Williams *et al. Fire Ecology* (2023) 19:46
https://doi.org/10.1186/s42408-023-00201-7

Fire Ecology

Association for Fire Ecology

ORIGINAL RESEARCH

Open Access

Fuel treatments in shrublands experiencing pinyon and juniper expansion result in trade-offs between desired vegetation and increased fire behavior

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Abstract

Background Native pinyon (*Pinus* spp.) and juniper (*Juniperus* spp.) trees are expanding into shrubland communities across the Western United States. These trees often outcompete with native sagebrush (*Artemisia* spp.) associated species, resulting in increased canopy fuels and reduced surface fuels. Woodland expansion often results in longer fire return intervals with potential for high severity crown fire. Fuel treatments are commonly used to prevent continued tree infilling and growth and reduce fire risk, increase ecological resilience, improve forage quality and quantity, and/or improve wildlife habitat. Treatments may present a trade-off; they restore shrub and herbaceous cover and decrease risk of canopy fire but may increase surface fuel load and surface fire potential. We measured the accumulation of surface and canopy fuels over 10 years from ten sites across the Intermountain West in the Sagebrush Steppe Treatment Evaluation Project woodland network (www.SageSTEP.org), which received prescribed fire or mechanical (cut and drop) tree reduction treatments. We used the field data and the Fuel Characteristic Classification System (FCCS) in the Fuel and Fire Tools (FFT) application to estimate surface and canopy fire behavior in treated and control plots in tree expansion phases I, II, and III.

Results Increased herbaceous surface fuel following prescribed fire treatments increased the modeled rate of surface fire spread (ROS) 21-fold and nearly tripled flame length (FL) by year ten post-treatment across all expansion phases. In mechanical treatments, modeled ROS increased 15-fold, FL increased 3.8-fold, and reaction intensity roughly doubled in year ten post-treatment compared to pretreatment and untreated controls. Treatment effects were most pronounced at 97th percentile windspeeds, with modeled ROS up to 82 m min^{-1} in mechanical and 106 m min^{-1} in prescribed fire treatments by 10 years post-treatment compared to 5 m min^{-1} in untreated controls. Crown fire transmissivity risk was eliminated by both fuel treatments.

Conclusions While prescribed fire and mechanical treatments in shrublands experiencing tree expansion restored understory vegetation and prevented continued juniper and pinyon infilling and growth, these fuel treatments also increased modeled surface fire behavior. Thus, management tradeoffs occur between desired future vegetation and wildfire risk after fuel treatments.

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Keywords *Artemisia tridentata*, Fuel Characteristic Classification System, Fire modeling, Fuel load, Great Basin, *Juniperus occidentalis*, *Juniperus osteosperma*, Pinyon-juniper woodland, *Pinus monophylla*, Sagebrush Steppe Treatment Evaluation Project

Resumen

Antecedentes Árboles de pinos nativos (*Pinus* spp.) y juníperos (*Juniperus* spp.) se están expandiendo a lo largo y ancho de comunidades de arbustos del oeste de los EEUU. Estos árboles frecuentemente compiten con la vegetación nativa de artemisia (*Artemisia* spp.) y especies asociadas, dando como resultado un incremento en los combustibles de los doseles y un decremento en los combustibles superficiales. La expansión de estos rodales resulta frecuentemente en largos períodos de retorno del fuego con el potencial de generar fuegos de copa de gran severidad. Los tratamientos de combustibles son comúnmente usados para prevenir el ingreso y crecimiento continuo de estos árboles y reducir así el riesgo de incendios, incrementar la resiliencia ecológica, mejorar la cantidad y calidad del forraje, y/o mejorar el hábitat para la fauna. Los tratamientos pueden presentar un mecanismo de compensación; así, se puede restaurar la cobertura de arbustos y herbáceas y disminuir el riesgo de incendios de copa, pero al mismo tiempo pueden incrementarse los combustibles superficiales y por ende el riesgo de fuegos de superficie. Por diez años, medimos la acumulación de combustibles superficiales y en los doseles en diez sitios ubicados dentro de la Red del Proyecto de Tratamiento de áreas leñosas en la Estepa de Artemisia en la zona Intermontana del Oeste de los EEUU (www.SageSTEP.org), que recibía tratamientos mecánicos (corte y apilado o dejado en el lugar) o quemas prescriptas, para reducir la biomasa de árboles. Usamos los datos de campo y el Sistema de Clasificación de Combustibles Característicos (FCCS) en la aplicación Herramientas de Combustibles y Fuegos (FFT), para estimar el comportamiento del fuego en superficie y en los doseles tanto en parcelas tratadas como en controles, en las fases I, II, y III de expansión de estos árboles.

Resultados En incremento en los combustibles superficiales luego del tratamiento con quemas prescriptas incrementó la tasa modelada de propagación del fuego superficial (ROS) en 21 veces y casi triplicó la longitud de llama (FL) el décimo año luego del tratamiento en todas las fases de expansión. En los tratamientos mecánicos, la tasa de propagación (ROS) modelada se incrementó 15 veces, la longitud de llama (FL) 3,8 veces, y la intensidad de reacción resultó casi el doble en el décimo año post tratamiento comparado con el pretratamiento o los controles no tratados. Los efectos de los tratamientos fueron más pronunciados para el 97mo percentil de la velocidad del viento, con la tasa de propagación modelada de hasta 82 m por minuto en los tratamientos mecánicos y 106 m min en los tratamientos de quemas prescriptas, diez años luego de los tratamientos, comparados con 5 m min en los controles no tratados. Ambos tratamientos eliminaron el riesgo de transmisividad a fuegos de copas.

Conclusiones A pesar de que las quemas prescriptas y los tratamientos mecánicos en arbustales en los que se expanden los pinos y juníperos restauraron la vegetación nativa y previnieron la expansión y el crecimiento de árboles de estas especies, estos mismos tratamientos también incrementaron el comportamiento modelado de los fuegos superficiales. Por ese motivo, los manejadores de recursos deberían considerar las pérdidas y ganancias entre la vegetación que se desea a futuro y el riesgo de incendio luego de los tratamientos.

Background

Climate change, increased anthropogenic land uses and fire ignitions, and invasive grasses have altered fire regimes in sagebrush (*Artemisia* spp.) shrublands. In areas experiencing the invasion and expansion of non-native, annual grasses, such as cheatgrass (*Bromus tectorum*), the frequency and extent of wildfires has increased (Bradley et al. 2018). However, wildfire suppression, overgrazing, and the expansion of juniper (*Juniperus* spp.) and pinyon (*Pinus* spp.) into shrublands, coupled with progressive increases in stand density, have lengthened fire return intervals (Chambers et al. 2014a; Miller et al. 2019). This departure from historical conditions is a concern in sagebrush ecosystems due to the loss of critical wildlife habitat, reductions in biodiversity (Davies et al. 2011; Mahood and Balch 2019), loss of perennial native shrub and herbaceous cover (Ellsworth et al. 2020, Pyke et al. 2022), and increased runoff and soil erosion (Pierson et al. 2010) associated with increased wildfire intensity and frequency.

In the Intermountain West, pinyon and juniper woodlands (hereafter, woodlands) are native tree communities that historically had lower rates of expansion and infilling than is characteristic of contemporary woodlands (Miller and Tausch 2001; Miller et al. 2019). Tree encroachment across all US rangelands has resulted in a 50% increase in tree cover from 1990 to 2019 (Morford et al. 2022), with tree cover in western cold deserts on Bureau of Land Management rangelands increasing by 40% between 1991 and 2020 (Kleinhesselink et al. 2023). Woodland expansion is occurring mainly at mid to high elevations at the more mesic and productive end of sagebrush shrublands due to a suite of biophysical (i.e., aspect, elevation, soil temperature, and moisture regimes) and anthropogenic factors (i.e., land use history, fire suppression) (Chambers et al. 2014a; Miller et al. 2019). Increased temperatures and reduced precipitation have contributed to the increase and spread of woodlands, invasion of annual grasses, and lengthened fire seasons (Miller et al. 2019). Woodland encroachment has environmental consequences, including loss of understory perennial shrubs and herbs (Dittel et al. 2018; Roundy et al. 2020), increased competition between trees and understory vegetation for water and other resources (Romme et al. 2009, Miller et al. 2019, Freund et al. 2021), increased runoff and soil erosion (Peterson and Stringham 2008, Pierson et al. 2010), altered wildlife habitat (Knick et al. 2013, Braun et al. 1977), altered plant community composition (Freund et al. 2021), and reduced biodiversity (Bates et al. 2000).

Pinyon-juniper expansion has been categorized into three distinct phases, with increasing tree dominance having a substantial impact on ecological processes over time (Miller et al. 2005, 2019). In phase I, the dominant vegetation is composed of shrubs and herbaceous vegetation with few trees present. In phase II, the understory shrubs and the trees become co-dominant. In phase III, trees are the dominant vegetation and have the most influence on ecological processes (Miller et al. 2005). As woodlands become increasingly dense, shrub and herbaceous vegetation decline, shifting the majority of fuel from the surface to the tree canopy by phase III (Miller et al. 2005).

Fuel treatments are management activities that reduce or redistribute the amount of burnable material with the ultimate goal of decreasing future fire intensity or severity (Reinhardt et al. 2008). In shrublands experiencing tree expansion, treatments focus on removing or redistributing woody fuels. These treatments may also increase desirable native perennial or invasive annual grasses, and thus fine fuel loading and fuel continuity, which may have unintended consequences in regard to surface fire behavior (Bernau et al. 2018; Chambers et al. 2021). Two common fuel treatments used in shrublands experiencing pinyon and juniper expansion are mechanical cut-and-drop (trees are felled and woody material is left on the ground) and prescribed burning (McIver and Brunson 2014, Miller et al. 2019). Prescribed burning is intended to reduce shrubs and trees (woody fuel) in the short term and promote recovery of perennial bunchgrasses and shrubs in the longer-term (Rau et al. 2008; Davies et al. 2011; Chambers et al. 2014a, McIver and Brunson 2014). Mechanical treatments aim to reduce tree competition, thus increasing available resources for the shrub and herbaceous understory (Boyd et al. 2017; Dittel et al. 2018). However, mechanical treatments that leave woody fuels on the ground may contribute to an increase in surface fuels (Bernau et al. 2018, Wozniak et al. 2020). Despite widespread application of woody fuels reduction treatments in shrubland ecosystems, the long-term effect of pinyon and juniper removal on post-treatment wildfire risk and ecological resilience remains relatively unknown.

In sagebrush shrublands, land managers are actively treating large areas of woodland expansion each year to reduce pinyon and juniper and to restore sagebrush shrublands (Davies and Bates 2019). However, most research on the effects of fuel treatments has been short-term (2–3 years) and focused on the response of the vegetation and not the fuels and fire behavior (Miller et al. 2019). To better understand the longer term effects of fuel treatments on fuel loads and potential fire behavior, as well as resilience to future wildfires and resistance to invasion by annual grasses, vegetation response and fuel data were collected for over 10 years as part of the Sagebrush Steppe Treatment Evaluation Project (SageSTEP; www.sagestep.org). SageSTEP is a broad-scale and long-term collaborative management and research project that includes ten study sites across four states in the Intermountain West that are characterized by pinyon and juniper expansion into sagebrush shrublands ("woodland network") across four states in the Intermountain West (McIver et al. 2010, McIver and Brunson 2014, Freund et al. 2021). Here, we describe the effects of prescribed fire and cut and drop mechanical fuel treatments on fuels and fire behavior relative to controls in years 0, 1, 2, 3, 6, and 10 for the woodland network. We used a fire modeling framework to derive three metrics of surface fire behavior: reaction intensity (heat per unit area of the flaming front), rate of spread, and flame length; and two metrics of crown fire behavior: crown initiation (likelihood that a surface fire will transition to crown fire, i.e., torching) and crown transmissivity (potential for fire to carry through the canopy) (Prichard et al. 2013). We hypothesized that (1) prescribed fire would reduce overall fuel loads and subsequent modeled surface fire behavior compared to untreated controls, at least in the short-term; (2) mechanical cut and drop treatments would result in additional burnable material on the ground surface and increase modeled surface fire behavior compared to untreated controls; (3) reductions in surface fire behavior due to treatment would diminish with time

Table 1 Site characteristics of the SageSTEP woodland experiment network

Site	Dominant tree species	Mean annual precipitation (mm)	Soil temperature / moisture regime	Elevation (m)	Year treated
Blue Mountain, CA	Western juniper	458	Frigid/Xeric	1500–1700	2007
Bridge Creek, OR	Western juniper	306	Mesic/Aridic-Xeric	800–900	2006
Devine Ridge, OR	Western juniper	373	Frigid/Xeric	1600–1700	2007
Walker Butte, OR	Western juniper	261	Frigid/Xeric	1400–1500	2006
Marking Corral, NV	Single-needle pinyon, Utah juniper	308	Mesic/Aridic-Xeric	2300–2400	2006
Seven Mile, NV	Single-needle pinyon, Utah juniper	292	Frigid/Xeric	2300–2500	2007
South Ruby, NV	Single-needle pinyon, Utah juniper	317	Mesic/Xeric	2100–2200	2008
Greenville Bench, UT	Utah juniper, Colorado pinyon	330	Frigid/Xeric	1750–1850	2007
Onaqui Mountain, UT	Utah juniper, Colorado pinyon	322	Mesic/Aridic-Xeric	1700–2100	2006
Scipio, UT	Utah juniper, Colorado pinyon	395	Mesic/Aridic	1700–1800	2007

post-treatment as vegetation recovers; and (4) crown fire metrics would decrease with both mechanical and prescribed fire treatments, with the greatest decreases in crown fire potential in treated phase III woodlands where greater woody biomass is removed. We discuss the implications of the results for selecting pinyon and juniper treatments to reduce fuels and fire behavior vs. restoring and conserving sagebrush ecosystems.

