LANDSCAPE CHARACTERISTICS OF SAGEBRUSH-STEPPE/ JUNIPER WOODLAND MOSAICS UNDER VARIOUS MODELED PRESCRIBED FIRE REGIMES

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ABSTRACT

Juniper (*Juniperus* spp.) encroachment into adjacent vegetation has been identified as an ecological and management issue for juniper woodlands. This encroachment has resulted in numerous site- and landscape-level ecosystem changes, such as reduced herbaceous biomass production, altered watershed characteristics, reduced site and landscape diversity, and increased soil erosion. The cause of the encroachment includes direct and indirect fire suppression, livestock grazing, and climatic change. Regardless of cause, the use of prescribed fire has been proposed in many ecosystems as a management technique to address juniper encroachment. The effects of prescribed burning programs on the characteristics of juniper woodland–dominated landscapes and their ability to meet long-term ecological and management objectives have not previously been assessed.

Three 6th-order watersheds were selected on the Owyhee Plateau, Idaho, because of the availability of prescribed fire effects data and other supporting data. The watersheds were classified into potential vegetation types and the area of each was classified into current successional stages utilizing Landsat 7 Enhanced Thematic Mapper (ETM+) and digital elevation model data. Landscape dynamics under four different prescribed fire regimes were modeled for each watershed with the Tool for Exploratory Landscape Scenario Analyses (TELSA). The fire regimes studied were prescribed burning levels affecting 0, 2, 5, and 7% of each watershed's area per decade. Ten simulations of the predicted landscape composition for each watershed under each fire regime were analyzed using FRAGSTATS.

Our results suggest that encroachment of western juniper (*J. occidentalis* ssp. *occidentalis*) is likely to continue into the future if no fire or even limited prescribed burning were implemented. Continued encroachment of western juniper is predicted to result in decreased landscape diversity. While the overall mean patch size within the landscapes decreased, mean patch size of juniper-dominated patches increased. This increase, together with decreases in Simpson's evenness indices, indicate a continued successional trajectory of simplification of watershed composition. Prescribed fire can be utilized to maintain landscape diversity or at least minimize the loss of diversity. Fire occurrence on at least 5% of the watersheds per decade will be required to maintain sagebrush-steppe vegetation at current levels.

keywords: FRAGSTATS, Idaho, *Juniperus occidentalis*, landscape diversity, landscape model, successional model, TELSA, western juniper.

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INTRODUCTION

Western juniper (*Juniperus occidentalis*) woodlands dominate approximately 3.6 million ha in the northwestern portion of the Great Basin and southern Columbia Basin (Miller et al. 2005). Western juniper has been classified into two subspecies: *J. occidentalis* ssp. *australis*, which occurs in the Sierra Mountains of California; and *J. occidentalis* ssp. *occidentalis*, which occurs in extreme northern California, eastern Oregon, and southwestern Idaho (Vasek 1966, Burkhardt and Tisdale 1969). Western juniper–dominated areas are considered to be the northwestern equivalent of the Intermountain pinyon–juniper zone (West 1999) but do not have an associated codominant pine (*Pinus* spp.). Widespread encroachment of western juniper appears to have accelerated during the last 150 y (Burkhardt and Tisdale 1969, 1976; Miller and Rose 1994; Miller and Wigand 1994; Miller et al. 2005), which coincides with the period of Euro-American settlement and development of western North America. Research on age-class distributions of western juniper stands indicates that recent expansion began during the period between 1860 and 1880 (Burkhardt and Tisdale 1969, Miller and Rose 1994), slowed during the 1930s and 1940s, and since 1960 has been progressing at an exponential rate (Miller and Rose 1994). The area currently dominated by western juniper in southwestern Idaho has more than doubled since 1960 (Burkhardt and Tisdale 1969).

During the pre–Euro-American settlement period, western juniper is thought to have primarily occurred as dense stands on the more dissected topography or to have occurred as open, savanna-like woodlands on canyon slopes and more regular topography (Burkhardt and Tisdale 1969, 1976; O'Rourke and Ogden 1969; Vasek and Thorne 1977; Miller and Rose 1994; Miller and Wigand 1994). Western juniper has pri-

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marily encroached into many adjacent vegetation types, but the encroachment has been most dramatic in vegetation types with deeper soils (Young and Evans 1981, Eddleman 1987, Miller and Wigand 1994, Miller et al. 1995). These sites include those dominated by mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana*) steppe, aspen (*Populus tremuloides*) woodlands, and Idaho fescue (*Festuca idahoensis*)/ bluebunch wheatgrass (*Pseudoroegneria spicata*) steppe. Encroachment has also occurred into low sagebrush (*Artemisia arbuscula*) steppe; however, the rate has been much lower due to less productive site conditions (Miller et al. 2005).

