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Factors Affecting Efficacy of Prescribed Fire for Western Juniper Control ☆

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ABSTRACT

Western juniper (*Juniperus occidentalis* Hook.) is a tree species occurring on 3.6 million ha in the northern Great Basin. This native species can be quite invasive, encroaching into sagebrush-grassland vegetation, forming woodlands, and dominating extensive landscapes. Control of encroaching juniper is often necessary and important. Efficacy of prescribed fire for western juniper control depends on many factors for which our understanding is still quite incomplete. This knowledge gap makes fire management planning for western juniper control more difficult and imprecise. Natural resource managers require a fire efficacy model that accurately predicts juniper mortality rates and is based entirely on predictors that are measurable prefire. We evaluated efficacy models using data from a fall prescribed fire conducted during 2002 in southwestern Idaho on mountain big sagebrush (*Artemisia tridentata* Nutt. ssp. *vaseyana* [Rydb.] Beetle) rangelands with early to midsuccessional juniper encroachment. A logistic regression model, which included vegetation cover type, tree height, fire type, and bare ground as predictors, accurately predicted (area under the receiver operating characteristic [ROC] curve [AUC] $= 0.881 \pm 0.128$ standard deviation [SD]) the mortality rate for a random sample of western juniper trees marked and assessed prefire and 5 yr post fire. Trees occurring in an antelope bitterbrush (*Purshia tridentata* [Pursh] DC.) type, which had a heavy fuel load, were 8 times more likely to be killed by fire than trees in a mountain big sagebrush type, where loading was typically lighter. Probability of mortality decreased by 28.8% for each 1-meter increase in tree height. Trees exposed to head fire were 3 times as likely to be killed as those exposed to backing fire. Findings from this case study suggest that with just four factors which are readily quantifiable prefire, managers can accurately predict juniper mortality rate and thus make better informed decisions when planning prescribed fire treatments.

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Introduction

Western juniper (*Juniperus occidentalis* Hook.) is a tree species that occurs on about 3.6 million ha in the northern Great Basin (Miller et al., 2005). Although it is a native species, western juniper can be quite invasive, encroaching into sagebrush-grassland vegetation, eventually forming woodlands, and functionally dominating extensive landscapes. However, western juniper, a nonsprouting species, can be killed by fire. Local distribution of western juniper can thus be strongly influenced by the periodic occurrence of wildfire. Before European settlement (i.e., before 1870), western juniper in the northern Great Basin was primarily confined to rocky ridgetops, shallow soils, and other areas with sparse fuels (Cottam and Stewart, 1940; Burkhardt and Tisdale, 1976; West, 1984; Miller et al., 1999). Juniper trees encroaching on shrub- and grass-dominated rangelands, where soils were deeper and fuels more abundant, were probably killed by periodic wildfires. However, during the nearly 150 yr since European settlement, western juniper populations have increased exponentially (Miller and Wigand, 1994; Miller et al., 2005). Climatic changes may account for some of this increase (Knapp and Soule, 1996), but anthropogenic reductions of wildfire occurrence and/or extent, through heavy livestock grazing and active fire suppression, are believed to be the principal cause of this dramatic growth in western juniper populations (Burkhardt and Tisdale, 1976; Miller and Rose, 1995, 1999).

Juniper encroachment and transition of shrub-grasslands into western juniper woodlands reduce shrubs and, in some cases, herbaceous plant cover (Burkhardt and Tisdale, 1969; Bunting et al., 1999; Miller et al., 2000), increase risk of soil nutrient loss (Miller et al., 2005), reduce effective precipitation through canopy interception (Young et al., 1984; Larsen, 1993), decrease site capacity to capture and store water (Bates et al., 2000; Roundy et al., 2014b; Kormos et al., 2017), increase the potential for runoff and erosion (Pierson et al., 2013; Williams et al., 2014), decrease the quality and/or diversity of habitat for some wildlife species

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(Willis and Miller, 1999; Noson, 2000; Schaefer et al., 2003; Baruch-Mordo et al., 2013; Coates et al., 2017); and adversely impact forage quantity and quality, livestock grazing opportunities, and ranch-level economics (Young et al., 1982, 1985; Bates et al., 2000; Aldrich et al., 2005; McClain, 2013). Juniper encroachment and woodland development on shrub-grasslands have been classified into three phases (Miller et al., 2005). Juniper trees are sparsely scattered, small in size, and subdominate to shrubs in phase I; more common and codominate with shrubs in phase II and dominate the site in phase III or woodland phase. Juniper control in phases I and II can prevent advancement of a site into phase III and avoid the adverse consequences of woodland development (Bates et al., 2014). Restoring phase III juniper sites back to a preinvasion shrub-grassland state is difficult, costly, and, often, not entirely possible (Young et al., 1982; Miller et al., 2005; Bates et al., 2017). Control of encroaching western juniper is, therefore, often necessary and important.

Approaches to juniper control generally include mechanical (e.g., chaining, cutting/felling, and mastication), chemical (e.g., tebuthiuron and picloram herbicides), and prescribed fire (Miller et al., 2005; Bates and Svejcar, 2009; Bates and Davies, 2016; Bates et al., 2017). All have their merits and drawbacks (Miller et al., 2013; Roundy et al., 2014a; Bates and Davies, 2016), but prescribed fire is often the most efficient and cost-effective approach, particularly for phase I encroachments into mountain big sagebrush associations when trees are smaller, less densely clustered, and ample shrubs are present as ladder fuels (Miller et al., 2000, 2014). Efficacy of prescribed fire for western juniper control, however, depends on many factors. Yet our understanding of the influence and ranked importance of these factors is still incomplete. This knowledge gap makes fire management planning for western juniper control more difficult and imprecise.

Factors affecting the efficacy of prescribed fire have been evaluated for other juniper and rangeland tree species. Tree size, wind speed, relative humidity, and total fuel load affect fire-induced mortality rates of honey mesquite (*Prosopis glandulosa* Torr. var. *grandulosa*) (Britton and Wright, 1971). Tree size is a factor in alligator juniper (*Juniper deppeana* Steud.) killed by fire (Johnson et al., 1962). Mortality rate of redberry juniper (*Juniperus pinchotii* Sudw.) decreases with increased tree height and burial of the bud zone (Steuter and Britton, 1983). Tree height also affects fire-induced mortality rate of Ashe juniper (*Juniperus ashei* J. Buchholz) (Noel and Fowler, 2007). Leaf moisture content and seasonal climatic conditions influence the flammability of redberry juniper (Bunting et al., 1983). Fuel load under and adjacent to the crown contribute to crown scorch severity in redberry and Ashe juniper (Twidwell et al., 2009). Factors affecting prescribed fire-mortality rates in western juniper, however, have received less research attention.

Fire efficacy and conifer mortality models commonly include tree-injury variables (e.g., crown volume scorched, bore char), which are measured following the fire (Ryan and Reinhardt, 1988; McHugh and Kolb, 2003; Thies et al., 2006; Hood et al., 2007). For planning prescribed fires, models dependent on tree-injury predictor variables would have much less utility than a model based on predictors that can be readily assessed in the field and/or at the geographic information system (GIS) workstation before burning operations. Natural resource managers planning prescribed fires for control of western juniper need a fire-efficacy model that accurately predicts juniper mortality rates and is based entirely on predictors that are measurable prefire. Consequently, the goal of this research was to develop and evaluate a preliminary version of this kind of fire-efficacy model based on data acquired from a prescribed fire conducted on sagebrush rangeland in phase I of western juniper encroachment. Specific objectives of this research included 1) determine the principal factors affecting the efficacy of fall prescribed fire for killing western juniper trees on sagebrush steppe landscapes and 2) evaluate the accuracy and predictive performance of simple, fire-efficacy models based only on predictors readily assessable in the field or with GIS before fire application.