Methods

Study sites

The Sagebrush Steppe Treatment Evaluation Project (SageSTEP; www.sagestep.org) is a network of study sites across four states in the Western US (Table 1). We used data from ten sites within the woodland network in Oregon, Northern California, Nevada, and Utah (McIver et al. 2010) (Fig. 1). Three types of woodlands are represented across the ten sites: Utah juniper (*Juniperus osteosperma*) and singleleaf pinyon-pine (*Pinus monophylla*) in Nevada, Utah juniper in Utah, and Western juniper *(J. occidentalis)* in Oregon and California (McIver et al. 2010; Wozniak and Strand 2019). Elevation across all sites ranges from 1400 to 2500 m, and annual mean precipitation ranges from 305 to 356 mm (Bernau et al. 2018; Wozniak and Strand 2019). There were no recorded fires at any of the sites for the 50 years before the study began. Study sites were fenced, and livestock grazing has been excluded in these areas since the start of the study. Further information about site characteristics and site selection can be found in McIver and Brunson (2014).

Experimental design

At each site, three 10–25 ha plots were delineated (10 sites × 3 plots) with size based on topography and site manager treatment needs. Plots were randomly selected for treatment (prescribed fire, mechanical, and untreated control) (Fig. 2). Treatments were applied using a staggered-start experimental design in 2006, 2007, or 2008 with the intention to remove all trees (mechanical and prescribed fire treatments) and to reduce the shrub layer (prescribed fire treatment only) (McIver and Brunson 2014). Mechanical and prescribed fire treatments were applied during the same year at each individual site. Treatments were applied to the broader area containing the individual plots, resulting in a buffer of the respective treatment type around each study plot.

Pre-treatment data were collected prior to treatment and are represented as Year 0 in the data. Prescribed fires were conducted during the fall with the intention of burning 100% of the plot; post-prescribed fire monitoring occurred to identify any surviving trees, which were individually ignited to achieve complete canopy consumption. Despite this, an average of 36 trees survived at each site in the prescribed fire plots, with the number of surviving trees by site ranging from 0 to 281. At the four Western juniper sites (Walker Butte, Blue Mountain, Devine Ridge, and Bridge Creek), trees were felled or girdled the year prior to prescribed burning. For the mechanical treatment, all trees > 0.5 m tall were cut at the base with a chainsaw and left where they fell in the plot (Wozniak and Strand 2019).

Within each plot, 15 measurement subplots of 0.1 ha were chosen randomly from a larger set of potential subplots (McIver and Brunson 2014). Subplots spanned a condition gradient of pretreatment tree density that was determined by the number of trees present on the site before treatment (McIver and Brunson 2014). Final subplot selection encompassed subplots across woodland phases I, II, and III (Miller et al. 2005; Wozniak and Strand 2019). Most sites contained all three phases in all three treatments, but the Bridge Creek site did not have a phase III control plot and the Walker Butte site did not have a phase III prescribed fire plot.

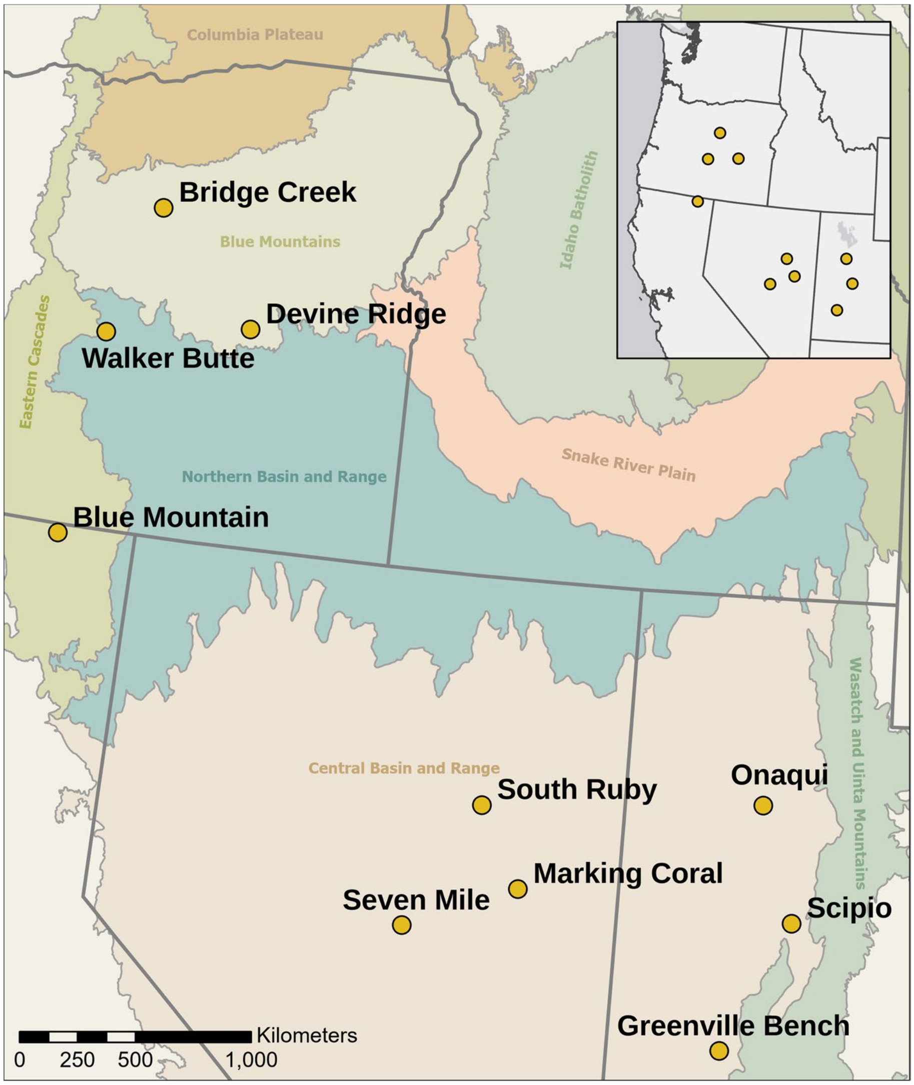

Fig. 1 Map of ten study sites from the Sagebrush Steppe Treatment Evaluation Project (SageSTEP) located within the Great Basin region using the Level III Ecoregions layer (US Environmental Protection Agency; Omernik and Griffith 2014)

Field data collection

Field measurements were collected from April to July during the peak growing season. Data were collected before treatments were applied (Year 0), as well as in years 1, 2, 3, 6, and 10 post-treatment. Canopy height, height to live crown, longest canopy diameter, and longest perpendicular canopy diameter of all trees > 0.5 m were quantified (Wozniak and Strand 2019). Tree and shrub fuel loads were estimated using site-specific allometric equations from tree and shrub height, canopy dimension, and volume (more details on the equations can be found in Wozniak and Strand [2019]). The equations were developed by Sabin (2008) and Tausch (2009) with refinements by Stebleton and Bunting (2009) and Bourne and Bunting (2011). Trees (> 50 cm) and seedlings (≤ 50 cm) were counted within each plot and averaged at the plot level, which was used to determine the number of trees per hectare. We collected herbaceous live, standing dead, and litter biomass (kg ha^{-1}) separately within 8, 0.25-m^2 quadrats. Biomass quadrats were placed at 4-m intervals rotating annually among the 11- and 19-m transects running perpendicular to the baseline. The starting position was advanced in 1-m increments every 2 years starting at the 0-m position. Tree litter and duff biomass were collected using 0.065 m^2 quadrats; litter and duff depth was not recorded (Wozniak and Strand 2019). Biomass collections were dried at 70 °C until a constant mass was achieved (approximately 48 h) and then weighed.

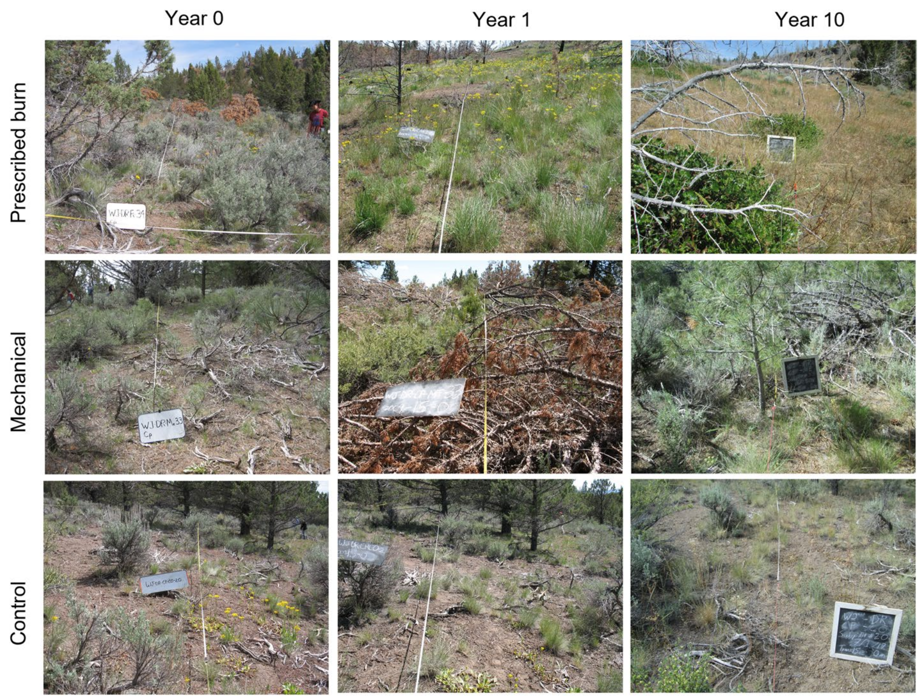

Fig. 2 Devine Ridge site photos of prescribed burn (top), mechanical (center), and control (bottom) treatment plots in years 0, 1, and 10 (SageSTEP). These pictures depict three different subplots over time (columns) by treatment (rows)

Fuel size classes of 10-h (0.6–2.5 cm), 100-h (2.5–7.6 cm), 1000-h (>7.6 cm; sound), and 1000-h (rotten) time lag fuels were collected for down woody debris using the planar-intercept method (Brown et al. 1982). All data were averaged to the plot level for analysis. South Ruby burned during the tenth year of the study; thus, there is no year 10 post-treatment data for this site. Fuel summaries for all plots for the first 10 years post-treatment are published in Wozniak and Strand (2019).

Fire behavior modeling

Surface fire behavior

To determine the impacts of fuel treatments on surface fire behavior over the 10-year study, we utilized the fire behavior modeling capabilities of the Fuel Characteristic Classification System (FCCS) in the Fuel and Fire Tools (FFT) application (Prichard et al. 2013). The FCCS predicts surface fire behavior using localized fuel data, wind speeds, and fuel moisture scenarios to reflect conditions throughout a typical fire season. We used field data to create custom fuel models representing fuel amount, structure, and arrangement for each treatment (mechanical, prescribed fire, and control) and year (0, 1, 2, 3, 6, and 10) combination. Custom fuel beds were initiated using the pre-set fuelbed 58: Western juniper/sagebrush savannah—post prescribed burn for all post-treatment prescribed burn plots and mechanical thinning plots, or 69: Western juniper/sagebrush-bitterbrush shrubland for all pre-treatment, control plots (Prichard et al. 2013). Then, in situ fuels data were entered to represent the amount and type of fuels at each site, generating a custom fuel model for treatment by year combination. For fuel categories not collected in the field (tree diameter at breast height [dbh], duff cover and depth, moss depth, and snag cover, density and dbh) and for one input that changes through a field season (live to dead herbaceous ratio), FCCS defaults were used. The custom fuel beds were used to predict surface fire behavior and crown ignition potential. Moisture scenarios were selected to mimic the progression of vegetation moisture content through the growing season from spring green-up through fall curing when vegetation has dried out and there is a higher risk of fire. The modeled moisture scenarios in FFT were fully green (D2L4), $^1/_3$ cured (D2L3), $^2/_3$ cured (D2L2), and fully cured (D2L1). Moisture scenarios are calculated with a moisture damping coefficient, which has a linear relationship with the model outputs, such that more extreme fire behavior is predicted with drier fuels and reduced fire behavior with moist fuels (Prichard et al. 2013). No field data on fuel moisture was collected at these sites, but the moisture scenarios used by the model fall within the range of field measurements of fuel moisture reported by Wright and Prichard (2006) for several locations within the sagebrush shrubland. The slope was set to 13%, the average slope of all woodland plots in the SageSTEP network, and wind speeds modeled represented mean 50th, 80th, 90th, and 97th percentile wind speed over the summer (June–September) from the nearest remote automated weather station (RAWS) across all sites and study years. Each custom fuel model was run at each moisture scenario to predict surface rate of spread (ROS; m/min), reaction intensity (RI; kW m^{-2} min^{-1}), and flame length (FL; m) at the 50th percentile wind speed, including wind gusts (19 kph). To estimate extreme fire conditions, we also modeled ROS and FL in the fully cured scenario at 80th (32 kph), 90th (38 kph), and 97th (48 kph) percentile wind speeds.

Crown fire behavior

The FCCS also outputs “crown fire initiation” (CI), an index of the potential for torching (the ignition of individual trees), and “crown transmissivity” (CT), an index of the potential for fire spread through the canopy. These metrics are expressed using a unitless index of 0 to 9, where 0 indicates no potential for crown-to-crown ignition or transmissivity, respectively, and 9 indicates certainty of ignition or spread. Crown fire metrics are entirely based on fuel loading, as FCCS does not support tests of multiple moisture scenarios for crown fuels and does not incorporate spotting potential (Sandberg et al. 2007; Johnston et al. 2021).

Analysis

For all models, treatment plots within sites were considered replicates, and site was treated as a random factor. To evaluate treatment impacts on fuel loads, linear mixed models were used to test for differences in shrub, herbaceous, litter, and downed wood fuel loads, tree, and seedling density as a function of treatments, woodland phase, and year post-treatment (hereafter, year). Interactions between year × treatment, year × phase, treatment × phase, and year × treatment × phase were used to test whether fuel treatment effectiveness differed by phase or through time. To evaluate treatment impacts on modeled fire behavior metrics, linear mixed models were used to test for differences in surface fire behavior and crown fire behavior as a function of treatments, woodland phase, environmental scenario, and year post-treatment. Interactions between year × treatment, year × phase, treatment × phase, and year × treatment × phase were used to test whether fuel treatment effectiveness differed by phase or through time. To evaluate treatment impacts on extreme fire behavior, linear mixed models were used to test for differences in ROS, FL, and RI in the fully cured moisture scenario only as a function of windspeed, treatments, woodland phase, and year. Interactions between year × treatment, year × phase, treatment × phase, treatment × windspeed, year × windspeed, phase × windspeed, treatment × year × wind, year × phase × wind, and treatment × phase × wind were used to test whether fuel treatment effectiveness differed by phase, through time, or with increasing wind. The Tukey-Kramer honest significant difference (HSD) post hoc analysis was used to determine the differences between groups. Analyses were performed using IBM SPSS 27 (IBM Corp 2020).