Causes for encroachment of western juniper are thought to be similar to those of juniper woodlands in general (Bunting 1993, Miller and Wigand 1994). Research has shown that changes in plant competition are probably not a factor in western juniper encroachment because plant composition did not affect rate of establishment (Burkhardt and Tisdale 1976, Eddleman 1987, Miller and Rose 1994). However, heavy utilization of rangelands by livestock could facilitate juniper establishment through secondary effects. The resulting reduction in fine fuel loads due to high forage utilization would decrease fire occurrence (Bunting 1993, Miller et al. 1995) and increase sagebrush seedling establishment (Ellison 1960, Tisdale 1969). Increased sagebrush density provides greater availability of safe sites for juniper seedling establishment (Burkhardt and Tisdale 1976, Eddleman 1987, Miller and Rose 1994, Miller et al. 1995). Miller et al. (2005) found that pre–Euro-American period fire-free intervals (FFI) in western juniper varied depending on local conditions but was usually less than 20 y for woodlands associated with mountain big sagebrush steppe. Young and Evans (1981) estimated, based on the growth rate of young western juniper seedlings, that a fire every 50 y or less would prevent their encroachment in northern California. The encroachment of juniper usually reduces the herbaceous production on the site (Tausch and Tueller 1990) and thereby greatly reduces fire potential (Bunting 1987, Everett 1987). Wildfires may burn only under the most severe weather conditions in dense stands of mature juniper.

Western juniper encroachment into sagebrush steppe or grassland communities decreases herbaceous and shrub biomass production (Vaitkus and Eddleman 1991, Bunting 1993, Miller and Wigand 1994, Miller et al. 1995). A reduction of plant species richness and species diversity has been documented for western juniper and other Great Basin juniper woodlands (Blackburn and Tueller 1970, West et al. 1979, Koniak and Everett 1982, Miller et al. 2000). Bunting et al. (1999) did not find a decrease in species richness. However, species diversity declined due to changes in species equitability of canopy coverage. Sites were increasingly dominated by a single species, western juniper, with associated declines in the canopy coverage of other species.

Studies have shown that fire occurrence can be effective in maintaining early seral vegetation communities associated with juniper woodlands (Miller et al. 2005). However, little information exists on the proportion of fire on the landscape required to stabilize composition or the effects of fires on landscape characteristics. Our study was designed to address these questions for the Owyhee Plateau in southwestern Idaho. We hypothesized that at some level of fire occurrence within the landscape, the current characteristics (i.e., proportions of cover types and landscape metrics) of the watersheds could be maintained into the future.

STUDY AREA

The Upper Owyhee River basin is located within the southern portion of the Owyhee Plateau in southwestern Idaho (lat 42°30′N, long 116°50′W) and was selected for this study because of the abundance of western juniper–dominated vegetation and availability of auxiliary data. Western juniper forms a noncontinuous woodland zone between 1,450 and 2,100 m in elevation within the Owyhee Plateau. The cool, semiarid climate is characteristic of the northern Great Basin, with precipitation varying from 25 to 50 cm. Three 6^{th}-order hydrologic code (HUC^6) watersheds, ranging in area from 6,380 to 6,910 ha, were chosen that had western juniper vegetation on at least 60% of the watersheds' area with a variety of successional stages present. The selected areas include Red Canyon Creek, Smith Creek, and Currant Creek watersheds. Currently, wildfires rarely occur and are suppressed. Prescribed fire has not been commonly applied in these watersheds during the past decade.

METHODS

The potential vegetation type (PVT) classification utilized was developed from a modified classification of the Interior Columbia Basin Ecosystem Management Project (ICBEMP) midscale analysis (Quigley et al. 1996, USDA 1996). PVTs are defined as groupings of habitat types that have similar overstory composition, structure, and environmental requirements, and consequently are broader than habitat types developed for the Columbia Basin. The ICBEMP classification of the watersheds was modified because, for example, many sites that currently supported juniper were not classified as a juniper PVT. We reclassified those areas misclassified to the appropriate PVT taxonomic unit.