Materials and Methods

Study Area

The study was conducted at the Breaks prescribed-fire study area (81.1 ha), which comprises private lands located within the Reynolds Creek Experimental Watershed (43°6′29″N, 116°46′37″W) and about 80 km south of Boise in southwestern Idaho (Fig. 1). This research centers around a prescribed fire conducted in the study area during fall 2002. Climate at the study area is continental with maritime influences. Winters are cold and wet, while summers are warm and dry. Long-term (1966 − 1975, 2002 − 2016) mean water-yr precipitation at the Breaks gauges (site ID 145) was 571 mm (NWRC, 2017), about one-third of which falls as snow (Hanson, 2001). Annual precipitation during the 2002 water yr was 525 mm and thus quite close to average. The growing season is about 100 d, but frost can occur during any month of the year. Long-term (2001 − 2016) mean daily maximum, minimum, and mean air temperatures at Breaks were 8.3°C, 3.8°C, and 7.8°C, respectively (NWRC, 2017). The daily mean air temperature during the 2002 study yr was 7.9°C.

Topography of the study area is an east-facing hillslope ranging from 1 542 to 1 763 m in elevation. Slope ranges from flat to steep (78% or 38° maximum). Aspects in all four cardinal directions are well represented. Four hillslope positions are present in the study area: summit, shoulder, backslope, and footslope. The toeslope on this landscape was below and outside the bounds of the study area. Soils are primarily derived from granitic parent materials and composed of a complex of Takeuchi (coarse, loamy, mixed, frigid Typic Haploxerolls) and Kanlee (fine, loamy, mixed, frigid Typic Argixerolls) soil series (Seyfried et al., 2001).

Three vegetation cover types dominate this landscape: 1) mountain big sagebrush–mountain snowberry (*Artemisia tridentata* Nutt. ssp. *vaseyana* [Rydb.] Beetle–*Symphoricarpus oreophilus* A. Gray), 2) antelope bitterbrush (*Purshia tridentata* [Pursh] DC.)–mountain big sagebrush, and 3) native bunchgrass grassland. This vegetation composition is typical of the mid- and higher-elevation portions of the sagebrush steppe throughout the northern Great Basin (see Fig. 1). All three of these vegetation types would be classified to the Loamy 16-22 Ecological Site with the two shrub-dominated types representing state 1, phase 1.5 and the bunchgrass type, state 1, phase 1.4 (R025XY022ID). In addition to the two codominant shrub species, the mountain big sagebrush − mountain snowberry type includes yellow rabbitbrush (*Chrysothamnus viscidiflorus* [Hook.] Nutt.), Saskatoon serviceberry (*Amelanchier alnifolia* [Nutt.] Nutt. ex M. Roem. *alnifolia*), bluebunch wheatgrass (*Pseudoroegneria spicata* [Pursh] A. Love), Sandberg bluegrass (*Poa secunda* J. Presl.), squirreltail (*Elymus elymoides* [Raf.] Swezey), Idaho fescue (*Festuca idahoensis* Elmer), basin wildrye (*Leymus cinereus* [Scribn. & Merr.] A. Love), mountain brome (*Bromus marginatus* Nees ex Steud.), silvery lupine (*Lupinus argenteus* Pursh), tapertip hawksbeard (*Crepis acuminata* Nutt.), western stoneseed (*Lithospermum ruderale* Douglas ex Lehm.), and western aster (*Symphyotrichum ascendens* [Lindl.] Nesom). Graminoid components of the antelope bitterbrush − mountain big sagebrush type include bluebunch wheatgrass, Sandberg bluegrass, and squirreltail. Arrowleaf balsamroot (*Balsamorhiza sagittata* [Pursh] Nutt.), western yarrow (*Achillea millefolium* L. var. *occidentalis* DC.), buckwheats (*Eriogonum spp.* Michx.), biscuitroots (*Lomatium spp.* Raf.), and tapertip hawkbeard are the principal forbs in this type. Bluebunch wheatgrass, Sandberg bluegrass, squirreltail, Idaho fescue, and needlegrasses (*Achnatherum* spp. Beauv.) dominate the native bunchgrass cover type. Cheatgrass (*Bromus tectorum* L.) has a minor to common presence in all three of these dominant vegetation types.

Two additional vegetation types occur in minor extents within the study area. Dry meadow grassland occurs in some swales on the footslope. Western rush (*Juncus occidentalis* Wiegand) and bluegrasses (*Poa* ssp. L.) are the principal vegetation in this type. Several small stands (< 1 ha) of quaking aspen (*Populus tremuloides* Michx.) occupy