Results

Surface fuel loads

Total surface fuels averaged 6.23 Mg ha^{-1} across all plots prior to treatments and did not change by year 10 in control plots (Fig. 3; $p > 0.05$). Total surface fuels increased to a mean of 11.13 Mg ha^{-1} across prescribed fire plots ($p < 0.01$) and to 21.9 Mg ha^{-1} across mechanical plots ($p < 0.01$) by year 10 (Fig. 3).

Prior to treatment, shrub fuels averaged 3.37 Mg ha^{-1} in phase I woodlands, 2.19 Mg ha^{-1} in phase II, and 0.89 Mg ha^{-1} in phase III woodlands (Table 2; $p < 0.01$). Prescribed fire reduced shrub fuels to 0.17–0.22 Mg ha^{-1} across all phases. Shrubs gradually recovered in all plots, though never reached pre-fire or control fuel levels. In phase I plots, shrub fuels were 1.02 Mg ha^{-1} in year 10 post-treatment, and in phase II and III plots, they reached 0.87 Mg ha^{-1} and 1.35 Mg ha^{-1}, respectively (phase × treatment effect, $p < 0.01$). Shrub fuels in mechanical plots were higher than fire or untreated controls (treatment effect, $p < 0.01$; treatment × year effect, $p = 0.02$) and slowly accumulated with time since treatment, reaching 4.49 Mg ha^{-1}, 4.07 Mg ha^{-1}, and 1.94 Mg ha^{-1} in phase I, phase II, and phase III woodlands, respectively.

Herbaceous fuels in untreated control plots averaged 0.30 Mg ha^{-1} across all treatment years and woodland phases ($p > 0.05$). Prescribed fire increased herbaceous fuels to 0.68 Mg ha^{-1} by year 2 and remained elevated through year 10 ($p < 0.01$). Mechanical treatments also increased herbaceous fuels to 0.62 Mg ha^{-1} by post-treatment year 3, and they remained high relative to the controls through year 10 ($p < 0.01$). The treatments effects on herbaceous fuel was only marginally different by phase ($p = 0.06$) though there was a trend towards more herbaceous fuel in phases I and II than in phase III.

Litter fuel was a small proportion of the total fuel load. Litter was lower in fire treatment plots (0.23 Mg ha^{-1}) than control (0.36 Mg ha^{-1}) or mechanical treatments (0.36 Mg ha^{-1}) across all years and phases ($p < 0.01$). Year was a significant predictor of litter fuel, with the lowest fuel loads occurring in year 2 (0.18 Mg ha^{-1}), and the highest litter fuel loads in year 10 (0.42 Mg ha^{-1}). There was no significant treatment by time interaction ($p = 0.21$). Woodland phase was not a significant predictor of litter variability ($p = 0.24$).

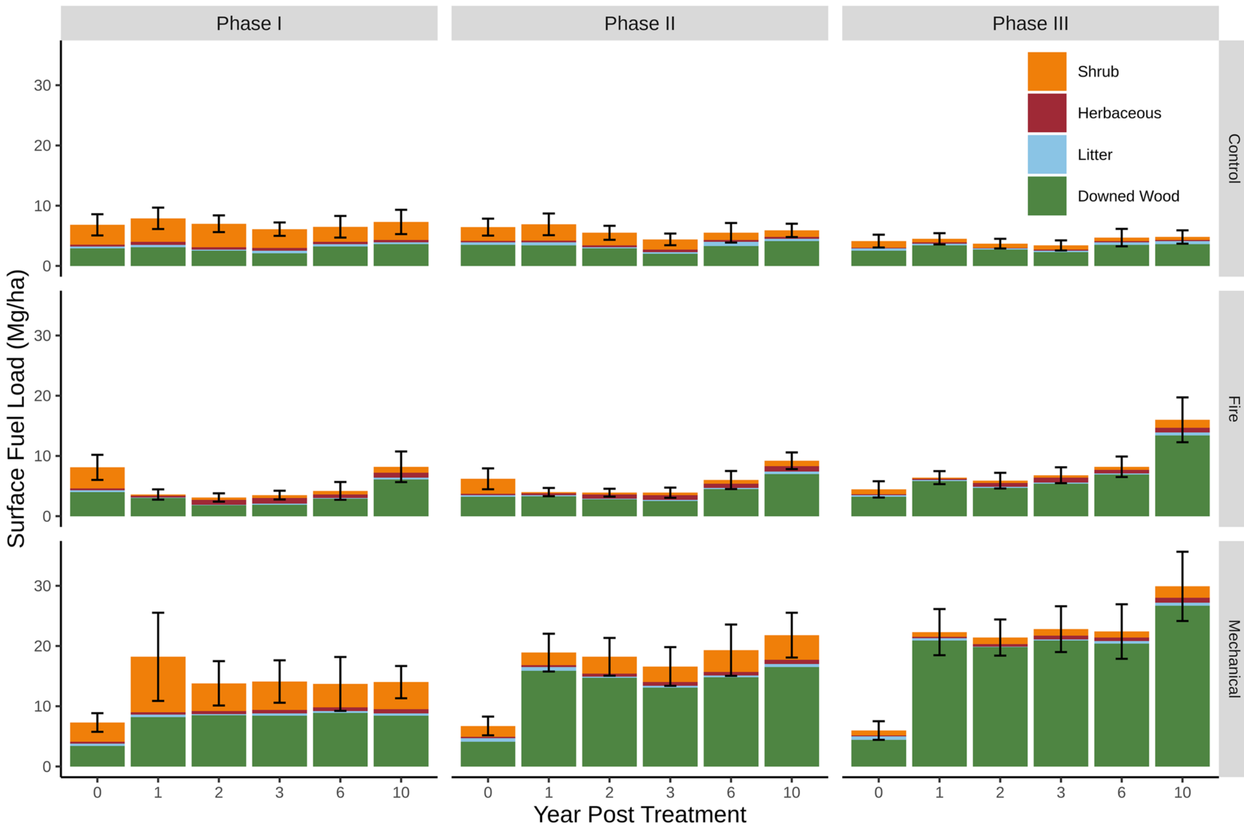

Fig. 3 Shrub, herbaceous, litter, and downed woody fuel (Mg ha^{-1}) in control (top), prescribed fire (center), and mechanical treatment (bottom) plots for woodland phases I (left), II (center), and III (right) in years 0, 1, 2, 3, 6, and 10 post-treatment. The sum of all colored bars represents mean total fuel for each year by phase, treatment combination, and error bars represent ± 1 standard error for the total fuel

Table 2 Linear mixed models predicting the shrub, herbaceous, litter, and downed woody debris fuel loads as a function of woodland phase, fuel treatment (prescribed fire, mechanical, or untreated control), and year (years 0, 1, 2, 3, 6, and 10 post-treatment) from ten replicate sites across the Intermountain West, USA

	Shrub		**Herbaceous**		**Litter**		**Downed wood**	
Source	***F***	***P***	***F***	***P***	***F***	***P***	***F***	***P***
Intercept	25.3	**< 0.001**	60.3	**< 0.001**	20.2	**< 0.001**	73.3	**< 0.001**
Year	0.4	0.83	47.3	**< 0.001**	4.4	**0.001**	23.2	**< 0.001**
Treatment	27.9	**< 0.001**	73.1	**< 0.001**	7.5	**< 0.001**	251.1	**< 0.001**
Phase	31.5	**< 0.001**	14.4	**< 0.001**	1.4	0.243	48.1	**< 0.001**
Year × treatment	2.2	**0.02**	10.4	**< 0.001**	1.3	0.21	9.9	**< 0.001**
Year × phase	0.9	0.54	0.9	0.58	0.9	0.57	2.8	**0.01**
Treatment × phase	5.4	**< 0.001**	2.3	0.06	0.3	0.87	23.3	**< 0.001**
Year × treatment × phase	1.1	0.32	0.4	0.99	0.3	0.99	1.1	0.36

Downed woody fuels averaged 3.48 Mg ha^{-1} across all plots prior to treatment. Prescribed fire plots had consistent downed wood fuels through year 6 but they increased to 8.93 Mg ha^{-1} in year 10. Mechanical treatments had an even larger effect on downed wood fuel. In year 1, downed woody fuels increased to 15.03 Mg ha^{-1} and they remained high (up to 16.5 Mg ha^{-1} in phase II to 26.70 Mg ha^{-1} in phase III; $p < 0.01$) through 10 years post-treatment.

Table 3 Linear mixed models predicting tree and seedling density as a function of fuel treatment (prescribed fire, mechanical, or untreated control), woodland phase, and year (years 0, 1, 2, 3, 6, and 10 post-treatment) from ten replicate sites across the Intermountain West, USA

	Tree density		Seedling density	
Source	***F***	***P***	***F***	***P***
Intercept	68.6	**< 0.001**	20.5	**< 0.001**
Year	67.4	**< 0.001**	2.3	**0.042**
Treatment	390.3	**< 0.001**	12.5	**< 0.001**
Phase	9.8	**< 0.001**	12.4	**< 0.001**
Year × treatment	19.7	**< 0.001**	1.82	0.06
Year × phase	1.0	0.42	0.19	0.99
Treatment × phase	1.1	0.38	1.58	0.18
Year × treatment × phase	0.5	0.96	0.52	0.96

Canopy fuels

Tree density was altered by both treatments (Table 3; Fig. 4). Total tree density (trees taller than 50 cm) prior to treatment averaged 324.15 ha^{-1} in phase I, 337.12 ha^{-1} in phase II, and 353.07 ha^{-1} in phase III woodlands ($p < 0.01$). Control plot density did not change over time ($p = 0.92$). Prescribed fire treatments reduced tree density to a mean of 36.03 ha^{-1} in the first year post-treatment ($p < 0.01$), and they remained reduced through the 10-year study period ($p < 0.01$). Mechanical treatments completely removed trees greater than 0.5 m ($p < 0.01$) but by year 10 there was a significant increase to 178.72 trees ha^{-1} ($p < 0.01$). There was no evidence that treatments differentially impacted tree density by phase ($p = 0.43$).

Prior to treatment, total seedling density (trees ≤ 50 cm) averaged 249.21 ha^{-1} in phase I, 379.62 ha^{-1} in phase II, and 420.84 ha^{-1} in phase III woodlands ($p < 0.01$; Table 3; Fig. 4). Prescribed fire treatments reduced seedling density overall compared to control and mechanical plots to an average of 135.70 ha^{-1} ($p < 0.01$). Seedlings in

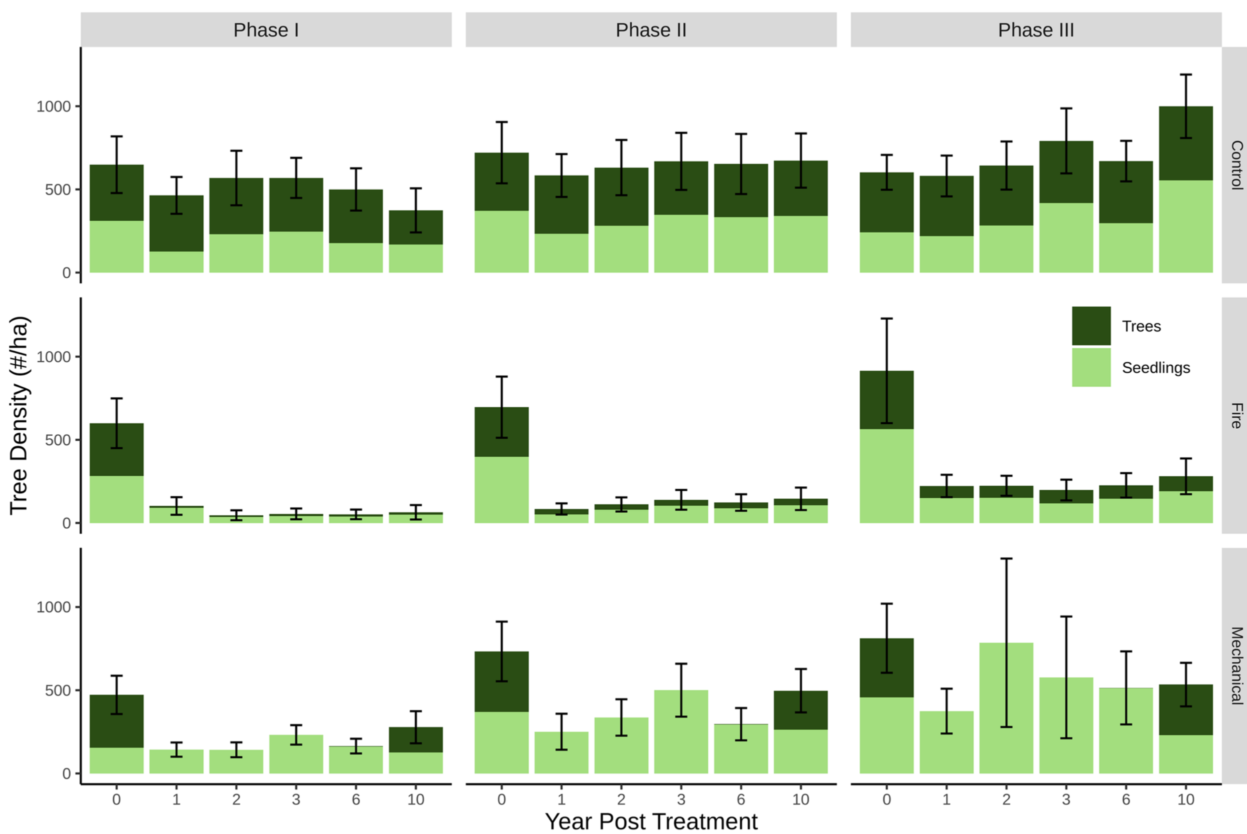

Fig. 4 Seedling and tree density (number of trees/hectare) in control (top), prescribed fire (center), and mechanical treatment (bottom) plots for woodland phases I (left), II (center), and III (right) in years 0, 1, 2, 3, 6, and 10 post-treatment. The sum of all colored bars represents the mean combined tree and seedling density for each year by phase and treatment combination and the error bars represent ± 1 standard error around the mean combined tree and seedling density

Table 4 Linear mixed models predicting the rate of surface fire spread, flame length, and reaction intensity as a function of moisture scenario (% moisture by fuel class), woodland phase, fuel treatment (prescribed fire, mechanical, or untreated control), and year (years 0, 1, 2, 3, 6, and 10 post-treatment) from ten replicate sites across the Intermountain West, USA

	Rate of spread		Flame length		Reaction intensity	
Source	***F***	***P***	***F***	***P***	***F***	***P***
Intercept	51.3	**< 0.001**	174.5	**< 0.001**	140.5	**< 0.001**
Year	68.6	**< 0.001**	228.0	**< 0.001**	38.5	**< 0.001**
Treatment	387.2	**< 0.001**	1041.8	**< 0.001**	268.4	**< 0.001**
Phase	10.2	**< 0.001**	75.8	**< 0.001**	253.4	**< 0.001**
Moisture Scenario	108.9	**< 0.001**	305.5	**< 0.001**	210.7	**< 0.001**
Year × treatment	21.7	**< 0.001**	57.2	**< 0.001**	20.8	**< 0.001**
Year × phase	3.0	**0.01**	1.2	0.26	5.8	**< 0.001**
Treatment × phase	22.5	**< 0.001**	2.1	0.08	45.6	**< 0.001**
Year × treatment × phase	3.1	**< 0.001**	1.1	0.34	4.8	**< 0.001**

mechanical treatments did not differ from control plots ($p = 0.37$). Additional data on the canopy fuels across all subplots, phases, and treatments were reported in Stebleton and Bunting (2009) and Wozniak and Strand (2019).