Two PVTs of primary interest for this study were defined for the three watersheds: western juniper/ mountain big sagebrush and western juniper/low sagebrush woodland. In addition, the following PVTs that covered a smaller portion of the watersheds were defined in order to create a continuous vegetation layer for the watersheds: western juniper–curl-leaf mountain mahogany (*Cercocarpus ledifolius*) woodland, western juniper woodland/rock, aspen woodland, broadleaf riparian woodland, mountain shrubland, low sagebrush steppe, and meadow.

A successional stage classification was developed for each PVT. The classifications were developed by modifying those previously described by Quigley et

Table 1. Successional stage descriptions of the three most prevalent potential vegetation types found within three watersheds in the Owyhee Plateau, Idaho.

Successional stage	Potential vegetation type	
	Western juniper/mountain big sagebrush	Western juniper/low sagebrush
Herbland	Shrub cover <5%, herbaceous cover <67%	Shrub cover <5%, herbaceous cover <67%
Shrubland	Canopy of low (<50-cm) and/or medium 50- to 200-cm shrubs with a canopy coverage <67%, tree coverage <2%	Canopy of low (<50-cm) shrubs with a canopy coverage <67%, tree coverage <2%. Sometimes very open stands of large mature juniper trees are present.
Woodland initiation (Phase 1)	Canopy (<5%) of usually young, sometimes mid-aged junipers present. Juniper is having only minor effects on competition and environment of the site, and the sagebrush community is intact except directly under juniper trees.	Juniper canopy (<3%) of usually young, sometimes mid-aged junipers present. Juniper is having minor effects on competition and environment of the site, and the sagebrush community is intact except directly under juniper trees.
Open young woodland (Phase 2)	Canopy (5–10%) of young and mid-aged junipers present. Juniper beginning to have an effect on the environment of the site. Sagebrush-steppe species declining and sagebrush skeletons often present.	Canopy (3–8%) of young and mid-aged junipers present. Juniper beginning to have an effect on the plant interspace environment of the site. Sagebrush-steppe species declining and sagebrush skeletons present. However, low sagebrush still common in interspaces.
Young multistory woodland (Phase 3)	Canopy (>10%) of young and mid-aged junipers present. Few or no mature junipers present. Sagebrush skeletons often numerous in understory.	Canopy (>8%) of young and mid-aged junipers present. Usually a few mature junipers present. Sagebrush skeletons often present in understory. However, low sagebrush still common in interspaces.
Mature woodland	Overstory canopy >15%, composed of primarily mature individuals (flat-topped trees and wolf lichen [*Letharia vulpina*] usually present). Few sagebrush remain except in larger openings. Some stands are completely dominated by old mature trees. Other stands may have open canopy of mature trees and a codominant layer of various aged mid-aged junipers.	Overstory canopy >8%, composed of primarily mature individuals (flat-topped trees and wolf lichen usually present). Trees may not necessarily be large sized. Sagebrush usually present in openings except in the denser stands of juniper.

al. (1996). For those areas supporting western juniper/mountain big sagebrush and western juniper/low sagebrush woodland PVTs, successional stages identified included herbland, shrubland, and initiation, open young, young multistory, and mature woodlands with 0, <2, <5, 5–10, >10, and >15% western juniper canopy coverage, respectively (Table 1). Stand initiation, open young, and young multistory woodlands correspond to the Phase 1, Phase 2, and Phase 3 successional stages, respectively, as described by Miller et al. (2005).

Landsat Classification Procedures

The land cover PVT/structural stage map was produced through a supervised classification (Roth 2004) of a Landsat 7 ETM+ image from 2 August 2002. Image preprocessing steps included radiometric correction and georeferencing to U.S. Geological Survey orthophotos. The classification of the Landsat spectral bands was complemented by the use of topographic variables, elevation and slope, and a solar radiation variable. The solar radiation variable was derived from the 30-m digital elevation model through processing in the Solar Analyst software (HEMI Solar Analyst 1999–2000). The image was classified into successional stages of each PVT using a supervised nonparametric nearest neighbor discriminant analysis method (SAS 8.02; SAS Institute, Cary, NC). The overall accuracy of the land cover classification was 72%. Post-processing steps included smoothing of the image through filtering in ArcInfo (ESRI, Redlands, CA) and conversion of the raster image to polygons.