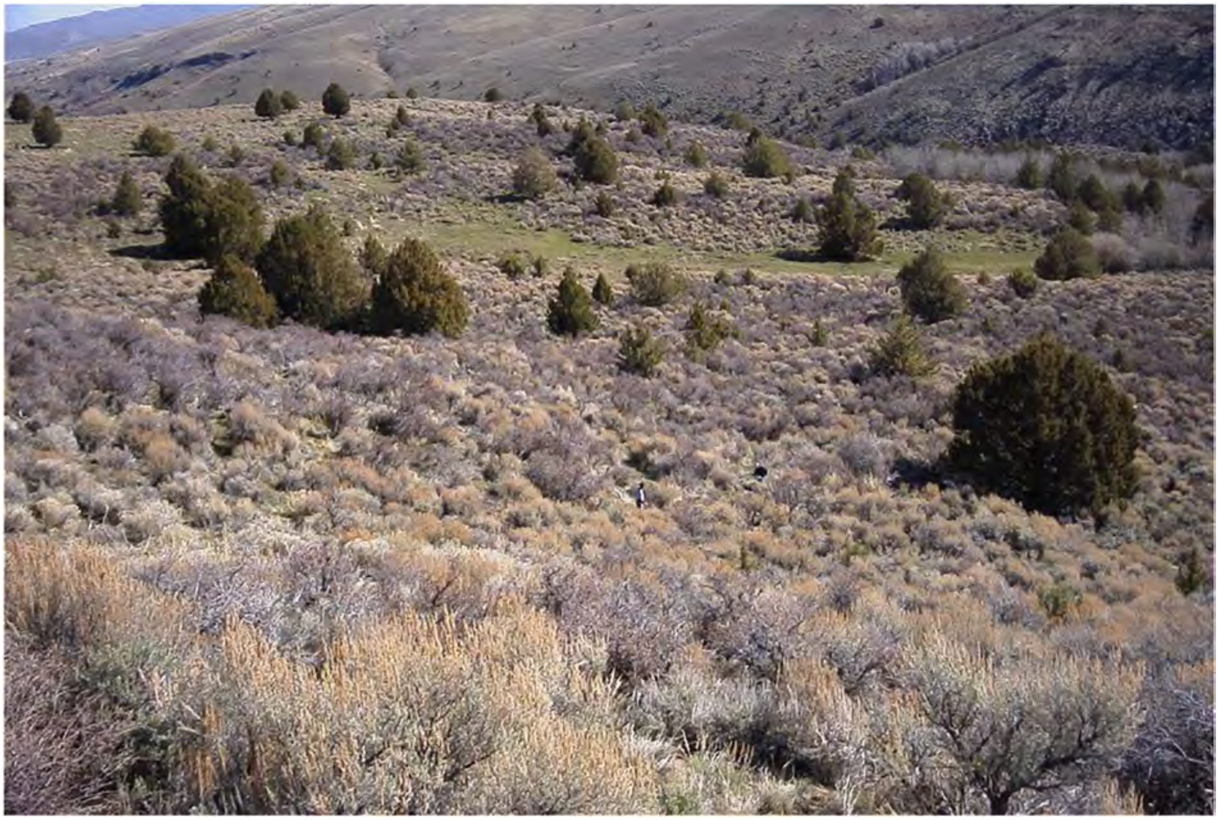

Figure 1. Photograph taken 9 April 2001 illustrating the vegetation cover types, fuel load conditions, and western juniper encroachment level at the Breaks prescribed-fire study area within the Reynolds Creek Experimental Watershed in the Owyhee Mountains of southwestern Idaho. View is from the backslope position, near the geographic center of the study area, looking northeast across the footslope (distant hills in background are outside the study area). Large green trees are western juniper. Smaller juniper trees can occasionally be seen just protruding above the mountain big sagebrush (tawny-colored tops) and antelope bitterbrush (gray tops) shrub canopy. A bunchgrass grassland is evident near the upper left corner. The green strip across the upper-center is a dry meadow grassland. The gray, leafless trees near the upper-right corner form a quaking aspen stand where encroached juniper trees are evident. As a scale reference, the person standing in the center of the photograph is 2 m tall.

swales and moist areas on the footslope. Forb productivity in the aspen type tends to be greater than in any of the four remaining types. Sweetcicely (*Osmorhiza berteroi* DC.), Rocky Mountain iris (*Iris missouriensis* Nutt.), white sagebrush (*Artemisia ludoviciana* Nutt.), and western yarrow are the principal herbaceous species in this type. Given their limited spatial extent and few juniper trees, the dry meadow grassland and quaking aspen types were later combined for statistical analyses into a single composite type named "Other" vegetation.

Based on an aerial photography time series and corroborating descriptions by the landowner, this landscape had not burned since before 1939. In the more than 63-yr absence of fire, shrubs in the mountain big sagebrush and bitterbrush vegetation types had grown rather large, densely spaced, and ecologically dominant. Bitterbrush plants on the deeper soils of the footslope were commonly $1.5-2$ m or more in height, and bitterbrush on the much less productive, rocky divides dissecting the backslope were often well over 1 m in height. This long interval without fire had apparently also allowed western juniper to encroach into all five vegetation types present. Juniper encroachment into the bitterbrush and mountain big sagebrush types was in the early successional or phase I stage (Miller et al., 2000, 2005). Cohorts of young juniper trees were just starting to exceed the height of the shrub canopy. Some small areas also contained clusters of larger, older trees, as evident in Figure 1, which were codominant with the shrub understory and thus would be classified as phase II encroachment (Miller et al., 2005). Juniper trees in the meadow grassland type were relatively young, small trees (<3 m height), while both small and large juniper trees ($4-5$ m in height) were present in some aspen stands.

Prefire Sampling

In the 2001 study yr, 200 western juniper trees were randomly selected from the population growing within the study area boundaries. Selected trees were individually marked at the base with numbered metal identification tags and measured for total height and diameter at breast height (DBH). Trees having DBH > 4 cm ($n = 118$) were cored at breast height (1.37 m) with an increment borer to determine their age (i.e., breast-height age). The remainder ($n = 82$) were considered juvenile trees and too small to core.

A GIS was used to determine the elevation (m), slope (%), terrain curvature (index value), aspect (four classes), and hillslope position (four classes) of each of the 200 tree locations based on raster elevational data (5-m ground sample distance [GSD]) derived from airborne LiDAR sampling. Locations of all juniper trees visible in digitized aerial photography (NAPP3, 1:40 000 scale, black/white, and acquired in 1998) of the study area were identified and recorded as points in the GIS (Fig. 2). Density (# ha^{-1}) of these visible trees was assessed within a 100-m diameter circular zone surrounding each of the 200 marked tree locations. The distance to the nearest visible juniper tree from each marked tree location was also assessed using the GIS. A vegetation cover type map, derived by classifying airborne hyperspectral imagery (5-m GSD) acquired 8 September 2001, was used to determine the vegetation type occurring at each tree location and dominant vegetation type within the 100-m circular zone surrounding the tree location.

Vegetation cover and biomass near each tree location were characterized during peak vegetation production (late June/early July) of the

