RUNOFF AND EROSION FOLLOWING A PRESCRIBED FIRE ON A SAGEBRUSH-STEPPE RANGELAND IN IDAHO, USA

William J. Elliot[1,*,†]

[1] USDA Forest Service, Rocky Mountain Research Station, Moscow, Idaho, USA
[*] Correspondence: welliot@moscow.com
[†] Retired.

HIGHLIGHTS

- A spike in erosion occurred the year following a prescribed burn on a sagebrush-steppe rangeland.
- Two years after a prescribed burn on a sagebrush-steppe rangeland, burned plots had less runoff and erosion than unburned plots.
- Snowmelt processes may be important contributors to erosion in some sagebrush-steppe rangelands.

ABSTRACT. *A study was carried out to compare runoff and erosion from natural rainfall on plots that had been treated with prescribed fire to unburned plots on a sagebrush-steppe rangeland in Southeast Idaho, U.S. Prescribed fire on rangeland sites is intended to maintain healthy shrub-steppe ecosystems but sometimes results in undesirable consequences such as increased runoff and soil erosion. Information is lacking on plot-scale erosion studies from natural precipitation in this ecosystem. Such plot-scale studies are needed to better understand sediment sources (uplands or channels) to support the management and modeling of rangeland watersheds. In this study, there were two treatments: four prescribed burn plots and three adjacent unburned control plots. Runoff and erosion were measured from natural rainfall for water years (WY) 2004 to 2010 following a prescribed burn in October 2003. Runoff and erosion were also measured from nearby unburned plots for WY 2005–2010. Ground cover on the burned plots averaged 57% (standard error (s.e. = 4%) in the summer of 2004, compared to 95% (s.e. = 5%) on the unburned plots. By the end of the study in 2010, ground cover had increased to 81% (s.e. = 4%) on the burned plots but decreased to 74% (s.e. = 5%) on the unburned plots. Annual runoff averaged 6 mm (s.e. = 7) from four burned plots, compared to 34 mm (s.e. = 8.7) from three unburned plots. In WY 2006, high rates of runoff from snowmelt on the unburned plots resulted in 122 mm (s.e. = 16) of runoff compared to only 12 mm (s.e. = 14) of runoff from the burned plots. An analysis of variance showed significant differences in runoff due to either precipitation (p = 0002) or year (p = 0.004) and treatment (burned vs. unburned; p=0.03). There were also significant differences in seasonal runoff (p = 0.05), as 90% of the measured runoff occurred in the spring, with all large runoff events associated with snowmelt. Erosion on the burned plots averaged 233 kg ha^{-1} (s.e. = 21) compared to 133 kg ha^{-1} (s.e. = 29) on the unburned plots. From two years after the burn and for the remainder of the study, there were no significant differences in erosion between burned and unburned plots (p < 0.05). Future studies are needed to link upland runoff and erosion with channel deposition, erosion, and sediment delivery, and more detailed studies on erosion associated with snowmelt on rangelands are needed to aid in the development of watershed modeling tools. Future studies should include observations of plant community regeneration in addition to ground cover, runoff, and sediment delivery.*

Keywords. *Erosion, Hydrology, Prescribed fire, Rangelands.*

An interdisciplinary research project was carried out following a prescribed burn in October 2003 on a shrub-steppe ecosystem in Southeast Idaho, U.S. The effects of the burn on canopy, soil quality, and insect populations were reported in Page-Dumroese et al. (2023). The effects of that prescribed burn on ground cover, runoff, and erosion compared to a nearby unburned site are reported in this article.

Rangelands encompass approximately 31% of the land area in the United States (U.S.), with the federal government managing 62 million ha (Lubowski et al., 2006), located primarily in the West. The Great Basin region (fig. 1a) of the

Submitted for review on 17 July 2023 as manuscript number NRES 15738; approved for publication as a Research Article by Associate Editor Dr. Devendra Amatya and Community Editor Dr. Kati Migliaccio of the Natural Resources & Environmental Systems Community of ASABE on 16 April 2024.

Mention of company or trade names is for description only and does not imply endorsement by the USDA. The USDA is an equal opportunity provider and employer.

Citation: Elliot, W. J. (2024). Runoff and erosion following a prescribed fire on a sagebrush-steppe rangeland in Idaho, USA. *J. ASABE, 67*(6), 1481-1498. https://doi.org/10.13031/ja.15738

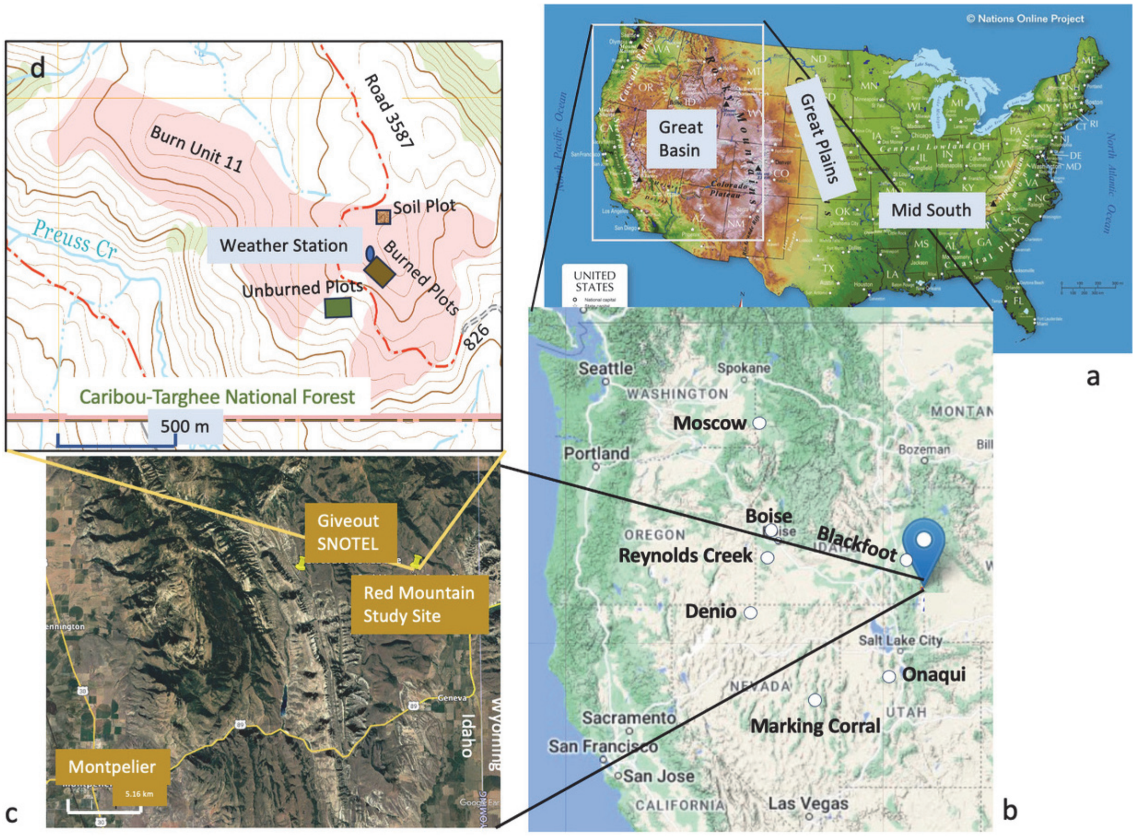


Figure 1. Locations of (a) geographic regions in the U.S.; (b) this study's research site and rainfall simulation sites near Boise, Reynolds Creek Experimental Watershed and Blackfoot, Idaho, Denio and Marking Corral, Nevada, and Onaqui, Utah; (c) the Red Mountain study site, Giveout SNOTEL site, and Montpelier, Idaho; and (d), the erosion research plots, one of the three soil quality plots (Page-Dumroese et al., 2023), weather station, and burn unit 11 (shaded in pink; Johnson, 2003) on the Red Mountain burn erosion study (sources of base maps are (a) Nationsonline.org; (b) Google Maps; (c) Google Earth; (d) U.S. Geological Survey).

western U.S. is located between the Rocky Mountains on the east and the Sierra Nevada Mountains on the west. Shrub-steppe (shrub and bunchgrass) plant communities dominate the land cover. The community is a biologically rich ecosystem, with sagebrush (*Artemesia* spp.) being the dominant shrub (Brooks et al., 2015; Dumroese et al., 2015). Wildfire is a natural occurrence in the sagebrush-steppe ecosystem and has become more frequent in recent decades (Brooks et al., 2015).

Prescribed fire is used on grasslands and rangelands for fuel and litter reduction, increasing available nutrients and soil water for desirable species, and ecosystem restoration, mimicking the beneficial properties of wildfire (Brooks et al., 2015; Bunting et al., 1987; Engle and Bidwell, 2001; Fuhlendorf et al., 2011; Harper et al., 2011). Prescribed fire has been beneficial for reducing invasive tree encroachment in both Mid-South U.S. and Great Basin U.S. ecosystems (Harper et al., 2011; Williams et al., 2020; fig. 1a). One of the main invasive plants in the Great Basin is cheatgrass (*Bromus tectorum* L.), but prescribed fire has increased the susceptibility of species establishment (Keely and McGinnis, 2007; Williams et al., 2020). In the Mid-South U.S. (fig. 1a), the timing of the burn depends on the goal, with spring burns encouraging the growth of desirable grasses, mid-summer burns reducing woody succession, and late summer or early fall burns encouraging forb coverage for wildlife (Harper et al., 2011). In the Great Plains (fig. 1a), prescribed fire has increased production of perennial grasses with early spring burns by reducing mulch layers that build up, particularly in the absence of wildfire or grazing (Engle and Bidwell, 2001). Later spring burns sometimes reduce forage production, depending on weather and other site conditions. Engle and Bidwell (2001) reported that fall burns may increase forb production, which may be desirable, or not, depending on the rangeland management goals. In the northern Great Basin shrub-steppe ecosystem (fig. 1a), prescribed burns are usually carried out in the early fall when grasses are dormant (Bunting et al., 1987; Williams et al., 2016b, 2020). Bunting et al. (1987) reported that in the spring, the moisture content of the vegetation was usually too high for an effective burn, and summer burns could be detrimental to the indigenous grass community, as can late fall burns if grasses have begun to resprout. Fall burns may be delayed until the onset of autumn rains to increase the soil water content and limit the severity of the burn (Johnson, 2003).

Prescribed fire may result in a loss of ground cover that may increase erosion (DeBano, 1981; Inbar et al., 1998; Meeuwig, 1971; Pierson et al., 2001, 2002, 2008; Vega et al., 2020; Williams et al., 2016a,b). Post-fire elevated runoff and erosion may also be attributed to postfire water repellency (DeBano et al., 1970; Pierson et al., 2001) and loss of vegetation canopy (Williams et al., 2016a,b). Increased runoff following a fire may lead to increased sediment transport capacity and sediment delivery to channels (Pierson et al., 2008, 2009; Williams et al., 2016b). Williams et al. (2020), however, reported that prescribed fire on rangeland steppe ecosystems may lead to an increase in herbaceous cover, resulting in less runoff and erosion from burned sites compared to unburned sites. The temporal and spatial extent of fire effects will likely be site-specific due to interactions among soil, vegetation, topography, and weather (Pierson et al., 2008; Vega et al., 2020; Williams et al., 2016b).

Fire can change the amount of soil surface and mineral organic matter (OM), carbon (C), and nitrogen (N), thereby altering soil properties (DeBano and Klopatek, 1988; Gustine et al., 2021; Page-Dumroese et al., 2023) and vegetation recovery (Fenn et al., 2010). After a fire, a reduction in the canopy and increased soil water content may increase the decomposition rates of organic ground cover (Bradford et al., 2014) and runoff rates (Castillo et al., 2003; Nearing et al., 1996), leading to increased erosion risk. Fire may decrease aggregate stability, resulting in finer particles being more easily detached and transported during rainfall and runoff events (Mataix-Solera et al., 2011).

