Project Title: Historic fire frequency in mountain big sagebrush communities of the eastern Great Basin and Colorado Plateau: A comparison of estimates based upon proxy fire scar records and predictions derived from post-fire succession rates

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I. Abstract

Knowledge of past fire regimes associated with mountain big sagebrush-dominated landscapes is inadequate for scientifically-based land management that requires assessment of departures from historic conditions. Widely utilized estimates of fire frequency for sagebrush ecosystems are largely based upon few studies using fire-scarred proxy trees positioned at the forest/shrubland ecotone. These studies, all conducted in the northern half of the species distribution, generally fail to adequately address questions of fire behavior across the fuels threshold at the forest/woodland-shrubland ecotone. Alternatively, post-fire rates of succession have been used to suggest fire frequencies compatible with big sagebrush recovery. Minimum and maximum fire-free intervals can be inferred based upon the time required for big sagebrush population recovery and succession to tree dominance, respectively. Published studies of mountain big sagebrush post-fire recovery are also limited primarily to higher latitudes and seldom consider the long timeframes required for trees to establish in burned areas on semi-arid landscapes. We had three objectives to address these deficiencies:

(1) We developed estimates of historical fire frequency for mountain big sagebrush communities at 10 sites in the eastern Great Basin, upper Colorado Plateau and intervening mountains and highlands (southern half of the species distribution) using fire chronologies from proximal fire-scarred trees. Proxy-derived estimates were evaluated based upon proximity of fire-scarred trees to mountain sagebrush communities. We assessed the likelihood that these estimates accurately reflect historic fire frequency for mountain sagebrush communities across the study region.

(2) We developed field-based estimates of recovery rate for mountain big sagebrush using a chronosequence approach (n = 27 burn sites of known fire year distributed throughout western and central Utah). Recovery time extrapolated from regression models averaged 37 years. The precipitation regime immediately following the fire year proved the single most important predictor of recovery. Recovery rate was positively associated with precipitation in the summer before fire and also with course textured soil.

(3) We developed a grid-based model of fire and sagebrush recovery via seed dispersal, establishment probability, and time to reproductive maturity to simulate the predicted long-term response of mountain big sagebrush to alternative fire regime scenarios. Model scenarios included a range of fire frequencies and fire extent. The response variable of interest was the percent of the virtual landscape dominated by mountain big sagebrush. Model outputs were highly sensitive to mountain big sagebrush life history trait parameters. Dispersal distance and time to reproductive maturity parameters were evaluated for their influence on model output. Slow rates of recovery were associated with the shortest dispersal distance (10 m) and the longest time to reproductive maturity (4 yrs). Fire regimes characterized by small fire sizes (e.g. 2 ha) are analogous to fire regimes with larger fires that are patchy and that leave unburned

islands every 2 ha with a viable seed source. In this regard it was redundant to model establishment from seedbank that survived the fire as this would effectively reduce the influence of fire size on persistence of sagebrush—our main response of interest. Under fire rotations commonly cited as typical of mountain big sagebrush (i.e. 30-80 years) (Baker 2006, 2011), simulations revealed that long-term convergence on 20% composition or greater is only compatible with small fire sizes, smaller than 5 ha for 30 yr rotations and smaller than 30 ha for 80-yr fire rotations.

II. Background and Purpose

Mountain big sagebrush distribution extends from southern British Columbia to northern Arizona and from California to the Dakotas. Across the southern half of this distribution it occurs on foothills, dry mountain slopes and ridges at elevations of 1,900 to 3,000 m. At higher elevations it occurs in forest openings of various sizes in association with quaking aspen and numerous coniferous tree species. At lower elevations mountain big sagebrush dominates many treeless landscapes and co-exists in shrubland/woodland mosaics with pinyon pines, Rocky Mountain, Utah and western junipers, and Gambel oak. Species composition and abundance of co-occurring shrubs and herbs vary with topography, soils, and disturbance history.

Although extreme weather, insects, and disease play a role in mountain big sagebrush population dynamics, fire is the dominant disturbance process in natural populations (Wright and Bailey 1982). Mountain big sagebrush plants are easily top-killed by fire and do not re-sprout from roots or crown. Population recovery can be relatively rapid when new plants establish from seeds that remain viable in the soil after burning (Wambolt et al. 1999, Wambolt et al. 2001). Recovery is delayed when dependent upon seed migration from unburned portions of the landscape (Welch 2005). Co-occurring shrubs and herbs vary in their ability to tolerate fire and in their strategies for post-fire recovery. Young coniferous trees are fire sensitive and are slow to re-colonize burned landscapes. Tree invasion is generally delayed until well after shrub recovery. On landscapes prone to tree invasion, mountain big sagebrush dominance is facilitated by fire-free intervals long enough to allow for big sagebrush recovery and short enough to prevent forest or woodland invasion and dominance (Miller & Rose 1999). The distribution of the mountain big sagebrush vegetation type is therefore largely dependent upon the spatial and temporal variation of fire frequency (Fig. 1).

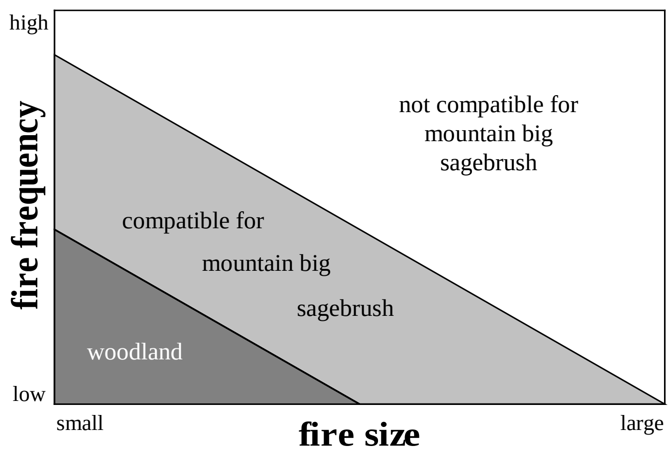


Figure 1. Mountain big sagebrush fire regime compatibility.

Estimates of historic fire frequency for mountain big sagebrush landscapes have been developed from fire-scarred trees growing in isolated forest pockets or near the forest-shrubland ecotone. Houston (1973) employed this approach to develop estimates of pre-1900 fire frequency for the sagebrush steppe of northern Yellowstone National Park. He calculated single-tree mean fire interval (MFI) values of 32 to 70 years. He considered this an overestimate of true MFI and estimated true MFI values to be 20 to 25 years. Arno and Gruell (1983) used a similar approach for forest-mountain big sagebrush shrubland ecotones of southwest Montana. They calculated pre-1900 MFI values of 41, 45, and 74 years for moist, dry, and hot-dry habitats. These estimates were adjusted to 35 to 40 years based on the assumption of probable missing fire evidence. Miller and Rose (1999) estimated MFI for a south central Oregon study using proxy fire-scarred ponderosa pine trees clusters imbedded in a mountain big sagebrush steppe landscape. Composite, pre-1900 MFI ranged from 12 to 15 years for three of the four clusters. Fires burned synchronously in three to four clusters during seven years between 1650 and 1880, resulting in an approximate MFI for these major fires of 38 years. The probability of replicating similar MFI values near forest – shrubland ecotones in the eastern Great Basin or upper Colorado Plateau was unknown. This study helps to fill this information void (project component 1).

Alternatively, historic fire frequency for mountain big sagebrush communities can be estimated indirectly from post-fire succession rates. We hypothesize that minimum and maximum fire intervals should correlate with the time required for recovery of the shrub component and for conifer invasion and dominance, respectively. Studies from Montana and Idaho sites have assessed changes in plant density and cover as measures of mountain big sagebrush recovery after fire (Harniss & Murray 1973, Humphrey 1984, Wambolt et al. 1999, Wambolt et al. 2001). Taken together, these studies suggest that mountain big sagebrush stand recovery requires fire-free intervals of 20 to 35+ years under favorable conditions and much longer intervals when conditions dictate a slower pace of recovery. Maximum intervals may be in the order of three to five times as long as minimum intervals (Miller & Tausch 2001).

Historic mountain big sagebrush fire frequency estimates based upon fire-scarred proxy tree chronologies have the advantage of temporal precision over long time periods. Unfortunately, sufficient quantities of suitable proxies are not available on many mountain big sagebrush landscapes. In addition, fuel matrices for forests and shrublands differ categorically, creating a fuels threshold at the forest – shrubland ecotone. Current understanding of fire behavior across this threshold is inadequate to accurately predict fire spread between forest and shrubland under historical, or even modern, conditions. Such difficulties lead to considerable uncertainty regarding the utility of this method for estimating historic fire frequency for mountain big sagebrush landscapes. We test the appropriateness of mountain big sagebrush fire frequency estimates generated from proxy tree fire chronologies by using simulation models to analyze, in a spatially-explicit sense, the implications of combining proxy tree-based estimates of fire frequency with observed rates of shrubland recovery and tree invasion. Results of these tests begin to address the question of fire behavior across the forest – shrubland ecotone.

III. Study Description and Location

Location

Post-fire sagebrush recovery study sites were co-located with proxy fire history sites and independently in the eastern Great Basin (Nevada and Utah), Colorado Plateau (Utah) and intervening highlands ranging between 1858 to 2810 m in elevation (Figure 2). These regions include high plateaus, canyons and mountains and big sagebrush comprises a dominant component of the landscape matrix for different communities distributed along a moisture and temperature gradient. Desert shrublands with Wyoming big sagebrush (*Artemisia tridentata* ssp. wyomingensis) and grasslands occur at lower elevations; pinyon (*Pinus edulis* and *P. monophylla*) and juniper (*Juniperus scopulorum* and *J. osteosperma*) occur at intermediate elevations with both mountain big sagebrush and Wyoming big sagebrush; Gambel oak (*Quercus gambelii*) and mountain mahogany (*Cercocarpus montanus* and *C. ledifolius*) occur in lower to mid elevations; ponderosa pine (*Pinus ponderosa*) occurs in drier montane forests and Douglas-fir (*Pseudotsuga menziesii*) in wetter sites; aspen occurs patchily in upper elevation zones; and subalpine forests consist of Engelmann spruce (*Picea engelmannii*) and fir (*Abies* spp.) above.