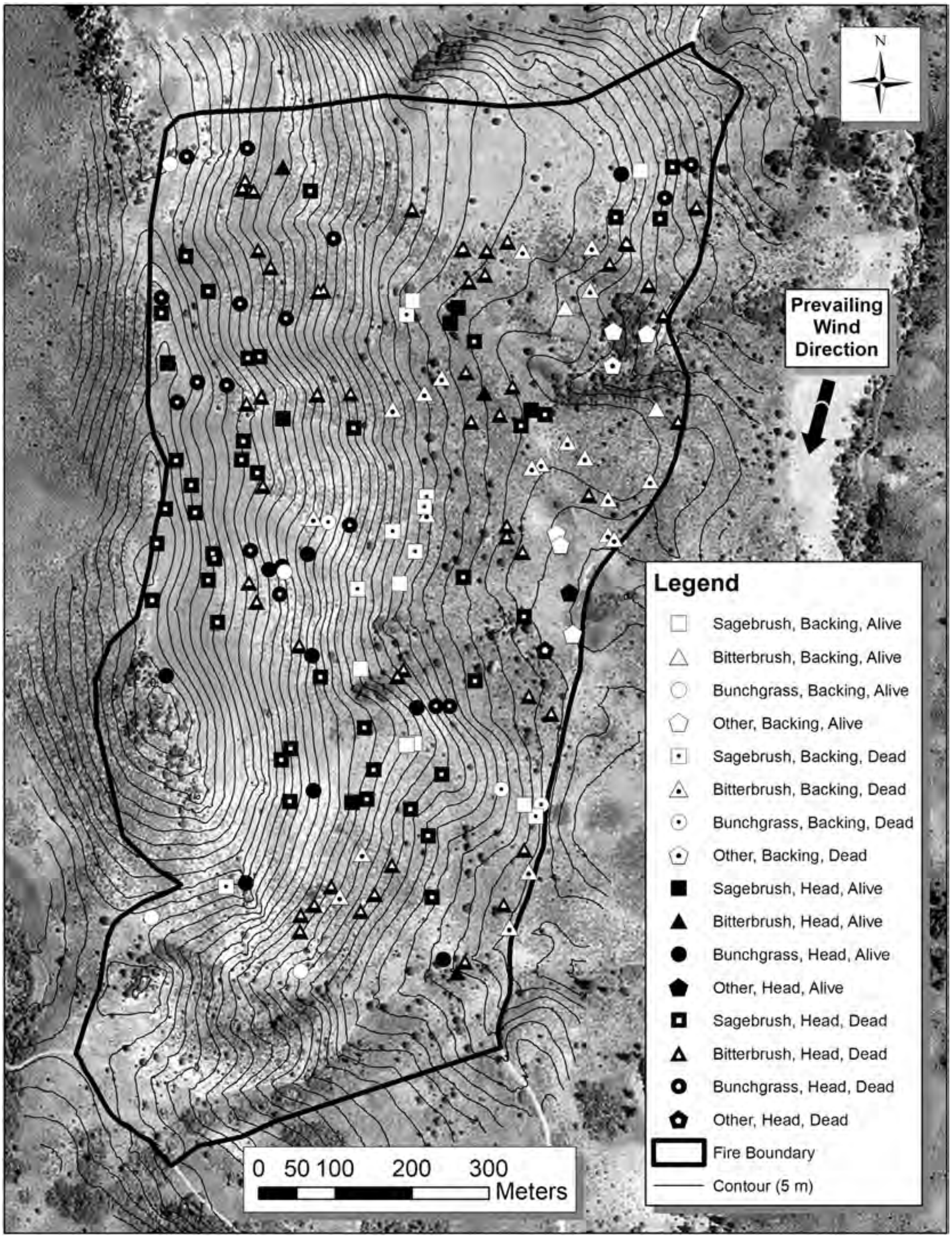

Figure 2. Map illustrating marked western juniper tree locations indexed by vegetation type, fire type (head or backing), and postfire status (alive or dead) at the Breaks study area. Vegetation types at tree locations include mountain big sagebrush/snowberry (Sagebrush), antelope bitterbrush/mountain big sagebrush (Bitterbrush), native bunchgrass grassland (Bunchgrass), and other vegetation (Other, which includes dry meadow grasslands and quacking aspen stands). Elevation contour lines illustrate terrain shape. Background image is an orthophotograph acquired in 1998, four yr before the Breaks prescribed fire.

2002 study yr. Canopy cover (%) values for graminoids, forbs, and shrubs were determined in randomly located 1-m^2 plots occurring in the same vegetation type and within 100 m of the tree location. Ground cover (%) by vegetation, soil, rock, moss, dung, and litter were also assessed in these plots. Aboveground biomass (kg ha^{-1}) of graminoids, forbs, shrubs, and litter were measured in the plots. Biomass was quantified by clipping (2.54-cm stubble), harvesting, sorting, and oven drying (50°C) all vegetation matter within the plots. Green and woody shrub biomass were quantified separately. In terms of fuel size, graminoid, forb, and green shrub biomass would be classified as live, fine fuels (< 0.64 cm). Woody shrub biomass samples included standing, nonphotosynthetic live and dead fuels but were not sorted by size class. Mean biomass and fuel loading by functional group or fuel type near marked juniper trees is provided in Table 1.

Prescribed-Fire Application

The Breaks prescribed fire was conducted on the study area during 24 September 2002 by the US Department of Interior Bureau of Land Management according to their burn plan (USDI-BLM, 2002). Principal objectives for the prescribed fire treatment were to remove 70 – 90% of the western juniper in the study area while maintaining and

Table 1
Aboveground biomass and fuel loading (mean ± SD kg m^{-1}) by functional group or fuel type as determined from samples collected near a random selection of western juniper trees before fall prescribed fire at the Breaks prescribed-fire study area.

Functional group or fuel type	Tree status[2]	Vegetation Type[1]			
		Mountain big sagebrush	Antelope bitterbrush	Bunchgrass grassland	Other
Graminoid (1-hr)	Alive	25.3 ± 29.8	58.6 ± 47.6	114 ± 97.4	68.0 ± 31.7
	Dead	42.3 ± 65.9	43.1 ± 34.8	75.9 ± 50.8	52.3 ± 19.4
Forb (1-hr)	Alive	80.0 ± 35.9	15.5 ± 16.3	45.9 ± 46.7	110 ± 121
	Dead	93.4 ± 60.6	39.6 ± 45.0	46.6 ± 37.7	138 ± 181
Shrub green (1-hr)[3]	Alive	45.6 ± 77.6	93.2 ± 35.6	2.49 ± 6.32	0
	Dead	85.3 ± 88.4	103 ± 78.2	0.09 ± 0.41	0
Litter (1-hr)	Alive	786 ± 370	528 ± 359	589 ± 384	128 ± 127
	Dead	632 ± 390	984 ± 656	356 ± 424	163 ± 183
Shrub wood (1-hr +)[4]	Alive	538 ± 635	1 085 ± 929	164 ± 417	0
	Dead	494 ± 643	1 482 ± 1 388	0	0

[1] Vegetation type at the marked tree location.
[2] Tree mortality status when assessed 5 yr post fire.
[3] Green, photosynthetic material, < 0.64-cm diameter, from shrubs.
[4] Woody, nonphotosynthetic, standing material, regardless of size, from shrubs.

improving the native bunchgrass stands, watershed function, and diversity of wildlife habitats. Fuels on site and adjacent were identified in the plan as being similar to those represented in Natural Fuels Photo Series code: PMS 830 WJ 02 (Ottmar et al., 1998) and Fuel Model 6 (Albini, 1976; Scott and Burgan, 2005). Prevailing wind direction on site during the day time is dominated by up-canyon flow moving from northeast to southwest. Wind direction reverses in the evening as down-canyon flow takes over. Fuel loading recorded in the plan included 3.4 Mg ha^{-1} (1.5 tons ac^{-1}) of fine, 1-hr fuels (< 0.64 cm or < ¼ in diameter), 2.2 Mg ha^{-1} (1.0 tons ac^{-1}) of 10-hr fuels (0.64–2.5 cm or ¼-1 in diameter), larger fuels were not recorded as present. Total dead fuel was listed as 0–10%. Duff depth was < 2.5 cm (< 1 in). Surface fuel depth was recorded as 0.61 − 1.8 m (2–6 ft). Total fuel loading (live and dead combined) was listed as about 5.6 Mg ha^{-1} (2.5 tons ac^{-1}).

Preliminary black-lining operations were conducted on the western (ridgeline with highest elevations) and northern boundaries during the evening of 16 September 2002. Additional black-lining was done on the morning of 24 September 2002 along a hand-made control line to form and reinforce the southern boundary of the study area. Main burn operations commenced at 12:07:22 Mountain Standard Time (MST) on 24 September 2002 and consisted of multiple staggered drip-torch ignition lines moving cross slope from the southern boundary toward the north. Ignitors on the upper hill slopes preceded those lower on the slope. Ignition operations were completed about 3 hr later at 15:23:52 of 24 September 2002 (MST). Wind direction during these operations remained quite consistently from the northeast with rare shifts to the north. Wind speeds averaged 2.2 m s^{-1} with a range of 0.66 − 3.5 m s^{-1}. Air temperature ranged from 20.8°C to 22.6°C and relative humidity 18.1 − 20.0%. Fuel moisture samples were collected at six sites distributed throughout the study area on the afternoon of 23 September 2001, the day before prescribed fire treatment application. Mean fuel moisture by type and size class are presented in Table 2.

Table 2
Fuel moisture (%) by functional group and fuel size class sampled on 23 September 2002 at the Breaks prescribed-fire study area within the Reynolds Creek Experimental Watershed in the Owyhee Mountains of southwestern Idaho.

