., 2020; Roundy, Miller, et al., 2014; Roundy, Young, et al., 2014; Williams et al., 2017, 2018; Williams, Pierson, Kormos, et al., 2019; Williams, Pierson, Nouwakpo, et al., 2019; Williams, Johnson, Pierson, et al., 2020; Williams, Pierson, Nouwakpo, et al., 2020). Although vegetation monitoring is reasonably easy to broadly deploy, rainfall simulation and other hydrology and erosion experiments are challenging logistically and require substantial physical and financial resources. Quantitative models provide an alternative to resource demanding and laborious field experiments. These models are commonly designed to predict physical process responses to user-specified changes in vegetation, ground cover, and soil attributes for given amounts/durations of water input or a specific climate (Al-Hamdan et al., 2013, 2017; Al-Hamdan, Pierson, Nearing, Stone, et al., 2012; Al-Hamdan, Pierson, Nearing, Williams, et al., 2012; Laflen et al., 1994; Robichaud et al., 2007).

The Rangeland Hydrology and Erosion Model (RHEM) is a hillslope-scale process-based event runoff and erosion model developed specifically for rangeland applications (Al-Hamdan et al., 2015; Hernandez et al., 2017; Nearing et al., 2011). The RHEM simulates hydrologic and erosion processes as a function of water input, amounts and types of vegetation and ground cover, soil texture, and hillslope angle and shape. The model applies specific infiltration and erodibility parameterization estimation equations (Al-Hamdan et al., 2017; Al-Hamdan, Pierson, Nearing, Williams, et al., 2012; Hernandez et al., 2017) that account for the influences of plant growth form (shrub, grass, etc.), associated microsite attributes, and amounts of cover by growth form and ground cover type on surface hydrology and erosion processes (see Pierson & Williams, 2016). The effects of soil texture and slope attributes on hydrologic and erosion processes are accounted for as variables in infiltration, erodibility, and various flow hydraulics model parameter estimation equations, as described by Al-Hamdan, Pierson, Nearing, Stone, et al. (2012), Al-Hamdan, Pierson, Nearing, Williams, et al. (2012), Al-Hamdan et al. (2013, 2017), and Hernandez et al. (2017). The RHEM was applied in a US national assessment of conservation practices effectiveness to reduce runoff and erosion on privately owned rangelands (USDA, 2011). The RHEM was also applied at the regional scale in the American Southwest to assess hydrologic vulnerability and relative soil erosion rates across different ecological sites (Hernandez et al., 2013). More recently, Williams et al. (2016a, 2016b) demonstrated application of the model to develop and enhance Ecological Site Descriptions for assessing hydrologic impacts of disturbances and targeting management practices. Woodland encroachment and tree removal present unique applications for RHEM (Williams, Pierson, Robichaud, Al-Hamdan, Boll, & Strand, 2016; Williams et al., 2016a) and similar models given hillslope-scale infiltration, runoff, and erosion processes on woodland ecosystems are regulated primarily by the structure of tree islands and intercanopy area and secondarily by amounts of the various cover types therein (Ludwig et al., 2005; Petersen et al., 2009; Pierson et al., 2010; Williams et al., 2018; Williams, Pierson, Al-Hamdan, et al., 2014). Given this challenge and the need for predictive tools, the aim of this study was to develop RHEM modeling approaches for quantifying hydrologic vulnerability and erosion potential of sagebrush-steppe rangelands in the later stages of woodland encroachment (Phases II and III; Figures 1 and 3) and with vegetation and ground cover conditions representative of the long-term effects of tree removal. Using multisite experimental data and process-based knowledge from decade-long vegetation and rainfall simulation studies by the authors (see Nouwakpo et al., 2020; Pierson et al., 2010, 2014, 2015; Williams, Pierson, Kormos, et al., 2019; Williams, Pierson, Kormos, et al., 2020; Williams, Pierson, Nouwakpo, et al., 2019; Williams, Pierson, Nouwakpo, et al., 2020), our specific objectives were to: (1) demonstrate RHEM's capability to accurately simulate runoff and erosion processes for tree canopy (tree islands) and intercanopy areas in untreated and treated (9 years after tree removal by burning) woodlands, and (2) develop and evaluate RHEM model approaches/frameworks to represent aggregated effects of tree canopy and intercanopy areas on patch- and hillslope-scale runoff and erosion in untreated and treated (burned, cut, and masticated) woodlands. Objective 1 was accomplished by comparing RHEM-predicted and experimentally measured runoff and erosion for a design storm applied to tree canopy and intercanopy areas or patches at two untreated and treated woodlands (Nouwakpo et al., 2020; Williams, Pierson, Nouwakpo, et al., 2019). Objective 2 was accomplished by assessing RHEM-predicted hydrologic and erosion response variables and derived parameters from modeled patch-scale and hillslope-scale approaches/frameworks of untreated and treated tree canopy and intercanopy aggregations in context with measured hydrologic and erosion responses and experimental-based process understanding of woodlands and tree-removal effects. This study is part of the larger SageSTEP study aimed at investigating the ecological impacts of invasive species and woodland encroachment on sagebrush rangelands and the effects of various sagebrush-steppe restoration practices (McIver et al., 2014; McIver & Brunson, 2014).

METHODS

Study area

Two sagebrush-steppe sites in the later stages (Phases II and III) of pinyon and juniper encroachment (Figure 1) and subjected to prescribed fire and mechanical (cutting and mastication/shredding) tree-removal treatments (autumn 2006) were selected for this study. Both locations are part of the SageSTEP network and are component sites for ongoing extensive long-term ecohydrologic and erosion research by the authors (see Williams, Johnson, Pierson, et al., 2020; Williams, Pierson, Kormos, et al., 2020; Williams, Pierson, Nouwakpo, et al., 2019, 2020). The Marking Corral site (39°27′17″ N latitude, 115°06′51″ W longitude; Figure 4) is a single-leaf pinyon (*P. monophylla* Torr. & Frém.)—Utah juniper woodland located in the Egan Range, about 27 km northwest of Ely, Nevada, USA. The Onaqui site (40°12′42″ N latitude, 112°28′24″ W longitude; Figure 5) is a Utah juniper woodland located in the Onaqui Mountains, approximately 76 km southwest of Salt Lake City, Utah, USA. The sites are managed by the US Department of Interior, Bureau of Land Management (BLM). Domestic cattle have been excluded from the sites since autumn of 2005 as part of the SageSTEP study. Summary geographic, climate, soils, and vegetation attributes of both sites are provided in Table 1. Field studies at the sites prior to tree-removal treatments assessed vegetation/community structure at the hillslope scale (990-m^2 site characterization plots) and vegetation, soil properties, and hydrology and erosion processes over fine (0.5 m^2) and patch (13 m^2) scales (see Pierson et al., 2019; Williams, Pierson, Kormos, et al., 2020). Rainfall simulation experiments in those studies quantified fine-scale microsite (tree canopy, shrub canopy, and interspace; see Figure 3b–e) and patch-scale tree canopy and intercanopy (Figure 3a) contributions to hillslope runoff and erosion and the respective controls on runoff generation and sediment detachment and transport (Pierson et al., 2010). Follow-up studies at both sites quantified short-term (1–2 years posttreatment) impacts of tree removal by prescribed fire and mechanical treatments (tree cutting and tree mastication) on vegetation, soils, and infiltration, runoff, and erosion processes at multiple spatial scales (Cline et al., 2010; Pierson et al., 2014, 2015; Williams, Pierson, Robichaud, Al-Hamdan, Boll, & Strand, 2016). More recently, a series of studies at the sites by Williams, Pierson, Kormos, et al. (2019); Williams, Pierson, Nouwakpo, et al. (2019, 2020); Williams, Johnson, Pierson, et al. (2020) and Nouwakpo et al. (2020) evaluated longer-term (9–13 years posttreatment) effects of prescribed fire and mechanical tree removal on vegetation, soils, infiltration, runoff, and erosion processes across point to hillslope spatial scales.

Field studies by the authors found tree removal substantially altered vegetation and ground cover structure at both sites over a 9-year period posttreatment and thereby reduced hillslope runoff and erosion rates (Nouwakpo et al., 2020; Williams, Pierson, Nouwakpo, et al., 2019; Williams, Pierson, Nouwakpo, et al., 2020). The vegetation structure at both sites prior to tree removal (measured summer 2006) consisted of isolated tree islands (~25% of area) surrounded by degraded intercanopy (~75% of area; Pierson et al., 2010; Williams, Pierson, Robichaud, Al-Hamdan, Boll, & Strand, 2016). Both sites exhibited high sagebrush mortality associated with pinyon and juniper encroachment. For untreated conditions, the intercanopy understory vegetation was dominated by shrub cover at Marking Corral (Figure 4a) and by a mixture of grasses and forbs at Onaqui (Figure 5a). The ground surface in the intercanopy at both sites was mostly bare, with bare soil and rock cover more than 60% at Marking Corral and near 80% at Onaqui (Pierson et al., 2010). Understory vegetation cover in tree canopy areas was minor at both sites and was primarily herbaceous (Pierson et al., 2010; Williams, Pierson, Robichaud, Al-Hamdan, Boll, & Strand, 2016). The ground surface underneath tree canopies (2.2- to 2.5-m canopy radius on average) was ~80%–100% covered with a ~5- to 9-cm-thick litter layer (Pierson et al., 2010).

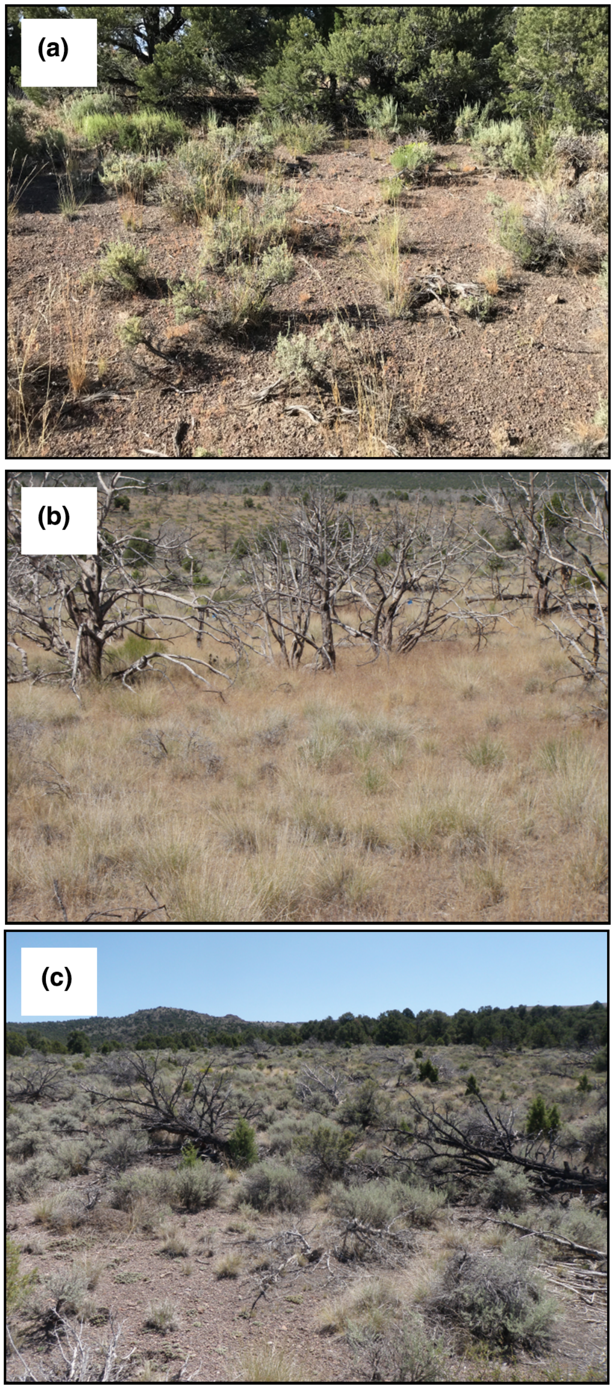

FIGURE 4 Photographs showing vegetation and ground cover at the Marking Corral study site in the untreated control (a) and in burned (b) and cut (c) treatment areas 9 years after tree removal. The images illustrate extensive degraded and bare intercanopy area in the untreated condition (a), enhanced herbaceous cover following burning (mainly *Achnatherum thurberianum* [Piper] Barkworth, *Pseudoroegneria spicata* [Pursh] Á. Löve, and *Hesperostipa comata* [Trin. & Rupr.] Barkworth in intercanopy areas, with *Bromus tectorum* L. around tree skeletons [former tree canopy]) (b), and retention and enhanced cover of shrubs and herbaceous plants following tree cutting (c). Pretreatment grass cover was limited (a) and mainly consisted of the same dominant species as posttreatment but with *B. tectorum* L. only in trace amounts (<1% foliar).