A total of >740 ground control points were collected in the watersheds in 1999–2003. Approximately 50% of the points were used as training points for the classification of satellite imagery, and the remaining points were reserved for accuracy assessment. The PVT and successional stage was recorded for each point. Location of ground reference points were recorded using Magellan (Santa Clara, CA) Global Positioning System units with an estimated maximum spatial error of 15 m. Large (>2 ha) homogeneous areas were selected for the ground control to ensure that the control area was larger than the minimum mapping unit and that the effects of the GPS spatial error would be minimized. Each ground control point was in the center of homogeneous vegetation with a minimum radius of 100 m.

Succession modeling is critical in order to predict broadscale long-term landscape trends (Jensen and Bourgeron 1993, Kaufmann et al. 1994). The Tool for Exploratory Landscape Scenario Analyses (TELSA; Version 3.3), a spatially explicit, deterministic succession modeling tool with stochastic properties, was utilized to predict future landscape compositions that would result from variety of fire management strategies (Kurz et al. 2000*a,b*; http://www.essa.com/downloads/telsa/index.htm). Changes in future landscape composition given different land management strategies were modeled using TELSA (Klenner et al.

2000). The model predicts post-fire succession from a process developed by Kessell and Fischer (1981). This approach links seral vegetation change along a multiple pathway model of successional development as initially suggested by Noble and Slatyer (1977). The length of time a given landscape unit is occupied by a particular successional stage is determined by lifespan and recruitment strategy of the dominant canopy species. This approach has been successfully used to model successional change in many forested vegetation types on the Northern Rocky Mountains (Crane and Fischer 1986, Fischer and Bradley 1987, Bradley et al. 1992, Klenner et al. 2000).

Succession was modeled on an individual polygon scale as a change in successional stage within each PVT. Following the initial image classification, each patch was tessellated into irregular-shaped polygons that were approximately 1 ha in area. The initial conditions for each polygon are set utilizing information on the PVT and the current successional stage. The occurrences of human- and natural-caused disturbances are stochastically simulated at the polygon level from a probability matrix. Disturbances result in an immediate change in successional stage, with a corresponding adjustment of the successional clock. Disturbance probabilities can be modified to assess landscape changes that would be affected by various fire management strategies (Klenner et al. 2000; Kurz et al. 2000*a*,*b*). Probability of the size class of wildfires and prescribed fires were defined and randomly assigned to each fire. The majority of wildfires were <1 ha (90%), while the majority of prescribed fires were 100–1,000 ha in area (70%). In the modeling process, tessellated polygons were randomly selected for burning based on the predetermined fire disturbance probability function assigned to each cover type–PVT combination. If a polygon was randomly selected for burning by either wildfire or prescribed fire, the fire was randomly assigned an area based on the predetermined fire size probability function. Bare rock or cover types such as those with shallow soils that do not burn were given a zero disturbance probability and thus did not burn in this process.

Future landscape compositions of the 3 HUCs were evaluated for 50, 100, and 200 y into the future. Fire management regimes assessed for each watershed included

1) Current fire management (suppressed wildfire only).
2) Use of prescribed fire to treat a combined average of 2% of the watershed area/decade focusing on Phase 1 and Phase 2 woodland successional stages.
3) Use of prescribed fire to treat an annual combined average of 5% of the watershed area/decade focusing on Phase 1 and Phase 2 woodland successional stages.
4) Use of prescribed fire to treat an annual combined average of 7% of the watershed area/decade focusing on Phase 1 and Phase 2 woodland successional stages.

Using a Monte Carlo approach, TELSA was run 10 times for each management regime per watershed. Fire occurrence was applied at random within each PVT and phase combination for each watershed. Means and variances were then calculated from these results.