Fire can complicate rangeland hydrology. The two dominant processes that cause surface runoff are infiltration excess runoff, when precipitation intensity or snowmelt rate exceeds soil hydraulic conductivity, or saturation excess runoff, when the soil pore spaces are near saturation, so additional infiltration is limited to soil water loss through deep seepage, lateral flow, or evapotranspiration (Ward and Elliot, 1995). In shrubland plant communities, these processes are further complicated by differences in infiltration rates beneath shrub canopies and between shrubs (Pierson et al., 2001, 2003). Pierson et al. (2003) reported infiltration rates beneath shrubs of 56–65 mm h^{-1} and between shrubs of 40–50 mm h^{-1} on sandy loam soils. Following wildfire, the infiltration rates beneath shrub sites dropped to 54–56 mm h^{-1}, while between the shrubs, rates increased to 60 mm h^{-1}. Fire, whether it be a prescribed burn or a wildfire, results in a mosaic of burn conditions that can range from unburned to high severity. One of the hydrologic effects of spatial variability is the concept of "partial area hydrology" or "variable source area hydrology," where the fraction of a hillslope with low hydraulic conductivity may generate runoff from low intensity precipitation or snowmelt rate, but it is only when the precipitation rate exceeds the highest hydraulic conductivity on the hillslope that the overall hydraulic conductivity for the site can be determined (Engmun, 1974). Partial area hydrologic processes can be influenced by multiple surface conditions, like variability in burn severity and soil properties (Robichaud et al., 2007), the presence of roads or other development (Engmun, 1974), and topographic features, like swales and variations in slope steepness (Boll et al., 2015). Partial area hydrology not only affects surface runoff but also lateral flow, a common process on steep lands (Boll et al., 2015), and deep seepage recharging groundwater resources and downstream base flows (Engmun, 1974). Variability in hydrologic processes on colder rangelands is further complicated by snow accumulation and melt. Areas with more dense shrub cover (Sturgis, 1977) or swales on the landscape (Luce, 2000; Vega et al., 2020) can trap and accumulate snow, leaving areas with less dense vegetation or ridge tops with less snow accumulation. The spatial variability in snow depth and runoff means that during a large precipitation or snowmelt event, some areas of the landscape will generate runoff while other areas may be sites of net infiltration. On sites with significant amounts of rock, soil hydraulic conductivity is reduced (Brakenseik and Rawls, 1994). The distribution of rock, both spatially and with depth, can impact both surface and subsurface hydrology. The complex variability in runoff will lead to variability in soil detachment and deposition within a given landscape (Cao et al., 2021).

Numerous hydrologic studies have been conducted on rangelands following prescribed fires with rainfall and runoff simulation (Al-Hamdan et al., 2015; Moffet et al., 2007; Pierson et al., 2008, 2009; Williams et al., 2020). These studies all concluded that prescribed fire reduced ground cover and canopy, and increased runoff and erosion compared to unburned plots. Williams et al. (2020) reported that nine years following a prescribed fire, rainfall simulation runoff was less from burned plots, but erosion was generally greater compared to the control plots. Vega et al. (2020) reported on a study using silt fences to measure erosion following the 2015 Soda wildfire on a sage-steppe landscape near Reynold's Creek, ID (fig. 1b). They reported erosion rates ranging from 0.0 to 28 Mg ha^{-1} from 24 plots and 415 mm of precipitation. About 150 mm of runoff was observed from the entire 1.29 km^2 Murphy Creek watershed where the plots were installed the year following the fire. However, studies exploring runoff and erosion following prescribed fire associated with natural weather events are lacking, particularly multiyear studies with diverse weather patterns and snowmelt interacting with recovering vegetation.

The objectives of this study were to quantify the impacts of prescribed fire on surface cover, runoff, and erosion on a sagebrush-steppe site in the five years immediately following a prescribed burn. Specifically, the hypotheses were that (1) surface runoff and soil erosion will be greater from burned plots than from nearby unburned plots, and (2) in the subsequent years following a prescribed fire, ground cover will increase and runoff and erosion on burned plots will decrease to unburned levels.

Materials and Methods

Study Area

The study was located at approximately 42.42°N latitude and -111.09°W longitude within the Red Mountain Prescribed Burn in the Montpelier Ranger District of the Caribou-Targhee National Forest (Johnson, 2003). The elevation was 2100–2140 m. The site was approximately 27 km northeast of Montpelier, Idaho, in Bear Lake County (fig. 1c). The Red Mountain Prescribed Burn consisted of 19 burn units covering 642 ha within the total project area of 1515 ha. The stated purpose of the burn was to "increase the diversity of mountain big sagebrush structure by targeting the sagebrush that is in a late successional stage (canopy cover greater than 15%)" (Johnson, 2003). Johnson (2003) stated that the sagebrush canopy cover ranged from 27% to 40% within the proposed burn units (fig 2). The Red Mountain prescribed burn was initiated on 3 October 2003, when soil water content

Figure 2. Rangelands in the vicinity of the Red Mountain burn soil erosion study (photo by I. S. Miller).

was near field capacity, resulting in a mosaic of unburned and low- to moderate- severity burns.

The average annual precipitation for this area was 520 mm, and the mean annual temperature was ~1.5°C. The average annual minimum temperature was -3°C, and the average annual maximum temperature was 13°C. The general climate had cold, wet winters and hot, dry summers, with most of the precipitation occurring as snow from October through April (Johnson, 2003). Figure 3 shows the average monthly temperatures and precipitation depths from the Giveout SNOpack TELemetry (SNOTEL) network site (https://wcc.sc.egov.usda.gov/nwcc/site?sitenum=493) located 6 km west of the research site (fig. 1c) for water years (WY) 1989–2023. The estimated RUSLE *R-Factor* for the site was about 600 MJ mm ha^{-1} h^{-1} y^{-1} (Huffman et al., 2013).

The dominant preburn shrubs in the 1500-ha burn area included little sagebrush (*Artemisia arbuscula* Nutt.; 35% cover) and mountain big sagebrush (*Artemisia tridentata* Nutt. var. *pauciflora* Winward and Goodrich; 30% cover). Major grass species were Idaho fescue (*Festuca idahoensis* Elmer; 25% cover) and bluebunch wheatgrass (*Pseudoroegneria spicata* (Pursh) Á. Löve; 5% cover) (figs. 2 and 4). The main forb present on the site was Jones' fleabane (*Erigeron jonesii* Cronquist; 10% cover) (Page-Dumroese et al., 2023).

When the study was initiated in 2003, the Caribou-Targhee National Forest soil scientist stated that the site soil series was the Squawval-Bischoff family complex (Page-Dumroese et al., 2023). A more detailed soil survey of the

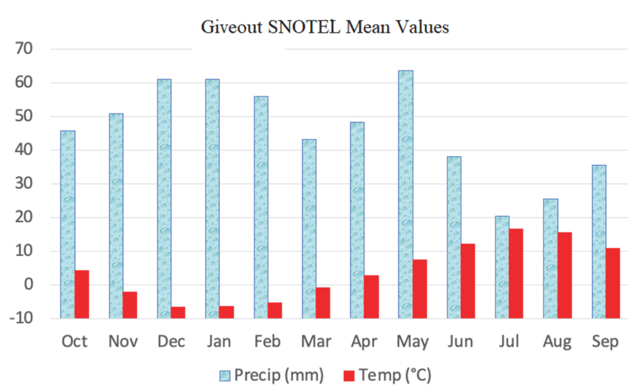


Figure 3. Average monthly precipitation and temperature from the Giveout SNOTEL site for WY 1989–2023 (https://wcc.sc.egov.usda.gov/nwcc/site).

Figure 4. Installation of an unburned plot. Note the upper plot border in the lower left corner and the cover over the tipping bucket installation at the bottom of the plot. The sagebrush canopy is the light green shrub, and the yellow flowering shrub is rabbitbrush (*Ericameria nauseosa*) on the Red Mountain burn erosion study (photo by I. S. Miller).

area was published in 2017, and the WEPP Cloud interface (Lew et al., 2022) to the U.S. Soil Survey Geographical Database (Reybold and TeSelle, 1989) showed that the soil beneath the burned plots was Poodle-Bishchoff family complex loam soil, but the nearby unburned plots were on a Yago stony silty clay loam. The textural details of these two soils are shown in table 1. In the spring of 2004, Page-Dumroese et al. (2023) installed soil quality plots, collected soil samples for nutrient analysis, and measured canopy cover on both burned and adjacent unburned sites. They found that soil nitrogen on the burned plots was significantly lower than nearby unburned plots ($p \leq 0.05$), but organic matter and bulk density were not significantly different between the two treatments (table 2).

A weather station was installed on the south side of a ridgeline near the burned plots (figs. 1d and 5). Precipitation, air temperature, humidity, solar radiation, and wind speed and direction were recorded continuously from 16 October 2003, through 27 August 2010. In April 2007, the rain gage malfunctioned, so precipitation data from the Giveout SNOTEL were accessed to support the study. The Giveout SNOTEL site was 6 km west of the site at an elevation of 2112 m (fig. 1c). Onsite air temperature data collection continued throughout the study.

Runoff and Erosion Plots

Four runoff and erosion research plots were installed on burn unit 11 (figs. 1d and 5) in October 2003, within days of the completion of the burn. The plots were located within the

Table 1. Soil properties from Lew et al. (2022) for the Red Mountain prescribed burn erosion study. *K Factor* estimated as per Huffman et al. (2013).

	Soil	
Property	Poodle-Bischoff	Yago
Treatment	Burned	Unburned
Clay (%)	16	30
Sand (%)	35	12
Silt (%)	49	58
Rock Content > 2 mm (%)	9	61
Organic Matter (%)	3	3
Depth (m)	1.2	1.6
K Factor (t ha h / ha MJ mm)	0.041	0.037

Table 2. Some post-burn soil chemical and physical properties for the 0- to 30-cm deep mineral soil core and vegetation cover for the burned plots and adjacent unburned sites 8 months after the burn on the Red Mountain prescribed burn site, mean values, and (standard error). Probability that the values were not significantly different ($p \leq 0.05$) are shown in bold (Page-Dumroese et al., 2023).

Soil and Vegetation Characteristics	Unburned	Burned	p-value[a],[b]
Soil			
Organic matter ($Mg•ha^{-1}$)	26.1 (7)	24.0 (6)	0.3
Carbon ($Mg•ha^{-1}$)	15 (5)	11 (4)	0.08
Nitrogen ($kg•ha^{-1}$)	1100 (21)	840 (20)	**< 0.001**
C:N	13 (0.2)	13 (1)	0.4
Bulk density ($Mg•m^{-3}$)	1.05 (0.3)	0.99 (0.4)	0.4
Vegetation Canopy Cover (%)			
Grass cover	30 (0.5)	35 (1)	**<0.001**
Forb cover	10 (0.5)	5 (0.4)	**<0.001**
Shrub cover	40 (3.2)	32 (4.0)	**0.01**

[a] Unburned versus prescribed fire soil properties were evaluated with a paired t-test.

[b] Bold p-values are significantly different ($p \leq 0.05$).

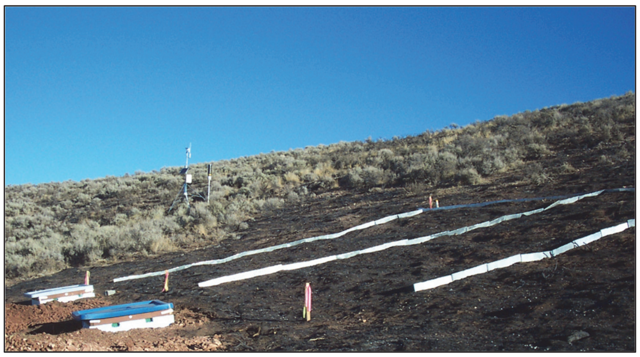


Figure 5. Installation of the plot borders on two of the burned plots on Burn Unit 11 in October 2003. On the lower left are the covers over the boxes containing the sediment basins and tipping bucket devices (fig. 7). The bottom gutters to divert runoff and sediment from the bordered plots to the sediment basins (fig. 6) have not yet been installed. The weather station is in the background still on Burn Unit 11, but the burn was less severe on the Red Mountain Burn Erosion study (photo by I. S. Miller).

burn unit, where the severity was considered moderate to high (Parsons et al., 2010), as there was little above-ground vegetation remaining (fig. 5). Snowfall prevented the installation of three unburned plots in 2003. In August 2004, three unburned plots were installed ~280 m southwest of the burned plots (figs. 1d and 4). Plots were 10-m long and 5-m wide (figs. 4, 5, and 6), with a southwest aspect (~215°). Slope steepness averaged 26% for burned plots and 31% for unburned plots.

Sheet metal borders were installed on three sides of each plot to isolate the plot from the surrounding hillslope (figs. 4, 5, and 6). Runoff was diverted by metal gutters on the downhill side of each plot into a 56-L sediment trap (figs. 6 and 7). Outflow from the sediment trap was directed to a tipping bucket runoff measuring device (Black and Luce, 2013; Wijayawardhana et al., 2021) fabricated in the Moscow Forestry Sciences Laboratory (MFSL) machine shop (figs. 7 and 8).

The tipping bucket devices used to measure runoff rate had a volume of ~1.0 L per tip (fig. 8). A magnetic reed switch on the frame of the device was connected to a data logger to record the period of each tip. Each device was individually calibrated at the MFSL for a range of flows, with intervals between tips from ~1 to 30 s (0.03 to 1.0 L s^{-1}). The data were summarized in tips per hour for high runoff rates and hours for a single tip representing approximately 0.02 mm of runoff from a 50 m^2 plot when runoff rates were low. To determine the plot runoff amount, the gutter was assumed to generate 100% runoff, so the fraction of runoff that was collected by the gutter was subtracted from the plot runoff volume measured by the tipping bucket. To estimate the suspended sediment concentration that was not collected in the sediment box, but carried through with the overflow runoff, a slot sampler was installed on one side of the tipping bucket to collect a small proportion of the runoff from each tip and divert it to a container further downslope (fig. 6).