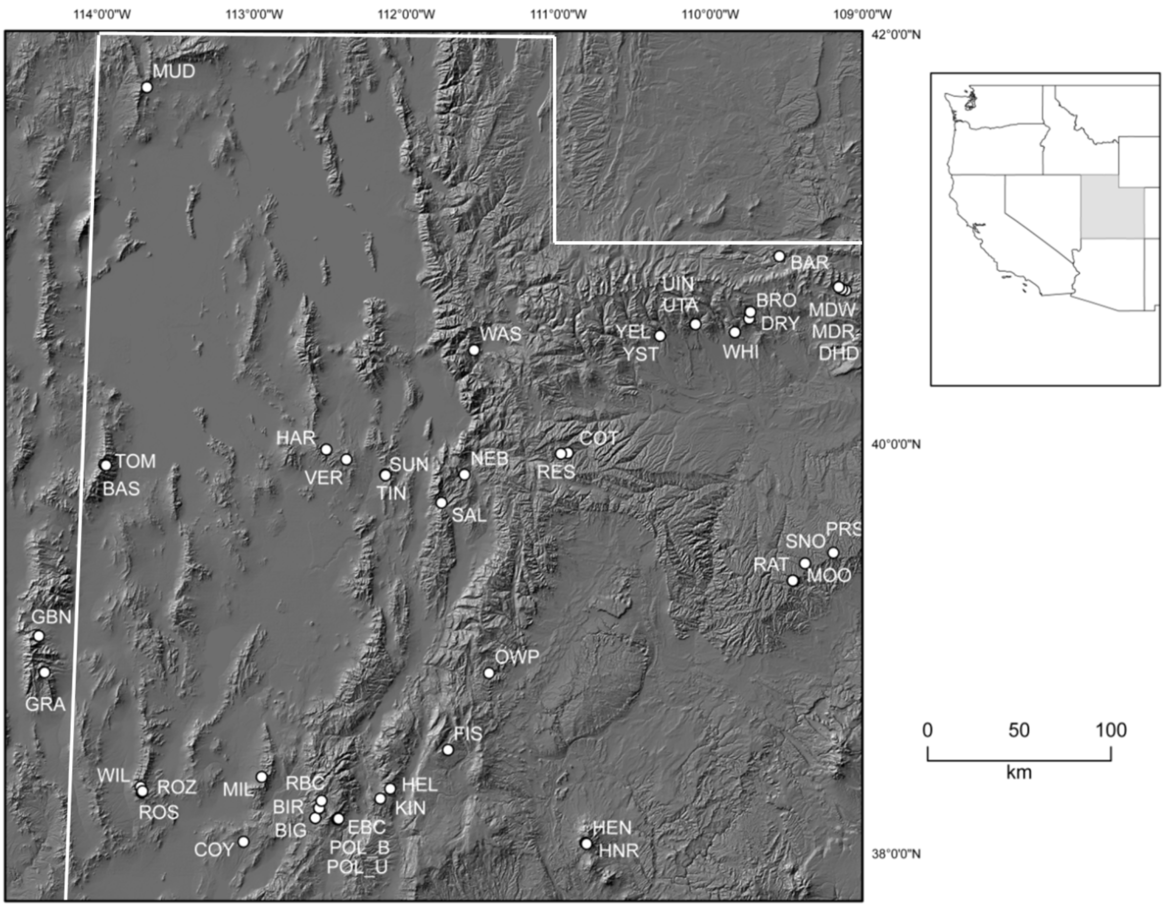


Figure 2. Locations of study sites in Utah and eastern Nevada

A. Study description: Historic Fire Frequency (Project component 1)

Partial cross-sections of 190 fire-scarred trees (stumps, logs and living) were assembled from 10 locations across the study area, four of which had not been sampled previously (Table 1). Sampled trees were primarily ponderosa pine with incidental numbers of Douglas fir, pinyon pine, limber pine, and Rocky Mountain juniper. For six sites, fire chronologies from samples collected as part of JFSP Project 01C-3-3-22 (Heyerdahl et al. 2011) were augmented by additional samples collected from fire-scarred trees selected for their proximity to sagebrush steppe communities. All samples were stabilized and surfaced using a band saw and belt sanders until individual cell structure was clearly visible under magnification.

Each cross-section was cross-dated using a combination of local master-ring chronologies and lists of indicator years. Calendar years were assigned to each fire scar based upon the location of the injury in the tree ring sequence. The earliest fire year was 1340 (ROS) and the most recent fire year was 1999 (BAR). Fire chronologies were developed for each tree individually. In order to minimize the probability of errors of omission or false inclusion, constrained composite chronologies were developed when contributing trees (2-16) were located within a circular area of no greater than 0.5 ha (40 m radius; Kitchen 2010). Fire frequency statistics including mean fire interval (MFI) and minimum and maximum fire-free intervals associated with 20 individual trees and 46 composite chronologies were treated as estimates of point fire frequency across all study areas. Average chronology length (period from first to last recorded fire) was 246 years (range 102-511 years).

Table 1. Mountain sagebrush fire history study sites and basic fire chronology information.

site name	site code	location	first and last fire years	number of fire chronologies by sagebrush proximity class				
				SB Imbed	SB/For Eco	For near SB	For dist SB	For, 2-sided
Bare Top	BAR	Eastern Uinta Mtns	1640-1999		2			
Beaver Creek	RBC	Tushar Mtns	1508-1898			3	5	1
Brownie Creek	BRO	Eastern Uinta Mtns	1419-1871	1		5	5	2
Great Basin NP	GBN	South Snake Range	1525-1907			9	5	1
Henry	HEN	Henry Mtns	1449-1928		2	1		
Old Woman Plateau	OWP	Wasatch Plateau	1520-1909		2	7		1
Rose Spring	ROS	South Wah Wah Range	1340-1864	6	1	7		
Salina Creek	SAC	Wasatch Plateau	1594-1872		1			
Snowshoe	SNO	East Tavaputs Plateau	1682-1971		2	1		
Tom's Creek	TOM	Deep Creek Range	1751-1916			1		
Total				7	10	34	15	5

Knowledge of the location of fire-scarred trees in relation to mountain sagebrush dominated communities is critical in assessing the value of derived fire frequency statistics to non-forested areas due to the unknown probabilities for fire crossing the forest-shrubland ecotone. Sampled trees and composite tree clusters and their corresponding fire chronologies were classified based upon their proximity to sagebrush communities as; 1) imbedded (surrounded by sagebrush steppe vegetation), 2) ecotonal (forested within 10 m of the apparent forest/woodland- shrubland interface), forested-near ecotonal (> 10 and ≤ ~500 m of the forest/woodland-shrubland ecotone), and forested (> 500 and < 1000 m of ecotone). A separate set of chronologies and fire statistics were generated for four sites which included only those years in which fire was recorded on trees located on opposing sides of sagebrush-dominated portions of the landscape. These more conservative estimates were developed based upon the assumption that if fire burned on two opposing sides of the sagebrush community there is a higher probability that it also burned through at least a portion of the intervening shrubland than if it had only burned on one side. Fire frequency statistics were then compared to time estimates for sagebrush recovery available from the literature and those generated in components 2 and 3 of this study.

A. Study description: Sagebrush recovery (Project component 2)

Sampling

We stratified sampling of historic fires between the Colorado Plateau, Great Basin, and intervening mountains. We selected ten sites from the Colorado Plateau. Time since fire for these sites ranged from three to 36 years and mean annual precipitation (± 95% CI) was 497 ± 43 mm while growing season (May-September) precipitation averaged 43 ± 7 mm. We selected nine sites from the Great Basin with time since fire ranging from 6 to 19 years and annual precipitation was 499 ± 37 mm with growing season precipitation averaging 36 ± 4 mm. We selected eight sites were from the Utah Highlands with time since fire ranging from 11 to 32 years, annual precipitation averaging 552 ± 63 mm and growing season precipitation averaging 36 ± 5 mm.

Field Methods

In 2007 and 2008, we collected cover data for mountain big sagebrush at 27 sites both within and outside of the burn perimeter. We used a 100-m line-intercept method to quantify cover of woody species directly beneath the transect line (Boyd et al. 2007). We defined mountain big sagebrush recovery as canopy cover from the burned transect divided by canopy cover recorded in an unburned control transect (sensu Lesica et al. 2007). This ratio scales recovery rate to site potential, allowing comparison across different site environments. Up to ten mature sagebrush plants at each transect were aged using standard dendrochronological approaches, to determine the time interval between the fire year and initial post-fire sagebrush establishment. We measured environmental variables in the field and derived climate variables from the PRISM data set (Daly et al. 1994). See Appendix 1 for a complete description of environmental variables used in the recovery analysis and their hypothesized effects on recovery time.

Data Analysis

We used Akaike's information criterion corrected for small sample size, AIC_c, to rank and weight a set of linear regression models representing our hypotheses for environmental influence on recovery rate (Akaike 1974, Burnham & Anderson 2002).

B. Study description: Simulation model (project component 3)

Overview

We simulated the landscape-level persistence of mountain big sagebrush as a function of parameterized life history traits under different fire size and fire rotation combinations. We ran simulations until a stable equilibrium of vegetation composition was reached assessed by the stability of a 50 to 100-year moving average proportion of sagebrush. We varied fire rotation by 10 year increments from 5 to 115 years and, for each fire rotation, simulated 11 different fire sizes from 2 ha to 50 ha. Fire size represented uniformly burned patches where no unburned islands remained in the burn interior; thus simulated burn patches had no viable seed sources within the patch. We used an artificial landscape simulated as a grid of 0.1 ha (10 x 10 m) cells. The landscape encompassed 4 km^2 or 40,000 grid cells. Sagebrush life history traits used as model inputs included seed dispersal distance, time to reproductive maturity, and establishment probability.

Succession

We modeled three broad composition classes including (1) early-seral woody and herbaceous resprouters and off-site colonizers (hereafter 'ruderal'), (2) sagebrush and (3) woodland (e.g. pinyon pine). In the early post-fire environment, the ruderal composition class dominated the grid cell. Sagebrush establishment was determined by the establishment probability but conditional on the seed present within the cell. If sagebrush propagules were unavailable, the stand remained in the ruderal class until sagebrush propagules arrived and establishment occured. Once sagebrush established, reproductive maturity occurred according to this parameter value (i.e. 2 to 4 years). Once cells hosted mature individuals, this cell then could disperse seed into the eight adjacent cells. Graduation to the woodland composition class was also based on life history traits (Appendix). The landscape was initialized at random with equal proportions of each of the three vegetation types. Models were developed in NetLogo version 5.01 (Wilensky 1999) (Appendix 8 for interface layout).

Fire

Fire size was incrementally varied from 2 ha to 50 ha (Table 2). Fire rotation was varied from 5 to 115 years by 10 year steps. Each year, an "area to burn" was calculated by multiplying the inverse of fire rotation (annual proportion to burn) by landscape extent (4 km^2). Next the number of fires to burn in a given year was calculated by dividing the area to burn by the particular fire size for that simulation. The number of fires to burn were then ignited randomly on the landscape and homogenous patches were burned of the given fire size, leaving no unburned patches within the burn interior. If the area to burn was smaller than the set fire size, then no fires burned in that year. However, the area to burn was carried over to the next year, doubling the area to burn and increasing the likelihood that area to burn was larger than the given fire size. This allowed tight control over the fire rotation and the fire size throughout a given simulation.

Output

Simulations were run until proportions of the different vegetation types stabilized. Model outputs shown in the results are the proportion of sagebrush that persisted on the landscape at the end of each of the simulations. We interpolated these data to show contour lines of persistence probability in relation to combinations of fire regime parameters.

IV. Key findings

The findings described herein are discussed in more detail in manuscripts currently in preparation and in review for publication in peer reviewed journals (Nelson et al. 2013, Nelson et al. in preparation).

A. Key findings: Historic Fire Frequency

Across all sites, estimates of point MFI (based upon proxy chronologies from fire-scarred trees) varied from 7.8 to 144.5 years. In general, fire intervals for the three northern Colorado Plateau sites (BAR, BRO, SNO) were notably longer (mean 78.8 years) than for one central Colorado Plateau site (HEN; 35.8 years), three Great Basin sites (GBN, ROS, TOM; mean 24.9 years), and three central highland sites (OWP, RBC, SAC; mean 35.5 years; Fig. 5) . Point MFI estimates for seven imbedded chronologies ranged from 12 to 58 years (mean 28.9, median 17.7 years; Fig. 6), while point MFI for 10 ecotonal chronologies ranged from 25.5 to 103 years (mean 47.7, median 44.2 years). A minimum fire-free interval of less than 10 years was observed for seven of these 17 fire chronologies. Conversely, maximum intervals of 58 and 103 years were observed for imbedded and ecotonal chronologies, respectively. Because of their close association with mountain sagebrush communities, we had hypothesized that values for these two classes of chronologies would most accurately portray the fire frequency and corresponding length of fire free intervals for mountain sagebrush communities under historical conditions.