Landscape metrics for the current and predicted vegetations were calculated using FRAGSTATS 3.3 (McGarigal and Marks 1994). This software performs landscape analysis using a number of indices. Four indices were selected that reflect the different types of changes in landscape composition and pattern. The indices selected to quantify the landscape changes for this study were Simpson's evenness index (SEI), mean patch size (MPS), contagion (C), and interspersion–juxtaposition index (IJI). The SEI is a measure of community diversity that is determined by abundance of each patch type within a watershed and decreases as a patch type becomes more dominant within a watershed. The SEI was selected rather than other diversity indices such as Simpson's diversity index or the Shannon–Wiener diversity index because community richness did not change over the modeled time periods. The SEI is a more sensitive measure of changes in the proportion of landscape cover types. The MPS is an average size of all patch types within a landscape. It averages the effects of both the aggregation of patches through succession and the creation of new patch types through disturbance. Patch size can have important effects on the modification of the local environment and the selection of habitat by many species. The C is related to the distribution of each patch type within the watershed and decreases as the spatial distribution of a patch type becomes more aggregated. This metric is affected by both disturbance patterns and more constant factors such as the distribution of soil type, geological parent materials, and aspect. The IJI is a measure of the randomness of patch type adjacency. IJI decreases as patch type adjacencies (cover type edges) become less random and as a patch type is associated more frequently with other patch types. The distribution of different types of edges influences some species' habitat selection and the degree of microclimate modification by edges. Together these four metrics measured the effects of disturbance on cover type diversity, mean patch size, patch type distribution, and diversity of edge types. Significant differences between management strategies were detected with analysis of variance and Tukey's pairwise mean comparisons test ($P \leq 0.05$) (SYSTAT 10.0; SPSS, Chicago, IL).

RESULTS

Under current fire management, western juniper woodlands were predicted to continue maturing as has been noted in previous field studies (Burkhardt and Tisdale 1969, Miller and Wigand 1994, Miller et al. 2005). Mature western juniper stages were predicted to increase about 30% in area over the next 100 y on both mountain big sagebrush and low sagebrush sites (Table 2). Phase 3 woodlands were predicted to in-

Table 2. Total predicted area (ha) in mountain big and low sagebrush steppe and Phase 1 juniper woodland cover types for three watersheds in the Owyhee Plateau, Idaho, under various fire management regimes (mean ± SD).

Watershed	Management regime	Time period			
		Present	50 y	100 y	200 y
Currant Creek[a]	Current fire management	1,087	740 ± 19 a[b]	714 ± 17 a	517 ± 11 a
	Prescribed fire (2%/decade)	1,087	900 ± 106 b	858 ± 109 b	636 ± 81 a
	Prescribed fire (5%/decade)	1,087	972 ± 185 b	1,108 ± 173 c	896 ± 209 b
Red Canyon Creek	Current fire management	1,908	445 ± 8 a	426 ± 9 a	305 ± 5 a
	Prescribed fire (2%/decade)	1,908	600 ± 12 b	504 ± 23 b	343 ± 15 b
	Prescribed fire (5%/decade)	1,908	1,188 ± 23 c	936 ± 27 c	900 ± 29 c
	Prescribed fire (7%/decade)	1,908	1,539 ± 28 d	1,194 ± 38 d	1,321 ± 178 d
Smith Creek	Current fire management	1,247	459 ± 7 a	395 ± 12 a	343 ± 13 a
	Prescribed fire (2%/decade)	1,247	647 ± 152 ab	470 ± 110 a	205 ± 47 a
	Prescribed fire (5%/decade)	1,247	832 ± 387 b	1,106 ± 367 b	837 ± 275 b
	Prescribed fire (7%/decade)	1,247	1,700 ± 312 c	1,848 ± 383 c	1,396 ± 255 c

[a] The low abundance of early successional stages (sagebrush steppe and Phase 1) communities did not permit modeling the 7%/decade fire management regime in this watershed.
[b] Mean values within a watershed/time period followed by the same letter are not significantly different ($P < 0.05$).

crease 200% and 120%, respectively, for mountain big and low sagebrush site over the next 100 y. The increased area of later successional stages was derived from earlier successional stages, such as sagebrush steppe, and Phase 1 and Phase 2 woodlands as they continue to advance in successional age. This process resulted in a reduction of SEI as the watersheds become increasingly dominated by later successional stages (Table 3). The MPS was predicted to decrease (Table 4) and C was predicted to increase (Table 5). In 100 y, sagebrush-dominated stages were predicted to nearly disappear from the watersheds. The decline of mountain big sagebrush communities was predicted to be greater than that of the low sagebrush-dominated communities because of faster rates of succession associated with that PVT.