Sediment delivery was measured by collecting sediment deposited in the gutter, collected in the sediment trap, and suspended in runoff from the tipping buckets. The annual amount of sediment collected was divided by the plot area (50 m^2) to determine the erosion rate (kg ha^{-1}). For burned plots, sediment that accumulated during the preceding 12 months was measured in August or September during 2004 through 2007. Sediment was allowed to accumulate from 2008 through 2010, and total accumulation for the three water years was measured in August 2010. For unburned plots, erosion was measured in 2005, 2006, and 2007, and a cumulative sample for 2008 through 2010 was collected in 2010.

When the plots were serviced each summer, sediment was removed from the sediment box and the gutter and dried at 105°C in the MFSL Soil and Water Engineering Laboratory. Runoff and sediment in the proportional runoff collectors, designed to collect a small sample of runoff from the sediment box, was insufficient to prevent the evaporation of the entire sample. To estimate the concentration of suspended sediment in the runoff leaving the sediment box, a ~2-L water sample was collected from the sediment box prior to the annual cleanout.

Ground Cover

Ground cover was measured within a 1.2-m square PVC gridded frame fitted with a string grid with 100 intersecting points (Lutes, 2006). The gridded frame was placed randomly at three locations within each of seven erosion plots. X and Y coordinates within each plot for each sample location were determined by a random number generator and were recalculated for each year. Ground cover was classified at each string intersection as bare mineral soil, rock, branches (≥ 1-cm diameter), charcoal, ash, or other organic material (including decomposing vegetation, twigs, moss, leaves, grass, or similar plant materials). In October 2003, ground cover was determined in each of the four burned plots within two weeks of burning. Additional ground cover measurements were carried out in August or September on burned plots–from 2004 through 2010, on unburned plots from 2004 through 2007, and again in 2010.

Statistical Analyses

The normality of the ground cover, runoff, and erosion were tested with the Shapiro-Wilk (S-W) test (R Core Team, 2021), where a p-value > 0.05 implied the data are not

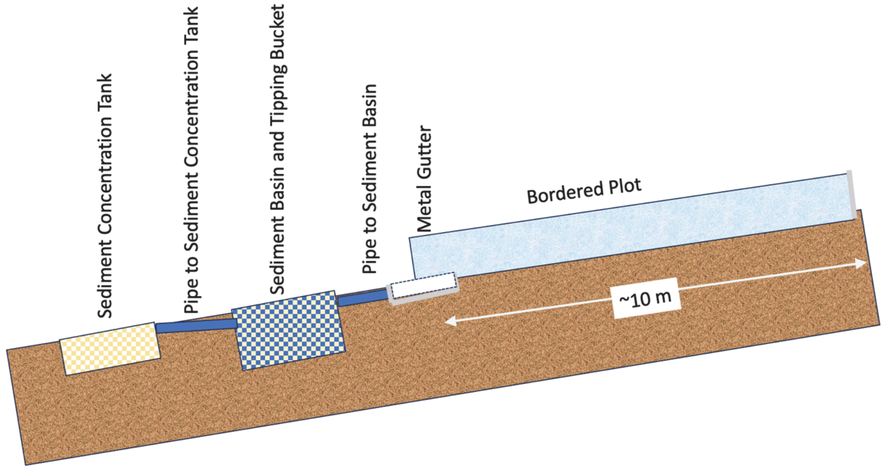


Figure 6. Diagram of plot setup showing the bordered plot, metal gutter to collect plot surface runoff and downslope boxes for the sediment basin (fig. 7), tipping bucket (figs. 7 and 8), and tank to collect a sample of runoff to determine sediment that was not deposited in the sediment basin on the Red Mountain Burn Erosion study.

Figure 7. Sediment basin and tipping bucket device installed in enclosure downhill from the bordered erosion plot (figs. 5 and 6) on the Red Mountain Burn Erosion study (photo by I. S. Miller).

Figure 8. Tipping bucket runoff measuring device like those installed in the pits below the erosion plots (figs. 6 and 7) to measure runoff rates on the Red Mountain Burn Erosion study (photo by A. Yuksel).

significantly different from a normal distribution. To achieve normality with bare mineral soil, the square root was calculated for each of the observed values (S-W p-value = 0.75). The annual runoff values needed to be log-transformed to improve normality (S-W p-value = 0.03). The normality for the erosion was improved with a log10(log10(*erosion*)) transformation (S-W p-value = 0.02). Numerous other transformations were evaluated for runoff and erosion, but none were identified that were better than the two that were selected. Following transformations, two-way analyses of variance (ANOVA) were applied to the three data sets with the independent variables treatment (burned or unburned) and a yearly variable, using either year or annual precipitation from the Giveout SNOTEL site, along with the interaction between treatment and year or precipitation. A least square means (LS Means) analysis protocol was applied to the data sets because of the imbalance of design (4 treated and 3 untreated plots, and no unburned data for one year; R Core Team, 2021). The WY 2008–2010 erosion observations were averaged for the three years and included as a single year with the other annual observations in the erosion analyses. A pairwise comparison was carried out on the LS means values for all three dependent variables. A difference was considered significant at $p \leq 0.05$.

Results

Climate at the Red Mountain Site

A "water year" (WY) is from October 1 of the previous year to September 30 of the water year. Annual precipitation totals for the Red Mountain Site and the nearby Giveout SNOTEL sites are summarized in table 3 for WY 2004 through 2006 for the study site and 2004 through 2010 for the Giveout SNOTEL site. The average annual precipitation (rain and snow) for WY 2004 through 2006 was 506 mm for the onsite rain gage when erosion risk from the prescribed burn was greatest (table 3). The average precipitation for the Giveout SNOTEL site for 2004–2010 was 509 mm. The correlation coefficient between the Red Mountain and the Giveout SNOTEL annual precipitation totals was 0.93. Distributions of precipitation and average daily air temperatures were typical for the climate (fig. 3), with summer highs between 20 and 25°C and winter lows around -20°C. The

Table 3. Annual precipitation measured at the Red Mountain prescribed burn erosion study site and the Givout SNOTEL site; average annual plot runoff from four burned plots for WY 2004–2010 and three unburned plots for WY 2005–2010 (standard error); and average annual erosion from four burned plots (WY 2004–2007 and WY 2008–2010 average) and unburned plots (WY 2005–2007 and 2008–2010 average) (standard error).

	Annual Precip (mm)		Runoff (mm)		Annual Erosion (kg ha^{-1})		
Water Year	Red Mtn	Giveout	Burned	Unburned	Burned	Unburned	LS Means
2004	439.4	502.9	6.0 (14)		629 (47) a[a]	--[b]	
2005	499.4	563.9	7.8 (14)	7.9 (16)	391 (47) b	219 (55) bf	305 (36)
2006	577.9	581.7	12.1 (14)	122.0 (16)[c]	75 (47) c	209 (55) bc	142 (36)
2007	--	424.2	6.2 (14)	24.4 (16)	41 (47) c	23 (55) c	32 (36.2)
2008	--	459.7	4.1 (14)	5.7 (16)			
2009	--	581.7	9.4 (14)	31.9 (16)			
2010	--	452.1	1.8 (14)[c]	2.0 (16)	30 (47) c	80 (55) c	55 (36)
Average	505.6	509.5					
LS Means			5.9 (7.2)	33.8 (8.7)	233 (21)	133 (29)	

[a] Different letters indicate significant differences in annual erosion ($p \leq 0.05$).
[b] -- = No data.
[c] The only significant difference among runoff observations was between 2006 unburned and 2010 burned plots.

greatest daily precipitation from 2004–2006 was 24 mm, which was measured on 9 January 2005 and was likely snow as the average temperature was -3.4°C and no runoff was observed from any of the 7 plots on that day. The greatest runoff (34 mm from unburned plot 15) that occurred from any of the plots was on 12 April 2006 when there was no precipitation. The greatest daily rainfall that resulted in measurable runoff (0.4 mm from burned plots and 0.2 mm from unburned plots) was 23 mm on 20 October 2004, which followed likely snowfall on 18 October with 11 mm of precipitation and an average daily temperature of -0.1°C.

Vegetation and Ground Cover

Following the prescribed burn, shrub and forb covers were significantly less on burned plots compared with unburned plots, but grass cover was significantly greater ($P<0.001$) on the burned plots, according to Page-Dumroese et al., 2023 (table 2). Sagebrush was the dominant shrub in unburned plots (figs. 2 and 4). Burned plots in this study had been purposely sited where there were few live shrubs remaining after the burn (fig. 5), but by 2005 they were primarily covered with grass (fig. 9). The unburned plots continued to support the original sagebrush-dominated plant community, as shown in figure 4.

Groundcover was measured immediately following the fire in October 2003 and in late summer from 2004 to 2010 on the burned runoff plots and nearby unburned sites (table 4) (see Ground Cover subsection in Methods). Ash covered 41% of the burned plots in 2003 but was zero in all subsequent years. The high ash cover resulted in only 24% bare mineral soil exposure immediately following the burn. In 2004, organic material (OM) covered 38% of the burned ground surface, compared with 75% on unburned plots (table 4). Conversely, bare mineral soil on burned plots averaged 43%, compared with only 5% of bare soil on unburned plots. On the unburned plots, the greatest component of ground cover was surface OM, averaging 72% for the study. The year after the fire (2004) was the only year when the bare mineral soil exposure on the burned plots was significantly greater than the unburned plots ($p \leq 0.05$) (table 4). There were no significant differences in bare mineral soil among years ($p = 0.67$), but there was a significant difference due to treatment ($p < 0.001$) (table 4). There was also a significant interaction between treatment and year ($p < 0.001$) because bare soil on burned plots declined but increased on the unburned plots during the study (table 4). The year-to-year and treatment differences in bare mineral soil exposure from year 3 (2006) until the end of the study were not significant.

Figure 9. Looking upslope toward burned plots in August 2005 from the access track (fig. 1d), two years after the fire, when the height of grass was more than 1 m. The height of the technician in the photo was ~1.6 m on the Red Mountain prescribed burn study (photo by I. S. Miller).

Runoff

The mean annual runoff depths for burned and unburned plots are summarized in table 3. The analysis of variance on the transformed annual runoff amounts showed that there were differences due to year ($p = 0.003$) or annual precipitation ($p < 0.001$), but not both, as the two factors were not independent. There was also a significant difference due to treatment (*unburned* vs. *burned*; $p = 0.02$). The interaction between treatment and precipitation ($p=0.99$) or treatment and year ($p = 0.87$) was not significant. When comparing pairwise differences for each year and treatment, the only significant difference was between the lowest annual runoff (WY 2010 burned) and the highest annual runoff (WY 2006 unburned, table 3). The LS mean runoff depths were 6 mm from the burned plots and 34 mm from the unburned plots.

Tipping bucket runoff rates (mm h^{-1}) for WY 2006 for all burned and unburned plots are shown in figures 10a and 10b. The magnitude of the peak runoff rate was more than 30 times greater on unburned plots compared with burned plots on 9 April 2006 (note the difference in the scale of runoff axes in fig. 10). The largest runoff event was due to snowmelt following 10 days of average daily temperatures above freezing and no measurable precipitation in the preceding 8 days. Smaller runoff events were associated with

Table 4. Mean ground cover values for individual categories for all years and bare mineral soil exposure (%) immediately after burning and each year after burning on burned and unburned plots on the Red Mountain prescribed fire site.

Year	Treatment	Surface OM	Branches	Rock	Ash	Charcoal	Bare soil[b] LS Means (SE)
2003[a]	Post Burn	23	2	11	41	0	22 (4) abcdfg
2004	Burned	38	6	2	0	2	43 (4) abcdfg
	Unburned	75	16	4	0	0	5 (5) efg
2005	Burned	61	2	3	0	0	34 (4) acfg
	Unburned	82	8	2	0	0	8 (5)afg
2006	Burned	73	1	5	0	0	21 (4) acdfg
	Unburned	75	11	5	0	0	10 (5) adfg
2007	Burned	67	7	0	0	0	26 (4) abcdfg
	Unburned	66	27	1	0	0	7 (5)adfg
2008	Burned	74	2	2	0	0	22 (4) abcdfg
	Unburned						
2009	Burned	82	1	2	0	0	15 (4) acdefg
	Unburned						
2010	Burned	81	1	0	0	0	19 (4) abcdfg
	Unburned	62	11	2	0	0	26 (5) abcdf
Average 2003–2010	Burned	62	3	3	5	0	25 (2)
	Unburned	72	14	3	0	0	11 (2)

[a] 2003 measurement was the post-burn ground cover in October; other annual measurements were made in August of each subsequent year.
[b] Different letters indicate significant differences in bare soil exposure ($p \leq 0.05$).

warm spells causing snowmelt or from rainfall in the winter, late spring, or early autumn. Single tips (~0.02 mm h^{-1}) from the tipping buckets were removed to simplify figure 10 but were included in annual runoff amounts (table 3). The maximum peak runoff rate on the burned plots was 0.7 mm h^{-1} and exceeded 0.5 mm h^{-1} only 17 times total for the four plots in the 7 years of recording. The maximum peak runoff on the unburned plots was 9.95 mm h^{-1} (fig. 10) and exceeded 0.5 mm h^{-1} more than 200 times from the three plots during the 6 years of recording. On average, the burned plots recorded runoff for 182 hours each year, compared to the unburned plots, where runoff was recorded for 140 hours each year.