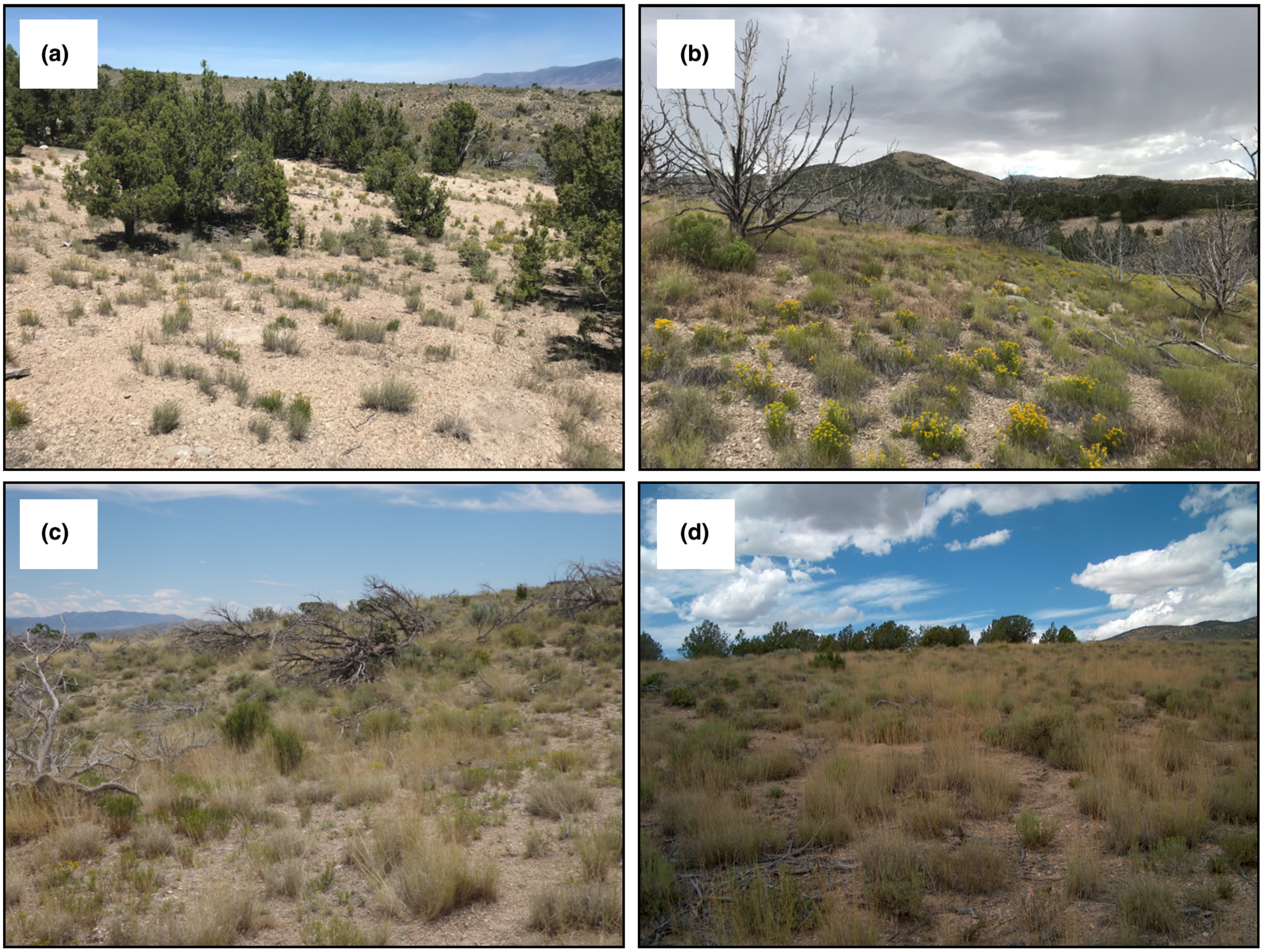

FIGURE 5 Photographs showing vegetation and ground cover at the Onaqui study site in the untreated control (a) and in burned (b), cut (c), and mastication (d) treatment areas 9 years after tree removal. The images show extensive degraded and bare area in the untreated condition (a), enhanced herbaceous cover following burning (b), and retention and enhanced shrub and herbaceous cover following tree cutting (c) and mastication (d). Grasses in treatment photos are primarily *Pseudoroegneria spicata* (Pursh) Á. Löve and *Poa secunda* J. Presl in intercanopy areas and *Bromus tectorum* L. in burned areas previously covered by tree canopy. The majority of shrub cover in burned areas (b) was *Chrysothamnus viscidiflorus* (Hook.) Nutt. (~10% foliar). Pretreatment grass cover was limited (a) and mainly consisted of the same dominant species as posttreatment but with *B. tectorum* L. only in trace amounts (<1% foliar).

Surface soils (≤5 cm depth) underneath thick tree litter layers were water repellent, but those underneath shrubs and in interspaces were wettable (Pierson et al., 2010, 2014). Surface soil bulk densities (0–5 cm depth) at both sites were generally lower in areas under tree (~0.97 $g\ m^{-3}$) and shrub (~1.07 $g\ m^{-3}$) canopies than in interspaces (~1.14 $g\ cm^{-3}$) (Pierson et al., 2010). The degradation-driven spatial heterogeneity in vegetation and surface conditions described above propagated high rates of runoff (51 $mm\ h^{-1}$ on average) and sediment discharge (1.3 $g\ s^{-1}$ on average) from intercanopies during rainfall simulation experiments (102 $mm\ h^{-1}$, 45 min, and 13-m^2 plots) at both sites (Pierson et al., 2010, Williams, Pierson, Robichaud, Al-Hamdan, Boll, & Strand, 2016). Interspaces exhibited poor infiltration relative to vegetated areas and generated higher mean runoff rates (51–56 $mm\ h^{-1}$) and sediment discharges (0.010–0.038 $g\ s^{-1}$) than shrub (~8 $mm\ h^{-1}$ and 0.001–0.007 $g\ s^{-1}$) and tree (3–31 $mm\ h^{-1}$ and 0.000–0.009 $g\ s^{-1}$) canopy areas during rainfall simulations (102 $mm\ h^{-1}$, 45 min, and 0.5-m^2 plots) (Pierson et al., 2010). Connectivity of bare ground throughout the extensive intercanopy area at both sites facilitated accumulation of interspace runoff and sediment sources during rainfall simulations that, over the patch scale, formed concentrated overland flow with high sediment detachment and transport capacities (Pierson et al., 2010, Williams, Pierson, Robichaud, Al-Hamdan, Boll, & Strand, 2016). As a result, intercanopy areas across both sites generated 4- to more than 10-fold more runoff and sediment yield than tree canopy areas during the high-intensity rainfall simulations

TABLE 1 Topography, climate, soil, and vegetation attributes at the Marking Corral and Onaqui sites prior to tree-removal treatments.

Site attribute	Marking Corral, Nevada, USA	Onaqui, Utah, USA
Woodland community	Single-leaf pinyon[a]/Utah juniper[b]	Utah juniper[b]
Elevation (m)—Aspect	2250—west to southwest facing	1720—north facing
Mean annual precipitation (mm)	306[c]	300[c]
Mean annual air temperature (°C)	6.5[c]	8.9[c]
Slope (%)	10–15	10–15
Parent rock	Andesite and rhyolite[d]	Sandstone and limestone[e]
Soil association	Segura-Upatad-Cropper[d]	Borvant[e]
Soil temperature/moisture regime[f]	Cool mesic/aridic–xeric	Warm mesic/aridic–xeric
Depth to bedrock (m)	0.4–0.5[d]	1.0–1.5[e]
Soil surface texture	Sandy loam, 66% sand, 30% silt, 4% clay	Sandy loam, 56% sand, 37% silt, 7% clay
Tree foliar cover (%)[g]	15[a], 10[b]	26[b]
Tree density (no. ha^{-1}) [g]	329[a], 150[b]	476[b]
Mean tree height (m)[g]	2.3[a], 2.4[b]	2.4[b]
Live shrub density (no. ha^{-1})	12,065	4914
Dead shrub density (no. ha^{-1})	2065	957
Intercanopy total foliar cover	39	19
Intercanopy shrub foliar cover	21	5
Intercanopy bare ground (%)[h]	64	79

Note: Data from Pierson et al. (2010) except where indicated by footnote. Common understory plants at the sites include *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; *Achnatherum thurberianum* (Piper) Barkworth; and various forbs.

[a]*Pinus monophylla* Torr. & Frém.

[b]*Juniperus osteosperma* [Torr.] Little.

[c]Estimated from 4-km grid for years 1971–2018 from PRISM Climate Group (2020).

[d]USDA NRCS (2007).

[e]USDA NRCS (2006).

[f]As reported in McIver and Brunson (2014).

[g]Tree data for trees ≥1-m height.

[h]Combination of bare soil and rock cover (fragments >5 mm in diameter).

and were the primary sources for water and sediment transport at the hillslope scale (Pierson et al., 2010, Williams, Pierson, Robichaud, Al-Hamdan, Boll, & Strand, 2016).

Over a 9-year period after tree removal by fire and mechanical treatments, elimination of competition for resources facilitated increased herbaceous cover in areas previously covered by tree canopy and throughout initially bare intercanopies at both sites (Williams, Pierson, Nouwakpo, et al., 2019, 2020; Figures 4 and 5). Across both sites, enhanced herbaceous cover posttreatment improved interspace infiltration by 1.5-fold and reduced interspace sediment yield by 2-fold as measured during rainfall simulations (102 mm h^{-1}, 45 min, and 0.5-m^2 plots) (Williams, Pierson, Kormos, et al., 2019; Williams, Pierson, Nouwakpo, et al., 2019, 2020). The overall structural shifts in vegetation patterns and enhanced vegetation and ground cover throughout intercanopies at both sites over 9 years after tree removal resulted in similar and low runoff rates (~14 mm h^{-1}) and sediment discharge (~0.12 g s^{-1}) during high-intensity rainfall simulations (111 mm h^{-1}, 45 min, and 12-m^2 plots) across burned plots (Nouwakpo et al., 2020; Williams, Pierson, Nouwakpo, et al., 2019). In the ninth year postfire, runoff rates and sediment discharge at a site for the high-intensity rainfall experiments were, on average, nearly equal for untreated litter-covered tree plots (12–16 mm h^{-1} and 0.091–0.170 g s^{-1}) and burned tree (8–20 mm h^{-1} and 0.020–0.166 g s^{-1}) and intercanopy (8–20 mm h^{-1} and 0.019–0.258 g s^{-1}) plots (Nouwakpo et al., 2020). No rainfall simulations were conducted in tree canopy and intercanopy areas at the patch scale within cut and mastication treatments 9 years after tree removal.

RHEM application at the patch scale

Measured data from field experiments by the authors (Nouwakpo et al., 2020; Pierson et al., 2010; Williams, Pierson, Nouwakpo, et al., 2019; Williams, Pierson, Robichaud, Al-Hamdan, Boll, & Strand, 2016) were utilized to populate comparable RHEM simulations of runoff and erosion from untreated and burned tree canopy and intercanopy plots at Marking Corral and Onaqui 9 years postfire. For this study, we employed the executable RHEM application, version 2.3 (https://apps.tucson.ars.ag.gov/rhem/docs), described by Hernandez et al. (2017). With the executable application, each RHEM simulation consists of four primary steps: (1) User inputs design storm rainfall intensity and duration, basic soils information (soil texture class for upper 4 cm), hillslope attributes (slope shape and steepness), and specific cover type amounts for foliar (bunch grass, forbs/annuals, shrubs, and sod grass) and ground (basal plant, rock, litter, and biological soil crusts) covers (Hernandez et al., 2017), (2) The RHEM utilizes user inputs to derive model required hydrologic and erosion parameters through a suite of empirical relationships and equations (see Al-Hamdan et al., 2013, 2015, 2017; Al-Hamdan, Pierson, Nearing, Stone, et al., 2012; Al-Hamdan, Pierson, Nearing, Williams, et al., 2012) (Appendix S1: Table S1), (3) the core engine models infiltration, overland flow, splash and sheet erosion, and concentrated flow erosion for the specified designed storm (Hernandez et al., 2017), and (4) the model outputs infiltration, runoff, and erosion rates and cumulative totals. We populated RHEM runs for 36 rainfall simulation experimental plots across untreated and burned tree canopy (10 untreated and 10 burned plots) and intercanopy (8 untreated and 8 burned plots) areas at the Marking Corral and Onaqui sites; each plot was originally established and sampled by Nouwakpo et al. (2020) as described below. Each rainfall simulation plot spanned an area 2 m wide by 6 m long, with the long axis perpendicular to the hillslope contour.

Each rainfall simulation plot was sampled in summer 2015 (9 years postfire) for foliar cover, ground cover, and slope gradient prior to rainfall simulation. All sampling protocols are described in detail in Nouwakpo et al. (2020) and are briefly summarized here. Overstory trees were removed from plots by chainsaw immediately prior to all sampling to allow for rainfall simulator placement. Understory foliar cover on each plot was quantified using laser-based line-point intercept methodologies on five evenly spaced transects spanning the width of the plot and with sample points spaced 10 cm apart. Foliar cover by plant life form at each point was recorded solely for the uppermost canopy layer intersected (i.e., first hit only). Ground cover by cover type was recorded at every sample point. Slope gradient for each plot was quantified by field survey methods using a Nikon NPR 352 total station.

Rainfall was applied to each plot using a Walnut Gulch Rainfall Simulator (Nouwakpo et al., 2020; Paige et al., 2004). The simulator was fitted with four Veejet 80–100 nozzles evenly spaced along an oscillating central boom, elevated 2.44 m above the ground surface, and pressurized to generate rainfall characteristics similar to that of natural high-intensity events (Paige et al., 2004). Two rainfall events were applied to each plot, a dry run and wet run. The dry run applied a rainfall intensity of 70 mm h^{-1} for 45 min on dry antecedent soil moisture conditions. The wet run was applied approximately 45 min after the dry run, at an intensity of 111 mm h^{-1} for 45 min on wet antecedent moisture conditions. Timed samples of plot runoff were collected in 1-L bottles at 3-min intervals during each rainfall simulation and were processed in the laboratory to determine respective plot-level rates and cumulative totals of runoff and erosion (see Nouwakpo et al., 2020).