Fire is frequently suggested as a potential management tool to counteract the influence of woodland encroachment (Wright and Bailey 1982, Miller et al. 2005). We compared the efficacy of using current management to three different levels of prescribed fire to maintain sagebrush-dominated vegetation. We simulated burning approximately 2, 5, and 7% of each watershed per decade, focusing burning the early stages of woodland succession (Phases 1 and 2). The 2% level was found to be inadequate to maintain the current amount of sagebrush steppe on the landscape (Table 2). Woodland continued to develop at the expense of sagebrush steppe. The 5% level of fire was adequate in maintaining low sagebrush steppe for all watersheds within the next 100 y. The 5% level of fire was also adequate in maintaining mountain big sagebrush steppe for the Smith Creek and Currant Creek watersheds. However, for Red Canyon Creek, which currently has 21% of mountain big sagebrush steppe, more than 7% per decade will be required to be burned. The 7% level resulted in a reduction in mountain big sagebrush steppe in the Currant Creek watershed. This decrease occurred because, at this level, the non-woodland areas were being burned so frequently that sagebrush steppe did not have adequate time to develop. Thus, grassland vegetation increased in the watersheds. In the long term (>200 y), Phase 3 woodlands continued to increase under all management regimes. This increase was due to the inability to burn all Phase 2 woodland areas. These areas advanced through succession into Phase 3 woodlands that were not subject to the prescribed fire regimes.

The changes in SEI and C through time for the four regimes were fairly consistent, although the mag-

Table 3. Predicted Simpson's evenness index for three watersheds in the Owyhee Plateau, Idaho, under various fire management regimes for three future time periods (mean ± SD).

Watershed	Management regime	Time period			
		Present	50 y	100 y	200 y
Currant Creek[a]	Current fire management	0.927	0.921 ± <0.001 a[b]	0.912 ± 0.001 a	0.899 ± 0.001 a
	Prescribed fire (2%/decade)	0.927	0.921 ± 0.001 ab	0.914 ± 0.002 b	0.894 ± 0.002 b
	Prescribed fire (5%/decade)	0.927	0.919 ± 0.002 ac	0.919 ± 0.002 c	0.894 ± 0.004 b
Red Canyon Creek	Current fire management	0.918	0.900 ± <0.001 a	0.827 ± 0.001 a	0.829 ± 0.001 a
	Prescribed fire (2%/decade)	0.918	0.912 ± 0.001 b	0.853 ± 0.002 b	0.830 ± 0.001 a
	Prescribed fire (5%/decade)	0.918	0.931 ± 0.001 c	0.912 ± 0.001 c	0.875 ± 0.001 b
	Prescribed fire (7%/decade)	0.918	0.928 ± 0.001 d	0.925 ± 0.001 d	0.895 ± 0.001 c
Smith Creek	Current fire management	0.957	0.910 ± 0.001 a	0.880 ± 0.003 a	0.881 ± 0.001 a
	Prescribed fire (2%/decade)	0.957	0.922 ± 0.001 b	0.889 ± 0.010 a	0.873 ± 0.005 a
	Prescribed fire (5%/decade)	0.957	0.936 ± 0.001 c	0.926 ± 0.032 b	0.907 ± 0.013 b
	Prescribed fire (7%/decade)	0.957	0.938 ± 0.001 c	0.938 ± 0.023 c	0.919 ± 0.008 c

[a] The low abundance of early successional stages (sagebrush steppe and Phase 1) communities did not permit modeling the 7%/decade fire management regime in this watershed.
[b] Mean values within a watershed/time period followed by the same letter are not significantly different ($P < 0.05$).

Table 4. Predicted mean patch size (ha) for three watersheds in the Owyhee Plateau, Idaho, under various fire management regimes for three future time periods (mean ± SD).

		Time period			
Watershed	Management regime	Present	50 y	100 y	200 y
Currant Creek[a]	Currant fire management	12.9	10.7 ± 0.1 a[b]	9.9 ± 0.2 a	9.4 ± 0.1 a
	Prescribed fire (2%/decade)	12.9	9.7 ± 0.6 b	9.2 ± 0.4 b	9.7 ± 0.6 ab
	Prescribed fire (5%/decade)	12.9	10.1 ± 0.4 b	9.1 ± 0.3 b	9.2 ± 0.5 ac
Red Canyon Creek	Current fire management	12.4	10.7 ± 0.1 a	10.7 ± 0.1 a	9.2 ± 0.1 a
	Prescribed fire (2%/decade)	12.4	8.9 ± 0.2 b	9.0 ± 0.1 b	9.1 ± 0.2 a
	Prescribed fire (5%/decade)	12.4	7.3 ± 0.1 c	6.2 ± 0.1 c	6.7 ± 0.2 b
	Prescribed fire (7%/decade)	12.4	7.7 ± 0.1 d	6.1 ± 0.1 c	5.8 ± 0.1 c
Smith Creek	Current fire management	10.6	10.7 ± 0.1 a	10.8 ± 0.3 a	10.4 ± 0.3 a
	Prescribed fire (2%/decade)	10.6	9.5 ± 1.2 b	10.2 ± 1.0 a	12.4 ± 0.8 b
	Prescribed fire (5%/decade)	10.6	7.9 ± 1.1 c	7.4 ± 1.0 b	8.1 ± 1.4 c
	Prescribed fire (7%/decade)	10.6	8.3 ± 0.9 c	7.0 ± 0.5 b	6.9 ± 0.7 d