One of the tipping bucket devices on a burned plot (TB-11) recorded no runoff in WY 2009 and WY 2010, and a second tipping bucket on the burned plots (TB-12) recorded no runoff in WY 2010. Whether this was because the devices failed or because there was no surface runoff from those plots in those years is not clear, especially in 2010, the year with the lowest observed runoff depths from all plots (table 3).

Runoff amounts from the burned plots ranged from 0.4% of precipitation in WY 2010 from both burned and unburned plots to 2% of the precipitation in 2006 from the burned plots and 21% of the precipitation from the unburned plots (table 3). Overall, the runoff from the burned plots was 1.3% of the precipitation, compared to the runoff accounting for 5.9% of the precipitation on the unburned plots. WY 2007 was the driest year during the study, with only 424 mm of precipitation recorded at the Giveout SNOTEL site, but the runoff from the unburned plots (24 mm) was greater (although not significant) than in WY 2005, 2008, and 2010, when the Giveout site recorded greater amounts of precipitation (table 3).

Figure 10 shows that in 2006, the annual runoff was overwhelmingly dominated by snowmelt events, with 90% of the runoff occurring during the spring season. Table 5 shows that the seasonal trend of increased runoff during the spring occurred throughout the study, just not to the extent that was observed in WY 2006. The seasonal runoff data were highly skewed with many zeros and could not be normalized, so an analysis of variance was not possible. However, a pairwise comparison of LS Means was carried out to identify differences within the analysis and found that runoff in the spring from unburned plots was different from all other values except the fall runoff from the unburned plots ($p \leq 0.05$).

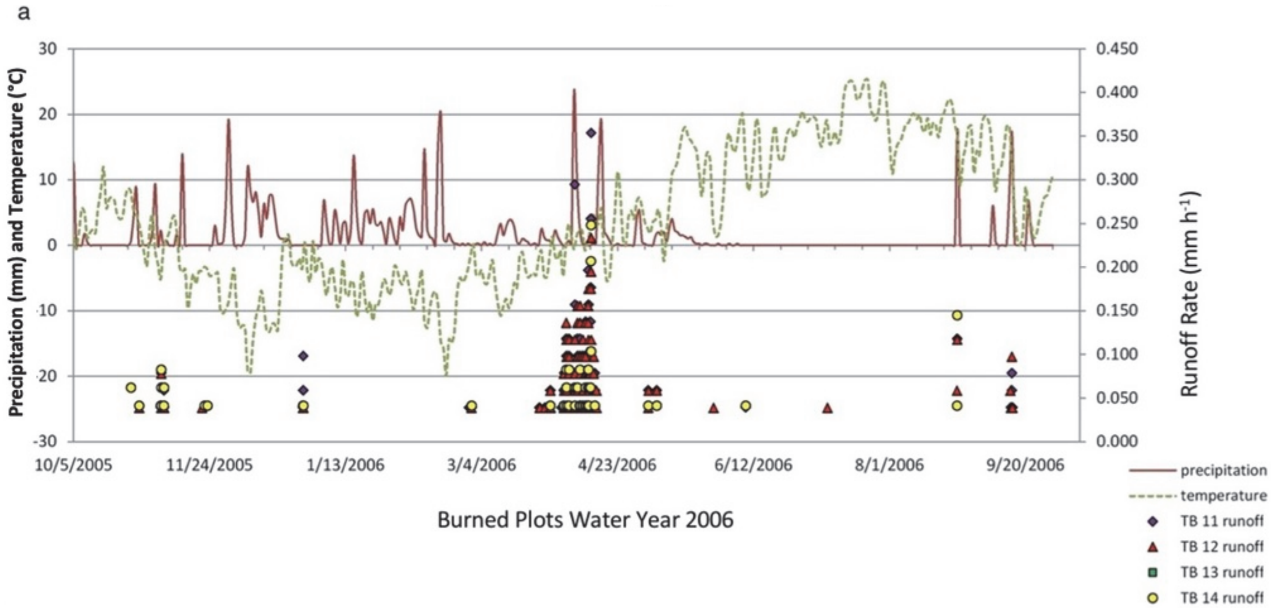
Soil Erosion

Total erosion, including sediment removed from the gutter, collected in the sediment trap, and estimated as lost with runoff, is presented in table 3. The gutter collected 80% of the sediment on the burned plots and 77% of the sediment on the unburned plots. The estimated sediment lost as suspended sediment was only 0.1% of the total measured sediment from the burned plots and 0.05% from the unburned plots.

The least square mean annual erosion for 6 years was 133 kg ha^{-1} y^{-1} from unburned plots and 233 kg ha^{-1} y^{-1} from burned plots (table 3). No samples were collected from the plots during water years 2008 and 2009, so the total sample for water year 2010 was the combined sample amount averaged for the three water years from WY 2008 through 2010. There was no significant difference in soil loss between the burned and unburned plots ($p = 0.44$) and no significant interaction between year and treatment ($p = 0.23$). However, there were differences between years ($p \leq 0.0001$), with erosion declining every year following the burn from 629 kg ha^{-1} to 30 kg ha^{-1} average in 2008–2010. A similar declination of erosion was observed following plot installation on the unburned plots, from 219 kg ha^{-1} to 23 kg ha^{-1} within three years, but then increasing in the last three years (2008–2010) to 80 kg ha^{-1} (table 3).

Discussion

This study was carried out to compare runoff and erosion from natural rainfall and snowmelt following a prescribed burn to unburned plots in a sagebrush-steppe ecosystem. The hypotheses were that (1) surface runoff and soil erosion will be greater from burned plots than from unburned plots, and



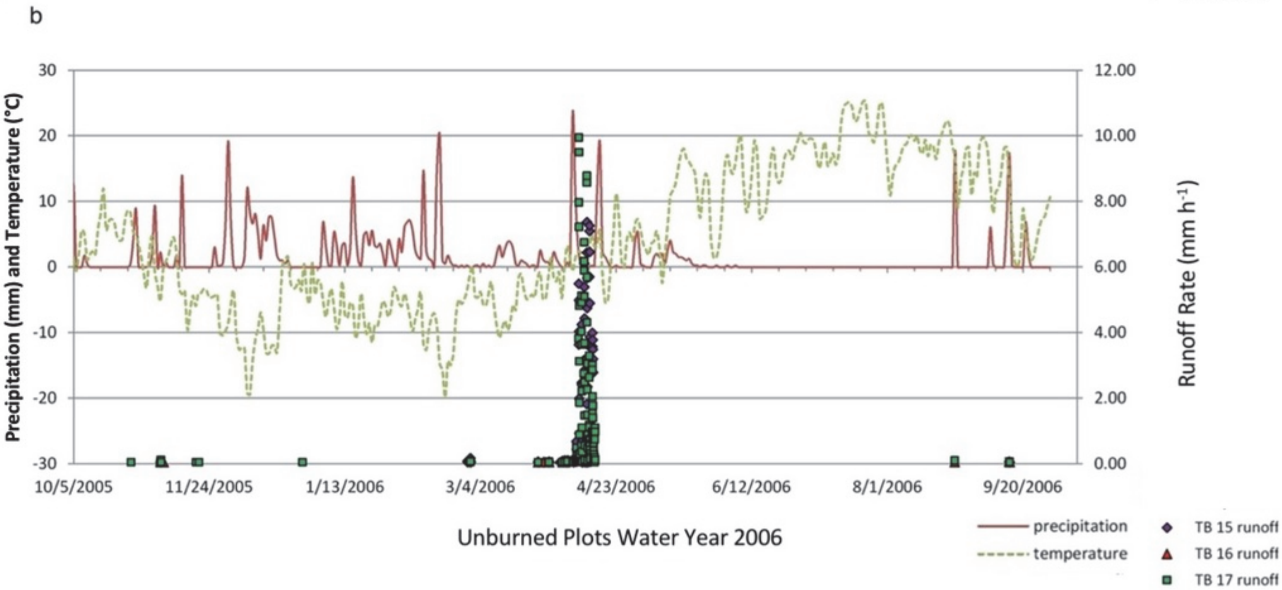


Figure 10. Temperature (°C), daily precipitation (mm), and hourly runoff rate (mm h^{-1}) for Water Year 2006 for the Red Mountain Prescribed Burn study: (a) burned plots and (b) unburned plots. TB number refers to "tipping bucket" runoff plot numbers. Note the difference scales for runoff and the within treatment similarities.

Table 5. LS mean distribution of runoff by season for WY 2004–2010 in the Red Mountain prescribed burn erosion study.

	LS Mean Plot Runoff and (s.e.) (mm)		
Season	Burned	Unburned	LS Means
Fall	1.1 (5.2) a[a]	2.1 (9.2) ab	1.6 (5.3)
Winter	0.5 (5.7) a	1.2 (6.5) a	0.8 (4.3)
Spring	5.7 (5.7) a	36.3 (6.5) b	20.9 (4.3)
Summer	0.6 (5.7) a	0.2 (6.5) a	0.4 (4.3)
LS Means	2.0 (2.8)	9.9 (3.6)	

[a] Different letters indicate significant differences in seasonal runoff depth ($p \leq 0.05$).

(2) in the subsequent years following a prescribed fire, ground cover will increase and runoff and erosion on burned plots will decrease to unburned levels. Both hypotheses can be accepted, with the exception that the prescribed burn did not significantly increase runoff.

Three weaknesses in this study were the lack of preburn data as the plots were not installed until after the burn, the failure of the onsite rain gage in the fourth year, and the failure of our collaborators to collect post-burn vegetation data beyond the first year. The failure of the onsite rain gage in the fourth year (table 3) coupled with large standard deviations in the observed runoff and erosion (table 3) limit the conclusions that can be drawn from this study. However, before the rain gage failed, the spike in runoff from the unburned plots due to snowmelt (fig. 10) and the spike in erosion the year following the wildfire on the burned plots and associated with plot installation on the unburned plots (table 3) stand out as significant despite the high variability within the data set.

The precipitation observed onsite was less than that observed at the Giveout SNOTEL site (table 3) for the three years that the onsite rain gage was operating (WY 2004–

2006). In those three years, the range of annual precipitation for the SNOTEL site was 503–582 mm, and the 2007–2010 annual precipitation values for the SNOTEL site were all within that range. This suggests that it was unlikely that there were any major climate-driven events in those years. The observed erosion rates in 2007–2010 were also relatively low and of similar magnitude compared to the installation disturbance and snowmelt driven higher rates in earlier years (table 3). This suggests that the experimental errors associated with weather data were likely minimal.

The post-fire landscape is one with considerable variability (Robichaud et al., 2007; Vega et al., 2020). Figure 5 underscores such variability where the plots and the weather station were both within burn unit 11, but the ground cover varied from blackened by the fire with no vegetation remaining where the plots were installed to still covered in shrubs with a green canopy where the weather station was sited. With only 3 plots for the unburned treatment and 4 plots for the burned treatment, capturing such variability is difficult. Even if additional plots were installed, the inherent variability would still be present, and the likelihood of capturing small differences due to treatment or burn severity is unlikely. Vega et al. (2020) reported on an erosion study following a wildfire on a shrub-steppe ecosystem using silt fences to measure soil erosion with 12 north-facing plots and 12 south-facing plots. Erosion rates in the first year of the Vega study varied from 0 to 29 Mg ha^{-1} the year following the fire. Despite having 24 plots with two years of data, Vega et al. (2020) were only able to show significant differences in erosion associated with aspect in the first year, otherwise no differences between years, and no difference in erosion in the second year between aspects. The Vega et al. (2020) results suggest that had a greater number of plots been installed in this study, the added inherent variability in such studies would likely not have improved the ability to discern significant differences due to treatment or time since the burn.

