Efficacy of Landscape Scale Oak Woodland and Savanna Restoration in the Ozark Highlands of Arkansas

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Abstract: The loss of historic ecosystem conditions has led forest managers to implement woodland and savanna ecosystem restoration on a landscape scale (≥10,000 ha) in the Ozark Plateau of Arkansas. Managers are attempting to restore and conserve these ecosystems through the reintroduction of disturbance, mainly short-rotation early-growing-season prescribed fire. Short-rotation early-growing season prescribed fire in the Ozarks typically occurs immediately before bud-break, through bud-break, and before leaf-out, and fire events occur on a three-to five-year interval. We examined short-rotation early-growing season prescribed fire as a restoration tool on vegetation characteristics. We collected vegetation measurements at 70 locations annually from 2011 to 2012 in and around the White Rock Ecosystem Restoration Area (WRERA), Ozark-St. Francis National Forest, Arkansas, and used generalized linear models to investigate the impact and efficacy of prescribed fire on vegetation structure. We found the number of large shrubs (>5 cm base diameter) decreased and small shrubs (<5 cm ground diameter) increased with prescribed fire severity. We found that horizontal understory cover from ground level to 1 m in height increased with time-since-prescribed-fire and woody ground cover decreased with the number of prescribed fire treatments. Using LANDFIRE datasets at the landscape scale, we found that since the initiation of a short-rotation early-growing season prescribed fire management regime, forest canopy cover has not reverted to levels characteristic of woodlands and savannas or reached restoration objectives over large areas. Without greater reductions in forest canopy cover and increases in forest-canopy cover heterogeneity, advanced regeneration will be limited in success, and woodland and savanna conditions will not return soon or to the extent desired.

Key words: Ozark Plateau, prescribed fire, restoration, savanna, woodland

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The aims of ecosystem restoration revolves around one central tenant, to return ecosystem conditions and functions to their historical or unaltered state after some significant change of those conditions has occurred (Hobbs and Norton 2006). When applied across large scales, restoration can conserve landscape characteristics and conditions necessary for system-adapted plant and animal species or communities. Previously, restoration at small scales such as the conservation of a single endemic species like the Table Mountain pine (*Pinus pungens*) in the Southern Appalachians has proven successful (Williams 1998), while restoration on the landscape scale such as in the Everglades ecosystem has been more of a challenge (Davis and Ogden 1994). Often these restoration efforts consist of the reintroduction of suppressed or altered natural disturbance regimes, such as fire or hydrologic cycles. Managers frequently try to restore ecosystems at the landscape level as this mimics the extent of historical ecosystem conditions, disturbances, and processes.

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Historically, oak woodlands and savannas covered large extents of central North America, creating a transition zone between closed canopy deciduous forest in the East and tall grass prairie in the Central Plains (Dyksterhuis 1957, Nuzzo 1986). Woodland and savanna ecosystems consisted of open canopies and diverse understories of grasses, forbs, and some woody shrubs (McPherson 1997, Anderson et al. 1999). Typically, fire disturbances prevented canopy closure, reduced shrub competition, promoted the presence and persistence of fire-adapted species, and maintained these ecosystems. After European settlement, conversion to agriculture and fire suppression significantly reduced the extent of oak woodlands and savannas. Nuzzo (1986) estimated that <1% of oak woodland and savanna ecosystems still exist in their historical conditions.

After the loss of woodland and savanna ecosystems managers realized that fire suppression and land use change had affected both plant and animal species that require the early successional and transitional characteristics of these systems. Managers are now attempting to restore former woodland or savanna sites to their historic structure and composition to create wildlife habitat, protect fire-adapted species, increase advance oak regeneration, and eliminate oak competitors (Sparks et al. 1998, Hutchinson et al. 2005). Restoration has mostly targeted sites currently in closed canopy forest with one or some combination of mechanical canopy removals, herbicide treatments, and/or prescribed fire (Jackson and Buckley 2004). In combination with restoration efforts, research studies have attempted to understand the role disturbance has in these systems and the effectiveness of restoration techniques. Many of these studies have addressed the reintroduction of fire, season of fire, intensity of fire, the effectiveness of fire as a system-maintaining disturbance, and the combination of fire and other restoration techniques (Sparks et al. 1998, Brose et al. 1999, Franklin et al. 2003, Jackson and Buckley 2004, Hutchinson et al. 2005, Albrecht and McCarthy 2006). Nuzzo et al. (1996), Sparks et al. (1998), and others have examined the understory vegetation responses to prescribed fire and oak woodland and savanna management. However, many of these research studies have addressed restoration and its components at relatively small scales (≤500 ha) compared to current landscape level implementation. Further, little research on how landscape scale restoration can affect forest structure commonly associated with wildlife habitat such as visual concealment has been completed.