Functional group	Fuel size	Type	Mean	SD	Max	Min
Grass	1-hr	Live/Dead	18.5	2.99	20.6	16.4
Forb	1-hr	Live/Dead	11.3	5.44	15.2	7.49
Shrub	1-hr	Green	36.1	5.14	39.7	32.4
Shrub	1-hr	Dead	7.30	0.427	7.60	7.00
Shrub	10-hr	Dead	7.25	1.08	8.02	6.49
Shrub	100-hr	Dead	9.02	0.280	9.22	8.83
Shrub	1 000-hr	Dead	7.08	NA	NA	NA
Downwood	1-hr	Dead	6.84	0.487	7.28	6.32
Downwood	10-hr	Dead	6.70	0.0756	6.76	6.61
Downwood	100-hr	Dead	7.22	0.593	7.72	6.56
Downwood	1 000-hr	Dead	11.1	1.34	12.1	10.1

Fire behavior was monitored throughout the ignition and active burning operation period using time-lapsed photography shot from three vantage points on the canyon wall opposite (east) of the study area. Digital cameras linked to laptop computers automatically acquired and stored images of the study area at 10-s intervals. Generally, despite smoke occlusion, any location on the study area landscape where active burning was occurring was visible from at least one of the vantage points. Scorch poles (3-m in height) provided a reference for monitoring flame dimensions as the fire progressed across the landscape. The staggered ignition pattern and variable terrain resulted in at least four different fire types, presented by increasing order of fire intensity: 1) backing fires moving downslope from ignition lines, 2) backing fires moving cross slope, 3) head fires moving upslope from ignition lines, and 4) head fires moving upslope and with the wind direction.

Postfire Sampling

Mortality rate of the marked trees was visually assessed in 2007, 5 yr post fire. Twenty-two of the 200 marked juniper trees were exposed to fire during the blacklining operation 1 wk prior to the main prescribed fire. Since the burning conditions differed during blacklining (i.e., higher humidity, lower temperature, and inversed wind direction) from that of the main fire, these 22 trees were excluded from the sample and statistical analysis was conducted on the remaining 178 marked trees (see Fig. 2). In 2007, 140 of these trees were assessed as "dead" and 38 as "alive" for an overall mortality rate of 78.7%. Further details regarding observed tree mortality rate relative to fire behavior, vegetation, and terrain are provided in Table 3.

Statistical Analysis

Efficacy of prescribed fire for killing western juniper trees was assessed using logistic regression modeling. Tree mortality (dead vs. live marked trees) was the binary response variable. The logistic regression models were developed from a list of 26 predictors known to influence fire behavior, burn severity, and/or fire-induced mortality rates in other conifers. This list included 3 tree attribute variables, 1 fire type variable, 3 fire weather variables, 5 terrain variables, 2 tree density variables, and 12 vegetation or fuel variables (Table 4). A principal intent for this analysis was to develop a predictive model for fire-planning purposes that relied on information which could be readily assessed prior to burning and did not rely upon fire intensity or postfire severity measurement data. Consequently, fire injury variables that are assessed post fire and commonly included in fire-mortality models for conifers (e.g., Ryan and Reinhardt, 1988; Hood et al., 2007) were not included

Table 3
Observed fire-induced mortality for a random sample of western juniper trees with height ≥ breast height (1.37 m) (labeled "taller trees") and < breast height ("shorter trees") under differing fire behavior, vegetation, and terrain conditions in the Breaks prescribed-fire study area.

Condition	Level	Taller trees		Mortality rate	Shorter trees		Mortality rate
		Dead	Alive		Dead	Alive	
		--#--	--#--	--%--	--#--	--#--	--%--
Fire type	Head fire	61	15	80.3	47	5	90.4
	Backing fire	17	13	56.7	15	3	75.0
Vegetation type[1]	Mountain big sagebrush	23	13	63.9	25	0	100
	Bitterbrush	45	5	90.0	26	0	100
	Bunchgrass grassland	9	7	56.3	10	7	58.8
	Other	1	3	25.0	1	3	25.0
Hillslope position	Footslope	35	15	70.0	34	6	85.0
	Backslope	38	11	77.6	20	2	90.9
	Shoulder	4	0	100	3	1	75.0
	Summit	1	2	33.3	5	1	83.3

[1] Vegetation type at the tree location.

here. The limiting sample size for this marked tree data set was the 38 nonevent observations (i.e., live marked trees that survived the fire) (Babyak, 2004). Consequently, we limited model complexity to four predictors based on the rule of 10 observations per predictor (Peduzzi et al., 1996) to avoid overfitting. The glmulti package in R (Calcagno, 2013) was used to fit all possible logistic regression models of four predictors or less, without interactions, based on the list of 26 predictors. Models received an initial ranking based on Akaike information criterion (AIC) score. Multicollinearities existed among some pairs of predictors, and these are indicated in Table 4. Model selection from this ranked set, consequently, was constrained to avoid including both members of a highly correlated ($r \geq 0.6$) predictor pair within any selected model. After this screening, the remaining top 10 scoring models were selected for further evaluation. The predictive performance of these top models was evaluated using a repeated k-fold cross-validation procedure implemented with the caret package in R (Kuhn, 2017). Ten folds and 100 repeats per fold were used. Classification performance of each model was evaluated using a ROC analysis, which compared true positive predictions of tree mortality to false positive predictions from the cross-validation procedure. An AUC statistic was calculated as a measure of predictive accuracy for each model. Sensitivity and specificity statistics were also calculated to evaluate model performance in terms of true-positive and true-negative classification rates, respectively. A final model was selected, based on performance and practicality, for further evaluation. An analysis of deviance was conducted on this final model to assess the relative importance of each predictor in the model.

Results

Top Models

A summary of the top 10 models, as ranked by AIC score, is provided in Table 5. Vegetation type and one of the three tree attributes occurred in all 10 top models with tree age occurring most frequently (8 of 10 models). Fire type was included in 8 of the 10 models. While all other models in the top 10 list consisted of 4 predictor variables, the seventh-ranked model consisted of only these 3 most prominent predictors, vegetation type, tree age, and fire type.

The best and runner-up models were similar in composition, differing only where tree age is included in the former and tree height in the latter, and yielded similar results (Tables 5 and 6). Prediction accuracy, assessed using the repeated k-fold cross-validation procedure, was also similar between the best model (AUC = 0.873) and the runner-up model (AUC = 0.881). Sensitivity scores for both these models were quite high while specificity scores were relatively low. Sensitivity or the true positive rate is, in this case, the proportion of marked trees killed by fire that were correctly predicted to be killed. The high sensitivity scores indicate both models were effective at avoiding false-

Table 4
Set of 26 predictor variables evaluated in logistic regression models of western juniper tree mortality following fall prescribed fire at the Breaks prescribed-fire study area. Multicollinearities ($r \geq 0.6$) were detected among some variables, and these are identified in the footnotes.

Condition	Predictor	Type	Levels	Units
Tree attributes	Age[1]	Continuous		yr
	Diameter at breast height	Continuous		cm
	Height, total	Continuous		m
Fire type	Fire type	Categorical	Head fire Backing fire	
Fire weather	Air temperature[2]	Continuous		°C
	Relative humidity[2]	Continuous		%
	Wind speed[2]	Continuous		$m\ s^{-1}$
Terrain	Aspect	Categorical	North East South Other[3]	
	Elevation[4]	Continuous		M
	Curvature	Continuous		index value
	Hillslope position	Categorical	Foot slope Back slope Shoulder Summit	
	Slope	Continuous		%
Tree density	Density of mature trees[5]	Continuous		trees ha^{-1}
	Nearest mature tree	Continuous		M
Vegetation type	Vegetation type[6]	Categorical	Bunchgrass grassland Mountain big sagebrush Bitterbrush Other[7]	
	Vegetation near[5]	Categorical	Bunchgrass grassland Mountain big sagebrush Bitterbrush Other[7]	
Cover	Graminoid cover[8]	Continuous		%
	Forb cover[8]	Continuous		%
	Shrub cover	Continuous		%
	Litter cover	Continuous		%
	Bare ground	Continuous		%
Biomass	Graminoid biomass	Continuous		kg/ha
	Forb biomass	Continuous		kg/ha
	Shrub green biomass[9]	Continuous		kg/ha
	Shrub wood biomass[10]	Continuous		kg/ha
	Litter biomass[8]	Continuous		kg/ha