The stated purpose of the prescribed burn was to increase the diversity of the site by reducing the sagebrush canopy from 27%–40% to a desired condition of 10%–15% (Johnson, 2003). Both the Johnson (2003) and Page-Dumroese et al. (2023) used a line transect method for measuring canopy cover (BLM, 1999). Table 2 shows that the sagebrush canopy reported by Page-Dumroese et al. (2023) for the study site was 40% preburn and reduced to 32% post burn. The burned erosion plots in this study, however, were purposely installed within Burn Unit 11 where there was no canopy remaining (fig. 5), whereas the burned area adjacent to the erosion plots but still within the burn unit where the weather station was established had retained much of the shrub cover as evident in figure 5, more in keeping with the post burn shrub canopy reported by Page-Dumroese et al. (2023) in table 2. Such a variability in burn severity was the goal of the burn plan where the objectives were that only 32% of the late seral plant community would be altered to early seral, and that where mountain big sagebrush canopy exceeded 20%, 70% of that canopy would be burned, leaving up to 30% unburned (Johnson, 2003). In other studies, the prescribed burn reduced shrub canopy in the year following the fire (Flerchinger et al., 2016; Fuhlendorf et al., 2011; Vega et al., 2020; Williams et al., 2020) and in subsequent years in some studies (Beck et al., 2009, 2011; Flerchinger et al., 2016; Fuhlendorf et al., 2011). Flerchinger et al. (2016) reported that following a prescribed fire, sagebrush was slow to recover compared to forbs, and that the burn did not reduce the leaf area index of grasses. Vega et al. (2020) reported that total canopy significantly increased the second year after wildfire to more than 100% due to forb and grass regeneration.

Bunting et al. (1987) reported that the fall burn before perennial grasses had begun to sprout had minimal impact on grass regrowth compared to shrubs and forbs, favoring grasses in the first few seasons following the burn. A large increase in grass regrowth was observed on the burned plots in this study in the second year after the fire, and by the third year after the burn, the burned plots were covered by 1-m high grass (fig. 9). Beck et al. (2009) reported a decrease in grass cover the year following a prescribed burn in Wyoming big sagebrush that is found on sites that tend to be drier than this site (Dumroese et al., 2015). In a literature review, Beck et al. (2011) reported that generally, grass cover was not affected by or decreased following prescribed fire in mountain big sagebrush communities. Fuhlendorf et al. (2011) reported in a review of prescribed fire studies that generally grass cover declined the year after the burn, but grass response was mixed in subsequent years, with some studies showing a decrease in grass, some showing no change, and some showing an increase in grass cover.

Other recent studies related to the hydrologic impacts of prescribed fire on soil erosion in sage-steppe ecosystems have all had fall burns. Williams et al. (2020), observed vegetation recovery similar to this study. They reported increased forb and grass covers following late summer prescribed burns on sites in Onaqui, UT, and Marking Corral, NV (fig. 1b) compared to the control plots, resulting in grass cover increasing from 5%–6% preburn to 40%–63% 8 years after the prescribed burn. Pierson et al. (2001) reported grass cover increasing from 4% following a wildfire to 30% within one year in their study near Denio, NV (fig. 1b). Pierson et al. (2009), observed rapid grass regeneration with a grass canopy not significantly different from unburned plots one year after a late September 2002 prescribed burn on a stage-steppe rangeland on the Reynolds Creek Watershed, ID (fig. 1b). Flerchinger et al. (2016) reported that grasses and forbs also dominated the plant community for more than 6 years following a September 2007 prescribed burn on the sage-steppe ecosystem at Reynolds Creek, Idaho (fig. 1). Rapid grass regrowth may have been due to reduced competition for water and nutrients from shrubs (Gustine et al., 2021; Williams et al., 2016b). An increase in grass cover was observed both in this study and on nearby plots by Page-Dumroese et al. (2023) two and three years after the fire, but neither study was focused on the details of vegetation recovery, which was a weakness in both studies. Both Page-Dumroese et al. (2023) and this study show the importance of including long-term, detailed plant community monitoring in any future rangeland studies (Williams et al., 2020). The increase in grasses and forbs on the burned plots likely resulted in similar runoff and erosion from burned plots compared to unburned in the final years (WY 2007–2010) of this study (table 3), similar to the findings reported in both the Williams et al. (2020) and the Pierson et al. (2001) simulated rainfall studies in the Great Basin.

Bare soil was 22% following the prescribed burn, with ash providing 41% ground cover and rock accounting for another 11% (table 4). Bare soil increased to 43% the following year due to the loss of ash cover from 41% to zero within a year. Similar first year rates of ash decomposition have been observed following wildfires on other rangeland and forest studies (Vega et al., 2020; Neris et al., 2017). The reduction in exposed rock cover from 11% to 2% for the burned plots from 2003 to 2004 was likely due to the increase in organic matter (23% to 38%) and branches (2% to 6%) killed by fire and subsequently shed by the canopy to cover the surface rock that fire had exposed in the previous year. Such canopy shedding is similar to needle cast observed in forests following low-severity fires (Pannkuk and Robichaud, 2003). Within three years post-burn, bare soil on the burned plots (34%) was not significantly different from unburned plots (8%) ($p \leq 0.05$; table 4). Beck et al. (2009) reported 32.5% bare ground on unburned plots compared to 36.5% on burned plots in a Wyoming big sagebrush study, greater than this study but likely reflecting the generally drier sites dominated by Wyoming big sagebrush compared to mountain big sagebrush (Dumroese et al., 2015). Ground cover of ~40% (bare soil <60%) protects soil from low-return interval storms (Pierson et al., 2008), which was the case on both the burned and unburned plots throughout the study (table 4). The low value for bare soil was likely due to the low severity of the prescribed fire that the managers had hoped to achieve (Johnson, 2003).

In 2006, there was increased animal disturbance observed on unburned plots when measuring ground cover. Tracks from elk (*Cervus elaphus nelsoni*), mule deer (*Odocoileus hemionus*), domestic sheep (*Ovis aries*), and cattle (*Bos taurus*) were observed on all plots, while no tracks had been observed during study installation in 2003 and 2004. Increased animal traffic may have contributed to the reduced ground cover on all the plots (Alshantiri, 2011). By the end of the study, ground cover on the unburned plots was 74%, compared to 95% at the start of the study (table 4). It is possible that the increase in grass cover that was observed following the prescribed burn (table 2) as well as anectodical observations of a greater height of the grass canopy on the burned plots compared to the unburned plots (figs. 9 and 4) attracted the grazers to both the burned and unburned areas surrounding the study site. Fuhlendorf et al. (2011) had listed "to facilitate the distribution of grazing and browsing animals" as one of the purposes for prescribed burning.

Comparisons of the results of this study to rainfall simulation studies provided some additional insights into post-burn hydrologic processes. Pierson et al. (2003) rainfall simulation studies were on sandy loam soils following medium- to high-severity wildfires on a sage-steppe landscape near Boise, ID, and Denio, NV (fig. 1b), where infiltration rates were 40–65 mm h^{-1} and were not greatly different between burned and unburned plots. On an unburned site nearer the Red Mountain study site, the Blackfoot, Idaho site in figure 1b, Franks et al. (1998) report an infiltration rate under a rainfall simulator as 49 mm h^{-1}, similar to Pierson et al.'s (2003) findings. The Red Mountain study site had finer textured soils with higher rock contents (table 1) compared to the Pearson et al. (2003) sites, likely reducing the infiltration rates (Huffman et al., 2013; Brakensiek and Rawls, 1994). However, the finer textures on this study site and the generally low-severity burn likely decreased the possibility of water repellency following the burn (Parsons et al., 2010; Sándor et al., 2021). The precipitation intensities in this study likely never exceeded the infiltration rates reported by Pierson et al. (2003) or Franks et al. (1998). The highest runoff rates occurred on 12 April 2006 as shown in figure 10, when there was no precipitation measured. The below freezing temperatures that started in November 2005 preceded the 12 April event (fig. 10) and allowed a snowpack to accumulate. That snowpack slowly melted starting 19 March 2006 with a few tips each day (1 tip = 0.02 mm runoff), increasing to nearly 40 mm in one day from one of the unburned plots on 12 April, and then decreasing slowly until 27 April. These observations strongly suggest that snowmelt, coupled with saturated soil conditions, dominated spring runoff in 2006. The greatest daily precipitation that resulted in runoff observed in the first three years of his study was on 20 October 2004, when 23 mm of rainfall was measured with a duration of more than 16 hours and an intensity well below the likely infiltration rates reported by Pierson et al. (2003) and Franks et al. (1998). This storm generated only 0.4 mm runoff spanning a 22–23 h runoff duration from the burned plots and only 0.2 mm during a 16–22 h duration from the unburned plots. As with the March 2006 event, the weather records show that in the 5 days before the event, 19 mm precipitation was measured on site with temperatures around freezing, so it was likely there was some snow accumulation just before the event as well as several days of infiltration preceding the event. This suggests that, as with the April 2006 event, the October 2004 runoff event was likely due to saturation excess runoff associated with a large rainfall event that was likely supplemented with snowmelt.

Table 3 showed that runoff did not vary with year for the burned treatment but did on the unburned plots, which resulted in a significant interaction between year and treatment in the statistical analysis. The largest runoff rates occurred in April 2006 from unburned plots (figs. 4 and 10b), but the higher rates were not observed on the burned plots (figs. 9 and 10a). These greater runoff rates from the unburned plots likely contributed to the higher erosion rates observed from the unburned plots compared to the burned plots that year (table 3). When comparing WY 2005 and 2007 on the unburned plots, the runoff in WY 2005 was 9 mm from 564 mm of precipitation compared to WY 2007, when 25 mm of runoff was measured from only 424 mm of precipitation at the Giveout SNOTEL site. This suggests that the timing of precipitation may be as important as the depth at this site. As summarized in table 5, the main runoff events occur in the spring, with the largest daily runoff in 2005 of 6.2 mm on 13 April (Giveout precipitation = 0.0; maximum temperature = 13.3°C), but much larger runoff events in 2007 with daily runoff depths of 5–10 mm on 12–18 April (total precipitation = 5 mm; maximum temperature = 5.2–14.8°C). These data tend to confirm that the distribution of precipitation and timing of snowmelt events dominate the hydrology of this site, while total precipitation is less important. One factor contributing to higher runoff rates from the unburned plots may have been sagebrush canopy cover, reducing earlier-season snowmelt rates from solar radiation (Sicart et al., 2004). The grass canopy on the burned plots likely flattened in the winter, resulting in increased low intensity solar radiation-driven snowmelt and runoff for several days preceding the big runoff event in mid-April (fig. 10a), when any melted snow would be readily infiltrated. Such radiation-driven snowmelt may have been less likely on the unburned plots because of canopy shading (fig. 10b). With less snowpack remaining on the burned plots before the large snowmelt event in mid-April, the observed peak runoff rate was less on the burned plots (fig. 10a) than on the unburned plots (fig. 10b). It is also possible that the sagebrush canopy captured and retained more blowing snow than adjacent burned plots. Sturgis (1977) reported that non-disturbed sagebrush plots collected more snow than plots where herbicide was used to reduce canopy. If there was more snow on the ground in the unburned plots when-temperatures rose above 0°C on 12 April (fig. 10), then greater snowmelt and runoff likely occurred compared to the burned plots (figs. 10a and 10b). Sturgis (1977) also reported that once snow melted sufficiently to expose the sagebrush canopy, melt rates could increase from about 20 mm h^{-1} to 60 to 70 mm h^{-1} as the canopy was able to absorb shortwave radiation and reradiate longwave radiation, thereby increasing snowmelt rates (Sicart et al., 2004). The Sturgis (1977) melt rates would support runoff rates of up to 10 mm h^{-1} (fig. 10b). Luce (2000) reported that when modeling snowmelt in a sage-steppe watershed in Reynolds Creek, ID (fig. 1b), it was necessary to increase snow albedo to increase the snowmelt rate to match observed melt rates of 0.4 mm^{-1} or less. The melt rates reported by Luce (2000) were lower than those reported by Sturgis (1977) or observed in April 2006 in this study, but more typical of runoff rates observed in other years. This sequence of snow accumulation followed by a high snowmelt rate did not occur in other years but likely contributed to the elevated runoff rates observed in 2007 and 2009 on the unburned plots (table 3). Even though the annual precipitation was less in 2007, the runoff rates from unburned plots were still greater than in 2005, 2008, and 2010, but not significantly different from burned plots ($p \leq 0.05$) (table 3). Vega et al. (2020) observed that snow accumulation tended to be greater in plots located in swales than on ridge tops, as did Luce (2000). Such an accumulation of snow on the lower-elevation unburned plots in this study may have also contributed to greater runoff and erosion from the control plots in 2006 as well as in the other years, but to a lesser extent.

The only significant difference in annual runoff depths was between WY 2006 from the unburned plots and WY 2010 from the burned plots (table 3). A detailed look at the onsite weather records showed that between 1 October 2005 and 10 April 10, 2006, 445 mm of precipitation were recorded. For that same time in WY 2010 (1 October 2009–10 April 2010), the Giveout SNOTEL station recorded a cumulative precipitation depth of only 267 mm. The average temperature from 1 April–10 April 2006 was 0.4°C, compared to the average temperature for 1 April – 10 April 2010, which was -2.3°C. It is likely the combination of more winter precipitation and warmer temperatures in April 2006 resulted in a deeper snowpack and greater runoff (122 mm) that year compared to the drier winter and colder April 2010 temperatures with lower snowmelt rates and a lower annual runoff of only 2 mm.