Point MFI estimates for forested chronologies located near forest-shrubland ecotones ranged from 7.8 to 144.5 years (mean 37.0, median 28.6 years), with a minimum interval of less than 10 years for 21 of 34 chronologies (Figs. 3 and 4). Mean point MFI for the 15 forested sites located more distant from forest-shrubland ecotones was 48.9 years. In addition, fire intervals were calculated for five, two-sided chronologies (fire-scarred trees on opposing sides of mountain shrubland communities) at four sites (GBN, BRO, OWP, RBC). Mean values ranged from 20.8 years at the GBN site to 109 years at the BRO site (mean 55.6, median 31.5 years). Across the four chronologies, minimum and maximum intervals recorded were 1 and 185 years.

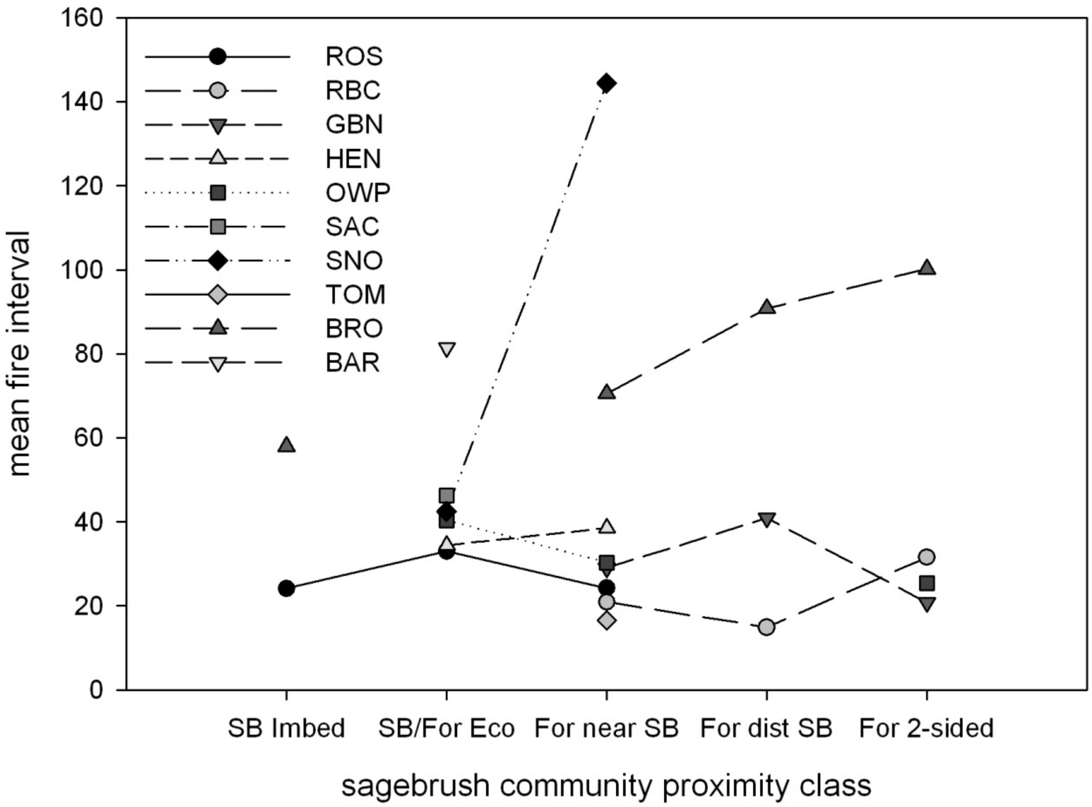


Figure 3. Point mean fire interval (MFI) estimates from proxy fire-scarred trees for 10 sites in the eastern Great Basin, Colorado Plateau, and intervening mountains and highlands. Sites codes are as follows: Rose Spring, Wah Wah Range (ROS); Beaver Creek, Tushar Range (RBC); Great Basin NP, South Snake Range (GBN); Henry Mountains (HEN); Old Woman Plateau, Wasatch Plateau (OWP); Salina Creek, Wasatch Plateau (SAC); Snowshoe Canyon, East Tavaputs Plateau (SNO); Tom's Creek, Deep Creek Range (TOM); Brownie Creek, Uinta Range (BRO); and Bare Top, Uinta Range (BAR). Estimates are separated by classes indicating the proximity of fire-scarred trees to mountain sagebrush communities. Proximity classes are defined as imbedded (SB Imbed; proxy trees located within sagebrush communities); ecotonal (proxy trees located within 10 m of forest-shrubland ecotone); forest-near ecotone (proxy trees located between 10 and 500 m of forest-shrubland ecotone); and forest (proxy trees located between 500 and 1000 m of forest-shrubland ecotone). A fifth classification is designated for fire-scarred trees located on opposing sides of a mountain sagebrush community having synchronous fire years (For 2-sided).

Within each site, estimates for point MFI were similar across sagebrush proximity classes, with one exception (Fig. 5). At the SNO site, the estimated point MFI for a single forested chronology suggested a three to four-fold increase in MFI when compared to the two ecotonal chronologies of the same site. Both the direction and magnitude of this pattern were not expected and with few trees sampled, are likely artifacts of an incomplete record. Otherwise, there is notable similarity between chronologies classified as imbedded, ecotonal or two-sided and those classified as forested, suggesting that forest-shrubland ecotones were relatively porous, allowing considerable cross-boundary fire spread.

Conversely, some high fire frequency values (low point MFI and short minimum fire intervals) are not consistent with the longer fire-free intervals required for mountain sagebrush dominance based upon analysis of results from components 2 and 3 of this study and a majority of existing literature (as reviewed in Kitchen and McArthur 2007). This would suggest that at locations where these chronologies were derived, fires either burned in patchy, incomplete patterns or that sagebrush was less abundant or even largely absent on portions of these landscapes historically, or both. The possibility of sagebrush infilling or expansion into fire-adapted, conifer-herb communities that occurred with 20th Century reductions in fire frequency may be the most plausible explanation for the current dominance of sagebrush on sites once exposed to frequent fire. This gradual shift towards increased woody composition would be similar to the encroachment of fire-intolerant conifer species into mountain shrublands as fire was excluded for even longer time periods.

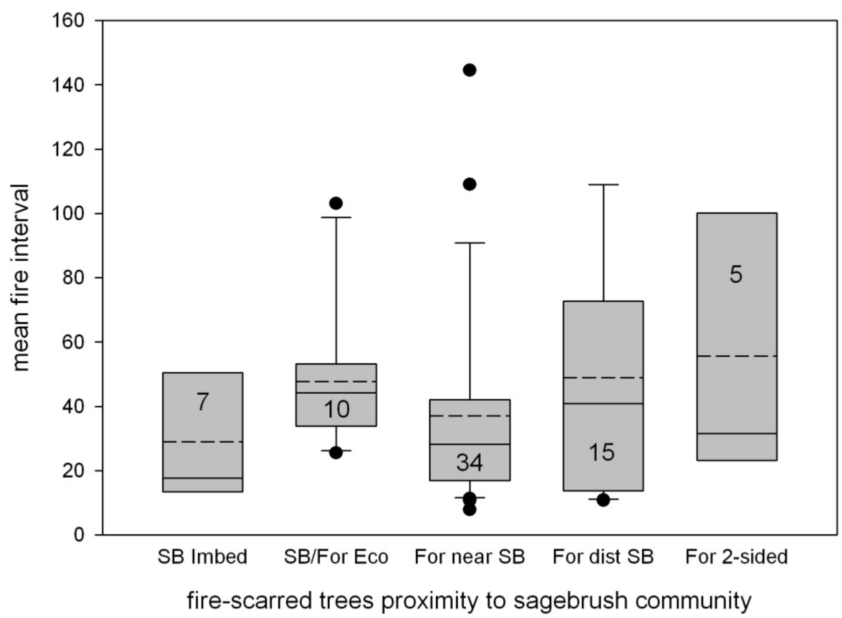


Figure 4. Point mean fire interval (MFI) estimates from proxy fire-scarred trees for 10 sites in the eastern Great Basin, Colorado Plateau, and intervening mountains and highlands. Estimates are separated by classes indicating the proximity of fire-scarred trees to mountain sagebrush communities. Proximity classes are defined as imbedded (SB Imbed; proxy trees located with sagebrush communities); ecotonal (proxy trees located with 10 m of forest-shrubland ecotone); forest-near ecotone (proxy trees located between 10 and 500 m of forest-shrubland ecotone); and forest (proxy trees located between 500 and 1000 m of forest-shrubland ecotone). A fifth classification is designated for fire-scarred trees located on opposing sides of a mountain sagebrush community having synchronous fire years (forest 2-sided). Within box plots, solid horizontal lines denote median values and dashed horizontal lines denote mean values. Numbers in boxes indicate the number of estimates assigned to each plot.

Figure 5. Fire-scarred ponderosa pine stump in a closed pinyon-juniper-mahogany woodland. Dead branches on the ground are primarily sagebrush skeletons. No live sagebrush plants were present. Composite mean fire interval (MFI) as recorded on two stumps (2nd stump is 15 m to the right and out of view) between 1340 and 1851 was 13.4 years.

Succession rate appears to have varied considerably among sites. For example, in portions of the ROS site, sagebrush skeletons are all that remain of the mountain sagebrush community that once occupied portions of the site that are now hidden in a dense pinyon-juniper-mahogany woodland (Fig. 5). Fire-scars on ponderosa pine stumps reveal MFI values of 15 years or less for a period of at least 500 years (and most probably much longer) ending in the mid-1800s, suggesting that a sagebrush-dominated community may have been short-lived and transitional between an open ponderosa pine-grass savanna and the closed woodland that persists today. At the other end of the spectrum, conifer encroachment into mountain sagebrush communities is minimal or only in early to moderate stages in at least portions of all sites even after apparently long periods without fire. It is clear that there are factors besides the length of the fire-free period that influence the rate of community succession.

A. **Sagebrush recovery (project component 2)**

Time since fire (TSF) explained 35% of the between-site variance in canopy recovery:

$$\% \textit{ Artemisia} \text{ canopy recovery} = 4.9\% + 2.6\% \text{ year}^{-1}(\text{TSF})$$

where $R^2 = 0.35$, $P = 0.002$, $n = 27$.

Extrapolation using this equation predicts 37 years for full recovery to unburnt cover values. The 95% confidence intervals (±2.05) for the estimate of the slope parameter bracket full recovery to between 24 and 84 years (Fig. 6). In addition to time since fire, winter precipitation in the year after fire (W1) positively influenced recovery (Appendix 2). Average winter precipitation (pptWint) and winter precipitation in the year before fire (W0) were highly correlated with this variable but explained less of the between-site variance in absolute cover, and had confidence intervals that overlapped zero.

Environmental controls on % Artemisia cover after fire

Similar to the recovery response variable scaled to reference values (above), time since fire explained 36% of the variance in % *Artemisia* cover calculated independent of the paired reference site:

$$\% \textit{ Artemisia} \text{ cover} = -0.02\% + 0.86\% \text{ year}^{-1}(\text{TSF})$$

where $R^2 = 0.36$, $P < 0.001$, $n = 36$.

Extrapolation of this equation to reach the mean % *Artemisia* cover for reference sites (24%) would take 27 years. The 95% confidence intervals (±0.42) for the estimate of the slope parameter bracket reaching 24% cover to between 19 and 51 years. The addition of W1 accounted for an additional 6% of the variance in *Artemisia* cover

$$\% \textit{ Artemisia} \text{ cover} = -8.47\% + 0.82\% \text{ year}^{-1}(\text{TSF}) + 0.36(\text{W1})$$

where $R^2 = 0.41$, $P < 0.0001$, $n = 36$ (Appendix 2, 3).