For the current study, comparative RHEM runs were populated for each of the Nouwakpo et al. (2020) rainfall simulation plots, applying the same design storm (70-mm h^{-1} rainfall for 45 min, followed by a 45-min hiatus without rainfall, followed by 111-mm h^{-1} rainfall for 45 min); a 2-m-wide by 6-m-long slope with uniform shape; sandy loam surface soil texture class; 0.10 initial soil saturation; and respective plot measured foliar and ground covers and slope gradient (Appendix S1: Table S2). Additionally, we compiled RHEM simulations representing area-weighted aggregated tree canopy and intercanopy plots for untreated and burned conditions at both study sites (Appendix S1: Table S3). For each site, input area-weighted patch (IAWP) RHEM simulations were populated for every possible untreated and burned tree canopy and intercanopy plot pair combination by area-weighting respective plot foliar cover, ground cover, slope gradient, and soil characteristics for a 2-m-wide × 6-m-long uniform slope as described above. Also, for each site, output area-weighted patch (OAWP) RHEM simulations were derived for every possible untreated and burned tree canopy and intercanopy plot pair combination by area-weighting outputs of RHEM simulations of the respective rainfall simulation plots. For both the IAWP and OAWP RHEM approaches, tree canopy data were weighted by 0.3 and intercanopy data by 0.7 given approximately 30% of the area at both sites was tree microsite prefire (Table 1). Measured overstory tree cover was approximately 25% at the sites, but tree microsite effects on woodlands commonly extend a short distance beyond the canopy dripline (Madsen et al., 2008). The tree canopy and intercanopy plot pairings at Marking Corral and Onaqui, respectively, yielded 48 (24 untreated and 24 burned) and 32 (16 untreated and 16 burned) RHEM patch-scale aggregated simulations for both the IAWP and OAWP approaches. IAWP and OAWP RHEM simulations were compiled to assess model predictions of hydrologic vulnerability and erosion potential for aggregations of tree canopy and intercanopy patches in context with those for individual (nonaggregated) tree canopy and intercanopy areas as sampled by Nouwakpo et al. (2020). The data used to populate the RHEM simulations of the Nouwakpo et al. (2020) plots are summarized in Table 2, and data used to compile all patch-scale RHEM simulations are provided in full detail in Appendix S1: Tables S2 and S3.

RHEM application at the hillslope scale

Four RHEM hillslope-scale modeling scenarios/frameworks were developed using field-measured cover, soils, and slope attribute data and were evaluated for untreated and burned, cut, and masticated (all 9 years posttreatment) woodland conditions: (1) input area-weighted hillslope (IAWH)—hillslope model compiled by area-weighting understory (no overstory tree cover) foliar and ground cover inputs for respective measured covers in tree canopy and intercanopy patches (12-m^2 plots) applying weighting factors of 0.3 for tree canopy area (30% of total area) and 0.7 for intercanopy area (70% of total area); (2) output area-weighted hillslope (OAWH)—hillslope-scale model output derived from separate patch-scale model runs for tree canopy and intercanopy areas, each built with respective measured understory (no overstory tree cover) foliar and ground covers (12-m^2 plots), and with outputs area-weighted to the hillslope scale applying the 0.3 tree canopy and 0.7 intercanopy weighting factors; (3) understory only hillslope (UH)—hillslope model with cover inputs from foliar and ground cover measures (990-m^2 plots) for understory only—measured overstory tree cover ignored; (4) understory hillslope + tree cover (UH + T)—hillslope model with cover inputs from measured understory foliar and ground covers plus measured overstory pinyon and juniper cover as additional shrub foliar cover (990-m^2 plots). Each scenario applied the same design storm (70-mm h^{-1} rainfall for 45 min, followed by a 45-min hiatus without rainfall, followed by 111-mm h^{-1} rainfall for 45 min), a sandy loam soil texture and 0.10 initial degree of soil saturation, and a 50-m-long hillslope with a uniform shape and slope steepness (10% for Marking Corral and 15% for Onaqui), typical of respective attributes at both sites (see Pierson et al., 2010), and foliar and ground cover inputs as described below.

Foliar and ground cover data for the IAWH and OAWH hillslope (50 m length) model runs were provided by aforementioned patch-scale understory foliar and ground cover measures from Nouwakpo et al. (2020; Table 2; Appendix S1: Tables S2 and S3). For these frameworks, RHEM hillslope-scale model runs were developed for every possible untreated and burned tree canopy and intercanopy plot pair combination at a site consistent with area-weighing approaches described for the aggregated patch-scale RHEM simulations (IAWP and OAWP). The tree canopy and intercanopy plot pairings at Marking

TABLE 2 Summary of ground surface slope, foliar cover, and ground cover conditions as measured on patch-scale rainfall simulation plots (12 m^2) in untreated and burned tree canopy and intercanopy areas at the Marking Corral and Onaqui sites 9 years postfire.

	Marking Corral				Onaqui			
	Untreated		Burned		Untreated		Burned	
Characteristic	Tree canopy	Intercanopy	Tree canopy	Intercanopy	Tree canopy	Intercanopy	Tree canopy	Intercanopy
Surface slope (%)	10	11	12	11	15	15	19	21
Foliar cover								
Total foliar (%)	24	29	25	36	10	14	63	41
Shrub (%)	17	17	1	6	2	6	12	14
Perennial grass (%)	6	10	12	23	6	4	15	14
Forbs and annual grass (%)	1	2	12	7	2	4	36	13
Ground cover								
Basal plant (%)	<1	1	<1	2	<1	<1	<1	<1
Soil crust (%)	0	0	0	0	<1	<1	<1	0
Litter (%)	77	39	93	54	70	14	75	34
Rock (%)	0	1	3	12	15	42	9	43
Bare soil (%)	22	59	4	33	15	43	16	23
No. plots	6	4	6	4	4	4	4	4

Note: Data from Nouwakpo et al. (2020). Full dataset provided in Appendix S1: Table S2.

Corral and Onaqui, respectively, yielded 48 (24 untreated and 24 burned) and 32 (16 untreated and 16 burned) RHEM hillslope-scale simulations for both the IAWH and OAWH frameworks (Appendix S1: Table S4).

Foliar and ground cover data required for the UH and UH + T model runs were acquired from hillslope-scale site characterization plots (33 m × 30 m) established and sampled at the Marking Corral and Onaqui sites in summer 2006 prior to tree removal (Pierson et al., 2010) and resampled in summer 2015 (Williams, Pierson, Kormos, et al., 2019; Williams, Pierson, Nouwakpo, et al., 2020), 9 years after tree removal by fire, cutting, and mastication treatments. Foliar cover and ground cover for the untreated and treated site characterization plots at both sites are summarized in Table 3 and provided in detail in Appendix S1: Table S4. For both the UH and UH + T frameworks, individual model runs were configured for each site characterization plot using respective plot measured foliar and ground cover, yielding six and nine model runs for untreated conditions at Marking Corral and Onaqui, respectively, and three model runs in each treatment (burn, cut, and mastication) at a site. The mastication treatment was only applied at Onaqui. The methodologies for foliar and ground data collection on site characterization plots are described in detail by Pierson et al. (2010) and Williams, Pierson, Nouwakpo, et al. (2020) and are briefly summarized here. Understory foliar and ground cover on each site characterization plot were measured using line-point intercept methods along five 30-m transects spaced approximately 5–8 m apart and oriented perpendicular to the hillslope contour. Foliar and ground cover on each plot were recorded at 60 points with 50-cm spacing along each of the 5 transects. A foliar cover hit was recorded for each plant encountered at each sample point, with the maximum number of foliar hits at a point never exceeding three. For the current study, percent foliar cover estimates for each plot were acquired based solely on the first hit foliar cover data from the historical site characterization datasets (Pierson et al., 2010; Williams, Pierson, Kormos, et al., 2019; Williams, Pierson, Nouwakpo, et al., 2020). The UH + T model runs also required overstory tree cover data collected on the site characterization plots. The density and crown cover of mature (>0.5-m height) pinyon and juniper trees were measured on each site characterization plot, as described by Pierson et al. (2010). The number of live mature trees within each plot border was tallied, and the height and crown radius of each mature tree were quantified. Individual tree crown area on each plot was assumed equivalent to the area of a circle and was calculated with the respective tree crown radius. Total overstory tree cover for each plot was calculated as the sum of measured tree crown areas on the respective plot. Since RHEM does not have an input category for tree cover, overstory tree cover for each of the UH + T model runs was entered as shrub foliar cover, additive to measured shrub foliar cover.

The different RHEM frameworks described above were compiled for multiple reasons. For woodland

TABLE 3 Summary of foliar cover and ground cover measured on hillslope-scale site characterization plots (990 m^2) in untreated areas and in treated (burned, cut, and masticated) areas 9 years after tree removal at the Marking Corral and Onaqui sites.

	Marking Corral			Onaqui			
Characteristic	**Untreated**	**Burned**	**Cut**	**Untreated**	**Burned**	**Cut**	**Mastication**
Overstory tree cover	25	4	3	26	0	0	0
Understory foliar cover							
Total understory foliar (%)	28	66	59	20	58	51	50
Shrub (%)	16	12	38	3	11	23	10
Perennial grass (%)	11	30	16	14	22	20	36
Forbs and annual grass (%)	1	24	5	3	25	8	4
Ground cover							
Basal plant (%)	<1	7	8	1	13	10	16
Soil crust (%)	<1	0	0	4	1	<1	3
Litter (%)	49	40	48	33	35	37	37
Rock (%)	24	13	1	26	22	17	14
Bare soil (%)	27	40	42	35	30	36	30
No. plots	6	3	3	9	3	3	3

Note: Data from Pierson et al. (2010), Williams, Pierson, Kormos, et al. (2019), and Williams, Pierson, Nouwakpo, et al. (2020). Full dataset provided in Appendix S1: Table S4.

ecosystems, inconsistencies in cover data sources and the typical spatial patterns in surface hydrology/erosion necessitate an evaluation of multiple potential model applications. Our model frameworks were developed in part based on the myriad of vegetation and ground cover data sources available to end users, such as remote sensing and local to regional databases (Jones et al., 2018; USDA NRCS, 2020a; USDA NRCS, 2020b; USDI BLM, 2020). Many RHEM end users will not actually measure the cover data required for model runs and will therefore rely on alternative data sources (Williams et al., 2016a). The various data sources often represent cover in different ways or at different horizontal and vertical spatial scales. For example, some data sources may not provide separate overstory (tree cover) and understory (shrubs and herbaceous) foliar cover, whereas others may accommodate separation of overstory tree cover and understory foliar cover in either or both vertical or horizontal dimensions. Additionally, RHEM currently has no input category for tree foliar cover, and therefore, an alternative representation of tree cover as shrub foliar cover requires investigation. Our extensive multiscale vegetation, hydrology, and erosion data and process-based knowledge from long-term field research at the Marking Corral and Onaqui study sites uniquely allow us to explore various scenarios and to develop one or more RHEM modeling frameworks for woodlands management that accommodate(s) a range of data availability.

Data analyses

All statistical analyses were conducted using SAS software, version 9.4 (SAS Institute, Inc., 2013). The RHEM-predicted versus measured runoff and erosion values for tree canopy and intercanopy rainfall simulation plots at the patch scale were evaluated using percent bias (PBIAS; Gupta et al., 1999):

$$\mathrm{PBIAS} = \frac{\sum_{i=1}^{n} (O_i - M_i)}{\sum_{i=1}^{n} O_i} \times 100,$$

where O_i is the ith observation evaluated, M_i is the simulated value for the corresponding ith observation, and n is the number of observations. Results from PBIAS were used to assess RHEM's capability to effectively predict hydrologic and erosion responses to applied rainfall as measured on untreated and burned tree canopy and intercanopy plots. Model performance was considered "very good" when $\mathrm{PBIAS} < \pm 15$, "good" when $\pm 15 \leq \mathrm{PBIAS} < \pm 30$, "satisfactory" when $\pm 30 \leq \mathrm{PBIAS} < \pm 55$, and "unsatisfactory" when $\mathrm{PBIAS} \geq \pm 55$ (Moriasi et al., 2007).

To assess RHEM detection of patch-scale tree-removal treatment effects at a site, a suite of RHEM-predicted hydrologic and erosion response variables and derived model parameters were evaluated using a mixed model with two treatment levels (untreated and burned) and two subtreatment or microsite levels (tree canopy and intercanopy). Similarly, within-site RHEM-predicted patch-scale hydrologic and erosion response variables and derived model parameters for tree canopy and intercanopy aggregated approaches were evaluated using a mixed model with two treatment levels (IAWP and OAWP) and two subtreatment levels (untreated and burned). For all analyses, plot location was considered a random effect and treatment and subtreatment or microsite were considered fixed effects. The variables/parameters evaluated include RHEM-derived hydraulic friction factor (F_t), effective saturated hydraulic conductivity (K_e), splash-sheet erodibility factor (K_{ss}), and cumulative runoff and sediment. Details on the RHEM model parameters and respective estimation equations are available in Al-Hamdan, Pierson, Nearing, Williams, et al. (2012), Al-Hamdan et al. (2013, 2015, 2017), and Hernandez et al. (2017) (see Appendix S1: Table S1). Data normality was tested prior to analyses of variance using the Shapiro–Wilk test, and log transformations were applied where necessary to address deviance. Back-transformed means are reported. Post hoc pairwise comparisons were conducted using Tukey's honestly significant difference. The Kruskal–Wallis test was applied to variables in which data transformation failed to achieve normality. Pairwise comparisons in such cases were conducted with the Dwass, Steel, Critchlow–Fligner (DSCF) post hoc test. All reported significant effects were tested at the $p < 0.05$ level.

Differences in RHEM-predicted hydrologic and erosion response variables and model parameters across treatments at a site for the various hillslope-scale model frameworks were evaluated using the Kruskal–Wallis method with the DSCF post hoc test at the $p < 0.05$ level. Treatments were untreated, burned, cut, and masticated, and frameworks were IAWH, OAWH, UH, and UH + T. Response variables and model parameters evaluated were consistent with those described above for patch-scale RHEM simulations.

Simple linear and nonlinear regressions were applied to explore explanatory relationships between variable pairs at the patch and the hillslope scales (e.g., runoff vs. bare soil and rock cover, sediment vs. bare soil). All reported significant effects were tested at the $p < 0.05$ level.