A confounding factor in the differences in runoff between the unburned and burned plots may have been the differences in soil (table 1). The unburned plot soils had more clay (30% vs. 16% on the burned), less sand (12% on the unburned plots vs. 35% on the burned plots), and more rock (61% unburned vs. 9% on the burned), properties that may have resulted in increased runoff from the unburned plots compared to the burned plots, especially 3 or 4 years after the burned plots had recovered (table 3; Brakensiek and Rawls, 1994; Huffman et al., 2013). Interestingly, Pierson et al. (2001) reported that, like this study, the burned plots following wildfire also had higher sand contents (83%) than unburned plots (68%). In this study, the unburned plots were within the proposed burn zone (Johnson, 2003), but the fire never reached the plots (fig. 1d). This suggests that areas with higher clay content soils may have vegetation with a higher water content and be less likely to carry a prescribed or wildfire. Another factor that may have contributed to elevated runoff and erosion from the unburned plots was slope steepness (Huffman et al., 2013). The average steepness of the burned plots was 26%, compared to 31% on the unburned plots. Steeper slopes are associated with less surface depression storage before runoff is initiated and tend to have higher erosion rates. The unburned plots were located lower on the landscape (fig. 1d), so there was also a possibility that shallow groundwater flow from further upslope may have increased the soil water content in the unburned plots, increasing surface runoff (tables 3 and 5; Boll et al., 2015), or even subsurface flow being forced to the surface during the spring period of soil saturation, increasing surface runoff from the unburned plots (Boll et al., 2015). The unburned soil, however, was deeper and steeper, increasing its ability to transfer upslope subsurface flow down the hill. More detailed studies incorporating multiple rain gages, winter snow depth monitoring, and soil water content probes are necessary to better understand these complex interactions among burn severity, surface vegetation, slope steepness, landscape features, soil properties, snow distribution, and snowmelt rates.

There was a consistent difference in seasonal runoff rates (table 5). These seasonal differences confirm that the greatest runoff depths occur in the spring from snowmelt and spring rainfall when soils are saturated and unlikely to be water repellant. Soils with low water content in the late summer and fall were more likely to be water repellant (Pierson et al., 2008). Although not statistically significant, summer runoff was greater from the burned plots than the unburned plots (table 5). This may reflect the higher evapotranspiration rates from the unburned plots, leading to drier soil conditions on the unburned plots and therefore decreasing the likelihood of saturation excess runoff (Srivastava et al., 2018), but more likely the reduced runoff from the unburned plots was because of the greater canopy cover (table 2) and ground cover (table 4) on the unburned plots the first two years following the burn. Nearing et al. (2011) stated that even though saturated hydraulic conductivity decreases as clay content increases, it also increases with increases in both ground cover (table 4) and canopy cover (table 2).

The reason for the failure of tipping buckets to record runoff from one of the burned plots in 2009 and from two of the burned plots in 2010 is not clear. It is possible that the tipping buckets failed, although no damage was reported when the devices were removed in 2010. It is also possible that there was no runoff measured from these plots because there was no surface runoff in those years. Some of the burned plots may have had only subsurface lateral low with their coarser textured soils (table 1). When the plots were serviced in August or September of each year, the sediment basins were filled with water. Sufficient water in the basin may have evaporated during the year so that some runoff would be stored in the sediment basin rather than overflowing onto the tipping bucket. Evaporation would be low, but low runoff rates observed overall in 2010 (table 3) suggest evaporation could have been a factor.

The observed erosion rate in this study the year following the burn was 629 kg ha^{-1} (table 3). This value can be compared to Vega et al. (2020) following a wildfire in a similar ecosystem near Reynold's Creek (fig. 1b) on a south-facing slope, who reported 993 kg ha^{-1} from 10–13-m long plots with 25%–45% gradients. The higher erosion rates in the Vega et al. study likely reflected the higher severity burn associated with wildfires and generally longer and steeper plots.

Since the 1980s, rainfall simulation has been the tool of choice to estimate soil erodibility (Al-Hamdan et al., 2015; Elliot and Flanagan, 2023; Franks et al., 1998; Nearing et al., 2011). Although not directly comparable, rainfall simulation studies may give additional insight into the findings of this study. Rangeland studies have shown that plant communities can play a role in soil erodibility (Al-Hamden et al., 2015; Elliot, 2004), so six rainfall simulation studies were selected for comparison to this study that were carried out in sage-steppe ecosystems in the Great Basin on 12%–40% slopes (fig. 1b; table 6). All six studies published an erosion:runoff ratio (kg ha^{-1} mm^{-1}), and those ratios were compared to the same ratio calculated for this study (table 6). The rainfall simulation rate was greater than 60 mm h^{-1} for all the simulation studies, compared to a maximum daily rainfall of 24 mm for this study. Table 6 shows that the erosion:rainfall ratio was similar for the Franks et al. (1998), the Pierson et al. (2002, 2003) studies and this study for the unburned or recovered conditions (ratio ~ 1–5 kg ha^{-1} mm^{-1}). Moffet et al. (2007) observed concentrated flow occurring on their 6.5-m long plots from the high intensity rainfall on 40% slopes, likely leading to the elevated ratio of 33 kg ha^{-1} mm^{-1}, whereas there was no obvious rill incision observed on the plots in this study nor on the other small simulation plots shown in table 6. The erosion:runoff ratios for burned plots from the Pierson et al. (2002, 2003) studies were lower than the Moffet et al. (2007) study in table 6. One reason for the lower rates in the Pierson et al. (2002, 2003) studies may be that sediment was limited on the smaller plots and the erosion rate declined with time. Figure 11 from Moffet et al. (2007) shows declines in sediment concentration with time, as frequently observed on small rainfall simulation plots on the other rangeland simulation studies (table 6), forest roads (Foltz et al., 2009), and in post-wildfire forests (Robichaud et al., 2016). The lower intensity of precipitation and runoff in this study and the larger plots with a greater source area for sediment suggest that sediment limitation likely did not influence the observed erosion rates in this study. However, the high erosion rate observed from the unburned plots the year following installation may have been due to the removal of sediment loosened by plot installation (27 kg ha^{-1} mm^{-1} in 2004), followed by a decline to less than 1 kg ha^{-1} mm^{-1} in 2007, a longer-term decline compared to the rapid decline in sediment under rainfall simulation in figure 11. The decreases in erosion rates on the burned plots were likely associated with recovery from plot installation, increases in ground cover (table 4), and natural armoring as fine particles loosened by the fire (Mataix-Solera et al., 2011) were removed, leaving coarser, less erodible particles and aggregates on the soil surface (Willgoose and Sharmeen, 2006). The two Williams et al. (2016b, 2020) rainfall simulation studies in table 6 were carried out on sites where evergreen trees were encroaching on sagebrush plant communities. Williams et al. (2016b, 2020) carried out rainfall simulation studies a year after a low to moderate autumn prescribed burn with plots on burned and nearby unburned sites. They installed 0.7 x 0.7 m plots on areas that were underneath

Table 6. Erosion:Runoff ratios for the Red Mountain burn erosion study (table 3) compared to rainfall simulation studies on sage-steppe ecosystems elsewhere in the Great Basin. Locations of all sites are shown in figure 1b.

Reference	Plot Length (m) and Slope (%)	Texture	Erosion:Runoff Ratio kg ha^{-1} mm^{-1} Unburned/ Recovered	Low Severity	High Severity
This study (2004)	10m, 26%			88.7	
This study (2004)	10 m, 30%	Silt loam	27		
This study (2007)	10 m, 30% & 26%		0.9 / 5.1		
Franks et al. (1998); Blackfoot, ID	10.1 m, 12%	Silt Loam	6.0–7.1		
Pierson et al. (2002); Boise, ID	1 m, 40%	Sandy loam	0.9–1.2		3.5–3.8
Pierson et al. (2003); Denio, NV	1 m, 40%	Sandy loam	5.1–6.4		8.7–13.6
Moffet et al. (2007); Reynold's Creek, ID	6.5 m, 40%	Sandy loam	33.3	644.	
Williams et al. (2016b);					
Marking Corral, Nevada	0.7 m, 12%	Sandy loam	7–71[a]	11–21[a]	
Onaqui, Utah			47–55[b]	71–104[b]	
Williams et al. (2020);					
Marking Corral, Nevada	0.7 m, 18%	Sandy loam	8–10 / 8–23[a]		
Onaqui, Utah			56–74 / 41–81[b]		

[a] Coppice
[b] Interspace

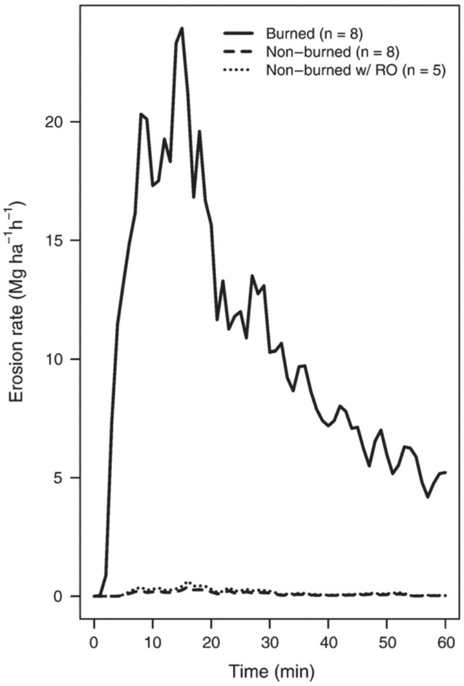


Figure 11. Mean erosion rates during 60-min simulated rainfalls on burned and non-burned plots (n = 8). Three non-burned plots yielded little runoff or sediment, and the mean sediment yield response for the field plots that did generate runoff is also shown (n = 5). Note that the units on erosion rate are in Mg ha^{-1} h^{-1}. From Moffet et al., 2007.

unburned and burned trees and sage brush and bare areas between plants, identifying those locations as individual "tree and shrub coppices" or "interspaces." The values shown for Williams et al. (2016b, 2020) cover the range of combined observations for coppices and interspaces for each treatment. Observations from the interspace tended to have higher ratios, but there were seldom significant differences in the ratio among the locations in those studies. The ratios reported by Williams et al. (2016b) following the low severity burn were similar to the ratios observed in this study, as were the ratios for the unburned plots on the Marking Corral site in Williams et al. (2020). The variability in ratios for the Williams et al. (2016b, 2020) sites is due to location and coppice vs. interspace, showing the complexity of rangeland sites, suggesting that not only is variable area hydrology influencing runoff, but spatial variability is likely influencing erosion rates (Robichaud et al., 2007). The greater ratios in the Moffett et al. (2007) study (33–664 kg ha^{-1} mm^{-1}) were likely due to the long plots in that study compared to the other steep site simulation studies, or the steeper plots compared to the Franks et al. (1998) study. One implication of figure 11 is that rainfall simulation studies may overpredict soil erodibility on rangelands because the plots are disturbed during installation, particularly on small plots were the fraction of the plot that is disturbed is relatively high.

When Nearing et al. (2011) developed the Rangeland Hydrology Erosion model (RHEM) from rainfall simulation data, including the Franks et al. (1998) data set, they concluded that rill erosion was not common on rangeland watersheds. However, when Al-Hamdan et al. (2017) reevaluated RHEM using more recent rainfall and runoff simulation data from steeper slopes, they determined that rill erosion was possible and presented an equation to estimate the probability of rill erosion that included terms for slope steepness, fraction of bare ground, and flow discharge. Al-Hamdan et al. (2017) stated that simulated runoff was routed along tortuous flow paths between shrub coppices, detaching and transporting more sediment than would have been generated by rainfall alone. In this study, the greatest observed runoff events tended to occur in the spring (table 5), associated with snowmelt only for some of the events (fig. 10), supporting Al-Hamdan et al.'s (2015) findings that concentrated flow was the main detachment mechanism on these plots, especially during runoff events when there was little to no rainfall.

One unexpected outcome of the Page-Dumroese et al. (2023) complementary study was that higher levels of termite activity were observed on the burned plots compared to the unburned plots. Termites were not anticipated at this latitude and elevation. Higher incidences of termite activity, potentially decreasing ground cover (Mando and Brussaard, 1999), and soil bulk density (Li and Su, 2008) may have contributed to greater infiltration rates and reduced runoff and erosion from the burned plots in the later years (WY 2008–2010) of the study (table 3) (Elkins et al., 1986; Mando et al., 1996).