Surface fire behavior

Rate of spread

Modeled ROS increased as fuel moisture decreased and the herbaceous fuel cured, mimicking the natural wildfire season regardless of treatment, phase, or year ($p < 0.001$; Table 4; Fig. 5). Prior to treatment, modeled ROS averaged 0.97 m min^{-1} across all treatments (year 0; $p > 0.05$) but increased to 11.14 m min^{-1} in prescribed fire and to 8.53 m min^{-1} in mechanical treatments by year 1 and remained higher than untreated controls through all 10 sampled years ($p < 0.05$; Fig. 5). Modeled ROS 10 years after mechanical treatments was 16× higher than in untreated controls in fully cured plots ($p < 0.01$), and prescribed fire treatments increased modeled ROS by 21-fold compared to untreated controls at year 10 regardless of phase ($p < 0.01$). The greatest differences among treatments occurred when fuels were fully cured ($p < 0.01$). While both treatments resulted in increases in modeled ROS relative to untreated controls, modeled ROS was higher for prescribed fire-treated plots in woodland phases I and II ($p < 0.01$), while modeled ROS was highest in mechanical treatment plots in phase III woodlands ($p < 0.01$; Fig. 5).

Flame length

Modeled flame lengths were lowest (mean 0.61 m) across all treatments, years, and phases when fuels were green and significantly increased to 1.33 m when fuels were fully cured ($p < 0.05$; Fig. 6). Pre-treatment woodland phase did not significantly affect modeled flame lengths for any treatment ($p = 0.08$). Modeled flame length depended on year and treatment (year × treatment interaction, $p < 0.05$; Table 4). Modeled flame lengths increased through time for both mechanical and fire plots with post-treatment year 10 having the highest flame lengths ($p < 0.05$), while control plot flame length remained consistently low. Modeled flame length averaged 0.45 m across all treatments, phases, and moisture scenarios prior to treatment (year 0). Post-treatment, flame lengths were 1.0 m higher in prescribed fire plots and 1.64 m higher in mechanical plots compared to the untreated control across post-treatment years, moisture scenarios, and woodland phases (Fig. 6). At year 10, fully cured mechanical plots had a mean 3.6× modeled increase and prescribed fire plots had a 2.8× modeled increase in flame lengths compared to fully cured controls (Fig. 6).

Reaction intensity

Moisture scenario increased modeled RI regardless of other factors: average modeled RI prior to treatment across all phases increased from 157.7 kW m^{-2} min^{-1} when fully green to 303.5 kW m^{-2} min^{-1} when herbaceous fuels were fully cured (Table 4; Fig. 7). Treatment effects varied by woodland phase and time since treatment. Untreated control plot RI did not change through time ($p > 0.05$), averaging 501.7 kW m^{-2} min^{-1} across years in phase I woodlands, 344.0 kW m^{-2} min^{-1} in phase II woodlands, and 185.7 kW m^{-2} min^{-1} in phase III woodlands under the fully cured moisture scenario. In phase I woodlands, prescribed fire reduced modeled RI by 76–78% and it remained reduced by ≥52% in year 10 ($p < 0.01$); however, mechanical treatments in phase I woodlands slightly increased RI ($p < 0.01$). In phase II woodlands, mechanical and control plots had similar

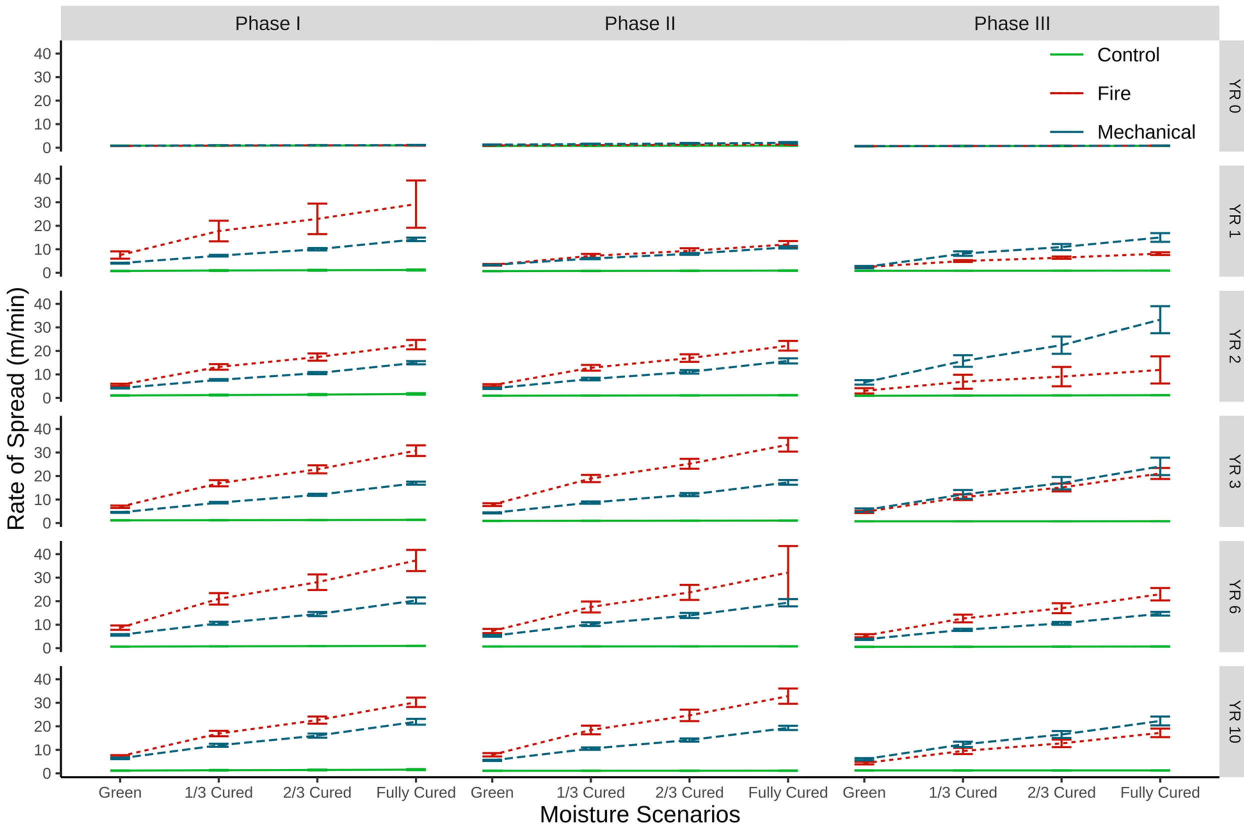

Fig. 5 Surface rate of spread by treatment type and woodland phase in years 0, 1, 2, 3, 6, and 10 following treatment as a function of moisture scenario (modeled fuel moisture content). The colored lines represent the mean by treatment type and the error bars represent ± 1 standard error around the respective mean

modeled RI through year 3 ($p > 0.05$), but by year 6 modeled RI in mechanical treatment plots were 58% higher than controls, and by year 10 modeled RI in mechanical plots was double that of controls ($p < 0.05$; Fig. 7). Prescribed fire in phase II woodlands initially reduced modeled RI compared to controls, but the effect lasted only through year 3 ($p < 0.05$). In phase III woodlands, modeled RI in prescribed fire and mechanical treatment plots was significantly higher than controls by year 3 ($p < 0.05$) and increased with time since treatment. By year 10, prescribed fire plots increased modeled RI by 78% compared to untreated controls and mechanically treated plots more than tripled modeled RI ($p < 0.05$; Table 4; Fig. 7).

Extreme fire behavior

Increases in wind speed, as expected, resulted in significant increases in modeled fire behavior ($p < 0.001$). Across all pre-treatment plots at the driest fuel moisture scenario (D2L1), modeled ROS averaged 1.1 m min^{-1}, 2.3 m min^{-1}, 2.9 m min^{-1}, and 4.1 m min^{-1}, and surface flame length averaged 0.61 m, 0.85 m, 0.96 m, and 1.12 m for the 50th, 80th, 90th, and 97th windspeed scenarios, respectively, when fuels are fully cured (Figs. 8 and 9). Both prescribed fire and mechanical treatments resulted in increases in modeled ROS and FL compared to untreated controls across all windspeed scenarios. This was most pronounced at the extreme windspeeds (treatment × windspeed, $p < 0.01$; Figs. 8 and 9; Table 5), with modeled ROS up to 82 m min^{-1} in mechanical treatments and 106 m min^{-1} in prescribed fire treatments by 10 years post-treatment, compared to 5 m min^{-1} in untreated controls. Similarly, modeled FL at year 10 was 5.2 m in mechanical treatments, 4.2 m in prescribed fire, but only 1.1 m in untreated controls at the 97th percentile windspeed scenario. The relationships between windspeed, treatment, and fire behavior were not strongly altered by pretreatment woodland phase (Table 5; Figs. 8 and 9).

Crown fire behavior

Crown initiation

Crown initiation is an index from 0 to 9 describing the potential for any seedlings or trees present to ignite (i.e., torching). Across all treatments, phases, and years, the

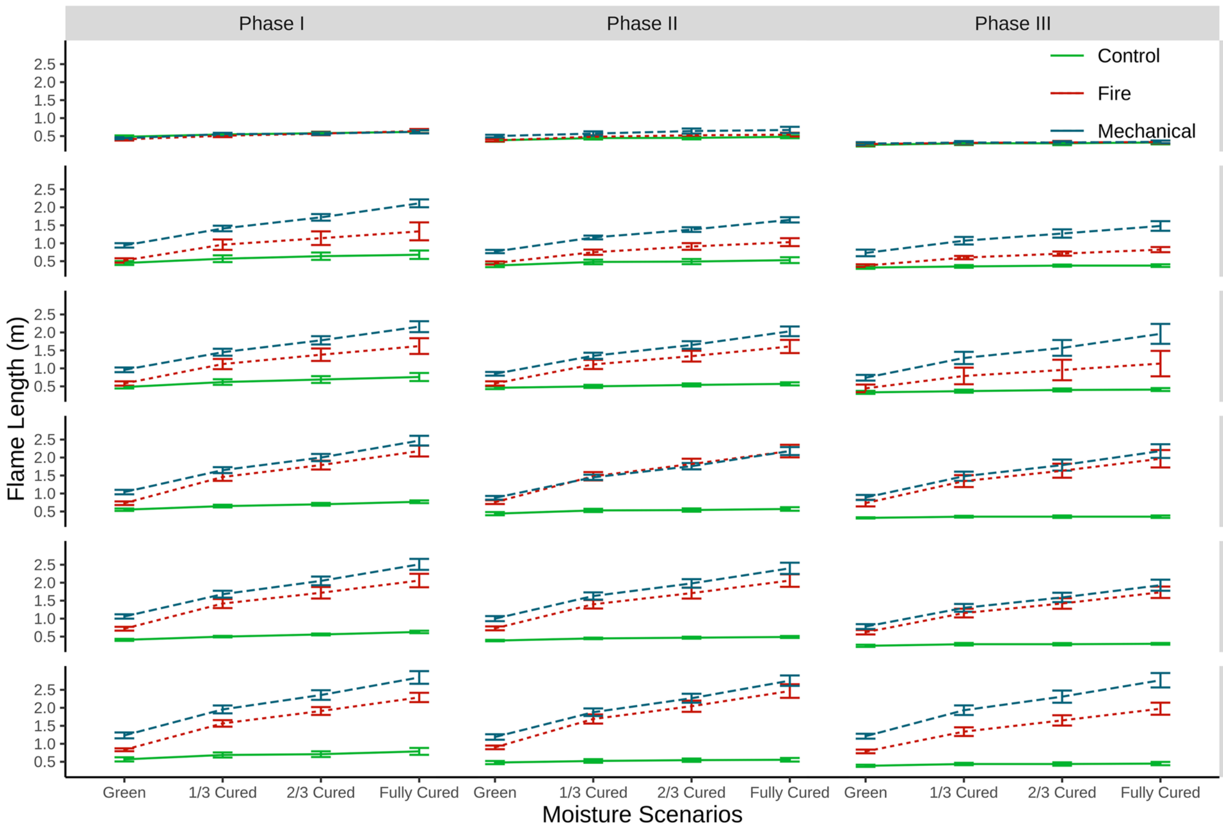

Fig. 6 Surface fire flame length (m) by treatment type in years 0, 1, 2, 3, 6, and 10 following treatment as a function of moisture scenario (modeled fuel moisture content). The colored lines represent the mean by treatment type and the error bars represent ± 1 standard error around the respective mean

mean crown initiation (CI) index was 4.8, but that risk significantly varied by treatment ($p < 0.01$), year ($p < 0.01$), and phase ($p < 0.01$), as well as an interaction between year × treatment and treatment × phase (Table 6). Control plots had more trees and seedlings overall than the treatment plots (Fig. 4), but those trees and seedlings were less likely to ignite (mean CI 3.7; Fig. 10A) due to sparse herbaceous fuels to carry surface fire. Mechanical treatments had significantly higher crown initiation potential (5.7) than prescribed fire treatments, likely due to retained shrub cover and increased downed wood (5.1; $p < 0.01$). Both treatments had higher CI potential than untreated controls ($p < 0.01$). Crown initiation significantly decreased from phase I through phase III plots; phase I plots had highest torching potential (5.3), and phase III plots were lowest (4.3), with intermediate risk in phase II (4.9; $p < 0.01$; Fig. 10A). Crown initiation potential increased with time since treatment for both prescribed fire and mechanical treatment plots, but untreated control risk remained constant (time × treatment, $p < 0.01$). Note that CI does not distinguish between seedling and tree density (Fig. 4).