The lack of substantial information on the restoration of oak woodland and savanna ecosystems at landscape scales motivated our study. We examined oak woodland and savanna restoration as implemented on a landscape level (≥10,000 ha) across multiple spatial and temporal scales to determine if this management strategy is restoring large areas of woodland and savanna ecosystem. Our objectives were to: (1) describe the vegetation structural changes over multiple spatial and temporal scales, (2) assess the effectiveness of current restoration techniques at a landscape level with a measurable criterion such as forest canopy cover, and (3) discuss potential obstacles, both natural and anthropogenic, to landscape-scale restoration efforts. We used both landscape scale vegetation data derived from various ground collected and remotely sensed sources and fine-scale vegetation data we collected on the study site. Our analyses cover immediate vegetation responses within the span of a single restoration treatment (≤6 yrs) and near-term responses to repeated treatments (~10 yrs). Our research should help managers make better decisions about the scale and techniques to consider using throughout the Ozark Plateau and other similar Central Hardwood regions when attempting to restore woodland and savanna ecosystems.

Study Area

The White Rock Ecosystem Restoration Area (WRERA) consists of 16,380 ha of upland hardwood and pine ecosystems in the Ozark Plateau of Northwest Arkansas. It is part of the larger main division of the Boston Mountain Ranger District (41,400 ha) on the Ozark-St. Francis National Forest (ONF). The WRERA is a high priority woodland and savanna restoration area for the U.S. Department of Agriculture (USDA) Forest Service. Historically the WRERA was dominated by woodlands and savannas maintained by frequent disturbances by fire (Foti 2004, Chapman et al. 2006, Guyette et al. 2006). After fire suppression in the 20th century, WRERA became dominated by closed canopy hardwood forests of various oak (*Quercus* sp.) and hickory (*Carya* sp.) species. Understories consisted of canopy species regeneration, blackgum (*Nyssa sylvatica*), flowering dogwood (*Cornus florida*), Carolina buckthorn (*Frangula caroliniana*), blackberry (*Rubus* spp.) and devil's walkingstick (*Aralia spinosa*) (USDA NRCS 2014). Pine ecosystem canopies were dominated by shortleaf pine (*P. echinata*) while understories consisted of hardwood and pine regeneration.

Since 2002, managers at WRERA have used large-scale (>500 ha in size) early growing season prescribed fire to restore historic woodland and savanna conditions. Management prescription for WRERA describes woodlands as having "open canopies, sparse mid-stories, and well-developed understories that are typically dominated by grasses and forbs but also may become shrubby between fires and have a significant woody component" with 40% to 60% canopy or overstory closure (Ozark NF Plan 2005). The prescription also calls for additional management techniques to achieve these objectives, including mechanical canopy removal, herbicide, and/or fire treatments. Mechanical and herbicide treatments have been mostly absent in the restoration area since 2002. The WRERA contains 16 prescribed fire units ranging from 467 to 1,670 ha in size. Early growing season prescribed fire occurs on a three-to five-year rotation in 15 of the 16 prescribed fire units with units receiving a range of one to four prescribed fire treatments since 2002. Prescribed fire units were burned in accordance to the ONF's Fire Management Plan and typically occurred from immediately before bud-break in late March to Mid-April just before full leaf-out. The ONF Fire Management Plan calls for the use of aerial ignition for large-scale prescribed fires and was intended to create low intensity fires. Each unit was allowed to burn as naturally as possible but ONF fire personnel would increase the number of ignition sources in areas where conditions prohibited or hampered the movement of fire. This practice typically resulted in >95% of a unit being burned.