In contrast, 30-year average pptWint only accounted for an additional 1% of the variance (Appendix 2.). The addition of % sand fraction in combination with W1 accounted for an additional 8% of the variance:

$$\% \textit{ Artemisia} \text{ cover} = -26.0\% + 0.85\% \text{ year}^{-1}(\text{TSF}) + 0.43(\text{W1}) + 0.31(\text{sand})$$

where $R^2 = 0.49$, $P < 0.001$, $n = 36$.

The effect of sand fraction should be interpreted with the knowledge that range of sand fraction for these sites ranged from 24 to 71%. Beyond these minimum and maximum values for this study, the relationship is unknown.

Environmental relationships with Artemisia cover in unburnt stands

In treeless plots of apparently unburnt *Artemisia* stands (n = 18), % sand fraction was negatively associated with *Artemisia* cover, explaining 37% of the variance:

$$\%\ \textit{Artemisia}\ \text{cover} = 43.6\% - 0.36(\text{sand})$$

where $R^2 = 0.37$, $P = 0.006$, $n = 18$ (Appendix 4).

The addition of August maximum temperature explained another 25% of the variance, with warmer sites having greater *Artemisia* cover:

$$\%\ \textit{Artemisia}\ \text{cover} = 11.31\% - 0.45(\text{sand}) + 2.36(\text{AugMax})$$

where $R^2 = 0.62$, $P < 0.001$, $n = 18$.

In locations where trees had established within *Artemisia* communities (tree cover range 1–35%), the presence of trees was negatively associated with *Artemisia* cover and explained 48% of the variance:

$$\%\ \textit{Artemisia}\ \text{cover} = 28.75\% - 0.58(\text{tree})$$

where $R^2 = 0.48$, $P < 0.001$, $n = 18$ (Appendix 4).

Age distributions

We sampled and dendrochronologically dated 1455 individual *Artemisia* plants from 72 transects (Appendix 5). All transects, located at sites that had burnt less than 40 years before sampling (n = 36) in 2007 and 2008, had established *Artemisia* cohorts within 7 years after the fire (Appendix 6). Most sites (30 of 36) had cohorts within the first 3 years after fire – including seven sites where individuals were found to have established before the year of the fire, presumably surviving within small unburnt islands near the transect location. This finding suggests that none of our sites were limited by seed availability for more than a few years after fire and that post-fire viable seed played an important role in recovery.

The concept of the regeneration niche is important in post-fire sagebrush recovery. Our data suggest that post-fire recovery is influenced strongly by windows of time favorable for regeneration (i.e. adequate deep soil water recharge, Appendix 7). Sagebrush regeneration and recovery are strongly limited by moisture in the more water-limited sites. We speculate that temperature or growing season length limit regeneration at colder-higher elevation sites where the influence of precipitation on recovery was less important. This pattern should be expected for any woody species, but is nevertheless important given the confusion surrounding attempts to generalize controls of post-fire recovery for such a widespread and ecologically adaptable species.

B. Key findings: Modeling (project component 3)

Sagebrush persistence was compatible with a wide range of fire regime parameters depending on the specific assumptions of life history trait parameter values and whether conifers were present in the landscape. For simulation interface illustration see Appendix 8. For parameter values see Appendix 9. In general sagebrush was excluded by fire on the short end of the fire rotation spectrum and excluded by woodland on the long end of the fire rotation spectrum (Fig. 6). Sagebrush persistence was low when dispersal distance was set to 10 m and reproductive maturity age was set to four years. This was the combination of life-history traits that most slowed the speed of post-fire recovery. Under fire rotations commonly cited as typical of mountain big sagebrush (i.e. 30-80 years) (Baker 2006, 2011)**,** simulations revealed that long-term convergence on 20% composition or greater is only compatible with small fire sizes, smaller than 10 ha for 50 year rotations and smaller than 30 ha for 80-year fire rotations (Fig. 6). Thus for large fires that are patchy in burn pattern, the area between unburned patches can be related to these simulated fire sizes in that the recovery of sagebrush into the burned areas depends on the spatial distribution of seed source. Appendix 10 shows how different life-history traits influence the proportion of the landscape that sagebrush is able to occupy under specific fire regime parameter values.

The simulations highlight the importance of assumptions about process parameters influencing the post-fire recovery. If it is believed that a certain fire regime was dominant over a historical period, there are caveats that should be acknowledged. If it is assumed that historical fire regimes for a given location were characterized by mean return intervals of 30 years for example, then there are additional assumptions that must be tied to this hypothesis: fire sizes were very small, on the order of 10 ha, or more realistically the burn mosaic was patchy in that the fire complex was effectively composed of many fires of small size; alternatively or in addition to this assumption, recovery from seed bank was a dominant process allowing for post-fire recovery within the interior of the burn, effectively reducing the effect of fire size on recovery from the burn perimeter. A third assumption of dispersal distance greater than 10 m may need to be acknowledged as this parameter is strongly related to the speed with which recovery can take place. Therefore, for any assertion of historical fire regime characteristics, it is also necessary to include assumptions of life history trait parameter values that could allow sagebrush to persist with the given fire regime.

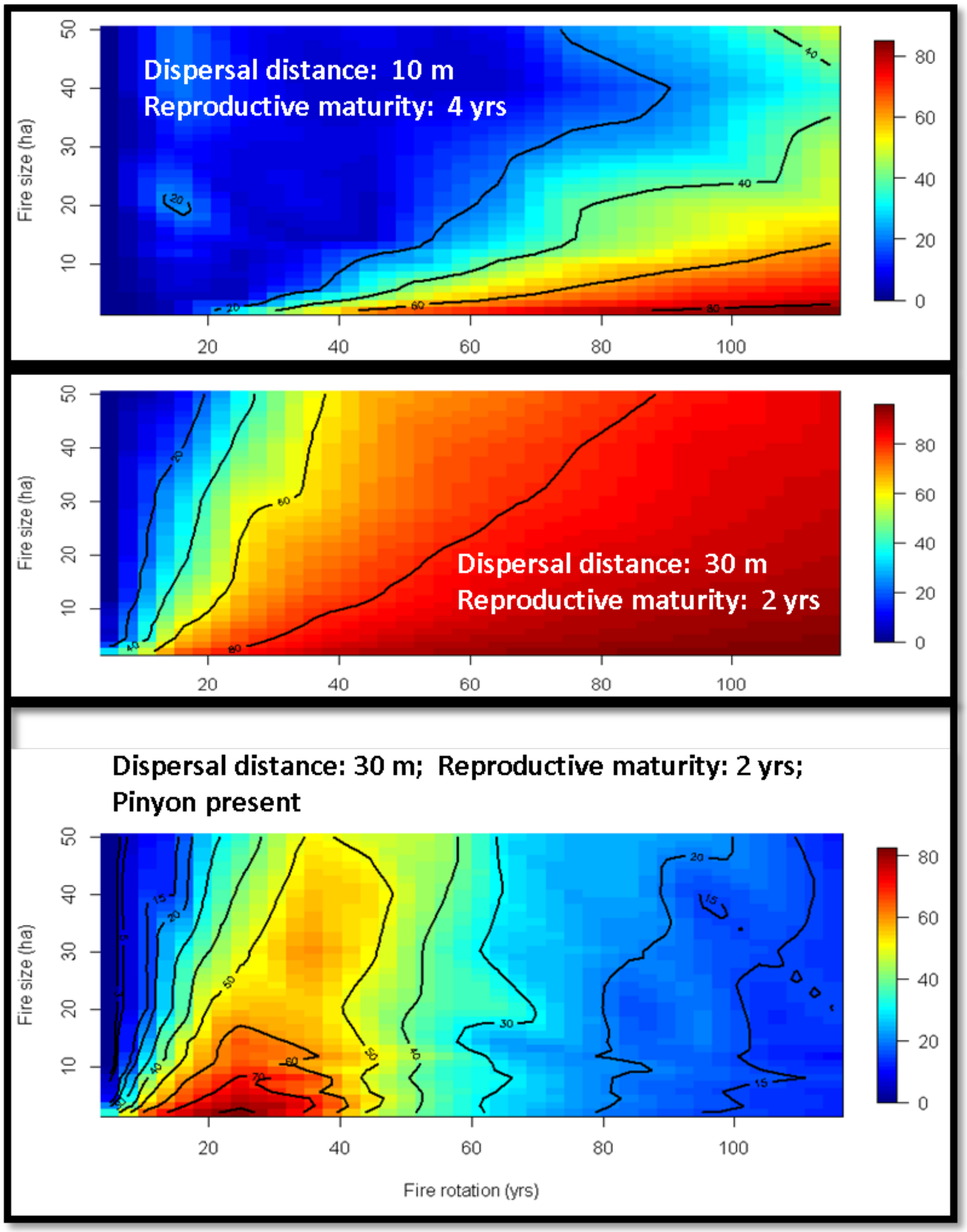


Figure 6. Fire regime parameter space compatible with long-term average percentage of sagebrush on the landscape. Sagebrush persistence is shown when dispersal distance = 10 m and reproductive maturity age = 4 year (top); when dispersal distance = 30 m and reproductive maturity age = 2 years (middle); and when pinyon is present (parameter values table 2) and sagebrush dispersal distance = 30 m and reproductive maturity age = 2 years (bottom).Contour lines represent the proportion of sagebrush on the landscape compatible with the different combinations of fire size (y-axis) and fire rotation (x-axis). Contours are interpolated over unique scenarios of fire size and fire rotation.

V. Management implications

A. Management implications: Historic Fire Frequency project component 1)

With the possible exception of the northern portion of the Colorado Plateau, fire was historically much more frequent than it is currently the case across the central Intermountain region (Utah and eastern Nevada) for which proxy fire chronologies were developed for this study. Mountain sagebrush may have been absent or present at much lower densities on some areas that are today dominated by sagebrush and in other areas sagebrush may now be absent where it was once abundant due to lengthening of fire-free periods allowing forest/woodland encroachment.

Evidence also suggests that fires burned relatively freely across vegetative ecotones shaping composition and structure of both forests and shrublands in dynamic patterns in time and space. For many locations, it appears that a rather broad window, or range in fire frequency is compatible for sagebrush persistence or dominance, though some areas (e.g. ROS) were susceptible to relatively rapid conifer encroachment without fire. Thus strategies for managing disturbance in favor of mountain sagebrush should be adaptive and tailored to site-specific conditions. Disturbance processes should be managed at landscape scales, allowing fire to cross vegetation boundaries in ways that mimic historical patterns.

B. Management implications: Sagebrush recovery (project component 2)

Our observations of big sagebrush recovery following fire over a broad regional scale suggest that an average recovery time of 25 to 35+ years, which many studies have corroborated (review in Baker 2011), is reasonable given that sagebrush adults or propagules survive within the burn interior at some reasonable level. However, there was much variation in recovery rate associated with the timing of precipitation relative to the particular fire event. We found that precipitation in the pre-fire growing season was positively related to post-fire recovery of mountain big sagebrush, and that precipitation in the late winter following fire was similarly positively related to recovery rate. This suggests that it is somewhat risky to use prescribed fire to “rejuvenate” mountain big sagebrush communities, as the rate at which sagebrush is able to recolonize the site will depend to large degree on the timing of precipitation events, something the land manager is unable to control. To the degree that fewer sagebrush plants or propagules survive the fire, recovery time will increasingly be a function of the distance to seed source. We explore this important process in project component 3.