The inclusion of the USLE *R* and *K Factors* in the site description was to aid readers in understanding the nature of the climatic and soil properties of this site. Their inclusion was not intended to suggest that the USLE/RUSLE tools were suitable for this ecosystem. The *R Factor* describes the ability of rainfall to detach and transport soil (Huffman et al., 2013), whereas this study clearly showed that runoff and sediment delivery were associated with snowmelt (table 5). The *K Factor* estimate (table 1) was based on regression relationships developed from tilled fallow cropland soils (Huffman et al., 2013), and applying those relationships to untilled rocky rangeland soils would be unwise. An implication of this study was that erosion models developed for this ecosystem need to incorporate winter processes, such as snow accumulation, redistribution, and melt. Current erosion models based on the Universal Soil Loss Equation (USLE), models that link a rainfall-runoff model to the USLE, such as SWAT (Arnold et al., 1998) and AGNPS (Young et al., 1989), and the runoff-based RHEM model (Nearing et al., 2011), do not have this capability. The only widely used erosion model that does attempt to incorporate snowmelt is the Water Erosion Prediction Project (WEPP) model (Srivastava et al., 2017), but it has not been evaluated for winter processes in sage-steppe ecosystems, and evaluation of snow distribution is limited (Shen, 2011).

CONCLUSIONS

A 6-year study evaluated the hydrologic impacts of prescribed fire on a sagebrush-steppe rangeland site in southeastern Idaho. The prescribed fire was intended to improve diversity on sage-steppe rangelands. The study found that the high erosion rates immediately following prescribed burning (629 kg ha^{-1}) were offset by lower erosion rates in subsequent years (< 100 kg ha^{-1}), likely associated with slower snowmelt rates, increased grass canopy and ground cover, differences in soil properties, and increased termite activity on burned sites. Erosion on the unburned plots was 219 kg ha^{-1} the first year, likely influenced by plot disturbance associated with plot installation, and decreased to 23 kg ha^{-2} in the third year following plot installation. Runoff rates up to 10 mm h^{-1} were measured from the unburned plots one spring compared to less than 0.5 mm h^{-1} from the burned plots during that same event, likely associated with winter snow accumulation in the unburned shrub canopy followed by high spring snowmelt rates. By the third year following the prescribed fire, there were no significant differences in runoff (~10 mm y^{-1}) or erosion (~30 kg ha^{-1} y^{-1}) between the burned and the unburned plots. This study showed the complex interactions among such diverse factors as spatial variability of soil properties, ground cover, vegetation regeneration rate and species, snow accumulation and melt, wildlife grazing, and even insects on the impacts of prescribed fire on rangeland erosion. The six-year duration of the study underscored the importance of monitoring post-fire hydrologic response for at least three years. The within-treatment variability among the plots was typical of post-fire studies and emphasizes the importance of a replicated experimental design, even though such replication is often limited by available personnel and financial constraints. Replication of rain gauges, however, is a relatively low-cost addition that would likely improve any remote study where close monitoring of weather data loggers is not possible. Unsuccessful attempts to publish this study combined with another from the same site as a single, more complex ecosystem analysis suggest that even though the scientific community often promotes the benefits of multidisciplinary research, reviewers often prefer papers of limited scope within their area of expertise. Future multidisciplinary rangeland ecosystem studies can be improved by regular monitoring of the diversity and density of recovering vegetation for several years following the prescribed fire in concert with other ecosystem attributes to better understand vegetation, soil water dynamics, and other factors that may be impacting snowmelt processes, runoff, and erosion. This study suggests that erosion models for this ecosystem will need to incorporate snow accumulation, redistribution, and melt. The results of this study, including the archived data, may be useful for developing, validating, or enhancing rangeland hydrologic and erosion models.

Acknowledgments

This research was supported by the USDA Forest Service National Fire Plan, Rocky Mountain Research Station (RMRS), and Intermountain Region (R4). The author acknowledges the considerable contributions of Ina Sue Miller, a retired hydrologist with RMRS for hydrologic data collection, analysis, and archiving; numerous summer students, former RMRS Hydrologist Brandon Glaza, and R4 employees Jeff Bruggink, John Lott, Kara Kleinschmidt, Ashelee Rasmussen, Jane Rushane, Dylan Johnson, Fred Wells, Richard Loetscher, Brooke Timothy, and K. C. Roberts for assistance in plot installation. RMRS engineer Sarah Lewis-Scholes assisted with some of the statistical analyses.

Research Data Archive

Data from the study are available from the USDA Forest Service Research Data Archive:
https://www.fs.usda.gov/rds/archive/Catalog/RDS-2020-0005

References

Al-Hamdan, O. Z., Hernandez, M., Pierson, F. B., Nearing, M. A., Williams, C. J., Stone, J. J.,... Weltz, M. A. (2015). Rangeland hydrology and erosion model (RHEM) enhancements for applications on disturbed rangelands. *Hydrol. Process., 29*(3), 445-457. https://doi.org/10.1002/hyp.10167

Al-Hamdan, O. Z., Pierson, F. B., Nearing, M. A., Williams, C. J., Hernandez, M., Boll, J.,... Spaeth, K. (2017). Developing a parameterization approach for soil erodibility for the Rangeland Hydrology and Erosion Model (RHEM). *Trans. ASABE, 60*(1), 85-94. https://doi.org/10.13031/trans.11559

Alshantiri, H. (2011). Effects of vegetation cover and grazing on rangeland runoff and erosion processes in the Asotin Creek Watershed, WA. MS thesis. Pullman, WA: Washington State University, Department of Biological & Agricultural Engineering. Retrieved from https://searchit.libraries.wsu.edu/permalink/01ALLIANCE_WSU/1rq08rk/alma99387103160101842

Arnold, J. G., Srinivasan, R., Muttiah, R. S., & Williams, J. R. (1998). Large area hydrologic modeling and assessment Part I: Model development. *JAWRA, 34*(1), 73-89. https://doi.org/10.1111/j.1752-1688.1998.tb05961.x

Beck, J. L., Connelly, J. W., & Reese, K. P. (2009). Recovery of greater sage-grouse habitat features in Wyoming big sagebrush following prescribed fire. *Restor. Ecol., 17*(3), 393-403. https://doi.org/10.1111/j.1526-100X.2008.00380.x

Beck, J. L., Klein, J. G., Wright, J., & Wolfley, K. P. (2011). Potential and pitfalls of prescribed burning big sagebrush habitat to enhance nesting and early brood-rearing habitats for greater sage-grouse. *Nat. Res. Environ. Iss., 16*(5). Retrieved from https://digitalcommons.usu.edu/nrei/vol16/iss1/5

Black, T. A., & Luce, C. H. (2013). Measuring water and sediment discharge from a road plot with a settling basin and tipping bucket. Gen. Tech. Rep. RMRS-GTR-287. Fort Collins, CO: USDA, Forest Service, Rocky Mountain Research Station. https://doi.org/10.2737/rmrs-gtr-287

Boll, J., Brooks, E. S., Crabtree, B., Dun, S., & Steenhuis, T. S. (2015). Variable source area hydrology modeling with the water erosion prediction project model. *JAWRA, 51*(2), 330-342. https://doi.org/10.1111/1752-1688.12294

Bradford, M. A., Warren Ii, R. J., Baldrian, P., Crowther, T. W., Maynard, D. S., Oldfield, E. E.,... King, J. R. (2014). Climate fails to predict wood decomposition at regional scales. *Nat. Clim. Change, 4*(7), 625-630. https://doi.org/10.1038/nclimate2251

Brakensiek, D. L., & Rawls, W. J. (1994). Soil containing rock fragments: Effects on infiltration. *CATENA, 23*(1), 99-110. https://doi.org/10.1016/0341-8162(94)90056-6

Brooks, M. L., Matchett, J. R., Shinneman, D. J., & Coates, P. S. (2015). Fire patterns in the range of the greater sage-grouse, 1984-2013 - Implications for conservation and management. USGS Open-File Report 2015-1167. USGS. https://doi.org/10.3133/ofr20151167

Bunting, S. C., Kilgore, B. M., & Bushey, C. L. (1987). Guidelines for prescribed burning sagebrush-grass rangelands in the northern Great Basin. Gen. Tech. Rep. INT-231. Ogden, UT: USDA, Forest Service, Intermountain Research Station. https://doi.org/10.2737/INT-GTR-231

Bureau of Land Management, (BLM). (1999). Sampling vegetation attributes, Interagency technical reference. Technical Reference 1734-4. Denver, CO: BLM National Business Center.
Cao, L., Elliot, W., & Long, J. W. (2021). Spatial simulation of forest road effects on hydrology and soil erosion after a wildfire. *Hydrol. Process., 35*(6), e14139. https://doi.org/10.1002/hyp.14139
Castillo, V. M., Gómez-Plaza, A., & Martínez-Mena, M. (2003). The role of antecedent soil water content in the runoff response of semiarid catchments: A simulation approach. *J. Hydrol., 284*(1), 114-130. https://doi.org/10.1016/S0022-1694(03)00264-6
DeBano, L. F. (1981). Water repellent soils: A state-of-the-art. Gen. Tech. Rep. PSW-46. Albany, CA: USDA, Forest Service, Pacific Southwest Forest and Range Experiment Station. https://doi.org/10.2737/psw-gtr-46
DeBano, L. F., & Klopatek, J. M. (1988). Phosphorus dynamics of pinyon-juniper soils following simulated burning. *Soil Sci. Soc. Am. J., 52*(1), 271-277. https://doi.org/10.2136/sssaj1988.03615995005200010048x
DeBano, L. F., Mann, L. D., & Hamilton, D. A. (1970). Translocation of hydrophobic substances into soil by burning organic litter. *Soil Sci. Soc. Am. J., 34*(1), 130-133. https://doi.org/10.2136/sssaj1970.03615995003400010035x
Dumroese, R. K., Luna, T., Richardson, B. A., Kilkenny, F. F., & Runyon, J. B. (2015). Conserving and restoring habitat for greater sage-grouse and other sagebrush-obligate wildlife: the crucial link of forbs and sagebrush diversity. *Native Plants J., 16*(3), 276-299. https://doi.org/10.3368/npj.16.3.276
Elkins, N. Z., Sabol, G. V., Ward, T. J., & Whitford, W. G. (1986). The influence of subterranean termites on the hydrological characteristics of a Chihuahuan desert ecosystem. *Oecologia, 68*(4), 521-528. https://doi.org/10.1007/BF00378766
Elliot, W. J. (2004). WEPP internet interfaces for forest erosion prediction. *JAWRA, 40*(2), 299-309. https://doi.org/10.1111/j.1752-1688.2004.tb01030.x
Elliot, W. J., & Flanagan, D. C. (2023). Estimating WEPP cropland erodibility values from soil properties. *J. ASABE, 66*(2), 329-351. https://doi.org/10.13031/ja.15218
Engle, D. M., & Bidwell, T. G. (2001). Viewpoint: The response of central North American prairies to seasonal fire. *J. Range Manag., 54*(1), 2-10. https://doi.org/10.2307/4003519
Engmun, E. T. (1974). Partial area hydrology and its application to water resources. *JAWRA, 10*(3), 512-521. https://doi.org/10.1111/j.1752-1688.1974.tb00592.x
Fenn, M. E., Allen, E. B., Weiss, S. B., Jovan, S., Geiser, L. H., Tonnesen, G. S.,... Bytnerowicz, A. (2010). Nitrogen critical loads and management alternatives for N-impacted ecosystems in California. *J. Environ. Manag., 91*(12), 2404-2423. https://doi.org/10.1016/j.jenvman.2010.07.034
Flerchinger, G. N., Seyfried, M. S., & Hardegree, S. P. (2016). Hydrologic response and recovery to prescribed fire and vegetation removal in a small rangeland catchment. *Ecohydrology, 9*(8), 1604-1619. https://doi.org/10.1002/eco.1751
Foltz, R. B., Copeland, N. S., & Elliot, W. J. (2009). Reopening abandoned forest roads in northern Idaho, USA: Quantification of runoff, sediment concentration, infiltration, and interrill erosion parameters. *J. Environ. Manag., 90*(8), 2542-2550. https://doi.org/10.1016/j.jenvman.2009.01.014
Franks, C. D., Pierson, F. B., Mendenhall, A. G., Spaeth, K. E., & Weltz, M. A. (1998). Interagency Rangeland Water Erosion Project report and state data summaries. Interagency rangeland water erosion team (IRWET) and national range study team (NRST). NWRC 98-1. Boise, ID: USDA-ARS, NRCS. Retrieved from https://upload.wikimedia.org/wikipedia/commons/2/21/Interagency_rangeland_water_erosion_project_report_and_state_data_summaries_%28IA_CAT11082292%29.pdf
Fuhlendorf, S. D., Limb, R. F., Engle, D. M., & Miller, R. F. (2011). Assessment of prescribed fire as a conservation practice, Chapter 2. In D. D. Briske (Ed.), *Conservation benefits of rangeland practices: Assessment, recommendations and knowledge gaps* (pp. 75-104). Washington, DC: USDA NRCS.
Gustine, R. N., Hanan, E. J., Robichaud, P. R., & Elliot, W. J. (2021). From burned slopes to streams: how wildfire affects nitrogen cycling and retention in forests and fire-prone watersheds. *Biogeochemistry, 157*(1), 51-68. https://doi.org/10.1007/s10533-021-00861-0
Harper, C. A., Bates, G. E., Hansbrough, M. P., Gudlin, M. J., & Gruchy, J. P. (2011). PB1752 Native warm-season grasses: Identification, establishment and management for wildlife and forage production in the Mid-South. Knoxville, TN: University of Tennessee. Retrieved from https://trace.tennessee.edu/utk_agexfish/9
Huffman, R. L., Fangmeier, D. D., Elliot, W. J., & Workman, S. R. (2013). *Soil and water conservation engineering* (7th ed.). St. Joseph, MI: ASABE. https://doi.org/10.13031/swce.2013
Inbar, M., Tamir, M., & Wittenberg, L. (1998). Runoff and erosion processes after a forest fire in Mount Carmel, a Mediterranean area. *Geomorphology, 24*(1), 17-33. https://doi.org/10.1016/S0169-555X(97)00098-6
Johnson, D. (2003). Red Mountain prescribed burn plan. Bear River Zone Fire Management. Ashton, ID: USDA Forest Service, Caribou-Targhee National Forrest.
Keeley, J. E., & McGinnis, T. W. (2007). Impact of prescribed fire and other factors on cheatgrass persistence in a Sierra Nevada ponderosa pine forest. *Int. J. Wildland Fire, 16*(1), 96-106. https://doi.org/10.1071/WF06052
Lew, R., Dobre, M., Srivastava, A., Brooks, E. S., Elliot, W. J., Robichaud, P. R., & Flanagan, D. C. (2022). WEPPcloud: An online watershed-scale hydrologic modeling tool. Part I. Model description. *J. Hydrol., 608*, 127603. https://doi.org/10.1016/j.jhydrol.2022.127603
Li, H.-F., & Su, N.-Y. (2008). Sand displacement during tunnel excavation by the Formosan subterranean termite (Isoptera: Rhinotermitidae). *Ann. Entomol. Soc. Am., 101*(2), 456-462. https://doi.org/10.1603/0013-8746(2008)101[456:Sddteb]2.0.Co;2
Lubowski, R. N., Vesterby, M., Bucholtz, S., Baez, A., & Roberts, M. (2006). Major uses of land in the United States, 2002. EIB-14. Washington, DC: USDA-ERS. Retrieved from https://www.ers.usda.gov/publications/pub-details/?pubid=43983
Luce, C. H. (2000). Scale influences on the representation of snowpack processes. PhD diss. Logan, UT: Utah State University, Department of Civil & Environmental Engineering. Retrieved from https://hydrology.usu.edu/dtarb/luce_dissertation.pdf
Lutes, D. C., Keane, R. E., Caratti, J. F., Key, C. H., Benson, N. C., Sutherland, S., & Gangi, L. J. (2006). FIREMON: Fire effects monitoring and inventory system. Gen. Tech. Rep. RMRS-GTR-164. Fort Collins, CO: USDA, Forest Service, Rocky Mountain Research Station. https://doi.org/10.2737/rmrs-gtr-164
Mando, A., Brussaard, L., 1999. Contribution of termites to the breakdown of straw under Sahelian conditions. Biol. Fert. Soils. 29, 332-334. https://doi.org/10.1007/s003740050561
Mando, A., Stroosnijder, L., & Brussaar, L. (1996). Effects of termites on infiltration into crusted soil. *Geoderma, 74*(1), 107-113. https://doi.org/10.1016/S0016-7061(96)00058-4
Mataix-Solera, J., Cerdà, A., Arcenegui, V., Jordán, A., & Zavala, L. M. (2011). Fire effects on soil aggregation: A review. *Earth*