Crown transmissivity

Crown transmissivity indicates risk of crown-to-crown fire spread and is similarly an index from 0 to 9. Prior to treatment, CT was zero for all phase I and phase II plots, but most phase III plots had some risk of crown fire spread (Fig. 10B). By year 10, the potential for crown fire spread in some phase II control plots was observed as infilling of trees increased density (Fig. 4). Both prescribed fire and mechanical treatments eliminated the risk of modeled crown fire spread entirely for all 10 post-treatment years (Fig. 10B; Table 6).

Discussion

Woody fuel treatments are commonly used throughout the Intermountain West to reduce pinyon and juniper woodland expansion and restore native shrub and herbaceous vegetation (Baughman et al. 2010, McIver and Brunson 2014, Miller et al. 2019). Across much of the Intermountain West, protecting core habitat for sage grouse and other species of conservation concern is a high priority (Paige and Ritter 1999, Maestas et al. 2022) and implementing measures to reduce wildfire risk are

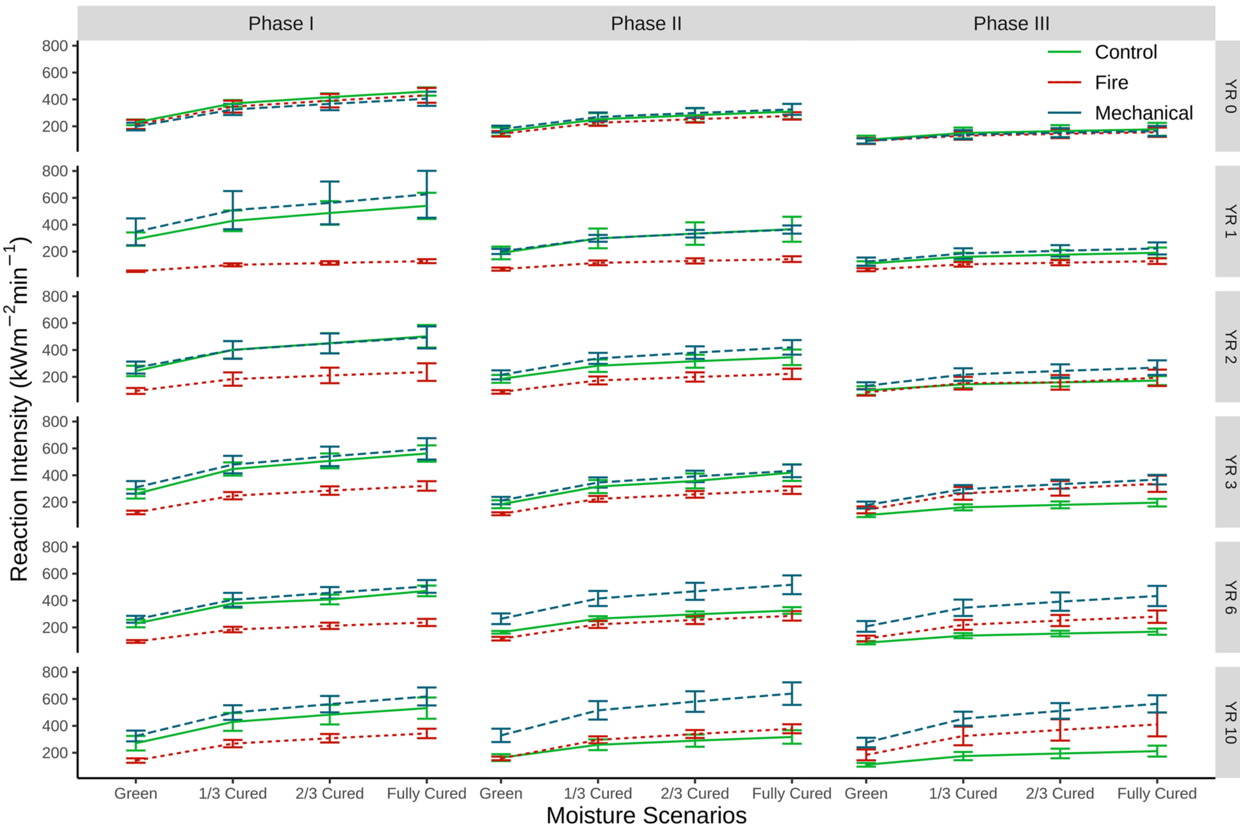

Fig. 7 Surface reaction intensity (kW m^{-2} min^{-1}) by treatment type in years 0, 1, 2, 3, 6, and 10 post-treatment as a function of moisture scenario (modeled fuel moisture content). The colored lines represent the mean by treatment type and the error bars represent ± 1 standard error around the respective mean

prioritized (Integrated Rangeland Fire Management Strategy Actionable Science Plan Team 2016; Chambers et al. 2017). While the goal of fuel treatments can be to increase sagebrush and herbaceous vegetation and improve habitat, our models showed that post-treatment increases in surface fuels were conducive to more rapidly spreading wildfire (prescribed fire treatment) and more intense wildfire (mechanical cut and drop treatment) when compared to untreated areas. These 10-year results indicated that fuel treatments result in trade-offs between increases in desired vegetation and undesirable surface fire behavior. However, a longer-term trade-off also exists for management consideration—not treating increases the likelihood of progressive tree infilling and growth. Increased infilling and growth of trees may compete with understory vegetation, increase the risk of high severity crown fires, and ultimately reduce the recovery potential of a site following wildfire.

Contrary to our first hypothesis, prescribed fire treatments resulted in increased modeled flame length and rate of surface fire spread when compared to the pretreatment and control plots due to the increase of surface fuel loading, especially the herbaceous and downed wood fuel categories. Prescribed fire also resulted in higher rates of modeled surface fire spread in phase I and II woodlands than mechanical treatments. By removing trees, available water, space, light, and nutrient levels likely increased, allowing native and non-native, herbaceous vegetation to increase (Rau et al. 2008; Miller et al. 2014; Roundy et al. 2014). The increase in herbaceous fuels (Figs. 2 and 3) was consistent with earlier results from these study sites (Bourne and Bunting 2011; Bernau et al. 2018; Wozniak and Strand 2019) and was responsible in part for the increase of modeled surface fire behavior post-treatment.

Previous studies on plot level fire behavior modeling in the sagebrush steppe have shown prescribed fire to decrease surface fire behavior (Ellsworth et al. 2022) emphasizing the importance of understanding the site composition before implementing treatment (Miller et al. 2019). For example, we did not see a parallel increase in modeled wildfire behavior after treatment at warmer and drier shrubland sites within the SageSTEP shrubland network (Ellsworth et al. 2022), where pinyon and

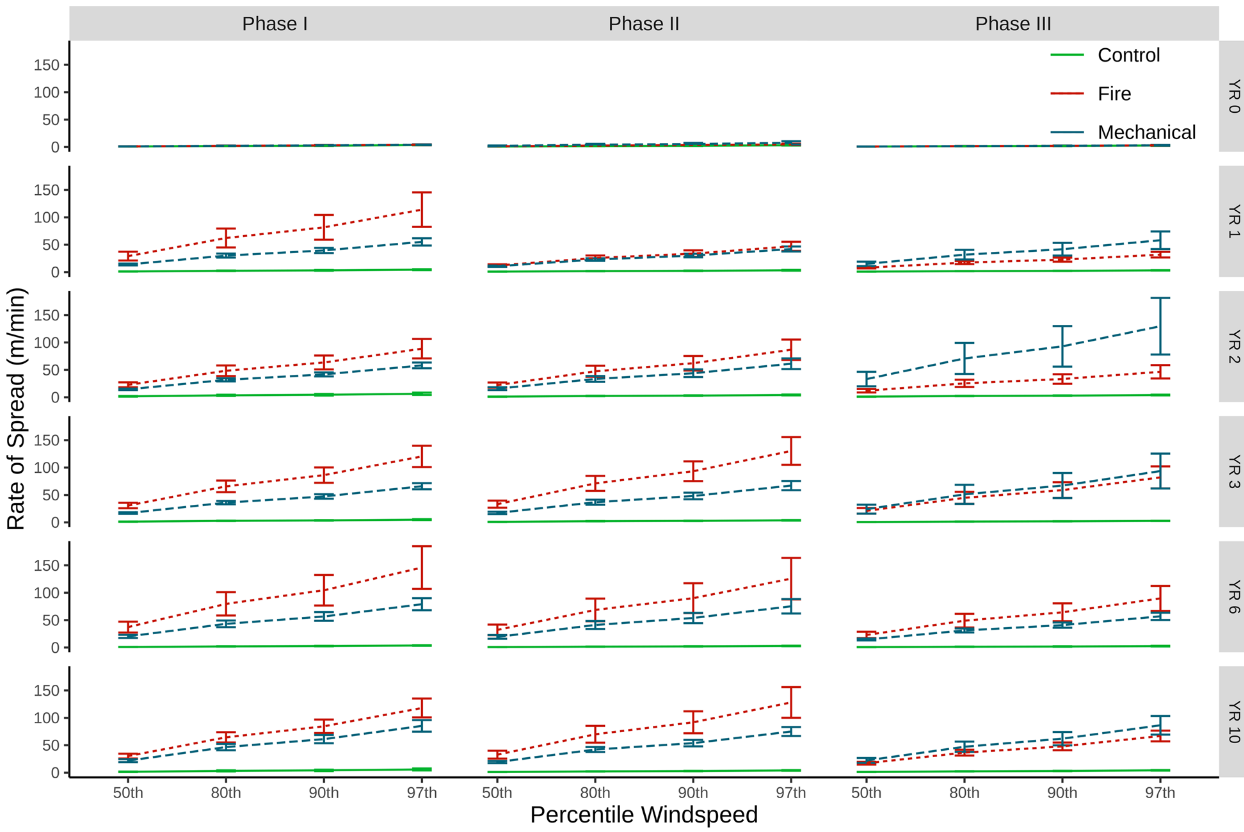

Fig. 8 Surface rate of spread by treatment type and woodland phase in years 0, 1, 2, 3, 6, and 10 following treatment as a function of percentile wind speed. The colored lines represent the mean by treatment type and the error bars represent ± 1 standard error around the respective mean

juniper expansion are not a concern. Prior research showed that prescribed burning in the SageSTEP woodland network sites resulted in the lowest sagebrush shrub cover 10 years post-treatment compared to mechanical and control plots (Freund et al. 2021). However, Freund et al. (2021) reported that prescribed burning increased cover of other shrubs when compared to control sites. Site characteristics impacted the post-treatment plant community response in the SageSTEP woodland network sites (Chambers et al. 2014b; Roundy et al. 2018; Freund et al. 2021). In particular, the relative resilience and resistance of the sites, as indicated by soil climatic regime, was a driving factor of post-treatment response, with sites located in cool and moist soil regimes having greater post-treatment recovery than sites in warm and dry soil regimes (Chambers et al. 2014a, b; Roundy et al. 2018; Freund et al. 2021). Warm and dry soils with lower resilience and resistance resulted in increased perennial graminoids but decreased sagebrush cover while cool and moist soils resulted in increases in both functional groups (Chambers et al. 2021; Freund et al. 2021).