C. Management implications: Modeling sagebrush persistence (project component 3)

Our landscape simulation model allows us to bracket the fire regime parameter space that is compatible with mountain big sagebrush persistence, given successional processes and a select set of life history traits including seed dispersal distance, seedbank persistence, and time to reproductive maturity. If fires were frequent in the past (e.g. 15 year MFI estimate over 500 years for a Ponderosa stand in project component 1 and other fire scar-derived fire chronologies (Miller & Rose 1999)) certain criteria would need to be satisfied for this MFI to be compatible with sagebrush: fires were patchy leaving unburned seed sources to facilitate reestablishment within the interior of burn; residual seedbank survived fire; or sagebrush was an ephemeral component in high fire frequency zones, establishing only when fire intervals temporarily lengthened as is the case now with extension of the fire free interval in the 20th century.

The multi-dimensional environmental context of a site influences life history trait performance and thus different site locations afford a different fire regime persistence space. For instance, steeper and windier sites may allow greater seed dispersal distances and more productive sites may allow an earlier reproductive age. Rather than mapping our simulation onto known sites, we simplified the simulation by focusing solely on the life history traits (seed dispersal distance and age to reproductive maturity). Probability of establishment was held constant at 30%. In our research paper (Nelson et al. in preparation), we synchronize this probability in time by including a cycling drought regime that influences this establishment. A soil layer spatial variable further modifies this probability so the implicit influence of local within-site heterogeneity (i.e. soil water) on the fire regime persistence space can be evaluated.

Simulation model results highlight the importance of fire patchiness (i.e. small fires or large patchy fires) for ensuring the long-term persistence of mountain big sagebrush communities especially where fire is relatively frequent. Post-fire refugia can refer to survival of reproductively mature plants, as well as to survival of viable seed bank where weather conditions for germination and establishment are favorable following fire. Prescribed fire should seek to maintain unburned patches and patches burned with low severity. Post-fire restoration efforts in sagebrush communities should introduce a patchy distribution of planted sagebrush that can act as distributed foci for subsequent natural establishment during climatically favorable years.

VI. Relationship to other recent findings and ongoing work on this topic

Ziegenhagen and Miller (2009) highlighted that for large burns recovery from seed bank is a critical determinant in the post-fire recovery trajectory. We take this finding one step further and quantify in a spatially explicit sense, the effect of fire size on recovery and long-term persistence when aggregated over time (project component 3). Our evaluation thus implicitly quantified what spatial structure of seed bank persistence within large burns could be compatible with sagebrush. Though we didn't report on results of explicit seed bank persistence in this report, the effect of recovery from seed bank within a large burn or the recovery from the burn perimeter in many smaller fires is effectively analogous.

Wijayratne and Pyke (2009) highlighted that sagebrush seed bank although short-lived can persist for several years. Shallow burial may prevent germination cues and thus in sandy sites where shallow burial may be more likely, recovery was enhanced perhaps owing to establishment from seed bank that persisted into the spring following the fire (project component

2, Appendix 7). Baker (2013) reviewed the MTBS data set and quantified the percent unburned within the fire perimeter in different sagebrush types. For the southern (n = 873) and northern Great Basin (n = 441) montane sagebrush type, Baker calculated approximately 20% of the area within burn perimeters went unburned. This unburned percent leaves seed sources within the interior of the burn and facilitates post-fire recovery. This unburned percent effectively reduces the effect of fire size (as computed from the convex hull of the fire perimeter) on recovery time. Although Baker did not quantify the spatial structure of the unburned proportion, this could be imported into our model and the spatial recovery process could be simulated under various scenarios including drought. In this regard, the simulation model could be used as an optimizer, in that the locations of restoration that maximize recovery rate could be mapped and used to inform restoration decisions.

VII. Future work needed

There is a need to apply our model to generate site specific output for specific purposes ranging from management applications to testing site-based assumptions of historic fire regime. The rate of post-fire sagebrush recovery depends on the spatial structure of seed source. The rate of and direction of seed spread depends on predominant wind direction, and slope. The resulting mosaic of propagule pressure that could be predicted on the basis of a wind model could be compared to the spatial structure of cohorts (e.g. project component 2). Cohort size should increase with favorable seasonal precipitation. Combining site-level reconstructions of cohort establishment with modeling, the aim would be to calibrate parameter values with respect to establishment timing in association with seasonal climate and underlying soil conditions. One application of this site-specific linkage of spatial processes of seed dispersal and fire spread with seasonal climate and edaphic characteristics would be to optimize the locations for reseeding and restoration. Given model predictions of where seed source foci should be located for a particular recovery objective, restoration plans could utilize these spatially explicit reseeding locations and use follow up monitoring to help refine the model parameters and the model structure.

The relative importance of seasonal precipitation coinciding with fire events and on post-fire regeneration from seed bank should vary along a productivity gradient. Post-fire treatments could include grazing exclosures, rain-out shelters, and snow increase fences to test the effects of precipitation timing on survival and mortality of both juvenile and adult plants and how the effect sizes vary along a productivity or temperature-moisture gradient. In association with climate-fire regime scenarios described in section VI, these effects could be extrapolated to locations with similar climate in the future.

Further modeling exercises are underway to investigate the influence of landscape heterogeneity at multiple scales. We are developing model scenarios to investigate how topographic and edaphic heterogeneity influence fuel loading and associated fire spread probability, and thus how landscape context might further constrain or expand the disturbance regime space suitable for persistence of mountain big sagebrush ecosystems.

As part of an NSF-funded (Nevada EPSCoR) project, we build on this JFSP funded work and explore the persistence of co-occurring woody species such as bitterbrush (*Purshia tridentata*) in a similar framework. By simulating the persistence of a number of different woody species with different fire persistence traits that commonly coexist in plant communities, we expect to further refine the disturbance regime space that is compatible with not only a single species (i.e. big sagebrush), but with species associations and communities.

Another important area of investigation that we are working on is climate change influence on both the expansion and contraction of suitable habitat for mountain big sagebrush and co-occurring woody species and the associated feedback that vegetation change will have on fuel mosaics, which in combination with longer fire seasons will change the fire regime parameters. We will explore several different scenarios of how climate influences spatial shifts in vegetation distribution and how both emergent vegetation mosaics and climate influence fire regime. Our model output will be the emergent vegetation communities on the landscape generated from climatic influences on ecophysiological limits to distribution, and from climate-influenced fire regime changes as these interact with species life-history traits. This will allow assessment of how scenarios of climate and fire regime change will influence the relative proportion of fire persistence traits expressed in future plant communities.

VIII. Deliverables Cross-Walk

Proposed	**Delivered**	**Status**
Project web site	Project description, progress, results, application	Updated as needed
JFSP progress reports	1 page summary of progress	Completed
JFSP final report	Data, analysis, model development, applications	Completed
oral presentations	Professional meetings (see citation database)	Completed
3 field workshops	Findings model development Website tutorial of model application (maybe from our website)	Completed Completed Completed
Drafts of 3-4 peer reviewed manuscripts	(1) Nelson et al. (2013) sagebrush recovery. International Journal of Wildland Fire (2) Nelson et al. (manuscript) spatial simulations (3) Nelson et al. (manuscript) drought simulations (4) Kitchen et al. (manuscript) mountain sagebrush fire history using proxy trees)	1. accepted (uploaded to JFSP website) 2. in preparation 3. in preparation 4. in preparation
Information for updating LANDFIRE	Succession model parameterization, FRCC descriptions	see appendix 11.

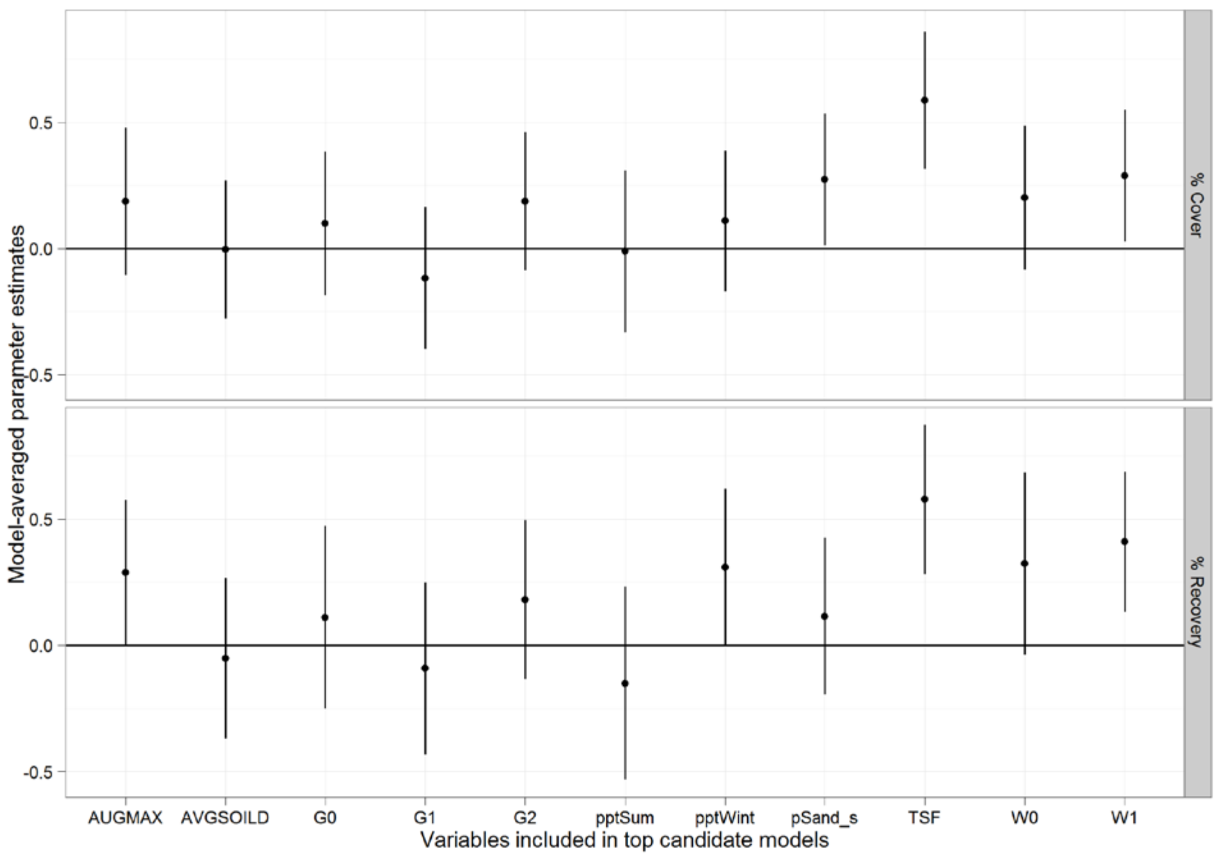
APPENDICES

Appendix 1. Predicted influence (+ or -) of each predictor variable on post-fire *Artemisia* recovery based on hypothesized direction of influence (↑ or ↓) on salient processes related to post-fire recovery. Spatial variables represent between-site averages of measured variables in the field, or derived through GIS analysis and the PRISM dataset. Precipitation timing variables were extracted from the PRISM climate surfaces for the months before and after the fire year.