Sci. Rev., 109(1), 44-60. https://doi.org/10.1016/j.earscirev.2011.08.002

Meeuwig, R. O. (1971). Soil stability on high-elevation rangeland in the intermountain area. *(94)*. Ogden, UT: USDA, Intermountain Forest & Range Experiment Station, Forest Service. https://doi.org/10.5962/bhl.title.69070

Moffet, C. A., Pierson, F. B., Robichaud, P. R., Spaeth, K. E., & Hardegree, S. P. (2007). Modeling soil erosion on steep sagebrush rangeland before and after prescribed fire. *CATENA, 71*(2), 218-228. https://doi.org/10.1016/j.catena.2007.03.008

Nearing, M. A., Liu, B. Y., Risse, L. M., & Zhang, X. (1996). Curve numbers and green-ampt effective hydraulic conductivities. *JAWRA, 32*(1), 125-136. https://doi.org/10.1111/j.1752-1688.1996.tb03440.x

Nearing, M. A., Wei, H., Stone, J. J., Pierson, F. B., Spaeth, K. E., Weltz, M. A.,... Hernandez, M. (2011). A rangeland hydrology and erosion model. *Trans. ASABE, 54*(3), 901-908. https://doi.org/10.13031/2013.37115

Neris, J., Elliot, W. J., Doerr, S., & Robichaud, P. R. (2017). Development of a model to predict ash transport and water pollution in fire-affected environments. EGU2017-18845. *19*. EGU General Assembly. Retrieved from https://meetingorganizer.copernicus.org/EGU2017/EGU2017-18845-1.pdf

Page-Dumroese, D. S., Cook, S. P., Kard, B. M., Jurgensen, M. F., Miller, C. A., & Tirocke, J. M. (2023). Prescribed burning alters insects and wood decay in a sagebrush-steppe rangeland in southwestern Idaho, United States. *Rangeland Ecol. Manag., 90*, 134-145. https://doi.org/10.1016/j.rama.2023.06.002

Pannkuk, C. D., & Robichaud, P. R. (2003). Effectiveness of needle cast at reducing erosion after forest fires. *Water Resour. Res., 39*(12). https://doi.org/10.1029/2003WR002318

Parsons, A., Robichaud, P. R., Lewis, S. A., Napper, C., & Clark, J. T. (2010). Field guide for mapping post-fire soil burn severity. Gen. Tech. Rep. RMRS-GTR-243. Fort Collins, CO: USDA, Forest Service, Rocky Mountain Research Station. https://doi.org/10.2737/rmrs-gtr-243

Pierson, F. B., Carlson, D. H., & Spaeth, K. E. (2002). Impacts of wildfire on soil hydrological properties of steep sagebrush-steppe rangeland. *Int. J. Wildland Fire, 11*(2), 145-151. https://doi.org/10.1071/WF02037

Pierson, F. B., Moffet, C. A., Williams, C. J., Hardegree, S. P., & Clark, P. E. (2009). Prescribed-fire effects on rill and interrill runoff and erosion in a mountainous sagebrush landscape. *Earth Surf. Process. Landf., 34*(2), 193-203. https://doi.org/10.1002/esp.1703

Pierson, F. B., Robichaud, P. R., & Spaeth, K. E. (2001). Spatial and temporal effects of wildfire on the hydrology of a steep rangeland watershed. *Hydrol. Process., 15*(15), 2905-2916. https://doi.org/10.1002/hyp.381

Pierson, F. B., Robichaud, P. R., Moffet, C. A., Spaeth, K. E., Williams, C. J., Hardegree, S. P., & Clark, P. E. (2008). Soil water repellency and infiltration in coarse-textured soils of burned and unburned sagebrush ecosystems. *CATENA, 74*(2), 98-108. https://doi.org/10.1016/j.catena.2008.03.011

Pierson, F. B., Robichaud, P. R., Spaeth, K. E., & Moffet, C. A. (2003). Impacts of fire on hydrology and erosion in steep mountain big sagebrush communities. *First interagency Conference on Research in the Watersheds* (pp. 625-630). USDA, ARS. Retrieved from https://www.fs.usda.gov/treesearch/pubs/23541

R Core Team. (2021). R: A language and environment for statistical computing. Vienna, Austria: R Foundation for Statistical Computing. Retrieved from https://www.R-project.org/

Reybold, W. U., & TeSelle, G. W. (1989). Soil geographic data bases. *J. Soil Water Conserv., 44*(1), 28-29. Retrieved from https://www.jswconline.org/content/44/1/28

Robichaud, P. R., Elliot, W. J., Pierson, F. B., Hall, D. E., & Moffet, C. A. (2007). Predicting postfire erosion and mitigation effectiveness with a web-based probabilistic erosion model. *CATENA, 71*(2), 229-241. https://doi.org/10.1016/j.catena.2007.03.003

Robichaud, P. R., Wagenbrenner, J. W., Pierson, F. B., Spaeth, K. E., Ashmun, L. E., & Moffet, C. A. (2016). Infiltration and interrill erosion rates after a wildfire in western Montana, USA. *CATENA, 142*, 77-88. https://doi.org/10.1016/j.catena.2016.01.027

Sándor, R., Iovino, M., Lichner, L., Alagna, V., Forster, D., Fraser, M.,... Fodor, N. (2021). Impact of climate, soil properties and grassland cover on soil water repellency. *Geoderma, 383*, 114780. https://doi.org/10.1016/j.geoderma.2020.114780

Shen, D. (2011). Simulating snow distribution using the Water Erosion Prediction Project (WEPP) model. MS thesis. Pullman, WA: Washington State University, Department of Biological and Agricultural Engineering. Retrieved from https://rex.libraries.wsu.edu/esploro/outputs/graduate/Simulating-snow-distribution-using-the-Water/99900525391601842#file-0

Sicart, J. E., Essery, R. L., Pomeroy, J. W., Hardy, J., Link, T., & Marks, D. (2004). A sensitivity study of daytime net radiation during snowmelt to forest canopy and atmospheric conditions. *J. Hydrometeorol., 5*(5), 774-784. https://doi.org/10.1175/1525-7541(2004)005<0774:ASSODN>2.0.CO;2

Srivastava, A., Wu, J. Q., Elliot, W. J., Brooks, E. S., & Flanagan, D. C. (2017). Modeling streamflow in a snow-dominated forest watershed using the Water Erosion Prediction Project (WEPP) model. *Trans. ASABE, 60*(4), 1171-1187. https://doi.org/10.13031/trans.12035

Srivastava, A., Wu, J. Q., Elliot, W. J., Brooks, E. S., & Flanagan, D. C. (2018). A simulation study to estimate effects of wildfire and forest management on hydrology and sediment in a forested watershed, Northwestern U.S. *Trans. ASABE, 61*(5), 1579-1601. https://doi.org/10.13031/trans.12326

Sturgis, D. L. (1977). Snow accumulation and melt in sprayed and undisturbed big sagebrush vegetation. RM-RN-348. Fort Collins, CO: USDA, Forest Service Rocky Mountain Research Station.

Vega, S. P., Williams, C. J., Brooks, E. S., Pierson, F. B., Strand, E. K., Robichaud, P. R.,... Roehner, C. (2020). Interaction of wind and cold-season hydrologic processes on erosion from complex topography following wildfire in sagebrush steppe. *Earth Surf. Process. Landf., 45*(4), 841-861. https://doi.org/10.1002/esp.4778

Ward, A. D., & Elliot, W. J. (1995). *Environmental hydrology.* Boca Raton, FL: CRC Press.

Wijayawardhana, L. M., Weerasinghe, K. D., & Navaratne, C. M. (2021). Tipping bucket device for measuring runoff in small catchments. *Hydrol. Sci. J., 66*(15), 2258-2266. https://doi.org/10.1080/02626667.2021.1977307

Willgoose, G. R., & Sharmeen, S. (2006). A one-dimensional model for simulating armouring and erosion on hillslopes: 1. Model development and event-scale dynamics. *Earth Surf. Processes Landf., 31*(8), 970-991. https://doi.org/10.1002/esp.1398

Williams, C. J., Pierson, F. B., Nouwakpo, S. K., Al-Hamdan, O. Z., Kormos, P. R., & Weltz, M. A. (2020). Effectiveness of prescribed fire to re-establish sagebrush steppe vegetation and ecohydrologic function on woodland-encroached sagebrush rangelands, Great Basin, USA: Part I: Vegetation, hydrology, and erosion responses. *CATENA, 185*, 103477. https://doi.org/10.1016/j.catena.2018.02.027

Williams, C. J., Pierson, F. B., Robichaud, P. R., Al-Hamdan, O. Z., Boll, J., & Strand, E. K. (2016a). Structural and functional connectivity as a driver of hillslope erosion following disturbance. *Int. J. Wildland Fire, 25*(3), 306-321. https://doi.org/10.1071/WF14114

Williams, C. J., Pierson, F. B., Spaeth, K. E., Brown, J. R., Al-Hamdan, O. Z., Weltz, M. A.,... Nichols, M. H. (2016b). Incorporating hydrologic data and ecohydrologic relationships into ecological site descriptions. *Rangeland Ecol. Manag., 69*(1), 4-19. https://doi.org/10.1016/j.rama.2015.10.001

Young, R. A., Onstad, C. A., Bosch, D. D., & Anderson, W. P. (1989). AGNPS: A nonpoint-source pollution model for evaluating agricultural watersheds. *J. Soil Water Conserv., 44*(2), 168-173. Retrieved from https://www.jswconline.org/content/jswc/44/2/168.full.pdf