[a] The low abundance of early successional stages (sagebrush steppe and Phase 1) communities did not permit modeling the 7%/decade fire management regime in this watershed.
[b] Mean values within a watershed/time period followed by the same letter are not significantly different ($P < 0.05$).

nitude of change varied by watershed (Tables 3, 5). The SEI decreased steadily though 100 y for all regimes. The decrease was greatest for the current management and 2% regimes. The lowest amount of decrease was predicted for the 7% regime. However, the low amount of Phase 1 and Phase 2 woodlands currently present in Currant Creek did not permit burning 7%. The C increased for the current management and 2% regimes up through 100 y into the future. It did not change greatly for any watershed under the 5% and 7% regimes.

The IJI was the most inconsistent pattern metric analyzed (Table 6). It increased for all fire management regimes at 50 y in Red Canyon Creek but remained relatively unchanged in Currant Creek and Smith Creek during this time period. At 100 y in the future IJI increased in Red Canyon, decreased in Currant Creek, and remained constant in Smith Creek under the 5% regime.

The MPS was predicted to decrease over the 100-y period for all watersheds with increasing fire occurrence (Table 4). The lowest decrease was predicted for Smith Creek. Smith Creek had the smallest initial patch size (10.5 ha) that may have influenced this effect.

DISCUSSION AND MANAGEMENT IMPLICATIONS

Results of this study suggest that encroachment of western juniper is likely to continue and will have significant effects on diversity and vegetation pattern of these watersheds. Under current fire management policy, SEI and IJI will continue to decline and C will continue to increase as juniper woodland becomes more dominant. Both mountain big sagebrush and low sagebrush steppe will decline to low levels as encroachment of juniper onto these sites continues. Rapid successional rates associated with more productive mountain big sagebrush PVT will result in <100 ha of this habitat available in each watershed within 100 y. Increased dominance of mature juniper woodland will influence future fire regimes, watershed characteristics and succession patterns, and increase the habitat for woodland species, particularly primary and secondary cavity-nesting animals. Juniper dominance will also have major negative implications for sagebrush steppe–dependent species such as the sage thrasher (*Oreoscoptes montanus*), sage sparrow (*Amphispiza belli*), and Brewer's sparrow (*Spizella breweri*).

Table 5. Predicted contagion index for three watersheds in the Owyhee Plateau, Idaho, under various fire management regimes for three future time periods (mean ± SD).

		Time period			
Watershed	Management regime	Present	50 y	100 y	200 y
Currant Creek[a]	Currant fire management	49.0	48.7 ± 0.1 a[b]	50.5 ± 0.2 a	51.8 ± 0.2 a
	Prescribed fire (2%/decade)	49.0	48.4 ± 0.4 a	49.9 ± 0.6 b	52.6 ± 0.6 b
	Prescribed fire (5%/decade)	49.0	49.1 ± 0.7 a	48.7 ± 0.5 c	52.4 ± 1.0 b
Red Canyon Creek	Current fire management	49.4	50.2 ± 0.1 a	55.8 ± 0.1 a	55.6 ± 0.2 a
	Prescribed fire (2%/decade)	49.4	47.6 ± 0.2 b	52.5 ± 0.2 b	55.5 ± 0.1 a
	Prescribed fire (5%/decade)	49.4	44.9 ± 0.2 c	45.7 ± 0.1 c	49.3 ± 0.2 b
	Prescribed fire (7%/decade)	49.4	46.4 ± 0.1 d	44.2 ± 0.4 d	47.0 ± 0.4 c
Smith Creek	Current fire management	44.9	49.5 ± 0.1 a	52.9 ± 0.2 a	53.4 ± 0.3 a
	Prescribed fire (2%/decade)	44.9	47.6 ± 1.8 ac	51.9 ± 1.6 ab	55.3 ± 0.9 a
	Prescribed fire (5%/decade)	44.9	45.5 ± 2.4 bc	46.5 ± 2.1 bc	49.2 ± 2.5 b
	Prescribed fire (7%/decade)	44.9	46.0 ± 1.8 bc	45.1 ± 1.0 bc	47.4 ± 1.7 b