Methods

Vegetation Data Collection

In our landscape scale analyses, we used 2001 and 2010 LANDFIRE vegetation datasets (Rollins 2009) for forest canopy cover. We used the raster calculator and focal statistics tools in ArcGIS to develop datasets for percent change between 2001 and 2010 for forest canopy cover and forest-canopy cover heterogeneity (ESRI 2013). Forest canopy cover heterogeneity is a metric derived from the forest-canopy cover datasets that represents the different values of percent forest-canopy cover immediately adjacent to a single location. A value of one indicates that there is no variability in forest canopy cover surrounding a location while values between one and eight indicate higher levels of variability surrounding a location. We incorporated all datasets into a geographic information system (GIS) with existing USDA Forest Service GIS data.

We collected fine-scale vegetation measurements at 70 locations during June and July from 2011 to 2013. We stratified locations by their time-since-prescribed-fire and cover type, and sampled each location once a year. We also collected vegetation measurements at reference locations with no history of fire according to the Impact/Reference Design of van Mantgem (2001). Vegetation measurements collected in 20 m diameter circular plots included four readings of horizontal understory cover board from 0 to 1 m in height (Nudds 1977), four estimates of percent ground cover type using 1 m^2 squares (Daubenmire 1959), tree counts, shrub counts, four measurements of understory height (m), forest canopy cover (Lemmon 1956), and a fire severity index (Cocking et al. 2014). Shrub classes included small shrubs, <5 cm ground diameter, and large shrubs, >5 cm ground diameter. The small shrub size category included advanced oak regeneration, a common measurement collected in studies examining the response of upland hardwood ecosystems to prescribed fire (Wendel and Smith 1986, Elliott et al. 1999. Tree categories were based on diameter at breast height (dbh) and were divided in three classes: small (<10 cm dhb), medium (10 to 20 cm dbh), and large (>20 cm dbh). We collected all tree and shrub counts using a line transect method on two 20-m transects within the plot. We calculated the fire severity index as the number of dead trees of medium and large classes still standing within the 20-m circular plot divided by the total of number of medium and large trees within the 20-m circular plot (Cocking et al. 2014). We only used dead standing trees with visible fire damage or scars to avoid attributing other sources of tree mortality such as wind-throw or ice damage as fire related mortality.

Data Analysis

We summarized all derived vegetation datasets for each of the 16 prescribed burn units. On these summarized data, we calculated the proportional area of each burn unit that fell into each respective vegetation variable cover/score category. We plotted these proportional areas for all prescribed burn units in bar plots to compare the distributions of area based on the number of prescribed fire treatments using package 'ggplot2' in program R (R Core Development Team 2014). We visually examined these plots for shifts in distributions explained by the number of prescribed fire treatments.

We used means and 95% confidence intervals for each ground-collected vegetation variable to determine if any trends or year effects existed, consistent with the Impact/Response Design (van Mangtem 2001). We fit generalized linear models (GLM) to each vegetation variable to determine what management factors influenced vegetation trends. Each model set consisted of 11 candidate models of non-collinear predictors including time-since-prescribed-fire (yrs), year the sample was collected (2011 to 2013), a plot fire severity index (zero to one), the number of prescribed fire treatments (one to three), and interactions. We used Akaike's Information Criterion (AIC_c) to rank candidate models and model averaged parameter estimates of the top model ($\Delta AIC_c \leq 2$) (Burnham and Anderson 2002). We only report model averaged parameter estimates for vegetation variable model sets that were within two ΔAIC_c of the top model. We performed all statistical analysis using R statistical language (R Core Development Team 2014).

Results

Landscape Scale

We examined the distributions of percent change in forest canopy cover for the 16 prescribed fire units and found in all units treated with prescribed fire that there was a negative shift in the percent change in forest canopy cover between 2001 and 2010 (Figure 1). During this same time, there was a positive shift in the percent change in forest canopy cover in the untreated unit (Figure 1). In the units treated with prescribed fire, the percent forest canopy cover dropped by about 25% (Figure 2). All of the treated units as of 2010 combined had approximately 1,000 ha (5%) of total area that met the ≤60% forest canopy cover criteria listed in management prescriptions for being considered a woodland. In 2001, before woodland restoration treatments began, a majority of each burn unit's areas had a canopy cover heterogeneity score of 2 ($\bar{x} = 58.2\%$, $\sigma = 2.5\%$). By 2010, after application of prescribed fire treatments, we observed a shift in the heterogeneity distributions of all units towards 1 (average proportion of unit area with a heterogeneity score of 1, $\bar{x} = 63.2\%$, $\sigma = 8.2\%$); i.e., the forest canopy cover either became more homogeneous or did not change. Therefore, the application of up to three prescribed fire treatments was not achieving the desired goal of producing woodland and savanna forested stands with ≤60% forest canopy cover that were heterogeneous in space.