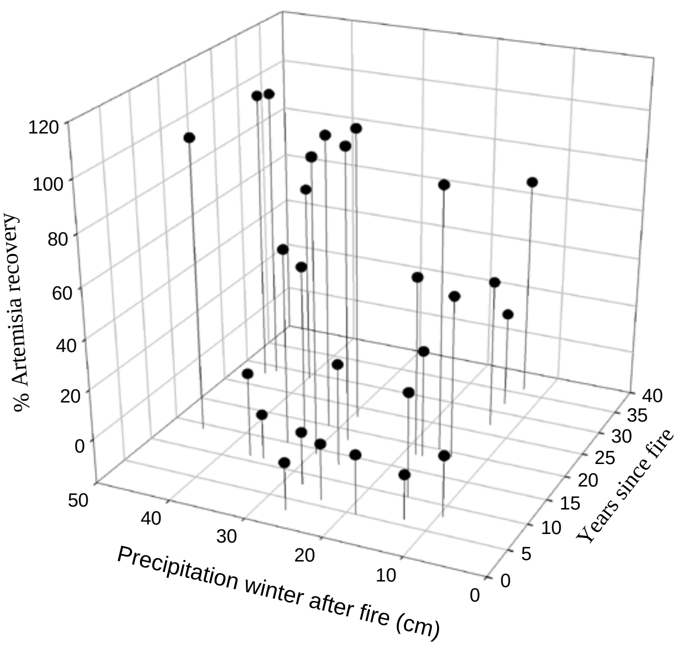
Variable code (value range)	Predictor Variable	Hypothesized influence (+ or -) of predictor variable on recovery; and influence on mechanism (↑ or ↓) by which recovery is influenced	Literature source
Spatial variables			
TSF (1-256+ years)	Time since fire	(+)↑ Recruitment	(Baker 2011)
Folded aspect (0.75-178.84˚)	Folded aspect around 225˚	(+) ↑growing season higher elevation; (-) ↓ soil moisture lower elevation	(Lesica *et al.* 2007; Ziegenhagen and Miller 2009)
AugMax (9.15-43.8 C˚)	Mean August maximum temperature (PRISM)	(+)↑ growing season (-) ↓ decrease water availability (+)↓ competition 1858-2810 m (6099-9220 ft.) elevation	(Daubenmire 1975; Cawker 1980)
AVGSOILD (5.75-44.1 cm)	Average soil depth	(+) ↑Soil moisture, ↑nutrients	(Lesica *et al.* 2007)
P (7.59-80.07 ppm)	Total Phosphorous	(+)↑Availability of potential limiting nutrient	
N (347.8-617.7 ppm)	Total Nitrogen	(+)↑Availability of potential limiting nutrient	
%Sand (19-73.28 %)	%Sand (soil texture)	(+)↑Seed burial, ↑dormancy, ↑seedbank ↑fire survival	(Meyer 1994; Wijayratne and Pyke 2012)
AVGROCK	% Rock cover	(-) ↓ Soil moisture, ↓nutrients	
Ca (1233-8268 ppm)	Calcium	(-) ↑pH, ↓ortho-P availability	

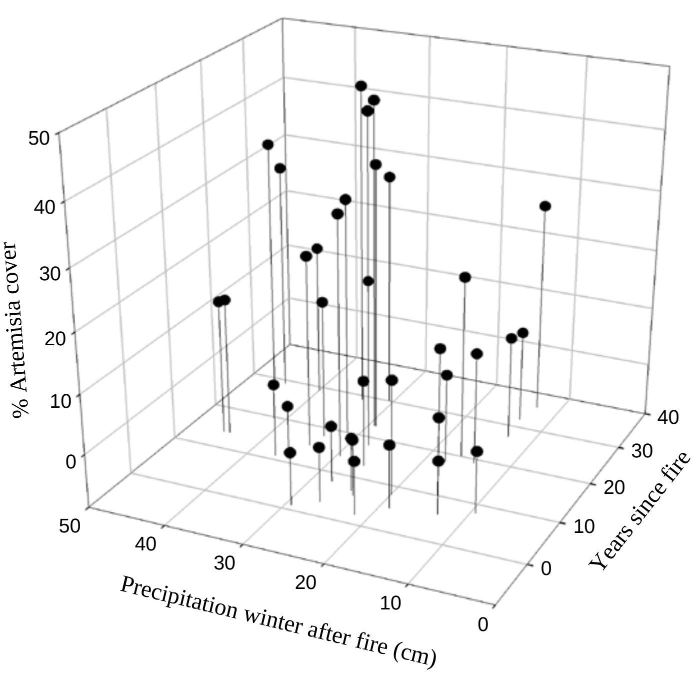
Na (30.52-99.18 ppm)	Sodium	(-) ↑Salinity stress, ↓recruitment	
pptWint (12.2-40.7 cm)	30-yr average winter (Oct-Mar) precipitation (PRISM)	(+) ↑Soil moisture, ↑recruitment	(Cawker 1980)
pptSum (16.4-29.6 cm)	30-yr average Summer (Apr-Sep) precipitation (PRISM)	(+) ↑Soil moisture, ↑recruitment	(Cawker 1980)
Precipitation timing			
G0 (10.59-42.96 cm)	Apr-Sep, year before fire	(+) ↑Seed production, ↑seed bank response	(Evans *et al.* 1991)
W0 (11.43-58.99 cm)	Oct-Mar, year before fire	(+) ↑ frost heaving, ↑seed burial,↑seed bank response	(Blaisdell 1953; Meyer 1994; Ziegenhagen and Miller 2009; Wijayratne and Pyke 2012)
G1 (11.87-45.93 cm)	Apr-Sep, year of fire	(+) ↑Fuel moisture, ↓fire severity on seed bank	
W1 (9.44-45.88 cm)	Oct-Mar, after fire	(+) ↑Soil moisture, ↑establishment	(Cawker 1980)
G2 (10.15-41.03 cm)	Apr-Sep, year after fire	(+) ↑Soil moisture, ↑establishment	(Cawker 1980; Daubenmire 1975)

Appendix 2. Standardised model-averaged parameter estimates for variables included in models explaining absolute percentage *Artemisia* cover (top) and % *Artemisia* recovery relative to unburned unburnt reference transects (bottom). Bars represent 95% unconditional confidence intervals. Time since fire was included in all models. Variable codes are as follows: AUGMAX, 30-year mean August maximum temperature; AVGSOILD, average soil depth; growing season precipitation (Apr–-Sept); G0, warm season in year before fire; G1, growing season of fire, and G2, growing season in year following fire; cool season (Oct–-Mar), before (W0), after (W1) and two cool seasons following fire event; pptSum, 30-year mean Apr–-Sept precipitation; pptWint, 30-year mean Oct–-Mar precipitation; pSand_s, percentage sand fraction; and TSF, time since fire.



Appendix 3. Positive influence of time since fire and precipitation in the year after fire on percentage *Artemisia* recovery to reference cover values (left) and % *Artemisia* cover (right).





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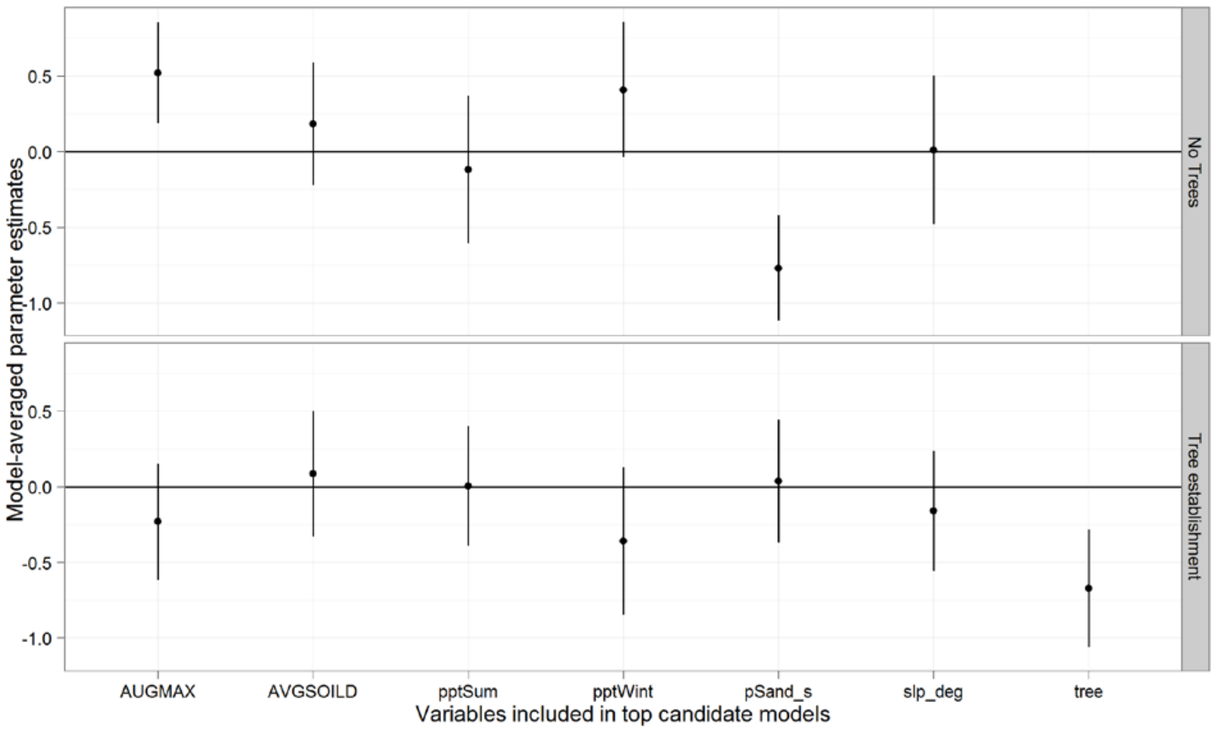
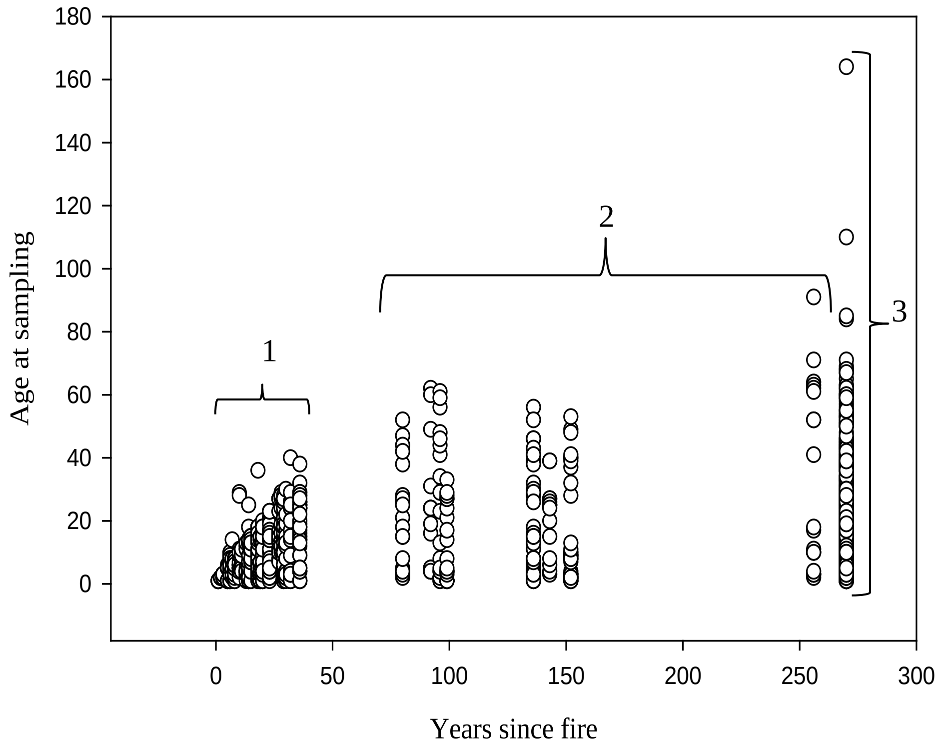
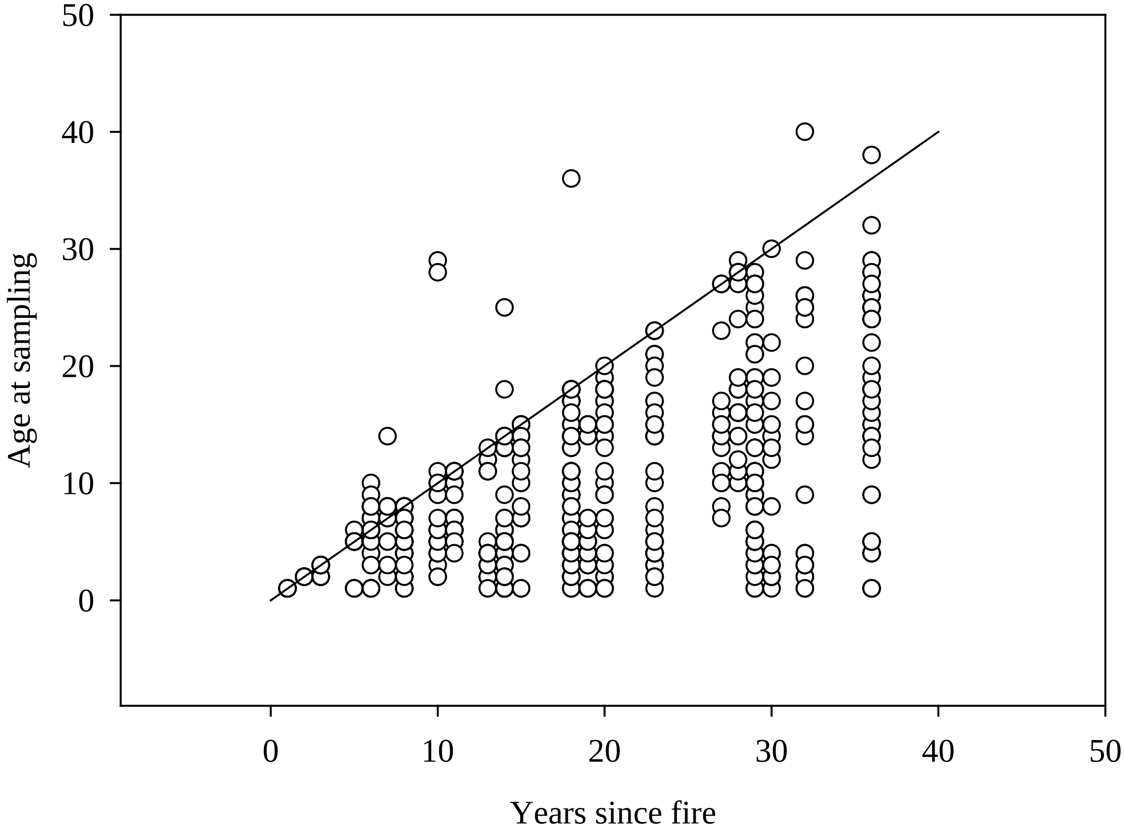
Appendix. 4. Standardised model-averaged parameter estimates for variables included in models explaining % *Artemisia* cover for unburned unburnt sagebrush stands without tree establishment (top) and with tree establishment (bottom). Bars represent 95% unconditional confidence intervals. Variable codes are as follows: AUGMAX, 30-year mean August maximum temperature; AVGSOILD, average soil depth; pptSum, 30-year mean Apr-Sept precipitation; pptWint, 30-year mean Oct-–Mar precipitation; pSand_s, percentage sand fraction; slp_deg, slope of site; and tree, tree canopy cover.