RESULTS

Model performance—Patch-scale rainfall simulation plots

The RHEM effectively predicted runoff and erosion from patch-scale rainfall simulation plots at both woodlands (Figure 6). Model performance for runoff and for sediment prediction was "good" (PBIAS = −21) and "very good" (PBIAS = 6.1), respectively (Figure 6). The RHEM-predicted and measured values exhibited a near 1:1 relationship through the full range of runoff, 1–67 mm (Figure 6a). Predicted and measured sediment ($R^2 = 0.65$) followed a 1:1 relationship throughout the data range (1–909 g m^{-2}) with exception of a single outlier, an untreated intercanopy plot at Marking Corral (Figure 6b). Omitting the single high leverage value ($x = 909$ g m^{-2} and $y = 325$ g m^{-2}), predicted and measured sediment definitively exhibited a 1:1 relationship over a range of 0–440 g m^{-2}, with a PBIAS = −14.8 ("very good") and $R^2 = 0.85$ (Figure 6c).

The RHEM also effectively predicted differences in tree canopy and intercanopy runoff and erosion and associated impacts of tree removal by prescribed fire (Table 4). Measured runoff was approximately 3- to 4-fold greater and sediment 4- to 14-fold greater for untreated intercanopy than tree canopy plots. The RHEM predicted runoff was two- to fourfold greater and sediment fivefold greater for untreated intercanopy than tree canopy plots. The lesser magnitude difference in predicted sediment yield for untreated intercanopy versus tree canopy areas is due largely to the single high sediment yield outlier (909 g m^{-2}) measured on an intercanopy plot at Marking Corral (Figure 6b). Measured data from rainfall simulation plots found burning at Marking Corral reduced runoff from tree canopy and intercanopy areas by three- to eightfold. The RHEM predicted runoff was two- to threefold less for burned versus untreated tree canopy and intercanopy plots at that site. Lesser treatment-induced reductions for RHEM-predicted versus measured runoff for intercanopy areas at Marking Corral are due to slight over predictions in runoff for burned intercanopy plots at that site (Table 4). The fire treatment at Marking Corral significantly reduced sediment yield by 5-fold in tree canopy areas and by more than 70-fold in intercanopy areas as measured by rainfall simulations. The RHEM predicted a significant fire-induced reduction in sediment yield for Marking Corral tree canopy and intercanopy areas, but predicted sediment reduction for the intercanopy was only sixfold, again largely due to one outlier untreated intercanopy plot with exceptionally high sediment yield. At Onaqui, prescribed fire had no significant impact on runoff and sediment yield as measured for tree canopy areas. The RHEM-predicted values likewise showed no fire effect on tree canopy runoff and sediment yield at that site. By contrast, the burn treatment at Onaqui reduced intercanopy runoff and sediment yield by two- to threefold as measured during rainfall

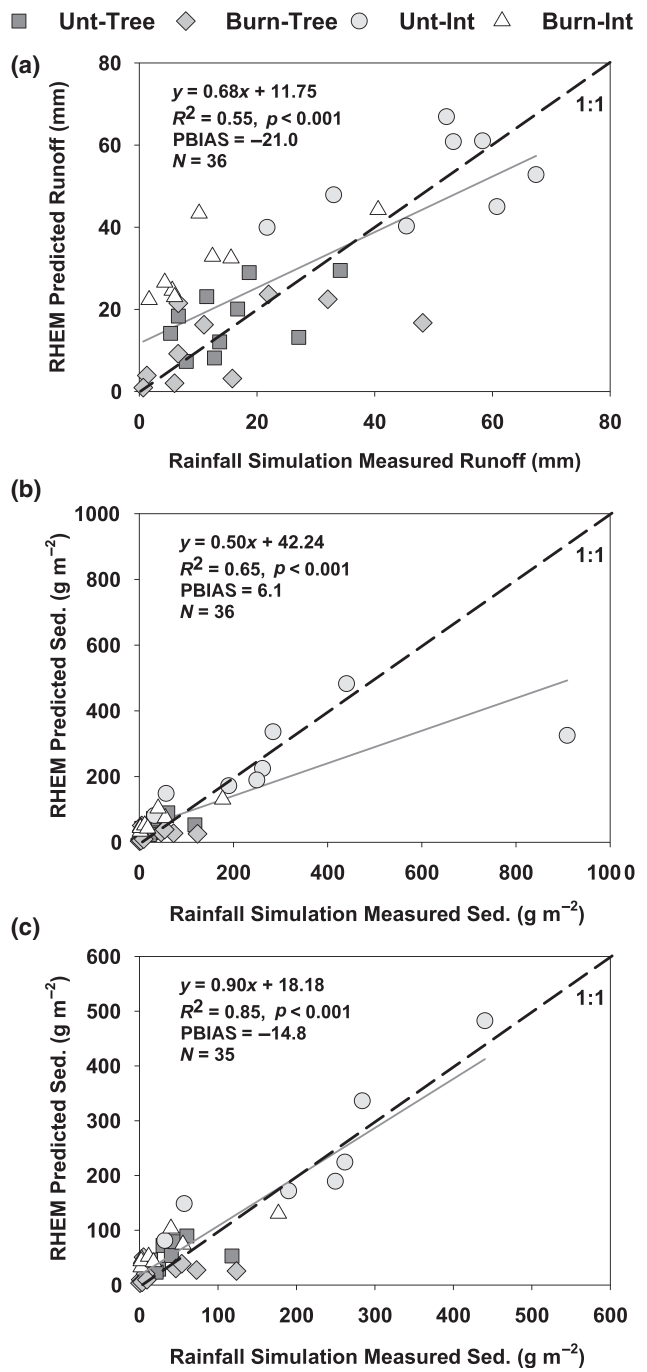

FIGURE 6 Measured rainfall simulation versus Rangeland Hydrology and Erosion Model (RHEM) predicted cumulative runoff (a) and sediment (Sed.; b and c) for untreated (Unt) and burned (Burn, 9 years postfire) patch-scale (12 m^2) tree canopy (Tree) and intercanopy (Int) plots at the Marking Corral and Onaqui study sites. Measured sediment versus predicted sediment in (c) excludes one outlier point ($x = 909$ g m^{-2} and $y = 325$ g m^{-2}), an unburned intercanopy plot at Marking Corral. Measured data are from Nouwakpo et al. (2020). Data and RHEM inputs and configuration are provided in Appendix S1: Table S2.

TABLE 4 Measured (rainfall simulations) and predicted (Rangeland Hydrology and Erosion Model, RHEM) runoff and sediment and RHEM-derived hydrologic and erosion parameters for patch-scale (12 m^2) plots in untreated and burned tree canopy and intercanopy areas at Marking Corral and Onaqui.

	Marking Corral				Onaqui			
	Untreated		Burned		Untreated		Burned	
Parameter	Tree canopy	Intercanopy	Tree canopy	Intercanopy	Tree canopy	Intercanopy	Tree canopy	Intercanopy
Rainfall simulation—measured								
Applied rain (mm)	151 bc	164 c	130 a	134 ab	124 a	143 a	140 a	133 a
Runoff (mm)	16 b	45 b	6 a	6 a	15 a	53 b	28 ab	18 a
Sediment (g m^{-2})	25 b	360 c	5 a	5 a	56 a	246 b	74 a	72 a
RHEM—predicted								
Runoff (mm)	21 b	50 c	7 a	27 b	13 a	53 c	20 a	36 b
Sediment (g m^{-2})	53 b	259 c	17 a	44 b	49 ab	230 c	31 a	88 b
RHEM—parameters								
F_t	16.5 c	5.3 a	30.7 d	9.5 b	19.5 bc	4.3 a	25.7 c	10.2 b
K_e (mm h^{-1})	50 b	27 a	57 c	32 a	42 b	17 a	44 b	23 a
K_{ss}	1178 b	2807 c	865 a	954 ab	1724 b	3161 c	751 a	1457 b
No. plots	6	4	6	4	4	4	4	4

Note: Rainfall simulation data from Nouwakpo et al. (2020). See Appendix S1: Table S2 for rainfall simulation data and respective model inputs. Within-site (Marking Corral or Onaqui) means for treatment (untreated and burned) and microsite (tree canopy and intercanopy) combinations in a row followed by different lowercase letters are significantly different ($p < 0.05$).

Abbreviations: F_t, hydraulic friction factor; K_e, effective hydraulic conductivity; K_{ss}, splash and sheet erodibility coefficient.

simulations, and RHEM predicted those reductions as two- to threefold. Overall, RHEM accurately represented microsite and treatment effects on runoff and erosion, particularly with exclusion of the one outlier point for measured sediment at Marking Corral (Figure 6, Table 4).

With few exceptions, RHEM-derived hydrologic and erosion parameters (F_t, K_e, and K_{ss}) followed similar trends as runoff and sediment variables for untreated tree canopy versus intercanopy microsites and for fire effects (Figure 7, Table 4). For untreated conditions, the hydraulic friction factor and effective hydraulic conductivity parameters were less for the mostly bare intercanopy plots than the litter-covered tree canopy plots at both sites (Figure 7d,e, Table 4). Likewise, the hydraulic friction factor was less for untreated plots at Marking Corral and for unburned intercanopy plots at Onaqui relative to respective burned plots (Table 4). Effective hydraulic conductivity was less for untreated than treated tree canopy areas at Marking Corral but was similar across untreated and treated conditions in tree canopy and intercanopy areas at Onaqui (Table 4). The splash and sheet erodibility coefficient was greater for the mostly bare untreated intercanopy plots at both sites relative to respective predictions on litter-covered untreated tree canopy plots (Figure 7f, Table 4). The erodibility coefficient was reduced by the fire treatment on tree canopy and intercanopy plots at both sites (Table 4).

Overall, measured and RHEM-predicted hydrologic and erosion responses and associated model parameters for untreated and burned conditions demonstrate the dominant controls on runoff and erosion processes for woodlands (Figure 7). Measured runoff from rainfall simulations was primarily controlled by the amount of bare ground (bare soil and rock cover) (Figure 7a) and secondarily by litter ground cover ($p < 0.001$). Measured sediment yield during rainfall simulation experiments was controlled by runoff (Figure 7b), bare soil (Figure 7c; increasing exponentially where bare soil exceeded 50%–60%), and litter ground cover ($p < 0.001$). Litter cover is the dominant ground cover type at the patch scale for both sites (Table 2) and therefore effectively is the inverse of bare ground for both woodlands. The RHEM-predicted hydraulic friction factor, effective hydraulic conductivity, and splash and sheet erodibility coefficient all varied consistently with changes in bare conditions (Figure 7d–f) and matched associated trends in measured and predicted runoff and erosion responses across untreated and burned microsite conditions (Figure 7a–c). The responses reflect the influence of bare area and ground cover on infiltration processes, as evident by the effective hydraulic conductivity versus bare ground relationship (Figure 7e), and on soil detachment and entrainment by raindrops and overland flow, as indicated by reduced

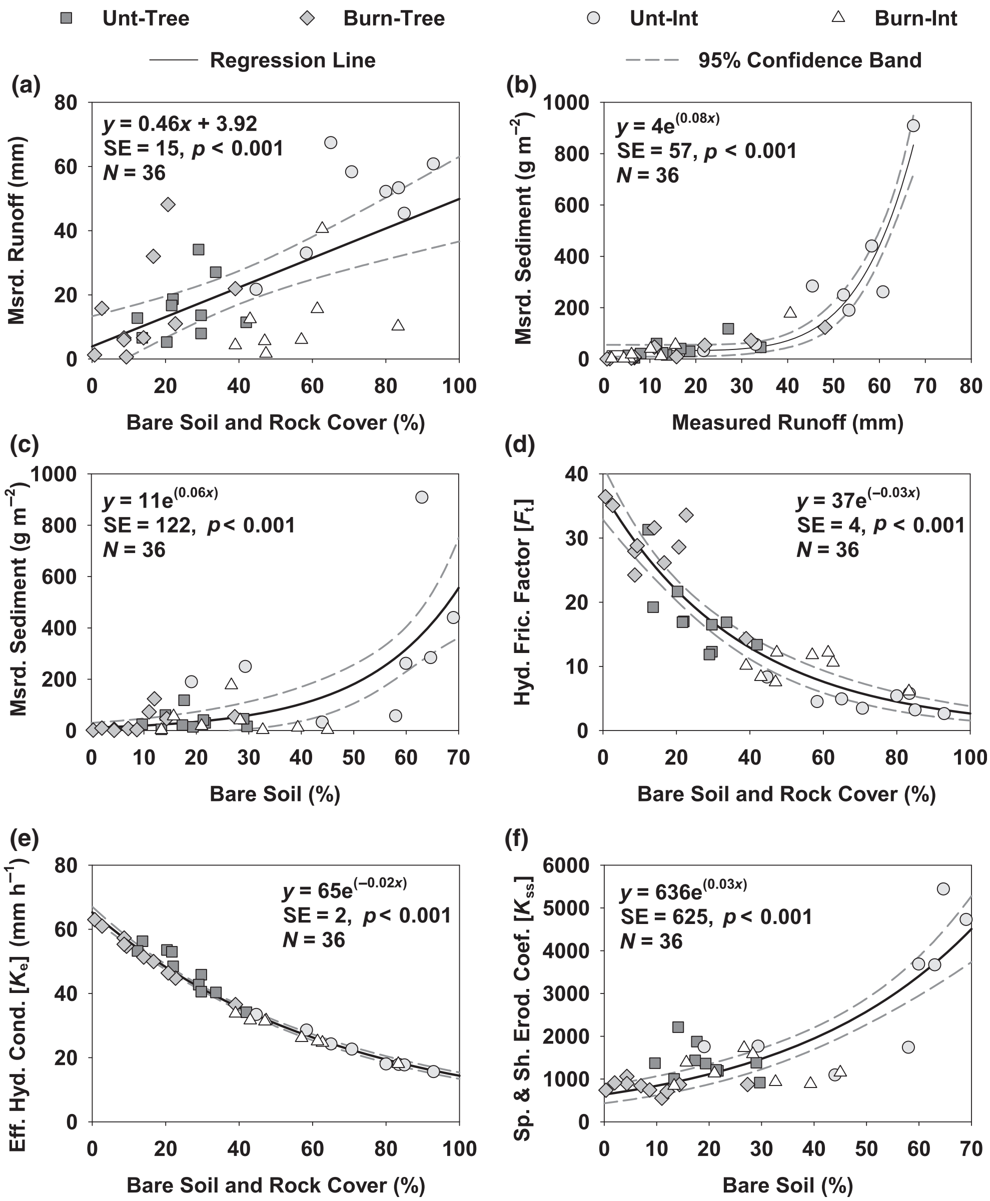

FIGURE 7 Correlations of measured (Msrd.) cumulative runoff (a) and sediment (b and c) and model derived (Rangeland Hydrology and Erosion Model [RHEM]) hydraulic friction factor (Hyd. Fric. Factor [F_t]) (d), effective hydraulic conductivity (Eff. Hyd. Cond. [K_e]) (e), and splash and sheet erodibility coefficient (Sp. & Sh. Erod. Coef. [K_{ss}]) (f) with respective plot attributes for patch-scale rainfall simulation plots (12 m^2) spanning untreated (Unt) and burned (Burn) conditions (9 years postfire) in tree canopy (Tree) and intercanopy (Int) areas at the Marking Corral and Onaqui study sites. Data and RHEM inputs and configuration provided in Appendix S1: Table S2.

hydraulic friction and increased sheet and splash erodibility with increasing bare ground (Figure 7d,f).