As predicted, mechanical treatments shifted most canopy fuels to surface downed woody fuel, which increased modeled surface wildfire spread, flame length, and reaction intensity when compared to the untreated controls. The modeled reaction intensity in the mechanical plots was roughly double that in the control plots (Fig. 7), due to the increase in large diameter woody surface fuel that both increases residence time and prolongs smoldering combustion (Brown et al. 2003). Tree density in the mechanical treatment plots remained lower than pretreatment and control levels across years and phases. By year 10, seedlings in the mechanical plots were beginning to reach the height threshold to shift into the tree category, indicating a need to consider appropriate retreatment intervals to prevent re-infilling of trees in shrublands. Herbaceous fuels increased in mechanical treatments with a strong response in early post-treatment years that leveled off in later years, which is consistent with previous results (Bourne and Bunting 2011; Bernau et al. 2018; Wozniak and Strand 2019). At 10 years post-treatment, sagebrush cover was higher in mechanical plots compared to prescribed fire and untreated control plots (Fig. 3; Freund et al. 2021). This increase varied by phase, with the greatest increase in sagebrush cover occurring at phase III plots. Unlike prescribed fire, the

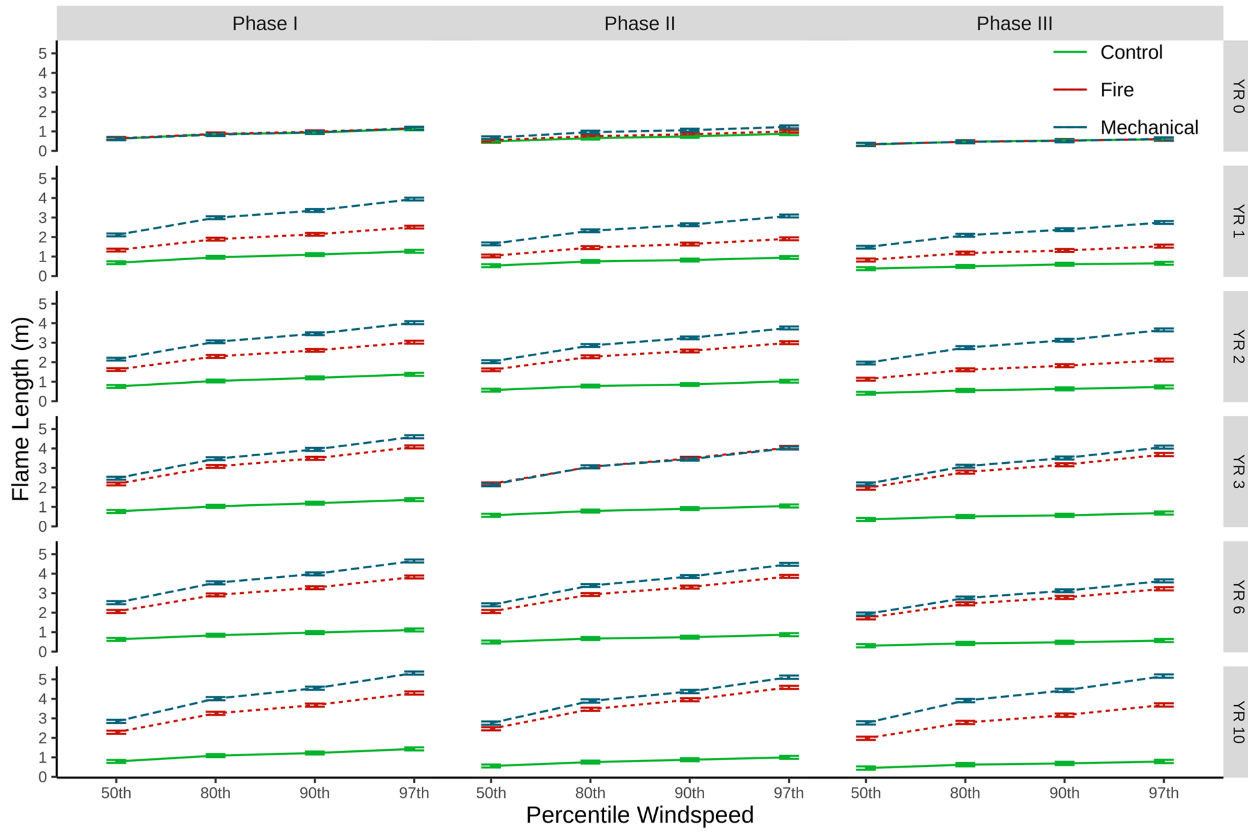

Fig. 9 Surface fire flame length by treatment type and woodland phase in years 0, 1, 2, 3, 6, and 10 following treatment as a function of percentile wind speed. The colored lines represent the mean by treatment type and the error bars represent ± 1 standard error around the respective mean

Table 5 Linear mixed models predicting the rate of surface fire spread and flame length as a function of windspeed (50th, 80th, 90th, or 97th percentile), woodland phase, fuel treatment (prescribed fire, mechanical, or untreated control), and year (years 0, 1, 2, 3, 6, and 10 post-treatment) from ten replicate sites across the Intermountain West, USA.

	Rate of spread		**Flame length**	
Source	***F***	***P***	***F***	***P***
Intercept	48.4	**< 0.001**	170.5	**< 0.001**
Year	78.0	**< 0.001**	355.7	**< 0.001**
Treatment	431.0	**< 0.001**	1655.1	**< 0.001**
Phase	9.3	**< 0.001**	96.1	**< 0.001**
Windspeed	118.9	**< 0.001**	253.8	**< 0.001**
Year × treatment	25.0	**< 0.001**	89.6	**< 0.001**
Year × phase	3.7	**< 0.001**	2.0	0.30
Treatment × phase	24.3	**< 0.001**	2.8	**0.03**
Treatment × windspeed	26.3	**< 0.001**	26.3	**< 0.001**
Year × windspeed	4.8	**< 0.001**	5.7	**< 0.001**
Phase × windspeed	0.5	0.79	1.3	0.23
Treatment × year × wind	1.6	**0.03**	1.4	0.06
Year × phase × wind	0.2	1.00	0.04	1.00
Treatment × phase × wind	1.5	0.12	0.04	1.00

Table 6 Linear mixed models predicting crown fire initiation and transmissivity as a function of fuel treatment (prescribed fire, mechanical, or untreated control), woodland phase, and year (years 0, 1, 2, 3, 6, and 10 post-treatment) from ten replicate sites across the Intermountain West, USA

	Crown initiation		**Crown transmissivity**	
Source	***F***	***P***	***F***	***P***
Intercept	1156.1	**< 0.001**	17.3	**0.002**
Year	90.4	**< 0.001**	4.3	**0.001**
Treatment	302.9	**< 0.001**	45.7	**< 0.001**
Phase	64.7	**< 0.001**	56.4	**< 0.001**
Year × treatment	12.4	**< 0.001**	4.8	**< 0.001**
Year × phase	1.7	0.08	3.0	**0.001**
Treatment × phase	3.9	**0.004**	32.6	**<0.001**
Year × treatment × phase	0.6	0.93	3.1	**<0.001**

mechanical treatment does not remove sagebrush; this increases the overall height of the surface fuels but also leaves a valuable seed source and releases water and nutrients for use by shrubs and herbaceous vegetation (Rau et al. 2008; Miller et al. 2014; Roundy et al. 2014).

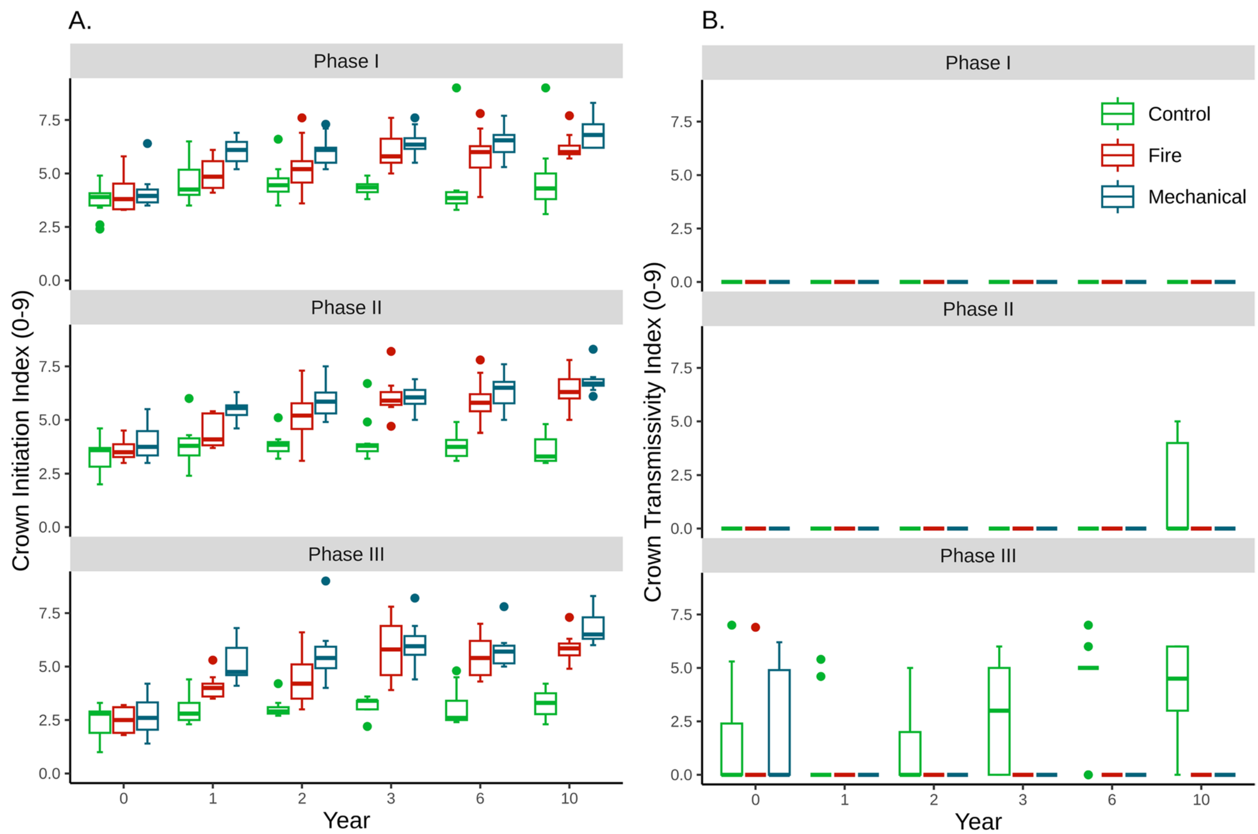

Fig. 10 Crown initiation (**A**) and crown transmissivity (**B**) by treatment type in years 0, 1, 2, 3, 6, and 10 following fire and mechanical treatments across three phases of woodland development (Miller et al. 2013). These indices are on a scale from 0 to 9, with 0 indicating no risk of tree ignition (initiation) or spread (transmissivity) and 9 indicating highest risk

The increase in shrub and downed woody fuels resulted in mechanical treatment plots having the highest modeled flame length and reaction intensity across years and woodland phases. In phase III plots, mechanical treatment resulted in the highest modeled rate of spread in all years except year 6. Fuel treatments in shrublands experiencing tree expansion seem to present a trade-off between restoration of critical shrubland ecosystems and an increase in risk of high intensity wildfire, at least in the short term. This trade-off underscores the need to either conduct cut and leave treatments only in the early phases of woodland expansion or to use cut and remove treatments to prevent large increases in downed woody fuels.

Fire behavior metrics increased with fuel curing. Fuel moisture scenarios were used to mimic the progression of curing throughout a fire season; as the season progressed, fuels contain less moisture. Fully cured fuels present the highest risk for fire ignition and rapid spread as drier fuels ignite with less energy, leading to more heat energy being contributed to the flaming front of a fire (Brown 1982) and resulting in more intense fire behavior (Wright 2013) compared to earlier in the fire season when fuels are fully green.

We hypothesized that treatment effectiveness at reducing modeled surface fire behavior metrics would decrease over time. Modeled flame length and rate of spread increased in year 1 and was sustained or further increased with time since treatment, indicating that mechanical and prescribed fire treatments were not effective at reducing modeled surface fire behavior. Increased modeled fire behavior over time was partly due to the increase of surface fuels, including increases in native shrubs, native perennial grasses, and annual grasses such as the invasive cheatgrass (Ellsworth and Kauffman 2017; Dittel et al. 2018; Freund et al. 2021). In woodlands with relatively low resistance to invasion of annual grasses, an increase of annual grass can result in a self-perpetuating invasive grass—fire cycle that can exacerbate surface fire behavior over time (Brooks et al. 2004, Pyke et al. 2016). Additionally, there was an increase in downed woody fuel across all years with a dramatic increase in year 10, that may have been caused by the accumulation of dead branches across study plots as dead trees breakdown. It is important to point out that increased surface fuels should be expected as a result of treatments; in many cases, increasing understory surface vegetation (particularly native grasses and shrubs) is the goal of treatments in conifer encroached woodlands and could contribute to maintaining a fire regime frequent enough to reduce risk of conifer encroachment (Bates et al. 2000, 2005; Miller et al. 2005). The increase in desired vegetation often comes with the risk of increased surface fire behavior, creating a trade-off between desired treatment results.

Wind speed has a significant impact on fire behavior, especially when compounded with low-moisture fuels (Figs. 8 and 9; Schroeder and Buck 1970). In mature pinyon-juniper stands, where there is separation between surface and canopy fuels and trees are the dominant vegetation on the landscape, higher wind speeds and low relative humidity are necessary for wildfire to spread into the tree canopy from the surface (Martin et al. 1977; Dicus et al. 2009). The impact of wind speed on wildfire behavior was documented in the 2020 Labor Day wildfires that ignited on September 7, 2020, and burned 393,315 ha throughout Oregon, Washington, and California and were driven in part by an unusual, but not unprecedented east wind event combined with dry fuels (Abatzoglou et al. 2021). The 2007 Tongue-Crutcher Wildland Fire Complex in Owyhee County, Idaho, burned 18,890 ha of phase III and mature woodlands in similarly dry and windy conditions (Strand et al. 2013). These events were an example of what can occur under extreme fire conditions, emphasizing the importance of including extreme wind speeds (i.e., 97th percentile wind speed) in fire behavior models to understand the full range of potential fire behavior that can result after implementing fuel or restoration treatments.

Contrary to our prediction, modeled crown initiation increased with both mechanical and prescribed fire treatments across all phases (Fig. 10A). The crown initiation metric represents the likelihood of an individual tree torching but does not account for tree density and size. Many of the conifers in these simulations are small seedlings that contribute to fire spread but do not necessarily indicate extreme crown fire behavior. Both prescribed fire and mechanical treatments decreased tree density overall, but mechanical treatments had increased tree density in year 10, as seedlings reached the size threshold to shift into the tree category (≤ 50 cm) (Fig. 4). Prescribed fire treatment decreased seedling density compared to control and mechanical plots, making it a strong choice of treatment for reducing risk of crown initiation in conifer encroached shrublands. The higher modeled crown initiation metrics across years and phases in the treatment plots is likely due to the increase in surface fuels, which provide the ladder fuels necessary for individual trees to torch or ignite. The low crown base height of seedlings made them particularly susceptible to torching (initiation) in the model (Prichard et al. 2013), which aligns with field observations; younger trees, particularly those shorter than 2 m, are typically more severely impacted by surface fire due to their thinner bark and proximity to surface fuels (Martin et al. 1977). As trees mature (> 50 years), they become more likely to survive surface fire of low and moderate severity (Burkhardt and Tisdale 1976), due to the lack of surrounding understory fuels to carry a surface fire (Miller and Tausch 2001, Martin et al. 1977).