[a] The low abundance of early successional stages (sagebrush steppe and Phase 1) communities did not permit modeling the 7%/decade fire management regime in this watershed.
[b] Mean values within a watershed/time period followed by the same letter are not significantly different ($P < 0.05$).

Table 6. Predicted interspersion–juxtaposition index for three watersheds in the Owyhee Plateau, Idaho, under various fire management regimes for three future time periods (mean ± SD).

Watershed	Management regime	Time period			
		Present	50 y	100 y	200 y
Currant Creek[a]	Currant fire management	78.9	80.2 ± 0.3 a[b]	77.0 ± 0.3 a	75.6 ± 0.3 a
	Prescribed fire (2%/decade)	78.9	80.1 ± 1.0 a	77.0 ± 1.1 a	74.2 ± 1.5 b
	Prescribed fire (5%/decade)	78.9	79.2 ± 1.3 a	79.5 ± 1.0 b	74.4 ± 1.4 b
Red Canyon Creek	Current fire management	75.5	76.9 ± 0.3 a	73.4 ± 0.3 a	75.1 ± 0.4 a
	Prescribed fire (2%/decade)	75.5	78.9 ± 0.2 b	75.5 ± 0.5 b	74.9 ± 0.4 a
	Prescribed fire (5%/decade)	75.5	79.1 ± 0.3 b	81.2 ± 0.2 c	79.0 ± 0.3 b
	Prescribed fire (7%/decade)	75.5	77.6 ± 0.4 c	83.1 ± 0.8 d	79.7 ± 0.8 c
Smith Creek	Current fire management	78.0	77.0 ± 0.2 a	72.6 ± 0.3 a	72.0 ± 0.5 a
	Prescribed fire (2%/decade)	78.0	77.4 ± 2.2 a	72.0 ± 2.4 a	67.9 ± 1.3 b
	Prescribed fire (5%/decade)	78.0	77.5 ± 2.1 a	76.8 ± 1.0 b	74.9 ± 2.6 c
	Prescribed fire (7%/decade)	78.0	77.3 ± 1.8 a	78.2 ± 2.1 b	75.7 ± 2.7 c

[a] The low abundance of early successional stages (sagebrush steppe and Phase 1) communities did not permit modeling the 7%/decade fire management regime in this watershed.
[b] Mean values within a watershed/time period followed by the same letter are not significantly different ($P < 0.05$).

In the three watersheds studied, prescribed fire could be implemented to alleviate this trend. Prescribed burning at least 5% of each watershed per decade is necessary to maintain the current level of low sagebrush steppe currently present. The amount of fire required to maintain mountain big sagebrush steppe is more variable among watersheds and dependent upon the proportion currently present. Generally application of 5–7% of the area per decade was adequate. However, in Red Canyon Creek, the watershed with the greatest proportion of mountain big sagebrush steppe, prescribed fire at the 7% level was not able to maintain current conditions.

The modeled prescribed fire was focused on the Phase 1 and Phase 2 woodland cover types. It was evident that this approach would not sustain either low or mountain big sagebrush steppe over the long term (>200 y). With this approach, there was a slow but continuous loss to Phase 3 communities that were then precluded from prescribed fire activity. Phase 3 woodlands were modeled with a low prescribed fire probability because they only burn under the most severe fire conditions. The level of prescribed fire in Phase 1 and Phase 2 community types only slowed the rate of decline. The solution appears to be either increasing the area of Phase 2 woodlands burned per decade or treatment of Phase 3 woodlands so they can be returned to earlier sagebrush-steppe portions of the successional sequence. However, Phase 3 communities are more difficult to prescribe burn due to low fine fuel loading and require greater amounts of time to recover (Miller et al. 2000, 2005). Other land treatments, such as mechanical removal of juniper, may have to be considered for Phase 3 woodland sites.

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