Tree canopy and intercanopy area-weighting approaches at the patch scale

The IAWP and OAWP approaches equally captured aggregated tree canopy and intercanopy microsite effects on runoff at the patch scale (Figure 8a). The RHEM predicted runoff for the untreated tree canopy and intercanopy plot pairs was consistent across IAWP and OAWP approaches and was, on average, 2.1-fold greater and 1.5-fold less than measured runoff on individual untreated tree canopy and intercanopy plots, respectively (Figures 8a and 9a,f, Tables 4 and 5). The RHEM runoff predictions with the aggregated approaches for burned conditions were, on average, 2.3-fold greater than measured tree canopy runoff but were also 2.6-fold greater than measured intercanopy runoff (Figure 10a,f, Tables 4 and 5). Regardless of the overpredictions for burned areas, both approaches effectively accounted for tree canopy and intercanopy differences in runoff at the patch scale (Figures 9 and 10, Tables 4 and 5). Similarities in runoff predictions across the two approaches indicate requisite capability in representing aggregated tree canopy and intercanopy effects on runoff at the patch scale (Figure 8a).

The IAWP and OAWP approaches both effectively represented aggregated microsite effects on patch-scale erosion processes, but the OAWP generally predicted greater sediment yield for the most degraded conditions (Figure 8b). As expected, RHEM IAWP and OAWP predicted sediment yields for untreated conditions were, on average, 3.0-fold greater and 2.5-fold less than measured sediment yields in untreated tree canopy and intercanopy areas, respectively (Figure 9b,f, Tables 4 and 5). The sediment predictions with the aggregated approaches for burned conditions at Marking Corral were, on average, 7.1-fold greater than measured sediment yields for tree canopy and intercanopy areas due to extremely low erosion on burned plots at that site (Tables 4 and 5). For Onaqui, sediment predictions with IAWP and OAWP approaches were similar to respective measured values for burned conditions across all plots (Tables 4 and 5). The RHEM-predicted sediment with the OAWP approach deviated from a 1:1 relationship with the same values for the IAWP approach for untreated conditions (Figure 8b). The deviation was due to data points for specific plot pairings with the four most erosive untreated intercanopy plots, measured sediment yields of 262–909 g m^{-2} (Figure 6b). The IAWP approach predicted sediment yields ranging 79–160 g m^{-2} for the associated 20 plot pair aggregations. The OAWP approach predicted sediment yields ranging 153–290 g m^{-2} for the same plot pair aggregations. For the IAWP approach, only seven of these plot pair aggregations had bare soil ≥52.5% (ground cover ≤ 47.5%), conditions in which RHEM applies a splash and sheet erodibility equation (K_{ss}) for simulating more erodible and degraded conditions (see Appendix S1: Table S1) (Al-Hamdan et al., 2017; Hernandez et al., 2017). K_{ss} as derived by RHEM increases gradually as bare soil increases from 0% to 52.5% (47.5%–100% ground cover) and then increases sharply with increases in bare soil beyond this threshold (Al-Hamdan et al., 2017). For the OAWP approach, five of the intercanopy plots simulated and used to compile RHEM output plot aggregations had bare soil ≥52.5%, yielding 26 plot pairs with intercanopy runs using the K_{ss} for more erodible degraded cover conditions. Twenty of these 26 plot pairs are the data points driving deviation from a 1:1 relationship for OAWP versus IAWP RHEM-predicted sediment yield (Figure 8b). These results indicate the OAWP approach likely predicts greater splash and sheet erodibility and associated sediment yield for woodlands with extensive bare intercanopy area (bare soil ≥52.5%; ground cover ≤ 47.5%) through greater weighting of the K_{ss} equation for degraded cover conditions (Appendix S1: Table S1). Regardless, there were no significant differences in IAWP and OAWP predicted sediment yield for untreated and for burned conditions in this study (Table 5). But, the noted deviation from a 1:1 relation for sediment predictions for the approaches (Figure 8b) and greater predicted erodibility coefficients for some OAWP models (untreated at Marking Corral; Figure 9e, Table 5) demonstrate potential for the OAWP approach to predict greater sediment yield than the IAWP approach at the patch scale.

The RHEM OAWP and IAWP runoff and sediment predictions for aggregated patch-scale intercanopy and canopy microsites mirrored treatment effects observed in tree canopy and intercanopy rainfall simulation experiments (Tables 4 and 5). Measured runoff and sediment for aggregated tree canopy and intercanopy plot pairs in rainfall simulation experiments were 2- to 6-fold and 2.6- to 51.8-fold less for burned than untreated conditions at the sites (Table 5). The differences reflect substantial

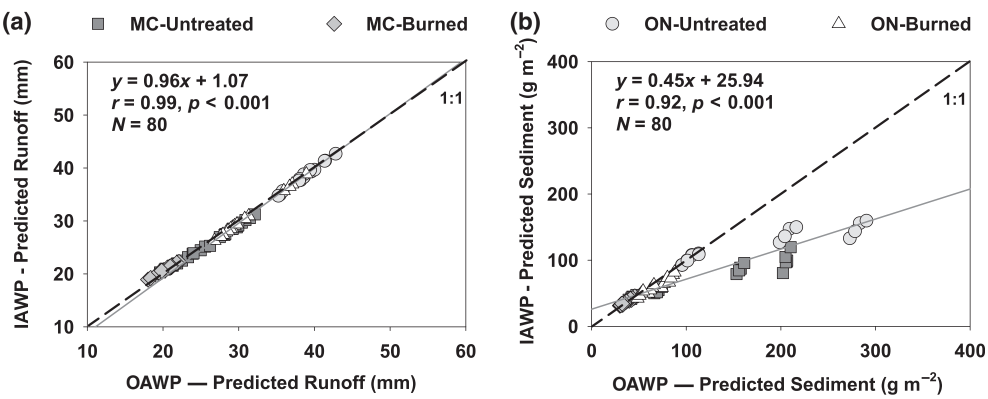

FIGURE 8 Rangeland Hydrology and Erosion Model (RHEM) predicted cumulative runoff (a) and sediment (b) for patch-scale (12 m^2) output area-weighted (OAWP) versus input area-weighted (IAWP) aggregated tree canopy and intercanopy model runs in untreated and burned areas 9 years postfire at the Marking Corral (MC) and Onaqui (ON) study sites. Data and RHEM inputs and RHEM configuration provided in Appendix S1: Table S3.

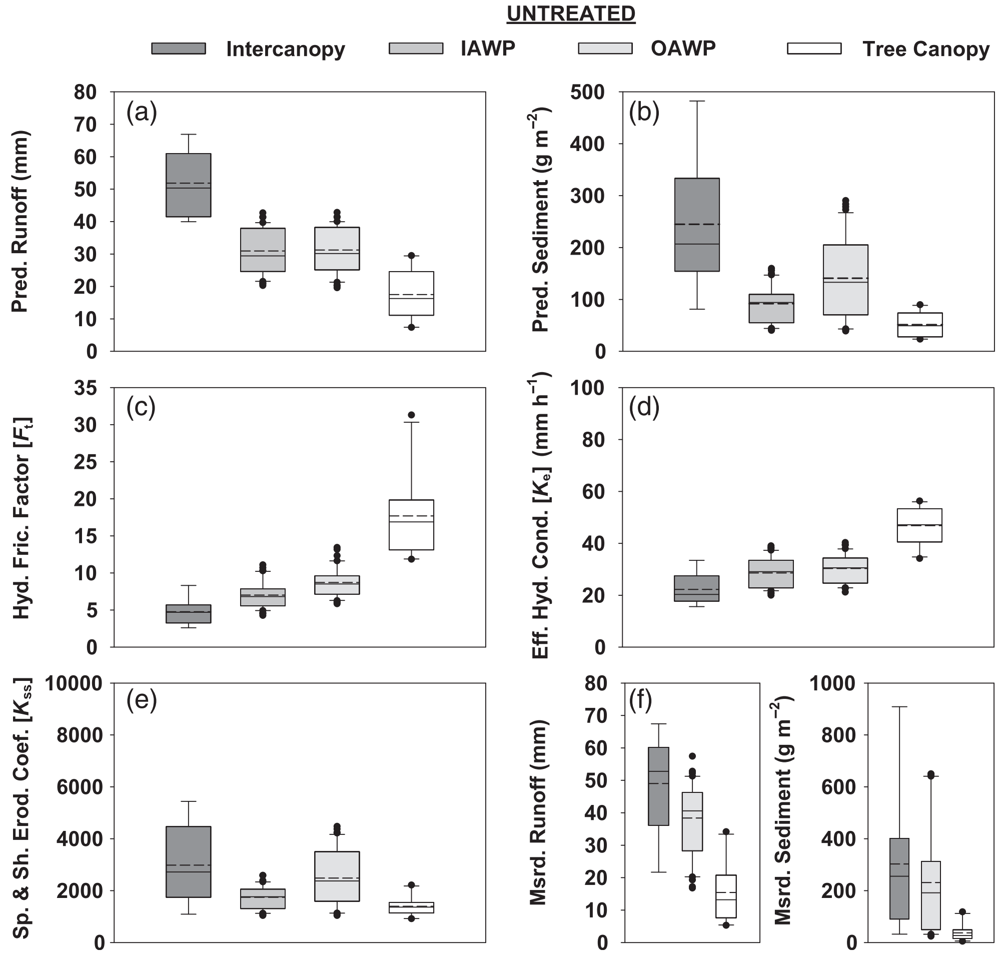

FIGURE 9 Rangeland Hydrology and Erosion Model (RHEM) predicted (Pred.) cumulative runoff (a) and sediment (b); hydraulic friction factor (Hyd. Fric. Factor [F_t]) (c), effective hydraulic conductivity (Eff. Hyd. Cond. [K_e]) (d), and splash and sheet erodibility coefficient (Sp. & Sh. Erod. Coef. [K_{ss}]) (e) derived RHEM parameters; and rainfall simulation measured (Msrd.) cumulative runoff and sediment (f) for patch-scale (12 m^2) intercanopy and tree canopy rainfall simulation plots and input area-weighted (IAWP) and output area-weighted (OAWP) tree canopy and intercanopy plot aggregations in untreated areas across the Marking Corral and Onaqui study sites. Data and RHEM inputs and configuration provided in Appendix S1: Tables S2 and S3. Mean value for each box indicated by dashed line; median value for each box indicated by solid line. Outlier data points (solid circles) are values outside the 10th and 90th percentiles.

runoff and sediment reductions associated with vegetation recovery in intercanopy areas (~70% of area) at both sites 9 years postfire (Table 4, Figures 4b and 5b). The RHEM IAWP and OAWP predicted runoff and sediment levels were generally less than measured values for rainfall simulations on aggregated tree canopy–intercanopy plot pairs with exception of the burned plots at Marking Corral (Table 5). Regardless, RHEM IAWP and OAWP predicted runoff and sediment for burned conditions across both sites were, on average, 1.3-fold and 2.5-fold less, respectively, than for untreated conditions (Table 5). Both area-weighted approaches predicted increased hydraulic friction (by 1.8-fold) and effective hydraulic conductivity (by 1.2-fold) for the burned treatment associated with postfire ground cover recruitment and reduced bare soil exposure at the patch scale (Tables 2 and 5). The IAWP and OAWP approaches also predicted twofold lower splash and sheet erodibility coefficients for burned than unburned conditions across both sites in association with fire-induced increases in vegetation and ground cover (Tables 2 and 5). The results indicate both approaches detected measured fire-induced treatment effects for tree canopy and intercanopy aggregations and provide good relative indication of improved hydrologic function and reduced erosion potential following tree removal.

Model approaches and treatment effects at the hillslope scale

With few exceptions, the various hillslope-scale RHEM frameworks predicted similar within-treatment runoff,

TABLE 5 Measured and predicted patch-scale (12 m^2) runoff and sediment and hydrologic and erosion model parameters for paired rainfall simulation plots in Rangeland Hydrology and Erosion Model (RHEM) runs of input area-weighted (IAWP) and output area-weighted (OAWP) aggregated tree canopy and intercanopy plots in untreated (Unt) and burned (Burn) areas at Marking Corral and Onaqui.