[1] Age at breast height (1.37 m from ground surface) based on increment coring. All 3 tree attribute variables including tree age were highly correlated ($r \geq 0.9$).
[2] Measured at 3-m height. Air temperature and relatively humidity were correlated ($r = 0.791$).
[3] Includes West aspects and flat areas (< 3% slope).
[4] Elevation and slope were correlated ($r = 0.679$).
[5] Number of trees and dominant vegetation type within a 100-m circular area at marked tree location.
[6] Vegetation type at the marked tree location.
[7] Includes dry meadow grassland and aspen types.
[8] Graminoid cover and graminoid biomass were correlated ($r = 0.806$), as were forb cover and biomass ($r = 0.724$) and litter biomass and bare ground and/or rock cover ($r = -0.850$).
[9] Green, photosynthetic material, less than 0.64 cm diameter, from shrubs. Shrub green and wood biomass were correlated ($r = 0.650$), and both were correlated with shrub cover ($r = 0.884$ and $r = 0.650$, respectively).
[10] Woody, nonphotosynthetic, standing material, regardless of size, from shrubs.

Table 5
Top 10 logistic regression models as ranked by Akaike information criterion score and their performance (area under the curve [AUC], sensitivity, and specificity), as assessed using a repeated k-fold cross-validation procedure, for predicting fire-induced mortality of western juniper trees in the Breaks prescribed-fire study area.

Rank	Predictors	AIC	AUC ± SD[1]	Sensitivity ± SD	Specificity ± SD
1	Vegetation type Tree age[2] Fire type Bare ground	135.00	0.873 ± 0.116	0.936 ± 0.0527	0.475 ± 0.299
2	Vegetation type Tree height[3] Fire type Bare ground	136.45	0.881 ± 0.128	0.936 ± 0.0527	0.500 ± 0.312
3	Vegetation type Tree age Fire type Forb biomass	136.79	0.862 ± 0.104	0.936 ± 0.0527	0.475 ± 0.322
4	Vegetation type Tree age Fire type Forb cover	136.93	0.862 ± 0.103	0.929 ± 0.0476	0.450 ± 0.284
5	Vegetation type Tree age Fire type Litter cover	137.45	0.848 ± 0.141	0.936 ± 0.0527	0.475 ± 0.299
6	Vegetation type Tree age Bare ground Relative humidity	137.48	0.844 ± 0.164	0.943 ± 0.0563	0.475 ± 0.343
7[4]	Vegetation type Tree age Fire type	137.48	0.844 ± 0.129	0.936 ± 0.0405	0.425 ± 0.313
8	Vegetation type Tree DBH[5] Fire type Bare ground	137.57	0.848 ± 0.157	0.943 ± 0.0563	0.450 ± 0.258
9	Vegetation type Tree age Forb cover Bare ground	137.65	0.867 ± 0.138	0.921 ± 0.0527	0.425 ± 0.290
10	Vegetation type Tree age Fire type Wind speed	137.74	0.855 ± 0.104	0.936 ± 0.0226	0.533 ± 0.315

[1] Area under the receiver operating characteristic curve.
[2] Age at breast height (1.37 m from ground surface) based on increment coring.
[3] Total height of tree from ground surface.
[4] Model with 3 predictor variables rather than 4.
[5] Diameter at breast height.

negative predictions (i.e., falsely predicting trees will survive the fire when they actually were killed). The false negative or type II error rate for both models was only about 3.7%. Specificity or the true negative rate is the proportion of all surviving marked junipers that were correctly predicted to survive the fire. Low specificity scores revealed that both models tended to produce some false-positive predictions and thus overpredicted the mortality rate. In fact, all 10 top models exhibited this kind of compromise between sensitivity and specificity (see Table 5).

From a practical, field-application point of view, a model containing tree height rather than age would be more useful to managers because tree height measurements are much more readily and efficiently attainable in the field than tree age. Consequently, given their close similarities in performance, we choose to focus the remainder of the results presentation on the runner-up rather than the top model (see Table 6). Hereafter, the runner-up model is referred to as the final model.

Final Model

Analysis of deviance (AOD) indicated all predictors in the final model significantly reduced deviance relative to the null model (i.e., intercept-only model) (Table 7). Vegetation type, however, was clearly the most important predictor in the model, followed closely by tree height, with fire type and bare ground (includes bare soil plus rock cover) providing relatively minor reductions in model deviance. In fact, a review of the composition of 500 of the top AIC-scoring models revealed that vegetation type was included in all 500 models, while tree height, fire type, and bare ground occurred in 173, 62, and 62 of these models, respectively, thus tending to support this AOD ranking of predictor importance.

The final model was fitted using the mountain big sagebrush type as the reference class for the vegetation type predictor. Calculating the odds ratios for this model revealed that a juniper tree in the bitterbrush type was about 8 times as likely to be killed by fire as a juniper tree in the mountain big sagebrush type. Trees in the bunchgrass and other vegetation types were about 0.2 and 0.05 times, respectively, as likely to be killed by fire as those in the mountain big sagebrush type. The predicted probability of a juniper in the mountain big sagebrush, bitterbrush, bunchgrass, or other type being killed by fire was 77.6%, 96.6%, 38.9%, or 13.8%, respectively.

With regards to the other predictors in the final model, the probability of being killed by fire decreased by 28.8% with each 1-m increase in tree height. A juniper tree exposed to a head fire had a probability of 75.1% of being killed and was about 3 times as likely to be killed as a

Table 6
Logistic regression model fit results for best and runner-up models, as ranked by Akaike information criterion scores, of fire-induced mortality of western juniper trees in the Breaks prescribed-fire study area.

Model	Predictor	Coefficient	St. Error	Z value	P value
Best	Intercept	1.150	0.6749	1.704	0.0884
	Vegetation type[1]—bitterbrush	1.915	0.6538	2.931	0.0034
	Vegetation type—bunchgrass	−1.645	0.5811	−2.830	0.0047
	Vegetation type—other	−3.089	0.9992	−3.092	0.0012
	Tree age	−0.05821	0.01380	−4.217	< 0.0001
	Fire type[2]—head	1.189	0.4931	2.411	0.0159
	Bare ground	0.02133	0.01048	2.035	0.0418
Runner-up	Intercept	1.245	0.6806	1.830	0.0673
	Vegetation type—bitterbrush	2.087	0.6627	3.148	0.0016
	Vegetation type—bunchgrass	−1.696	0.5790	−2.930	0.0034
	Vegetation type—other	−3.079	0.9958	−3.092	0.0020
	Tree height	−0.3402	0.08182	−4.158	< 0.0001
	Fire type—head	1.106	0.4898	2.258	0.0240
	Bare ground	0.02283	0.01052	2.169	0.0301

[1] Vegetation type where the mountain big sagebrush-snowberry type is the reference class.
[2] Fire type where the backing fire type is the reference class.

tree exposed to a backing fire. Interestingly, for each 1 percentage point increase in bare ground near the juniper tree location, the probability of being killed by fire was predicted to increase by 2.31%. Post-hoc mean comparisons indicated this effect was primarily limited to the bunchgrass type within which bare ground was greater ($P = 0.0125$) at locations where trees were killed by fire (54.6% bare ground) than at those where trees survived (31.3% bare ground) (Fig. 3).