Efficacy of Woodland and Savanna Restoration

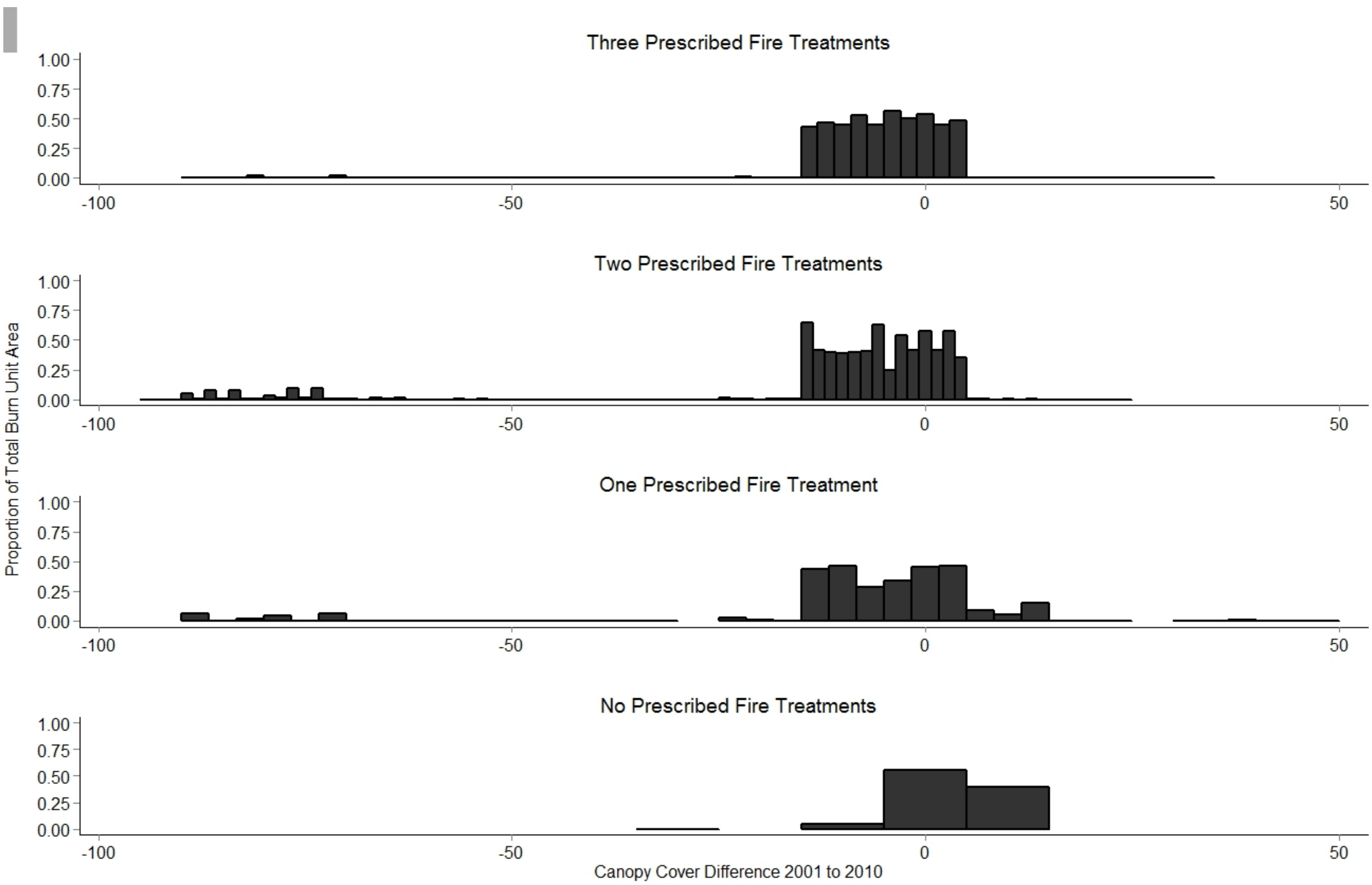

Figure 1. Average percent change in forest canopy cover (from 2001 to 2010) on 16 prescribed fire units on the White Rock Ecosystem Restoration Area, Arkansas. Units are divided based on the number of prescribed fire treatments received (0 treatments, $n = 1$; 1 treatment, $n = 3$; 2 treatments, $n = 7$; 3 treatments, $n = 5$).