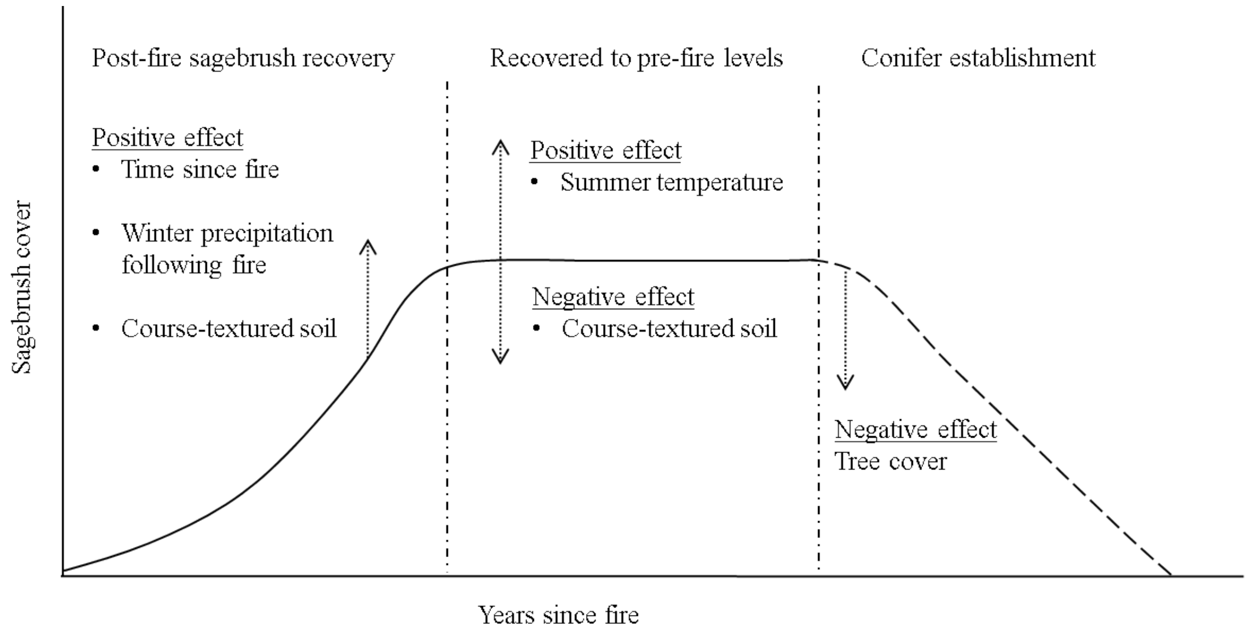
Appendix 5. (a) Relationship between the age in years of *Artemisia* at time of collection (2007–2008) (n = 1455), from 72 transects, in relation to the number of years since fire at the sampling time. The first group of sites (1) are recently burnt stands in the process of recovering (years since fire < 40). The second group (2) represents mature stands where the last fire date was estimated from nearby fire-scarred ponderosa pine trees. The third group (3) represents age distributions from unburnt reference sites where we did not estimate a last fire date from nearby fire-scarred trees and thus they were placed to the right of the other groups, for convenience.



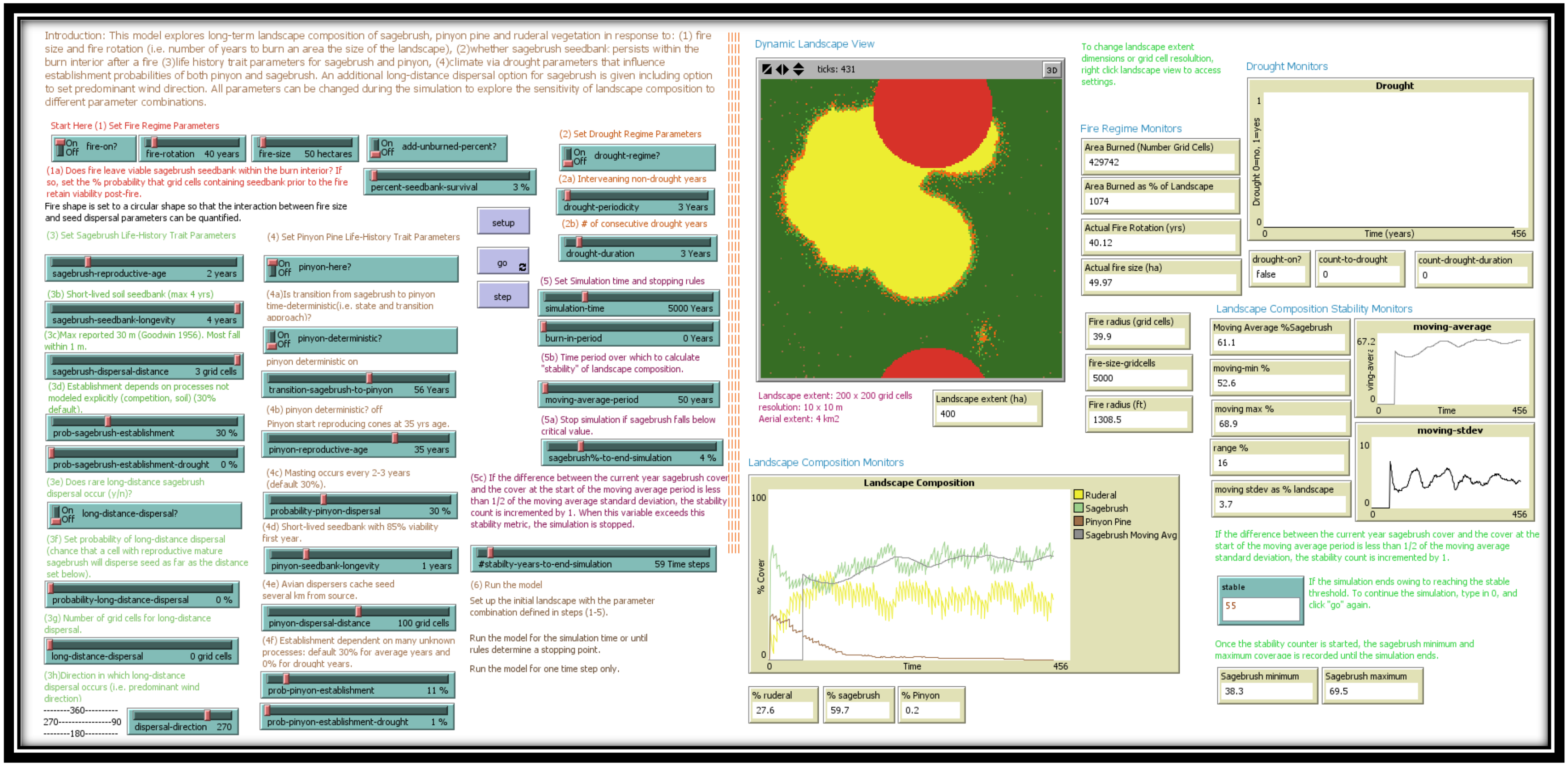
Appendix 6. Age of sagebrush estimated from 25 samples for each site including 10 samples of reproductively mature plants. Age distributions are shown in relation to the line $y = x$ (i.e. establishment in the spring after the burn).



Appendix 7. Conceptual model of environmental influences on *Artemisia* cover across a successional sequence. Different environmental variables were associated with *Artemisia* cover at different temporal phases of the chronosequence (recovery, mature, tree in-filling). The directional effects of the measured soil, climate, or vegetation characteristics on *Artemisia* cover are denoted by arrows. The last phase is dependent on the probability of tree establishment in particular landscape and climate scenarios (dashed line). Variables listed had model-averaged regression coefficients with 95% unconditional confidence intervals that excluded zero.



Appendix 8. Simulation model interface. Model was developed using NetLogo 5.01.