	Marking Corral				Onaqui			
	IAWP		OAWP		IAWP		OAWP	
Parameter	Unt	Burn	Unt	Burn	Unt	Burn	Unt	Burn
Rainfall simulation—measured								
Runoff (mm)[a]	-	-	36 b	6 a	-	-	42 b	21 a
Sediment ($g\,m^{-2}$)[a]	-	-	259 b	5 a	-	-	189 b	73 a
RHEM—predicted								
Runoff (mm)	26 b	21 a	26 b	20 a	38 b	31 a	39 b	31 a
Sediment ($g\,m^{-2}$)	71 b	36 a	119 b	35 a	123 b	61 a	176 b	69 a
RHEM parameters								
F_t	7.3 a	13.5 c	8.7 b	15.9 d	6.6 a	13.2 c	8.8 b	14.8 c
K_e ($mm\,h^{-1}$)	33 a	38 b	34 a	40 c	23 a	28 c	25 b	30 c
K_{ss}	1556 b	927 a	2319 c	927 a	2025 b	1178 a	2730 b	1245 a
No. plots	24	24	24	24	16	16	16	16

Note: See Appendix S1: Table S3 for RHEM inputs and configuration. Within-site (Marking Corral or Onaqui) means for approach (IAWP and OAWP) and treatment (Unt and Burn) combinations in a row followed by different lowercase letters are significantly different ($p < 0.05$).

Abbreviations: F_t, hydraulic friction factor; K_e, effective hydraulic conductivity; K_{ss}, splash and sheet erodibility coefficient.

[a]Area-weighted (30% tree canopy and 70% intercanopy) means for measured runoff and erosion from paired tree canopy and intercanopy rainfall simulation plots in untreated and burned areas at Marking Corral and Onaqui. Data from Nouwakpo et al. (2020; Appendix S1: Table S2).

and the area-weighted hillslope frameworks (IAWH and OAWH) predicted the highest within-treatment sediment yields (Tables 6 and 7). Predicted sediment, effective hydraulic conductivity, and erodibility coefficients were commonly greatest for the OAWH framework, followed by IAWH, UH, and the UH + T frameworks (Figure 11b–f). For cut and mastication treatments, there were no significant differences in predicted within-treatment runoff, sediment, or RHEM model parameters across the UH and UH + T frameworks. Regressions of predicted runoff and sediment and model parameters with ground cover attributes indicate all frameworks effectively represent the controls on runoff and erosion processes observed in patch-scale rainfall simulations and as expected for the various model parameter estimation equations (Figure 11). Across both sites and all treatments and frameworks, predicted runoff and sediment were largely controlled by bare ground (Figure 11a,c), and predicted sediment was strongly related to predicted runoff (Figure 11b). Predicted runoff and sediment were both generally highest for untreated bare conditions (Figure 11a–c). As expected, RHEM model parameters (F_t, K_e, and K_{ss}) were all also well predicted by bare ground (Figure 11d–f), owing to inclusions of the inverse, ground cover variables, in the associated parameter estimation equations (Appendix S1: Table S1). As at the patch scale, predicted runoff, sediment, the hydraulic friction factor, and the effective hydraulic conductivity were all well predicted by litter cover ($p < 0.001$). Although the regression results are intuitive given the parameter estimation equations (Appendix S1: Table S1), the presented relationships demonstrate RHEM responded to the primary controls on runoff and erosion processes at the study sites and effectively parameterized for untreated and treated conditions (Figure 11).

The effects of tree removal on hydrologic vulnerability and erosion potential were detected across all treatments with each of the four model frameworks (Tables 6 and 7). Predicted runoff was significantly lower (1.3-fold) and effective hydraulic conductivity higher (1.2-fold) for burned than untreated conditions across both sites for all but the UH and UH + T framework at Marking Corral (Tables 6 and 7). Predicted runoff was similar for the UH and UH + T frameworks for burned conditions at Marking Corral and 1.6-fold greater than that predicted by the IAWH and OAWH scenarios for burned conditions at that site (Table 6). We attribute the greater UH and UH + T predicted runoff on burned plots at Marking Corral to differences in litter cover in plot configurations for the various frameworks. Litter cover for burned conditions at Marking Corral averaged approximately 40% for the UH and UH + T frameworks and 65% for the IAWH and OAWH framework models. Litter cover across the same conditions at Onaqui averaged approximately 35% for the

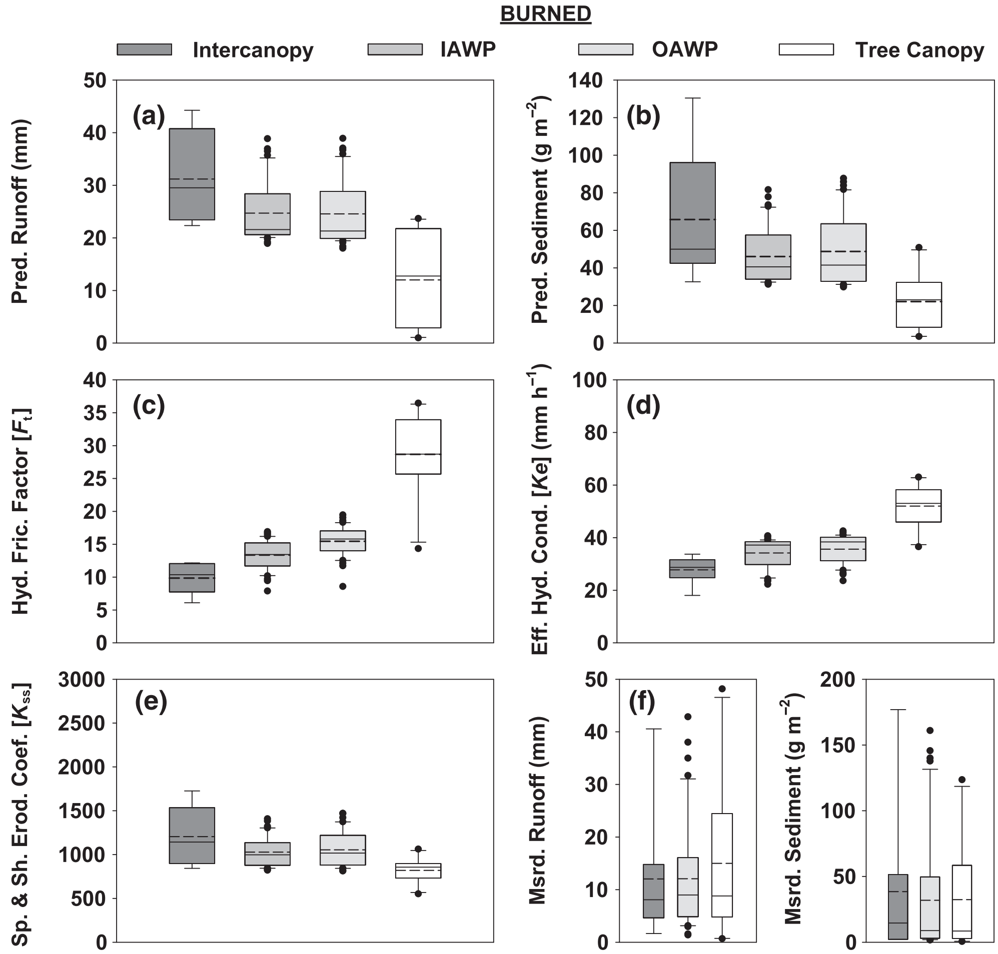

FIGURE 10 Rangeland Hydrology and Erosion Model (RHEM) predicted (Pred.) cumulative runoff (a) and sediment (b); hydraulic friction factor (Hyd. Fric. Factor [F_t]) (c), effective hydraulic conductivity (Eff. Hyd. Cond. [K_e]) (d), and splash and sheet erodibility coefficient (Sp. & Sh. Erod. Coef.) [K_{ss}] (e) derived RHEM parameters; and rainfall simulation measured (Msrd.) cumulative runoff and sediment (f) for patch-scale (12 m^2) intercanopy and tree canopy rainfall simulation plots and input area-weighted (IAWP) and output area-weighted (OAWP) tree canopy and intercanopy plot aggregations in burned areas 9 years posttreatment across the Marking Corral and Onaqui study sites. Data and RHEM inputs and configuration are provided in Appendix S1: Tables S2 and S3. Mean value for each box indicated by dashed line; median value for each box indicated by solid line. Outlier data points (solid circles) are values outside the 10th and 90th percentiles.

UH and UH + T frameworks and 46% for the IAWH and OAWH framework models, and predicted runoff for that site was consistent for burned conditions across all model frameworks (Table 7). Litter cover and basal plant cover are the two input variables in the effective hydraulic conductivity parameter estimation equation for RHEM (Appendix S1: Table S1). Across all frameworks, predicted sediment and erodibility coefficients for burned areas were, on average, twofold less than the same predicted variables in untreated areas. And, hydraulic friction factors, on average, increased 1.7-fold by burning relative to those predicted for untreated areas in all frameworks with exception of the UH and UH + T frameworks at Marking Corral (Tables 6 and 7). The preponderance of evidence across frameworks for burned conditions 9 years postfire indicates burning reduced runoff and sediment yield (Tables 6 and 7) through recruitment of understory vegetation and associated ground cover (Tables 2 and 3). Tree removal by cutting as assessed by the UH and UH + T frameworks also reduced runoff (by 1.3-fold on average) and resulted in 1.2-fold higher predicted effective hydraulic conductivities across both sites (Tables 6 and 7). The UH framework predicted 1.8-fold less sediment and 1.5-fold lower erodibility coefficients for the cut treatment than untreated areas across both sites, but there were no significant effects on these variables for the cut treatment with the UH + T framework (Tables 6 and 7). Predicted sediment and erodibility coefficients were generally less for untreated conditions in the UH + T than UH

TABLE 6 Predicted runoff and sediment and hydrologic and erosion parameters for paired rainfall simulation plots and site characterization vegetation plots in hillslope-scale Rangeland Hydrology and Erosion Model (RHEM) runs for untreated (Unt), burned (Burn), and cut (Cut) treatments 9 years post-tree removal at Marking Corral.

	Input area-weighted—IAWH		Output area-weighted—OAWH		Understory only—UH			Understory plus tree cover—UH + T		
Parameter	Unt	Burn	Unt	Burn	Unt	Burn	Cut	Unt	Burn	Cut
Runoff (mm)	22 Ba	17 Aa	22 Ba	16 Aa	25 Ba	26 Bb	19 Aa	24 Ba	26 Bb	19 Aa
Sediment ($g\ m^{-2}$)	65 Bb	33 Aa	109 Bb	31 Aa	51 Bb	29 Aa	29 Aa	34 Aa	27 Aa	28 Aa
F_t	7.3 Aa	13.5 Bb	8.7 Aa	15.9 Bc	6.8 Aa	7.1 Aa	8.6 Aa	6.8 Aa	7.1 Aa	8.6 Aa
K_e ($mm\ h^{-1}$)	33 Aab	38 Bb	34 Ab	40 Bc	29 Aa	28 Aa	35 Ba	30 Aa	28 Aa	35 Ba
K_{ss}	1556 Bbc	927 Ab	2319 Bc	927 Ab	1108 Cb	556 Aa	741 Ba	732 Ba	526 Aa	705 Ba
No. plots	24	24	24	24	6	3	3	6	3	3

Note: See Appendix S1: Table S4 for RHEM inputs and configuration. Within-framework (IAWH, OAWH, UH, or UH + T) treatment means (across Unt, Burn, and Cut) in a row followed by different uppercase letters are significantly different ($p < 0.05$). Within-treatment (Unt, Burn, or Cut) framework means (across IAWH, OAWH, UH, and UH + T) in a row followed by different lowercase letters are significantly different ($p < 0.05$).
Abbreviations: F_t, hydraulic friction factor; K_e, effective hydraulic conductivity; K_{ss}, splash and sheet erodibility coefficient.

TABLE 7 Predicted hillslope-scale runoff and sediment and hydrologic and erosion parameters for paired rainfall simulation plots and site characterization vegetation plots in hillslope-scale Rangeland Hydrology and Erosion Model (RHEM) runs for untreated (Unt), burned (Burn), cut (Cut), and mastication (Mast) treatments 9 years post-tree removal at Onaqui.