The final model predicted that 148 of the marked juniper trees would be killed by fire and 30 would survive. Compare this to the observed 140 dead and 38 live marked trees. As noted earlier, sensitivity of the final model was thus quite high (see Table 5). Only 8 false-negative predictions were made compared with 132 true positive predictions. Four of these eight trees occurred in the bunchgrass grassland type, two in the other type, and one tree each in the mountain big sagebrush and bitterbrush types. The tree occurring in the bitterbrush type, where the probability of mortality was otherwise high, was rather tall (11.5 m height), exposed to a backing fire, and had less bare ground than the marked sample mean. In fact, seven of these eight false-negative cases involved trees taller than breast height (1.37 m). In the remaining case, although the tree was shorter than breast height and thus likely more vulnerable to fire, it occurred in the least hazardous, other vegetation type; was exposed to a backing fire; and, interestingly, its location had less bare ground than the mean of all 178 trees. As such, predictions of survival for these eight trees do seem well justified. However, there is a factor not accounted for in the final model which would seem to explain, in most cases, why these trees were killed despite predictions to the contrary. Vegetation within a 100-m buffer of the tree location (i.e., the "vegetation near" predictor; see Table 4) was dominated by the bitterbrush type for six trees and mountain big sagebrush for one tree. In these cases, it is likely the effects of these hazardous vegetation conditions surrounding the tree locations overwhelmed the moderating influence of the modeled factors, which otherwise promoted predictions of survival.

Although specificity of the final model would be considered low (see Table 5), it was based on a rather small sample size of 38 marked trees that survived the fire and thus this finding should be interpreted with some caution. The model made 16 false-positive and 22 true-negative predictions. Five of the 16 false-positive cases occurred in bitterbrush and 5 in the mountain big sagebrush, the most hazardous fuel types. The remaining six cases occurred in the less hazardous, bunchgrass type but all six of these trees were of less than breast height and five were exposed to a head fire. Predictions of mortality, therefore, do seem sensible for all 16 false-positive cases. These prediction errors might, however, be explained by a paucity of litter and forb and/or graminoid fuels. Litter biomass for 13 false-positive cases and forb biomass for 11 false-positive cases were less than the respective means for all 178 marked trees. All 10 false-positive cases occurring in the bitterbrush and mountain big sagebrush types had less litter and forb biomass than the respective means for these vegetation types. Most trees involved in the remaining six false-positive cases were located where graminoid (five of six trees) and litter biomass (four of six trees) were less than the respective means for the bunchgrass vegetation type.

Discussion

Vegetation Type

On this study area, western juniper trees in the bitterbrush type were more likely to be killed by fall prescribed fire than those in the mountain big sagebrush type. Shrub cover, shrub wood biomass, and litter cover were all greater in the bitterbrush than mountain big sagebrush type. Consequently, the bitterbrush type generally had a heavier and more continuous fuel load than the mountain big sagebrush type. This heavier fuel load had the potential of producing greater fire intensity in the bitterbrush type (Byram, 1959; Rothermel, 1972). Furthermore, bitterbrush stemwood is typically a denser fuel than stemwood of mountain big sagebrush and this difference likely promoted longer fire residence time and deeper flaming front in the bitterbrush than sagebrush type (Kaitpraneet, 1982).

Vegetation on the study area had not burned since before 1939; consequently, the bitterbrush and mountain big sagebrush stands present during the 2002 fire were at least 63 yr old (see Fig. 1). Based on fire scars and other chronological evidence, presettlement fire-return intervals in mountain big sagebrush vegetation ranged from 15 to 25 yr

Table 7
Analysis of deviance results for an intercept-only model and the final predictive model of fire-induced mortality of western juniper trees in the Breaks prescribed-fire study area.

Model	Predictor	DF	Deviance	Resid. DF	Resid. deviance	P value
Null	Intercept			177	184.6	
Final	Vegetation type	3	30.53	174	154.1	$1.065e^{-6}$
	Tree height	1	21.32	173	132.7	$3.885e^{-6}$
	Fire type	1	5.157	172	127.6	0.0232
	Bare ground	1	5.141	171	122.4	0.0234

DF stands for Degrees of Freedom.

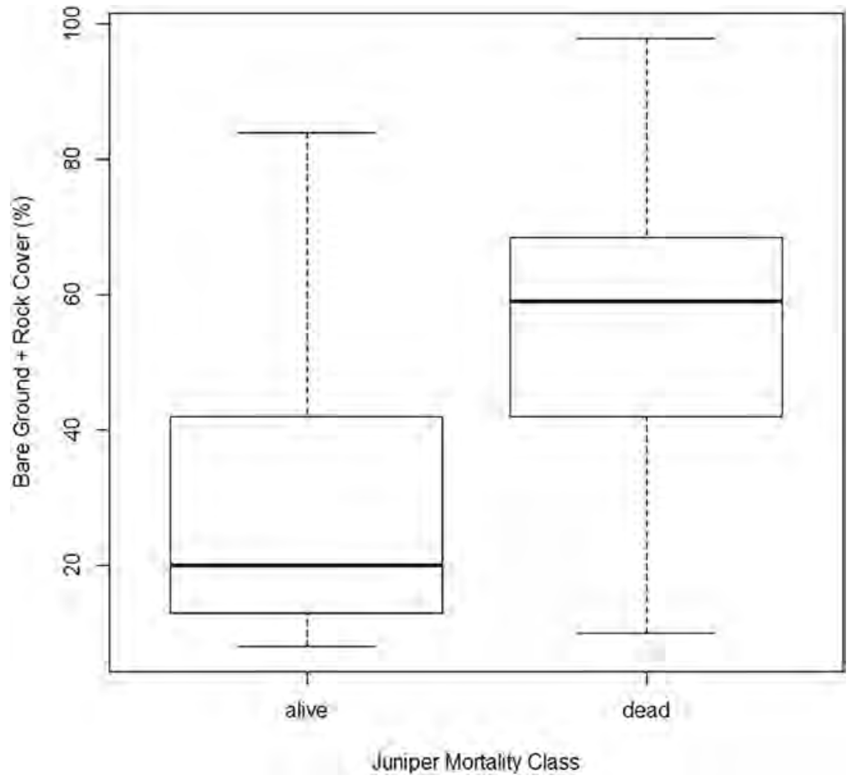

Figure 3. Box-plot of the bare ground percentages at locations of western juniper trees, in the bunchgrass grassland vegetation type, which were killed by prescribed fire and those which survived the fire.

(Burkhardt and Tisdale, 1976; Miller and Rose, 1999). Clearly, vegetation in the Breaks study area was long overdue for a fire event, as are many areas throughout the northern Great Basin where western juniper encroachment is occurring (Miller and Rose, 1999; Miller et al., 2005). Nevertheless, observed differences in fuel loads and juniper tree mortality for the bitterbrush and mountain big types in the current study may not hold true for landscapes where stands are younger and shrubs smaller in size (e.g., Martin et al., 1981).

The much lower predicted probability of mortality for juniper trees in the bunchgrass type is explained by the generally lower fuel loads and the absence of ladder fuels to carry fire into the juniper canopy. It is surprising that the observed juniper mortality rates in the bunchgrass type were about the same for taller, mature trees (≥ 1.3-m height) and shorter, juvenile trees (see Table 3). Lacking ladder fuels, one might expect greater survival of taller than shorter trees in the bunchgrass type. It is difficult to explain this departure from expectation, however, and it may simply be a spurious result stemming from the relatively small sample size of marked trees in the bunchgrass type ($n = 33$). A principal difference, in the bunchgrass type, between juniper trees killed by fire and those that survived was the amount of bare ground. This relationship is explored in a separate subsection later. The sample size for marked juniper trees ($n = 8$) in the other vegetation type is still too small, despite combining the dry meadow grassland and aspen vegetation types into one composite type, to draw any meaningful conclusions.