Modeled crown transmissivity results better captured the low tree density in treated plots, as canopy fire spread was only predicted in control plots that were in phase II and III at study initiation. However, in the 10 years since treatment, infilling and growth of young trees has occurred, increasing crown fire spread risk through time as those plots continue to move through woodland development (Miller et al. 2005). The crown transmissivity metrics are highly variable, which is a direct result of variability of seedling and tree density at the plot level; the majority of plots, specifically those in phase I and II, did not result in modeled crown transmissivity due to low tree or seedling density.

Crown fire metrics should be interpreted carefully, as fire behavior models are somewhat limited in the calculation of these metrics (Alexander and Cruz 2013). The FFT crown initiation metric is driven by the vertical gap between canopy and surface fuels (ladder fuels), surface reaction intensity, and flame residence time (Prichard et al. 2013 for further details). In the model, wind speed and fuel moisture indices do not impact this metric, which does not necessarily reflect real world conditions. In areas dominated by mature trees, high wind speeds and low relative humidity (modeled here as 97th percentile winds and fully cured moisture scenario) were necessary for fire to spread through the tree canopy (Martin et al. 1977, Miller et al. 2013). To model crown fire transmissivity, FFT utilizes canopy cover and assumes a wind speed of 4 mph (~ 6.4 kph, much lower than the wind speeds representative of the study sites) (Prichard et al. 2013). Because of the way these indices are calculated in the model, wind-driven canopy fires (including crown initiation or transmissivity as a result of embers carried via wind) are not captured, despite occasionally being observed in the field. Additionally, the model does not predict spotting behavior, which is a key factor in real life crown initiation; increased windspeeds lead to an increased spotting distance, resulting in a greater likelihood that the lofted embers will ignite a new fire downwind (NWCG 2014). As a result, the crown initiation and crown fire transmissivity metrics predicted by the model are likely underestimates when compared to the conditions that might occur in a wildfire.

Custom fuel models using FFT have been used in Wyoming big sage and basin big sage ecosystems, with good agreement to results generated in previous, validated fire behavior modeling (Ellsworth et al. 2020, 2022). While there are existing FCCS models for woodland systems, suggesting that others have also used these tools, we acknowledge that this may be the first publication using FFT in sagebrush ecosystems experiencing pinyon and juniper expansion. The SageSTEP network data provide an exceptional opportunity to examine the long-term effect of commonly used fuel treatments on both fuel and fire behavior. While the modeled output may or may not exactly replicate real-world scenarios, it provides a robust data set to compare among management options. The FFT custom fuel models used in this study are mathematical models that reflect the tools most widely accessible to and utilized by land managers. Unlike newer physics-based models that can predict fire behavior based on total heat flux metrics, FFT models have only indirect relationships with heat flux and vegetation mortality relationships. While physics-based models are an exciting tool that will improve our ability to predict fire behavior, they are not yet widely available.

To lessen negative impacts, fuel treatments should be implemented in the early phases of expansion to ensure that sufficient understory species remain to promote recovery and to minimize the amount of downed surface woody fuels resulting from cut and leave treatments. Additionally, selecting sites for treatment using resilience and resistance concepts can help prevent or minimize invasion of annual grasses and ensure that the site will recover, even if it subsequently burns in a wildfire (Chambers et al. 2014a, b; Roundy et al. 2018; Miller et al. 2019; Freund et al. 2021). In higher elevation sites within sagebrush shrublands, we are seeing increases of pinyon and juniper and reduced wildfire frequency compared to historical wildfire regimes (Miller and Rose 1999). Although still below historical frequency, wildfire frequency has begun to increase at higher elevations in recent decades (Alizadeh et al. 2021) at sites with high resilience and resistance, and this more frequent low intensity wildfire activity may be desirable for preventing pinyon-juniper woodland encroachment and maintaining sagebrush shrublands.

It is important to note that the increase of surface herbaceous fuels as a result of treatments is similar to what has occurred after historical wildfires. Currently, this increase is a potential concern due to the risk of invasive annual grasses and the recent increase in frequency and size of wildfires. However, without treatment the expansion of woodlands into sagebrush shrublands could continue and result in a worst-case scenario: high intensity crown fires with little or no residual understory to promote recovery. Despite the risks of increased invasive annual grasses and surface fire behavior, these treatments have been successful in reducing pinyon and juniper cover (Miller et al. 2014; Boyd et al. 2017; Bernau et al. 2018), creating or restoring sagebrush habitat for sage-grouse (Boyd et al. 2017; Severson et al. 2017), increasing forage for wildlife and livestock (Knick et al. 2011; Bates et al. 2019), and decreasing modeled crown fire behavior (Fig. 10). With these trade-offs in mind, desired management outcomes must be considered when evaluating fuel management treatment choices.

Conclusions

This paper analyzes long-term (10-year) data on fuel treatments in sagebrush ecosystems experiencing juniper and pinyon expansion that were conducted to (1) decrease canopy or overall woody fuels to lower the intensity of future fires and (2) increase the amount of herbaceous and/or shrub vegetation and thus maintain or increase ecological resilience and resistance to invasion (Reinhardt et al. 2008; McIver et al. 2010, McIver and Brunson 2014). Desirable increases in shrub and herbaceous vegetation was observed following both prescribed fire and mechanical (cut and drop) treatments. However, modeled surface and canopy fire behavior indicated that both treatments also resulted in increased surface fuel and thus elevated fire behavior metrics, including flame length, rate of spread, and fire intensity. These findings demonstrate a significant management tradeoff between short-term increases in surface fire behavior for restoration of shrubland plant communities and long-term reductions in the potential for canopy spread.

Abbreviations

CT	Crown transmissivity
CI	Crown initiation
DBH	Diameter at breast height
ROS	Rate of spread
RI	Reaction intensity
FL	Flame length

Acknowledgements

We thank the SageSTEP team and their management partners for development, implementation, and maintenance of the experimental network, as well as the many field technicians who collected and processed the data. This is contribution number 152 of the Sagebrush Steppe Treatment Evaluation Project (SageSTEP), funded by the US Joint Fire Science Program, the National Interagency Fire Center, and the Bureau of Land Management. Any use of trade, firm, or product names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

Authors' contributions

LME, BAN, JCC, EKS, and SS contributed to the study design and implementation. CLW did the fire behavior modeling. CLW and LME lead the writing of the manuscript. LME, BAN, EKS, JCC, KCS, CT, SS, and MR contributed to writing the manuscript. All authors read and approved the final manuscript.

Funding
Funding for implementation and data collection for this project was provided by the National Interagency Fire Center, US Bureau of Land Management, and the US Joint Fire Science Program. Funding for modeling and analysis was provided by the US Joint Fire Science Program Project 19-2-02-11.

Availability of data and materials
The datasets used and analyzed are available from the corresponding author by request.

Declarations

Ethics approval and consent to participate
Not applicable.

Consent for publication
Not applicable.

Competing interests
The authors declare that they have no competing interests.

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Received: 10 February 2023 Accepted: 14 June 2023
Published online: 07 August 2023

References

Abatzoglou, J. T., D. E. Rupp, and L. W. O'Neill, and M Sadegh. 2021. Compound extremes drive the western Oregon wildfires of September 2020. Geophysical Research Letters 48(8) https://doi.org/10.1029/2021GL092520.

Alexander, M. E., and M. G. Cruz. 2013. Limitations on the accuracy of model predictions of wildland fire behaviour: A state-of-the-knowledge overview. *The Forestry Chronicle* 89 (3): 372–383. https://doi.org/10.5558/tfc2013-067.

Alizadeh, M. R., J. T. Abatzoglou, C. H. Luce, J. F. Adamowski, A. Farid, and M. Sadegh. 2021. Warming enabled upslope advance in western US forest fires. Proceedings of the National Academy of Sciences 118(22): e2009717118. https://doi.org/10.1073/pnas.2009717118.

Bates, J. D., R. F. Miller, and T. J. Svejcar. 2000. Understory dynamics in cut and uncut western juniper woodlands. *Rangeland Ecology & Management/Journal of Range Management Archives* 53 (1): 119–126.

Bates, J. D., R. F. Miller, and T. J. Svejcar. 2005. Long-term successional trends following western juniper cutting. *Rangeland Ecology & Management* 58 (5): 533–541. https://doi.org/10.2111/1551-5028(2005)58[533:LSTFWJ]2.0.CO;2.

Bates, J. D., K. W. Davies, J. Bournoville, C. Boyd, R. O'Connor, TJ and Svejcar. 2019. Herbaceous biomass response to prescribed fire in juniper-encroached sagebrush steppe. *Rangeland Ecology & Management* 72 (1): 28–35. https://doi.org/10.1016/j.rama.2018.08.003.

Baughman, C., T. A. Forbis, and L. Provencher. 2010. Response of two sagebrush sites to low-disturbance, mechanical removal of piñyon and juniper. *Invasive Plant Science and Management* 3 (2): 122–129. https://doi.org/10.1614/IPSM-D-09-00020.

Bernau, C. R., E. K. Strand, and S. C. Bunting. 2018. Fuel bed response to vegetation treatments in juniper-invaded sagebrush steppe. *Fire Ecology* 14 (2): 1. https://doi.org/10.1186/s42408-018-0002-z.

Bourne, A., and S. C. Bunting. 2011. *Guide for quantifying post-treatment fuels in the sagebrush steppe and juniper woodlands of the Great Basin. Technical Note 437*. Denver: Bureau of Land Management. https://digitalcommons.usu.edu/sagestep_reports/13/.

Boyd, C. S., J. D. Kerby, T. J. Svejcar, J. D. Bates, D. D. Johnson, and K. W. Davies. 2017. The sage-grouse habitat mortgage: Effective conifer management in space and time. *Rangeland Ecology & Management* 70 (1): 141–148. https://doi.org/10.1016/j.rama.2016.08.012.

Bradley, B., C. Curtis, E. Fusco, J. Abatzoglou, J. Balch, S. Dadashi, and M. N. Tuanmu. 2018. Cheatgrass (*Bromus tectorum*) distribution in the intermountain Western United States and its relationship to fire frequency, seasonality, and ignitions. *Biological Invasions* 20 (6): 1493–1506. https://doi.org/10.1007/s10530-017-1641-8.

Braun, C. E., T. Britt, and R. O. Wallestad. 1977. Guidelines for maintenance of sage grouse habitats. *Wildlife Society Bulletin* 5 (3): 99–106.

Brooks, M.L., C.M. D'Antonio, D.M. Richardson, J.B. Grace, J.E. Keeley, J.M. DiTomaso, R.J. Hobbs, M. Pellant, and D. Pyke. 2004. Effects of invasive alien plants on fire regimes. *BioScience* 54 (7): 677–88. https://doi.org/10.1641/0006-3568(2004)054[0677:EOIAPO]2.0.CO;2.

Brown, J. K., R. D. Oberheu, and C. M. Johnston. 1982. Handbook for inventorying surface fuels and biomass in the interior West. General Technical Report INT-129. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experimental Station. https://doi.org/10.2737/INT-GTR-129.

Brown, J. K., E. D. Reinhardt, and K. A. Kramer. 2003. Coarse woody debris: Managing benefits and fire hazard in the recovering forest. General Technical Report RMRS-GTR-105. Ogden, UT: US Department of Agriculture, Forest Service, Rocky Mountain Research Station. https://doi.org/10.2737/RMRS-GTR-105.

Burkhardt, J. W., and E. W. Tisdale. 1976. Causes of juniper invasion in Southwestern Idaho. *Ecology* 57 (3): 472–484. https://doi.org/10.2307/1936432.

Chambers, J. C., R. F. Miller, D. I. Board, D. A. Pyke, B. A. Roundy, J. B. Grace, E. W. Schupp, and R. J. Tausch. 2014a. Resilience and resistance of sagebrush ecosystems: implications for state and transition models and management treatments. *Rangeland Ecology & Management* 67 (5): 440–454. https://doi.org/10.2111/REM-D-13-00074.1.

Chambers, J. C., B. A. Bradley, C. S. Brown, C. D'Antonio, M. J. Germino, J. B. Grace, S. P. Hardegree, and R. F. Miller, and DA Pyke. 2014b. Resilience to stress and disturbance, and resistance to *Bromus tectorum* L. invasion in cold desert shrublands of western North America. *Ecosystems* 17 (2): 360–375. https://doi.org/10.1007/s10021-013-9725-5.

Chambers, J.C., J.D. Maestas, D.A. Pyke, C.S. Boyd, M. Pellant, and A. Wuenschel. 2017. Using resilience and resistance concepts to manage persistent threats to sagebrush ecosystems and greater sage-grouse. *Rangeland Ecology & Management* 70 (2): 149–64. https://doi.org/10.1016/j.rama.2016.08.005.

Chambers, J. C., A. K. Urza, D. I. Board, R. F. Miller, D. A. Pyke, B. A. Roundy, E. W. Schupp, and R. J. Tausch. 2021. Sagebrush recovery patterns after fuel treatments mediated by disturbance type and plant functional group interactions. *Ecosphere* 12 (4): 435. https://doi.org/10.1002/ecs2.3450.

Davies, K. W., C. S. Boyd, J. L. Beck, J. D. Bates, T. J. Svejcar, and M. A. Gregg. 2011. Saving the sagebrush sea: An ecosystem conservation plan for big sagebrush plant communities. *Biological Conservation* 144 (11): 2573–2584. https://doi.org/10.1016/j.biocon.2011.07.016.

Davies, K. W., R. C. Rios, J. D. Bates, D. D. Johnson, J. Kerby, and C. S. Boyd. 2019. To burn or not to burn: comparing reintroducing fire with cutting an encroaching conifer for conservation of an imperiled shrub-steppe. *Ecology and Evolution* 9 (16): 9137–9148. https://doi.org/10.1002/ece3.5461.

Dicus, C. A., K. Delfino, and D. R. Weise. 2009. Predicted fire behavior and societal benefits in three eastern Sierra Nevada vegetation types. *Fire Ecology* 5 (1): 67–78. https://doi.org/10.4996/fireecology.0501067.

Dittel, J. W., D. Sanchez, L. M. Ellsworth, C. N. Morozumi, and R. Mata-Gonzalez. 2018. Vegetation response to juniper reduction and grazing exclusion in sagebrush-steppe habitat in Eastern Oregon. *Rangeland Ecology & Management* 71 (2): 213–219. https://doi.org/10.1016/j.rama.2017.11.004.