	Input area-weighted—IAWH		Output area-weighted—OAWH		Understory only—UH				Understory plus tree cover—UH + T			
Parameter	Unt	Burn	Unt	Burn	Unt	Burn	Cut	Mast	Unt	Burn	Cut	Mast
Runoff (mm)	33 Ba	26 Aa	34 Ba	26 Aa	34 Ba	25 Aa	25 Aa	24 Aa	31 Ba	25 Aa	25 Aa	24 Aa
Sediment ($g\ m^{-2}$)	119 Bbc	61 Aab	168 Bc	70 Ab	98 Bb	40 Aa	52 Aa	44 Aa	67 Ba	40 Aa	51 ABa	44 Aa
F_t	6.6 Aa	13.2 Ba	8.8 Aa	14.8 Ba	7.4 Aa	11.1 Ba	9.3 ABa	12.1 Ba	7.4 Aa	11.1 Ba	9.3 ABa	12.1 Ba
K_e ($mm\ h^{-1}$)	23 Aa	28 Ba	25 Ab	30 Ba	22 Aa	28 Ba	29 Ba	30 Ba	24 Aab	28 Ba	29 Ba	30 Ba
K_{ss}	2025 Bc	1178 Ab	2730 Bc	1245 Ab	1633 Bb	775 Aa	1045 Aa	922 Aa	1160 Ba	775 Aa	1039 ABa	921 Aa
No. plots	16	16	16	16	9	3	3	3	9	3	3	3

Note: See Appendix S1: Table S4 for RHEM inputs and configuration. Within-framework (IAWH, OAWH, UH, or UH + T) treatment means (across Unt, Burn, Cut, and Mast) in a row followed by different uppercase letters are significantly different ($p < 0.05$). Within-treatment (Unt, Burn, Cut, or Mast) framework means (across IAWH, OAWH, UH, and UH + T) in a row followed by different lowercase letters are significantly different ($p < 0.05$).
Abbreviations: F_t, hydraulic friction factor; K_e, effective hydraulic conductivity; K_{ss}, splash and sheet erodibility coefficient.

framework due to the additional cover of overstory trees. We attribute the lack of cut treatment effects on sediment yield and erodibility for the UH + T framework to the overstory tree cover added as shrub cover in that approach. There were no significant effects of tree cutting detected for the hydraulic friction factor as assessed by the UH and UH + T frameworks (Tables 6 and 7). Lastly, both the UH and UH + T frameworks predicted reductions in runoff (1.4-fold), sediment yield (1.9-fold), and erodibility coefficients (1.5-fold) and increases in hydraulic conductivity (1.3-fold) and the friction factor (1.6-fold) for the mastication treatment relative to untreated conditions at Onaqui (Table 7). Collectively, the UH and UH + T framework results indicate both tree cutting and mastication reduced hydrologic vulnerability and erosion potential at both sites as assessed 9 years posttreatment. The assertions on fire and mechanical-based treatment effects are based on relative differences in predicted responses to treatments with respect to responses for untreated areas. The propensity for the OAWH framework to generate the highest levels of sediment indicates that method may best capture processes as observed and measured in patch-scale bare intercanopy areas at both

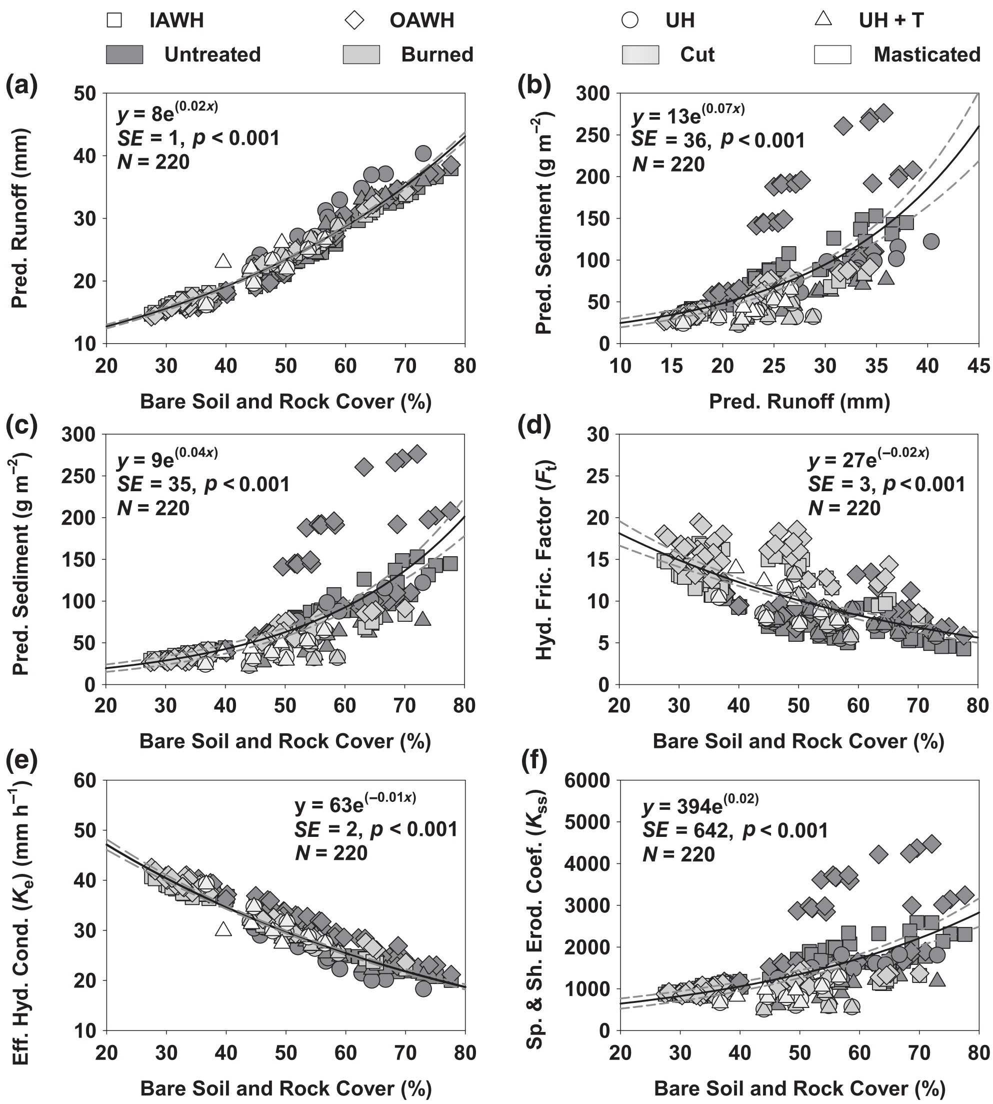

FIGURE 11 Correlations of predicted (Pred.; Rangeland Hydrology and Erosion Model (RHEM)) cumulative runoff (a), sediment (b and c), and model derived hydraulic friction factor (Hyd. Fric. Factor [F_t]) (d), effective hydraulic conductivity (Eff. Hyd. Cond. [K_e]) (e), and the splash and sheet erodibility coefficient (Sp. & Sh. Erod. Coef. [K_{ss}]) (f) with respective plot attributes for hillslope-scale model runs applying the input area-weighted (IAWH), output area-weighted (OAWH), understory only (UH), and understory only with tree cover (UH + *T*) frameworks in untreated and in treated (burned, cut, and masticated) areas 9 years after tree removal at Marking Corral and Onaqui. Dashed gray lines show 95% confidence interval around the solid black regression line. Data and RHEM inputs and configuration provided in Appendix S1: Table S4.

sites. More than 70% of the area at both sites is intercanopy, and those areas generate the most sediment at the patch scale, associated with formation of concentrated overland flow. Trends in OAWH sediment predictions at the hillslope scale are consistent with those for OAWP (patch scale) and with measured responses for intercanopy areas. Collectively, these results indicate the IAWH, UH, and UH + *T* frameworks may underestimate sediment yield where associated tree canopy and intercanopy aggregations underrepresent bare ground connectivity and thereby dampen the RHEM splash and sheet erodibility coefficient.

DISCUSSION

RHEM representation of patch-scale hydrologic and erosion processes for woodlands

Hydrologic and erosion responses (Table 4) and associated relationships with cover (Figure 7) quantified in this study are typical of woodland tree canopy and intercanopy hydrologic functional units, and the associated model results suggest RHEM effectively responds to these specific patch-scale attributes. Working with data across diverse western rangelands, Pierson et al. (2002) suggested hydrology and erosion models should organize around rangeland types and vegetation states according to similarities in relationships and responses and that model integration of such functional units would facilitate more accurate model predictions. Accordingly, for RHEM to effectively predict hydrologic and erosion responses in pinyon and juniper woodlands, the model must effectively represent tree canopy and intercanopy hydrologic functional units common to woodland landscapes. The RHEM was designed in part to predict how changes in vegetation and ground cover impact runoff and erosion processes on rangelands (as depicted in Figures 1 and 2) (Al-Hamdan et al., 2015; Hernandez et al., 2017; Williams et al., 2016a). Within the untreated woodlands in this study, measured and predicted runoff were 2- to 4-fold greater and sediment 4- to 14-fold greater for sparsely vegetated bare intercanopy areas relative to litter-covered tree canopy areas (Table 4). Such responses are typical for woodlands due to formation of concentrated overland flow through extensive bare intercanopy areas. Working at the same sites in this study, Pierson et al. (2010) found patch-scale runoff and erosion from rainfall simulations on untreated tree canopy and intercanopy plots were strongly correlated with percent bare soil and rock cover and that both response variables increased exponentially where combined bare soil and rock cover exceed 50%. The authors attributed exponential increases in runoff and sediment above this bare ground threshold to observed formation of high-velocity concentrated overland flow in intercanopy bare patches. Further, Pierson et al. (2010) reported similar relationships in litter cover with measured runoff and sediment yield as quantified for patch-scale plots in the current study. At a juniper-dominated sagebrush site in Idaho, USA, Pierson et al. (2013) found erosion during rainfall simulations in untreated tree canopy and intercanopy areas increased exponentially where bare ground exceeded 60% and attributed the relationship to concentrated flow processes in intercanopy bare patches. At a woodland-encroached sagebrush site in Utah, Roundy et al. (2017) found measured runoff from natural rainfall events increased exponentially where intercanopy bare area (inverse of vegetation foliar cover plus litter cover) exceeded about 40% and that sediment yield from natural rainfall events decreased rapidly where litter cover was more than 16% and bare ground was less than about 40%. Pierson et al. (2007) reported that high levels of runoff and sediment from rainfall simulations in degraded intercanopies at a juniper woodland in Oregon, USA, were attributable to concentrated flow through interconnected bare patches. Intercanopy bare ground exceeded 80% in that study (Pierson et al., 2007). The studies cited above all document structural and functional relationships for vegetation, ground cover, bare ground, and runoff and erosion processes common to woodland-encroached sagebrush sites associated with tree canopy and intercanopy hydrologic functional units. The RHEM's effective partitioning of patch-scale runoff, sediment yield, and model parameters for tree canopy and intercanopy plots at sites in this study clearly indicates the model can accurately predict responses associated with conditions representative of these hydrologic functional units (Figures 6 and 7).

Applicability of RHEM to assess tree-removal treatment effects on runoff and erosion processes in woodlands depends on model parameterization that reflects treatment-induced changes in vegetation and ground cover in tree canopy and intercanopy hydrologic functional units (Williams et al., 2016a, 2016b). Over a 9-year period, the prescribed fire treatments at sites in this study transitioned the woodlands to herbaceous-dominated landscapes (Figures 4a,b and 5a,b). Bare ground on burned tree canopy plots was generally greater at Onaqui (25%) than Marking Corral (7%), and total foliar was unchanged or increased on tree plots postfire. Bare ground was 1.3-fold less and total foliar cover 1.2- to 2.9-fold more on burned than unburned intercanopy plots across the sites (Table 2). The RHEM effectively predicted threefold reductions in runoff and sediment yield for burned relative to unburned tree canopy plots at Marking Corral and a lack of significant fire effects for the same measures across unburned and burned tree canopy plots at Onaqui (Table 4). For intercanopy plots, RHEM predicted near twofold less runoff and two- to sixfold less sediment yield for burned than unburned conditions (Table 4). The RHEM-predicted responses were consistent with observed fire impacts on runoff and sediment from rainfall simulation plots as assessed the ninth year postfire (Table 4). Further, RHEM-derived hydrologic and erosion parameters (F_t, K_e, and K_{ss}) exhibited similar treatment effects as measured and predicted runoff and sediment yield (Table 4). The observed shifts in vegetation structure from woodland to herbaceous-dominated 9 years following the prescribed fires (Figures 4a,b and 5a,b) are typical on woodland-encroached sagebrush steppe (Miller et al., 2013), as are the measured and predicted runoff and sediment responses (Table 4) in association with the respective cover transitions (Figure 1) (Pierson & Williams, 2016; Williams, Pierson, Al-Hamdan, et al., 2014; Williams, Pierson, Nouwakpo, et al., 2019; Williams, Pierson, Robichaud, Al-Hamdan, Boll, & Strand, 2016). The sagebrush and the pinyon and juniper species at sites in this study are commonly killed by fire and replaced by an initial flush of perennial and annual herbaceous species (Miller et al., 2013). Sagebrush and the trees can take 50+ and 70+ years to reestablish to pretreatment levels (Miller et al., 2013; Moffet et al., 2015; Ziegenliagen & Miller, 2009). Runoff and sediment yield generally decline over time postfire as ground cover increases and runoff transitions to more diffuse processes (Pierson & Williams, 2016; Williams, Pierson, Robichaud, & Boll, 2014). Williams, Pierson, Al-Hamdan, et al. (2014), Williams, Pierson, Nouwakpo, et al. (2019), and Roundy et al. (2017) found tree removal on woodland-encroached sagebrush sites can enhance hydrologic function by transitioning the intercanopy vegetation to shrub and herbaceous cover and recruiting litter ground cover. In this study, RHEM effectively parameterized similar patch-scale treatment effects on runoff and erosion processes associated with changes in vegetation and ground cover for tree canopy and intercanopy hydrologic functional units (Table 4, Figure 7).