Tree Height, Age, and Other Features

Fire-caused mortality was predicted to decrease with increased tree height. Research on Ashe juniper in Texas found shorter trees (e.g., 50 cm tall) were more likely to be killed by fire than taller trees (Noel and Fowler, 2007). Research with redberry juniper, alligator juniper, and honey mesquite also indicates that tree size or height strongly influences fire-induced mortality rates (Johnson et al., 1962; Britton and Wright, 1971; Steuter and Britton, 1983). These findings are intuitive and have a straightforward, physical basis. Shorter trees are prone to more extensive crown injury than taller trees because shorter trees tend to have a larger proportion of their crowns inside or in near proximity to the flame zone. Taller trees can potentially have some or all their crown positioned well above the flame zone of a low-intensity surface fire (i.e., burning primarily herbaceous fuels). In the current study, trees shorter than breast height (1.37 m) were particularly susceptible to fire in shrub-dominated vegetation where, potentially, the entire crown was within the flame zone of shrub-fed combustion and thus the observed mortality rate was 100% (see Table 3).

As noted earlier, models based on tree age performed somewhat better than those based on tree height but we focused on the latter given of the relative ease of measuring height in the field. Tree height, age, and other morphological or status factors can be physically related. Shorter stature often equates to younger tree age, smaller DBH, and thinner bark. Trees of thinner bark have greater susceptibility to cambium injury (Gill, 1995). Cambium heating or scorch can be a proximate cause of fire-induced conifer mortality (Michaletz and Johnson, 2007). However, not all short juniper trees have thin bark. Adverse site conditions, injury, etc. can stunt tree growth. Thus, some short trees can be relatively old and have, consequently, developed thick, insulating bark. A short juniper tree with thick bark might be somewhat protected from a low-intensity, ground fire with relatively short flame heights, but this same fire would likely produce cambium injury in thin-barked trees. Fire-efficacy models based on tree age rather than tree height performed better, probably because the former provided a more robust reflection of bark-thickness effects while still retaining an effective surrogate for tree height effects. As such, a model based on tree age would better account for the contributions of both crown and cambium injury to fire-induced juniper mortality than a model based on tree height.

Fire Type

It is not surprising that juniper tree mortality rates were predicted to be higher for head than backing fires. All else held constant, a head fire has a larger propagating heat flux and thus greater rate of spread than a backing fire (Rothermel, 1972). A head fire moving upslope effectively preheats upslope fuels through radiation. A backing fire moving downslope is much less effective at preheating fuels. A wind-driven head fire also preheats fuels in its path through radiation because its angled flame front arches over these fuels. Preheating also occurs, in this case, by wind-driven convection as hot air from the fire is blown ahead. The greater rate of spread, all else held equal, results in greater fire-line intensity for head than backing fires. Greater fire intensity, in turn, leads to greater crown scorch or consumption, bore charring, and mortality-causing injury.

Bare Ground

Juniper mortality was predicted to increase with increasing bare ground. Examination of other fitted models containing litter cover as a predictor instead of bare ground indicated mortality increased with decreasing litter cover. These findings are certainly counterintuitive as increasing bare ground and decreasing litter cover would seem to adversely impact the continuity of surface fuels, fire intensity, and the potential to cause tree mortality. There is obviously some complexity here, which requires additional exploration. In most vegetation types, the amount of bare ground was similar between marked trees killed by fire and those that survived. However, in the bunchgrass type, there was greater bare ground at locations of dead than surviving marked trees. What would cause this result? Is the significance of this factor specific to the Breaks study area alone? Examining tree mortality in the bunchgrass type using a GIS display revealed that the importance of bare ground predictor in the final model likely reflects the composited influence of several factors, not just one. Trees killed in the bunchgrass type at locations where the bare ground percentage was greater than the mean for all bunchgrass locations (44.7% ± 27.5% SD), tended to occur on relative steep slopes with shallow, rocky soils and were in close proximity (< 50 m) to stands of bitterbrush. In our study area, rocky divides dissect hillslopes and separate swales of deeper soil. These rocky divides are vegetated by bitterbrush, as is common in the northern Great Basin, and this vegetation can burn rather intensely (Kaitpraneet, 1982). In our study area, the bunchgrass vegetation type can occur at the border fringe of these bitterbrush stands, where soils are typically shallow and sparsely vegetated and there is much bare ground. Juniper trees growing in these fringe areas seem to have been more likely to be killed than trees growing in bunchgrass areas of deeper soil and less bare ground (e.g., upland benches and swales). We speculate that this increased tree mortality rate was due to close proximity of these trees to intensely burning shrub fuels, particularly bitterbrush, despite the tree locations actually being in the bunchgrass type. In fact, of the 11 trees killed in the bunchgrass type at locations where the bare ground percentage was greater than the type mean, 7 trees were within 50 m of bitterbrush stands and the remainder were near mountain big sagebrush vegetation. Trees located near and upslope and/or downwind from intensely burning shrub fuels would potentially receive substantial preheating from radiation and air movement associated with this combustion (Rothermel, 1972). These trees would also be potentially exposed to canopy ignition sources such as fire brands lofted out of burning shrub fuel beds or in some marginal cases, direct ignition by long flame lengths extending up and out of these shrub fuels (NWCG, 2017). Conversely, trees located within the interior of bunchgrass stands associated with deeper soils and less bare ground would tend to be more distant from shrub fuels, intense burning, and associated fire-brand ignition sources. Burning bunchgrass vegetation at these interior locations would likely expose juniper trees to relatively short flame lengths (Albini, 1976; Brown, 1982). Consequently, the likelihood of canopy ignition and fire-induced mortality would tend to be lower for these trees than for those at the border fringe locations with much bare ground. As such, we argue that the influence of the bare ground variable in the top model was probably more about position on the landscape relative to shrub fuels than about continuity of ground fuels.

Sensitivity-Specificity Compromise

While sensitivity of the 4-variable final model was quite high, specificity or the true negative rate was rather low, indicating overestimation of fire-caused mortality rates. Although our sample size limitations did not permit it, future researchers intending to further pursue this preliminary modeling effort will likely find that inclusion of additional predictors, specifically vegetation nearby and forb and/or litter biomass, potentially reduces false-positive predictions and thus increases model specificity compared with what we obtained.

Implications

Not surprisingly, this modeling work confirms the type of vegetation, and thus the type of fuels plays a principal role in determining the efficacy of fall prescribed fire for controlling western juniper on sagebrush-steppe rangelands. While western juniper trees in the bitterbrush and mountain big sagebrush types were much more likely to be killed by fire than trees in the bunchgrass grassland or other types, at this study area, bitterbrush fuels were clearly the most hazardous to western juniper trees. The study area landscape had not burned for more than 63 yr and was thus long overdue for a fire. The landscape was heavily loaded with shrub fuels, which is probably quite typical for encroached, mesic sagebrush rangelands throughout the northern Great Basin. Our model may not be applicable, however, to rangelands where fire has been more recent and/or shrub fuel loading is much lighter. Nevertheless, results from this case study suggest that with just four factors, readily assessed with field and GIS measurements, it is possible to accurately predict fire efficacy and juniper mortality rate, thus providing a more informed basis for prescribed fire planning.

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