Ellsworth, L. M., and J. B. Kauffman. 2017. Plant community response to prescribed fire varies by pre-fire condition and season of burn in mountain big sagebrush ecosystems. *Journal of Arid Environments* 144: 74–80. https://doi.org/10.1016/j.jaridenv.2017.04.012.

Ellsworth, L. M., J. B. Kauffman, S. A. Reis, D. Sapsis, and K. Moseley. 2020. Repeated fire altered succession and increased fire behavior in basin big

sagebrush–native perennial grasslands. *Ecosphere* 11 (5): e03124. https://doi.org/10.1002/ecs2.3124.
Ellsworth, L. M., B. A. Newingham, S. E. Shaff, C. L. Williams, E. K. Strand, M. Reeves, D. A. Pyke, E. W. Schupp, and J. C. Chambers. 2022. Fuel reduction treatments reduce modeled fire intensity in the sagebrush steppe. *Ecosphere* 13 (5): e4064. https://doi.org/10.1002/ecs2.4064.
Freund, S. M., B. A. Newingham, J. C. Chambers, A. K. Urza, B. A. Roundy, and J. H. Cushman. 2021. Plant functional groups and species contribute to ecological resilience a decade after woodland expansion treatments. *Ecosphere* 12 (1): e03325. https://doi.org/10.1002/ecs2.3325.
IBM Corp. 2020. *IBM SPSS Statistics for Windows, Version 27.0. Armonk*. NY: IBM Corp.
Integrated Rangeland Fire Management Strategy Actionable Science Plan Team. 2016. *The integrated rangeland fire management strategy actionable science plan*, 128. Washington D.C.: U.S. Department of the Interior. https://www.fs.usda.gov/research/treesearch/53265.
Johnston, J. D., J. H. Olszewski, B. A. Miller, M. R. Schmidt, M. J. Vernon, and L. M. Ellsworth. 2021. Mechanical thinning without prescribed fire moderates wildfire behavior in an Eastern Oregon, USA ponderosa pine forest. *Forest Ecology and Management* 501: e119674. https://doi.org/10.1016/j.foreco.2021.119674.
Kleinhesselink, A. R., E. J. Kachergis, S. E. McCord, J. Shirley, N. R. Hupp, J. Walker, J. C. Carlson, S. L. Morford, M. O. Jones, J. T. Smith, B. W. Allred, and D. E. Naugle. 2023. Long-term Trends in Vegetation on Bureau of Land Management Rangelands in the Western United States. *Rangeland Ecology & Management* 87: 1–12. https://doi.org/10.1016/j.rama.2022.11.004.
Knick, S. T., S. E. Hanser, R. F. Miller, D. A. Pyke, M. J. Wisdom, S. P. Finn, E. T. Rinkes, and C. J. Henny. 2011. Ecological influence and pathways of land use in sagebrush. In *Greater sage-grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology*, eds. S. T. Knick, and J. W. Connelly, vol. 38, 203–251. Berkeley, CA: University of California Press.
Knick, S.T., S.E. Hanser, and K.L. Preston. 2013. Modeling ecological minimum requirements for distribution of greater sage-grouse leks: Implications for population connectivity across their western range. *USA Ecology and Evolution* 3 (6): 1539–1551. https://doi.org/10.1002/ece3.557.
Maestas, J. D., M. Porter, M. Cahill, and D. Twidwell. 2022. Defend the core: Maintaining intact rangelands by reducing vulnerability to invasive annual grasses. *Rangelands* 44 (3): 181–186. https://doi.org/10.1016/j.rala.2021.12.008.
Mahood, A. L., and J. K. Balch. 2019. Repeated fires reduce plant diversity in low-elevation Wyoming big sagebrush ecosystems (1984–2014). *Ecosphere* 10 (2): e02591. https://doi.org/10.1002/ecs2.2591.
Martin, R.E., J.E. Dealy, and D.L. Caraher. 1977. *Proceedings of the western juniper ecology and management workshop. General Technical Report PNW-GTR-074*, 1–177. Portland: US Department of Agriculture, Forest Service, Pacific Northwest Research Station. https://www.fs.usda.gov/research/treesearch/25144.
McIver, J., and M. Brunson. 2014. Multidisciplinary, multisite evaluation of alternative sagebrush steppe restoration treatments: the SageSTEP Project. *Rangeland Ecology & Management* 67 (5): 434–438. https://doi.org/10.2111/REM-D-14-00085.1.
McIver, J. D., M. Brunson, S. C. Bunting, J. Chambers, N. Devoe, P. Doescher, J. Grace, D. Johnson, S. Knick, R. Miller, M. Pellant, F. Pierson, D. Pyke, K. Rollins, B. Roundy, E. Schupp, R. Tausch, and D. Turner. 2010. *The Sagebrush Steppe treatment evaluation project (SageSTEP): A test of state-and-transition theory*. Ft. Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.
Miller, R. F., and J. A. Rose. 1999. Fire history and western juniper encroachment in sagebrush steppe. *Rangeland Ecology & Management/Journal of Range Management Archives* 52 (6): 550–559.
Miller, R. F., and R. J. Tausch. 2001. The role of fire in pinyon and juniper woodlands: a descriptive analysis. In *Proceedings of the invasive species workshop: the role of fire in the control and spread of invasive species*, eds. K. E. M. Galley, and T. P. Wilson, vol. 11, 15–30. Tallahassee: Tall Timbers Research Station Miscellaneous Publication.
Miller, R. F., J. D. Bates, T. J. Svejcar, F. B. Pierson, and L. E. Eddleman. 2005. Biology, ecology, and management of western juniper (Juniperus occidentalis). Agricultural Experiment Station Technical Bulletin 152, Oregon State University, Corvallis.
Miller, R. F., J. C. Chambers, D. A. Pyke, F. B. Pierson, and C. J. Williams. 2013. *A review of fire effects on vegetation and soils in the Great Basin region: response and ecological site characteristics. RMRS-GTR-308. USDA General Technical Report*. Fort Collins: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. http://pubs.er.usgs.gov/publication/70057895.
Miller, R. F., J. Ratchford, B. A. Roundy, R. J. Tausch, and A. Hulet, and J Chambers. 2014. Response of conifer encroached shrublands in the Great Basin to prescribed fire and mechanical treatments. *Rangeland Ecology and Management* 67 (5): 468–481. https://doi.org/10.2111/REM-D-13-00003.1.
Miller, R. F., J. C. Chambers, L. Evers, C. J. Williams, K. A. Snyder, B. A. Roundy, and F. B. Pierson. 2019. The ecology, history, ecohydrology, and management of pinyon and juniper woodlands in the Great Basin and Northern Colorado Plateau of the western United States. General Technical Report RMRS-GTR-403. Fort Collins, CO: US Department of Agriculture, Forest Service, Rocky Mountain Research Station. https://doi.org/10.2737/RMRS-GTR-403.
Morford, S. L., B. W. Allred, D. Twidwell, M. O. Jones, J. D. Maestas, C. P. Roberts, and D. E. Naugle. 2022. Herbaceous production lost to tree encroachment in United States rangelands. *Journal of Applied Ecology* 59 (12): 2971–2982. https://doi.org/10.1111/1365-2664.14288.
National Wildfire Coordinating Group (NWCG). 2014. *Fire Behavior Field Reference Guide PMS* 437: 192 pages.
Omernik, J. M., and G. E. Griffith. 2014. Ecoregions of the conterminous United States: Evolution of a hierarchical spatial framework. *Environmental Management* 54 (6): 1249–1266. https://doi.org/10.1007/s00267-014-0364-1.
Paige, C., and S. A. Ritter. 1999. *Birds in a sagebrush sea: Managing sagebrush habitats for bird communities. Partners in Flight*. Boise, ID: Western Working Group.
Petersen, S. L., and TK Stringham. 2008. Infiltration, runoff, and sediment yield in response to western juniper encroachment in Southeast Oregon. *Rangeland Ecology & Management* 61 (1): 74–81. https://doi.org/10.2111/07-070R.1.
Pierson, F. B., C. J. Williams, P. R. Kormos, S. P. Hardegree, P. E. Clark, and B. M. Rau. 2010. Hydrologic vulnerability of sagebrush steppe following pinyon and juniper encroachment. *Rangeland Ecology & Management* 63 (6): 614–629. https://doi.org/10.2111/REM-D-09-00148.1.
Prichard, S. J., D. V. Sandberg, R. D. Ottmar, E. Eberhardt, A. Andreu, P. Eagle, and K. Swedin. 2013. Fuel Characteristic Classification System version 3.0: technical documentation. General Technical Report PNW-GTR-887. Portland, OR: US Department of Agriculture, Forest Service, Pacific Northwest Research Station. https://doi.org/10.2737/PNW-GTR-887.
Pyke, D. A., J. C. Chambers, J. L. Beck, M. L. Brooks, and B. A. Mealor. 2016. Land uses, fire, and invasion: exotic annual Bromus and human dimensions. In *Exotic brome-grasses in arid and semiarid ecosystems of the western US: causes, consequences, and management implications*, eds. M. J. Germino, J. C. Chambers, and C. S. Brown, 307–337. Springer International Publishing. https://doi.org/10.1007/978-3-319-24930-8_11.
Pyke, D. A., S. E. Shaff, J. C. Chambers, E. W. Schupp, B. A. Newingham, M. L. Gray, and L. M. Ellsworth. 2022. Ten-year ecological responses to fuel treatments within semiarid Wyoming big sagebrush ecosystems. *Ecosphere* 13 (7): e4176. https://doi.org/10.1002/ecs2.4176.
Rau, B. M., J. C. Chambers, R. R. Blank, and D. W. Johnson. 2008. Prescribed fire, soil, and plants: Burn effects and interactions in the central Great Basin. *Rangeland Ecology & Management* 61 (2): 169–181.
Reinhardt, E. D., R. E. Keane, D. E. Calkin, and J. D. Cohen. 2008. Objectives and considerations for wildland fuel treatment in forested ecosystems of the interior western United States. *Forest Ecology and Management* 256 (12): 1997–2006. https://doi.org/10.1016/j.foreco.2008.09.016.
Roundy, B. A., F. R. Miller, R. J. Tausch, K. Young, A. Hulet, B. Rau, B. Jessop, J. C. Chambers, and D. Eggett. 2014. Understory cover responses to piñon–juniper treatments across tree dominance gradients in the Great Basin. *Rangeland Ecology & Management* 67 (5): 482–494. https://doi.org/10.2111/REM-D-13-00018.1.
Roundy, B. A., J. C. Chambers, D. A. Pyke, R. F. Miller, R. J. Tausch, E. W. Schupp, B. Rau, and T. Gruell. 2018. Resilience and resistance in sagebrush ecosystems are associated with seasonal soil temperature and water availability. *Ecosphere* 9 (9): e02417. https://doi.org/10.1002/ecs2.2417.
Roundy, B. A., R. F. Miller, R. J. Tausch, J. C. Chambers, and B. M. Rau. 2020. Long-term effects of tree expansion and reduction on soil climate in a semiarid ecosystem. *Ecosphere* 11 (9): e03241. https://doi.org/10.1002/ecs2.3241.

Sabin, B. S. 2008. *Relationship between allometric variables and biomass in Western Juniper (Juniperus occidentalis). MS Thesis*. Corvallis: Oregon State University. https://ir.library.oregonstate.edu/concern/graduate_thesis_or_dissertations/x346d7083.

Sandberg, D. V., C. L. Riccardi, and M. D. Schaaf. 2007. Fire potential rating for wildland fuelbeds using the fuel characteristic classification system. *Canadian Journal of Forest Research* 37 (12): 2456–2463. https://doi.org/10.1139/X07-093.

Schroeder, M., and C. Buck. 1970. *Fire weather: a guide for application of meteorological information to forest fire control operations*, vol. 360, 1–234. USDA Forest Service, Agriculture Handbook. https://handle.nal.usda.gov/10113/CAT87208488.

Severson, J. P., C. A. Hagen, J. D. Maestas, D. E. Naugle, J. T. Forbes, and K. P. Reese. 2017. Effects of conifer expansion on greater sage-grouse nesting habitat selection. *The Journal of Wildlife Management* 81 (1): 86–95. https://doi.org/10.1002/jwmg.21183.

Stebleton, A., and S. Bunting. 2009. *Guide for quantifying fuels in the sagebrush steppe and juniper woodlands of the Great Basin. Technical Note 430*. Denver: Bureau of Land Management. https://digitalcommons.usu.edu/sagestep_reports/7/.

Strand, E. K., S. C. Bunting, and R. F. Keefe. 2013. Influence of wildland fire along a successional gradient in sagebrush steppe and western juniper woodlands. *Rangeland Ecology & Management* 66 (6): 667–679. https://doi.org/10.2111/REM-D-13-00051.1.

Tausch, R. J. 2009. A structurally based analytic model for estimation of biomass and fuel loads of woodland trees. *Natural Resource Modeling* 22 (4): 463–488. https://doi.org/10.1111/j.1939-7445.2009.00045.

Wozniak, S., and E. Strand. 2019. *Fuels guide for sagebrush and pinyon-juniper treatments: 10 years post-treatment. Technical Note 451*. Boise: Bureau of Land Management. https://digitalcommons.usu.edu/sagestep_reports/19/.

Wozniak, S.S., E.K. Strand, T.R. Johnson, A. Hulet, B.A. Roundy, and K. Young. 2020. Treatment longevity and changes in surface fuel loads after pinyon–juniper mastication. *Ecosphere* 11 (8): e03226. https://doi.org/10.1002/ecs2.3226.

Wright, C. S. 2013. Models for predicting fuel consumption in sagebrush-dominated ecosystems. *Rangeland Ecology & Management* 66 (3): 254–266. https://doi.org/10.2111/REM-D-12-00027.1.

Wright, C. S., and S. J. Prichard. 2006. Biomass consumption during prescribed fires in big sagebrush ecosystems. In *Fuels management–how to measure success: Conference proceedings*, eds. P. L. Andrews, B. W. Butler, comps., 28–30.

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