The RHEM IAWP and OAWP approaches to aggregate tree canopy and intercanopy microsite responses predicted similar runoff and sediment yield responses as quantified by individual tree canopy and intercanopy rainfall simulations (Tables 4 and 5), but variability in sediment prediction across the two methodologies suggests the OAWP approach may more effectively attribute intercanopy concentrated flow processes. Consistent with the rainfall simulations by Nouwakpo et al. (2020), RHEM IAWP and OAWP simulations similarly predicted less patch-scale runoff for burned than untreated conditions at Marking Corral and Onaqui (Tables 4 and 5). Both approaches also predicted less sediment yield for burned than unburned conditions at the sites (Table 5), but sediment yield was generally greater for the OAWP approach (Figure 8b). A review of ground cover inputs for IAWP (Appendix S1: Table S3) and OAWP (Appendix S1: Table S2) model runs indicates that, with respect to the OAWP approach, fewer IAWP aggregated plot pairs had bare soil ≥52.5%. The OAWP approach compiled outputs from RHEM simulations of individual tree canopy and intercanopy plots (Appendix S1: Table S2), and those compilations generated 26 plot pair aggregations using untreated intercanopy plots with bare soil ≥52.5%. By contrast, only seven untreated plot pair aggregations for the IAWP approach had bare soil ≥52.5%. There are several ramifications from the discrepancy in representing bare soil in untreated areas. First, the splash and sheet erodibility coefficient (K_{ss}) as predicted in RHEM increases sharply where bare soil exceeds 52.5%. The OAWP predicted K_{ss} values for untreated areas were generally greater than those for IAWP, and the OAWP predicted K_{ss} values in untreated areas were the highest across the two approaches (Table 5, Figure 9e). Second, the higher K_{ss} values for OAWP than IAWP plot aggregations of untreated areas resulted in generally higher predicted sediment yields for OAWP across the two approaches (Figure 9b, Table 5). As such, the IAWP approach predicts a lesser, but still significant, treatment-induced reduction in sediment yield relative to the OAWP approach (Table 5). The OAWP approach may best simulate sediment yield at the sites. Earlier field studies at the sites found intercanopy runoff was generally similar across the fine ($0.5\ m^2$) to patch scales for untreated conditions, but that the dominant runoff processes in these areas shifted from splash and sheet at the fine scale to concentrated overland flow at the patch scale (Williams, Pierson, Robichaud, Al-Hamdan, Boll, & Strand, 2016). The runoff processes transition from the fine to patch scales resulted in increasing sediment yield across spatial scales attributable to well-connected bare ground throughout the intercanopy (Williams, Pierson, Robichaud, Al-Hamdan, Boll, & Strand, 2016). Similar cross-scale process transitions were reported in another woodland study by Williams, Pierson, Al-Hamdan, et al. (2014). Of the two aggregation approaches evaluated in the current study, OAWP better represents effects of these cross-scale process transitions through more effective weighting of intercanopy bare soil, associated erodibility (K_{ss}), and sediment yield. On this basis, the OAWP approach likely best quantifies patch-scale sediment yields for woodlands with extensive bare soil and/or where there is field evidence of extensive and well-connected concentrated overland flow patterns. For these conditions, we suggest the OAWP approach may provide a better assessment of treatment effects when applied in a relative difference context.

Assessment of hillslope model scenarios

With some caveats, each of the four RHEM hillslope model frameworks evaluated in this study were effective for assessing tree-removal treatment effects on woodland-encroached sagebrush steppe (Tables 6 and 7). Predictions of runoff and sediment yield for untreated conditions were statistically similar across all but $UH + T$ framework. Sediment yield for untreated conditions was generally highest with the OAWH framework, consistent with OAWP at the patch scale, followed by the IAWH and UH frameworks. In $UH + T$ framework, vegetation cover model inputs included the addition of measured overstory tree cover as shrub cover. Adding overstory tree cover as shrub cover limited predicted sediment yield for the untreated $UH + T$ model runs. It is difficult to discern plausibility of the limitation without comparable patch- or hillslope-scale runoff and sediment data for those conditions. For untreated woodlands, the areas underneath tree canopies can contain understory shrub and grass foliar layers and commonly have 90%–100% ground coverage by a thick litter layer. These litter-covered patches are hydrologically stable and typically, with respect to intercanopy areas, generate limited soil erosion regardless of the tree canopy (Pierson et al., 2010; Williams, Pierson, Al-Hamdan, et al., 2014). The RHEM model parameter estimation equations were developed without inclusion of overstory tree cover given the source datasets were from rainfall simulation experiments without tree cover (trees removed for simulations). We suggest omission of overstory tree cover for woodland model configurations on that basis but tested the effect of overstory tree cover here to assess model response. For burned conditions, runoff prediction was similar for all but the frameworks compiled with hillslope-scale vegetation and ground cover data on plots at Marking Corral (UH and UH + T frameworks; Table 6). We attribute the differences in predicted runoff for burned conditions with UH and UH + T versus the area-weighted frameworks (IAWH and OAWH) at Marking Corral to variations in litter cover across the patch- and hillslope-scale input datasets (Tables 2 and 3). The lesser runoff predicted by the UH and UH + T frameworks did not impact interpretation of fire treatment effects on erodibility and sediment yield at the hillslope scale (Table 6). Sediment yield for burned conditions was generally consistent across all frameworks (Tables 6 and 7). For mechanical treatments, runoff and sediment yield predictions with the UH and UH + T frameworks were each similar at a site. Trends across within-treatment RHEM-derived model parameters (F_t, K_e, and K_{ss}) were generally consistent with predicted runoff and erosion trends at a site (Tables 6 and 7).

As discussed above for patch-scale approaches, the OAWH framework likely best represents cross-scale process transitions and sediment delivery for sites with extensive bare intercanopy area, particularly for sites with documented field evidence of extensive and well-connected concentrated overland flow patterns. In such conditions, the OAWH framework likely best quantifies relative differences in untreated and treated conditions with respect to the other frameworks through effective weighting of intercanopy erodibility and sediment yield. Data sources for RHEM simulations likely dictate framework selection for most applications. For example, if only hillslope-scale understory cover data are available, then UH is the only practical framework. An end user could apply either of the area-weighted frameworks in cases in which tree canopy and intercanopy patch-scale cover data are available. In our assessment here, the OAWH framework generated the highest sediment yield, but both the IAWP and OAWP frameworks were effective in testing fire effects with patch-scale rainfall simulation plot data. This suggests either the IAWH or OAWH framework is plausible where patch-scale cover data are available, but we caution that the IAWH framework may underestimate relative differences between simulated treatments spanning extensive bare conditions. We therefore suggest the OAWH framework for applications where separate tree canopy and intercanopy data are available. A final and more intensive alternative is to run multiple frameworks and use a "preponderance of evidence" approach in interpreting RHEM results.

Assessment of hydrologic vulnerability and erosion potential

The various modeling approaches evaluated in this study serve as RHEM frameworks for evaluating tree-removal impacts on hydrologic vulnerability and erosion potential. All of the RHEM frameworks applied indicate prescribed fire-reduced hydrologic vulnerability and erosion potential at the study sites through recruitment of vegetation and ground cover into previously bare intercanopies over a 9-year period after treatment (Tables 6 and 7, Figures 4b and 5b). The RHEM-based inferences are consistent with field studies at both sites (Nouwakpo et al., 2020; Williams, Pierson, Nouwakpo, et al., 2020). Williams, Pierson, Nouwakpo, et al. (2020) found increased herbaceous cover in 0.5-m^2 interspace plots at both sites 9-year postfire improved infiltration and reduced erosion by approximately twofold for high-intensity rainfall simulations. In the companion study, Nouwakpo et al. (2020) further determined that the enhanced interspace infiltration at both sites reduced requisite intercanopy runoff (by more than 4-fold) and sediment yield (by 3- to >70-fold) during similar rainfall simulation experiments at the patch scale. In context, the RHEM simulations applied in this study clearly represent the fine- to patch-scale improved infiltration and reduced erosion potential quantified in the field experiments at both sites.

The RHEM frameworks applied for mechanical treatments (UH and UH + T) at Marking Corral (Figure 4c) and Onaqui (Figure 5c,d) suggest tree removal by cutting and mastication reduced hydrologic vulnerability and erosion potential through retention of existing shrub cover and increases in foliar and ground cover in intercanopies. Predicted runoff and sediment yield were generally similar across the two treatments and were less than that predicted for untreated conditions (Tables 6 and 7). A field study by Williams, Pierson, Kormos, et al. (2019) 9 years after the mechanical treatments reported increased hillslope-scale shrub and herbaceous foliar covers at Marking Corral and Onaqui. Rainfall simulation experiments in that study found interspace infiltration rates in the mechanical treatments were slightly greater than those in untreated interspaces, but the differences were not significant due to high variability in the measures. Williams, Pierson, Kormos, et al. (2019) inferred that infiltration rates in degraded interspace microsites were improving following mechanical treatments at both sites but to a lesser degree relative to measured interspace infiltration rates in burned treatment areas, as reported by Williams, Pierson, Nouwakpo, et al. (2020). Williams, Pierson, Kormos, et al. (2019) further found that tree debris in intercanopy areas within the mechanical treatments at both sites was effective in reducing flow velocity, runoff, and erosion rates during concentrated overland flow experiments, but the study did not measure runoff and sediment yield from rainfall application at the patch scale. The RHEM hillslope simulations in the current study indicate the hillslope cover enhancements, patch-scale reduced overland flow erosion, and slight fine-scale increases in infiltration in mechanical treatment areas, as quantified by Williams, Pierson, Kormos, et al. (2019), likely indeed were indicators of reduced hydrologic vulnerability and erosion potential relative to untreated conditions. Pierson et al. (2007) reported similar results 10 years following a cutting treatment at juniper site in Oregon. In that study, tree removal by cutting on a juniper-encroached sagebrush site stimulated intercanopy herbaceous plant cover, improved infiltration, and reduced erosion rates as assessed by rainfall simulation experiments. In the aforementioned 5-year study at a woodland-encroached sagebrush site in Utah, Roundy et al. (2017) found that tree removal using a combined chaining and seeding treatment increased intercanopy vegetation and ground cover within the first year posttreatment and reduced intercanopy runoff and sediment yield from natural rainfall events in the second to fifth years after treatment. Collectively, the studies by Williams, Pierson, Kormos, et al. (2019), Pierson et al. (2007), and Roundy et al. (2017) demonstrate mechanical treatments are effective in improving vegetation, ground cover, and hydrologic function as also predicted for the Marking Corral and Onaqui sites applying the UH and UH + T frameworks.

CONCLUSIONS

Results from our comparisons of measured and predicted patch-scale runoff and sediment yield clearly demonstrate RHEM's effectiveness in parameterizing landscape attributes that regulate hydrologic and erosion processes in untreated and treated woodland-encroached sagebrush steppe. The RHEM performance in predicting runoff and sediment yield was "good" to "very good," the model accurately represented differences in tree canopy versus intercanopy ecohydrologic function, and RHEM replicated respective measured microsite long-term responses to tree removal by burning. Magnitudes in predicted versus measured runoff and sediment yield were variable, but measured and predicted hydrologic and sediment responses clearly indicated tree removal by burning reduced patch-scale runoff and sediment yield as assessed 9 years postfire. Measured runoff and sediment responses to applied rainfall were clearly and strongly regulated by the amount of bare ground and ground cover for both untreated and burned conditions. The RHEM parameter estimation equations for hydraulic friction, hydraulic conductivity, and splash and sheet erodibility effectively represented these controls, and predicted runoff and sediment yield exhibited similar relationships with bare ground and ground cover as requisite measured response variables. The RHEM model approaches to aggregate patch-scale tree canopy and intercanopy microsite effects predicted similar trends in hydrologic and erosion responses as measured for untreated and burned conditions in rainfall simulation experiments. The RHEM aggregated approaches suggest tree removal by fire-reduced patch-scale runoff and sediment yield, but reductions, as expected, were less than those measured and predicted for nonaggregated microsite effects. Although both RHEM patch-scale approaches detected treatment impacts on hydrologic vulnerability and erosion potential, the magnitude of treatment effects was best captured through the RHEM output area-weighted (OAWP) approach that most effectively weighted intercanopy bare ground, erodibility, sediment yield, and connectivity of runoff and erosion sources and processes.

Model results for the hillslope scale indicate RHEM is effective in representing the dominant controls on runoff and erosion processes in woodlands and as a tool to assess the impacts of woodland encroachment and tree removal on hydrologic vulnerability and erosion potential. Within-treatment predicted runoff and sediment responses across RHEM frameworks showed some variability, but all frameworks captured treatment effects associated with tree removal by burning, cutting, and/or mastication treatments. Area-weighted aggregated tree canopy and intercanopy RHEM frameworks (IAWH and OAWH) both showed burning reduced runoff and sediment yield. The RHEM frameworks compiled with hillslope-scale measured understory vegetation and ground cover (UH) and addition of overstory tree cover as shrubs (UH + T) likewise indicate burning reduced runoff and sediment yield at one site, but there was either no change or lesser reductions predicted for the other site. These results are associated with variability in measured litter cover across patch and hillslope scales and illustrate the complexity of cross-scale assessments. Predicted RHEM runoff and sediment responses for mechanically treated versus untreated hillslopes clearly indicate tree removal by cutting and mastication reduced hydrologic vulnerability and erosion potential at the sites. Fire and mechanical treatment effects for some response variables were dampened by the inclusion of tree overstory cover as a model input (UH vs. UH + *T* frameworks). The dampening is assumed associated with a duplication effect of multiple foliar layers and the associated litter cover already included as a ground cover input in RHEM. The model results suggest application of the UH framework without tree cover may better represent treatment effects at the hillslope scale, but both scenarios could be applied where desired to evaluate for "preponderance of evidence." As with the patch scale, relationships in predicted hillslope-scale responses with changes in bare ground and ground cover across conditions illustrate RHEM's effective parameterization of the dominant controls on runoff and erosion processes for woodlands. All RHEM hillslope frameworks captured treatment effects, but the OAWH framework most effectively quantified and weighted effects of intercanopy bare ground, erodibility, sediment yield, and process connectivity on hydrologic function and erosion potential. We therefore suggest use of the RHEM OAWH framework for model applications on untreated woodland landscapes with well-connected degraded surface conditions and associated physical processes and for assessing treatment effects on woodlands in general when separate tree canopy and intercanopy input data sources are available. For most users, selection of a preferred RHEM framework is likely dictated by the scale and type of data available for model input. All the RHEM frameworks presented in this study are effective in assessing hydrologic vulnerability and erosion potential on woodlands when applied in a relative difference approach.

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CONFLICT OF INTEREST

The authors declare no conflict of interest.

DATA AVAILABILITY STATEMENT

Data are previously published and publicly available, with those items properly cited in the article.

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SUPPORTING INFORMATION

Additional supporting information may be found in the online version of the article at the publisher's website.

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