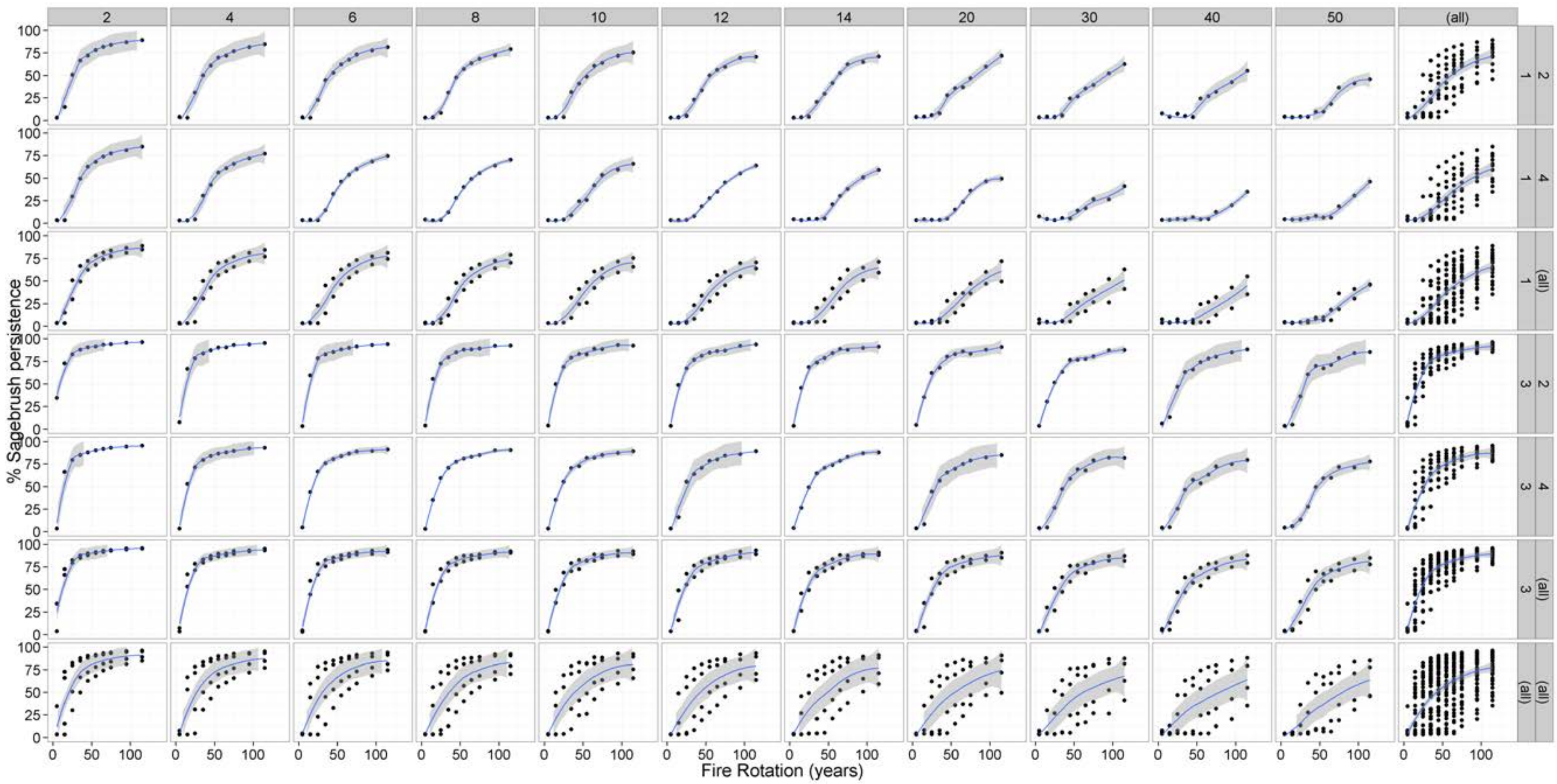


Appendix 9. Parameter values discussed in report and those addressed in the full simulation manuscript.

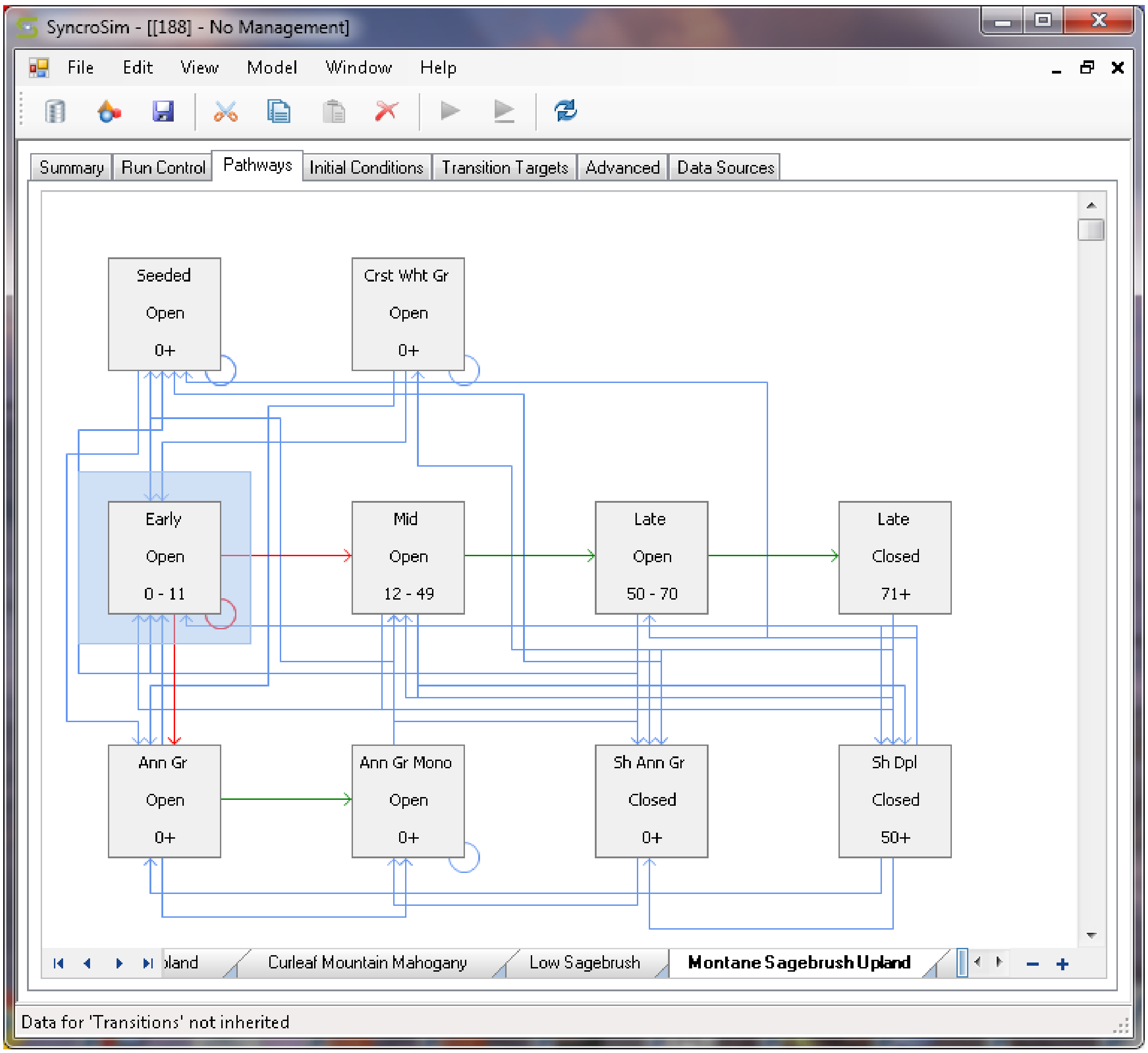
	Species characteristics	Parameter value [units]	Parameter value ranges evaluated in full manuscript	Description/justification and source if available.
Artemisia tridentata (Big sagebrush)				
	Reproductive age	2, 4 [years]	2,3,4,5[years]	2-5 years (Welch 2005)
	Probability of establishment	30 [%]	1, 10, 30, 60 [%]	Conservative (one out of three years)
	Dispersal distance	10, 30 [m]	10, 20, 30[m]	Goodwin (1965)
	Seed bank longevity	4 [years]	1, 2, 3, 4 [years]	Wijayratne and Pyke (2012)
	Probability of long distance dispersal	Off	0, 1, 5[%]	Long-distance dispersal may have low probability. Given all cells in burn perimeter
	Long-distance dispersal distance	Off	60, 100[m]	Max dispersal distances are reported at 30 m, but this is likely not the true maximum.
	Probability of establishment drought	Off	0, 1, 5[%]	Drought years have low likelihood of establishment.
Pinus monopylla (Pinyon pine)				
	Reproductive age	35 [years]	25, 35[years]	P. edulis = 25 yrs (Tueller et al. 1975), P. monophylla = 35 yrs (Meeuig et al. 1990)
	Probability of dispersal	30 [%]	30[%]	30% approximates masting every three years
	Probability of dispersal drought	Off	20[%]	Lagged effect of weather on seed crop
	Dispersal distance	1000[m]	30, 60, 120, 150[%]	up to 5 miles (set to ½ width of landscape) but most seed is cached nearby seed source.
	Seed bank longevity	1 [years]	1, 2 [years]	short lived seedbank
	Probability of establishment	30 [%]	1, 10, 30, 60 [%]	conservative

App.9 continued	**Species characteristics**	**Parameter value [units]**	**Parameter value ranges evaluated in full manuscript**	**Description/justification and source if available.**
	Probability of establishment drought	Off	0, 1, 5[%]	Drought years have low likelihood of establishment.
Fire Regime				
Fire Size	complete burned leaving no unburned islands	2, 4, 6, 8, 10, 12, 14, 20, 30, 40, 50[ha]	10, 50, 100, 200, 400, 800 [ha]	Larger fire sizes are modeled in combinations with a percent unburned
Fire Rotation	time required to burn an area the size of the landscape extent	5, 15, 25, 35, 45, 55, 65, 75, 85, 95, 105, 115[years]	10, 30, 50, 70, 90, 120, 150[years]	Fire rotation is varied from 10 to 150 years, the range of MFI from project component 2.
	Percent-seed bank-survival	Off	1[%]	The probability of cells with seedbank to maintain the seedbank viability through fire
Drought Regime				
	Drought-periodicity	Off	2, 3, 4, 5, 6[years]	Interval of time between drought
	Drought-duration	Off	1, 2, 3, 4, 5, 6, 7, 8, 9[years]	Length of drought

Appendix 10. Sagebrush % of landscape (y-axis) as a function of fire rotation (x-axis) grouped by fire size (2-50 ha) (top), reproductive maturity (2 or 4 years) (far right) and dispersal distance (1 grid cell = 10 m, 3 grid cells = 30 m) (near right).



Appendix 11. A state and transition model for "montane sagebrush upland" in the ST-Sim platform (funded in part by the LANDFIRE program); the latest version of version the Vegetation Dynamics Development Tool (VDDT), the Tool for Exploratory Landscape Scenario Analysis (TELSA), and the Path Landscape Model (Path). State and transition models are empirically based. Instead of explicitly accounting for seed limitation as in our model, a cell will transition to the next state as a function of time and not seed availability. STMs have proven useful for management planning. Our model could inform transition probabilities for a given fire regime. When fire sizes are large, the transition from "early open" to "mid open" is likely much longer than 12 years, especially when little unburned area remains in the burn interior and when recovery from seed bank fails given unfavorable weather. It would appear that hybrid approach using both process-based models and STMs could aid scientific decision making regarding land management and post-fire recovery plans.



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