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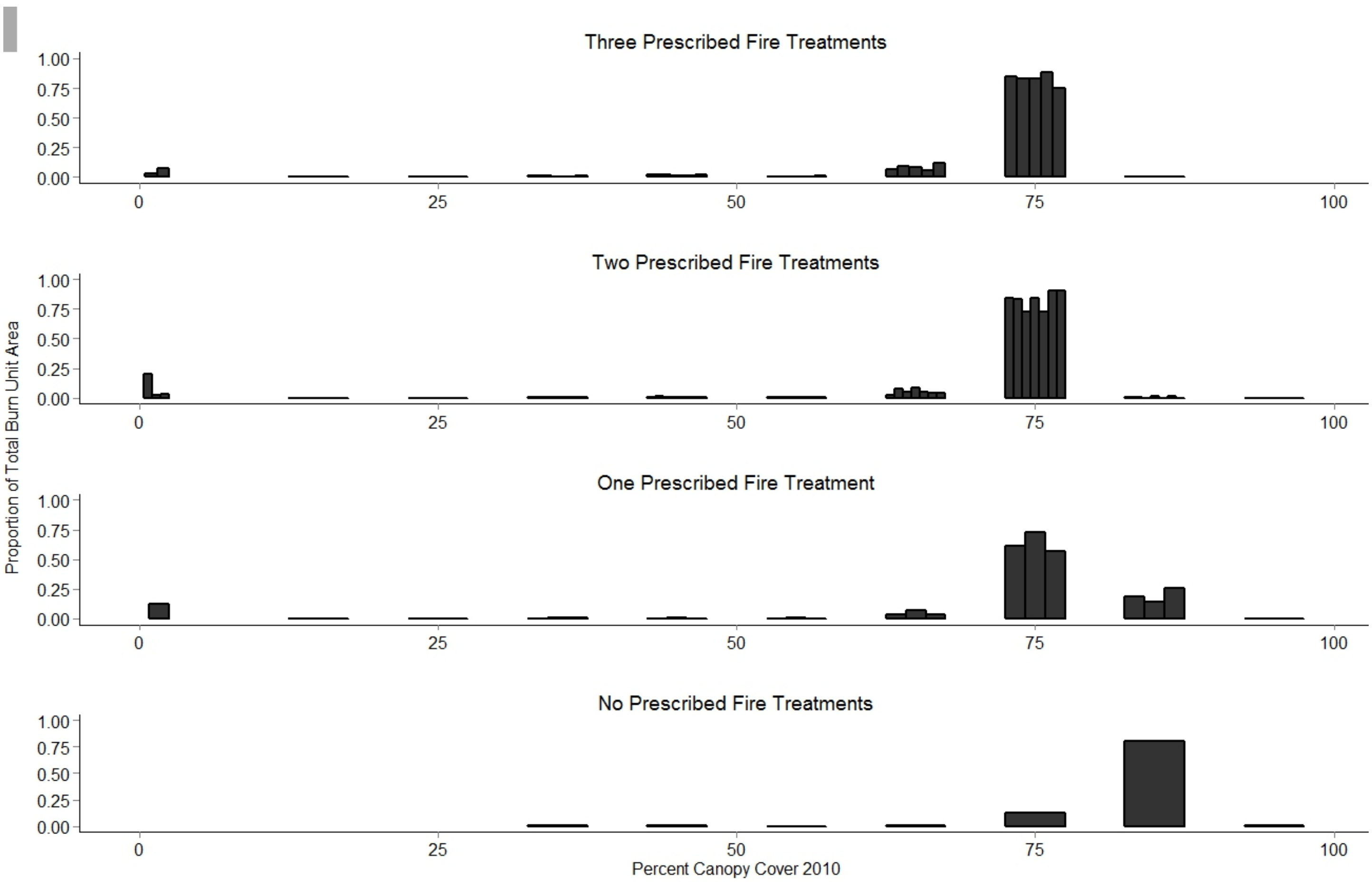

Figure 2. Percent forest canopy cover in 2010 on 16 prescribed fire units on the White Rock Ecosystem Restoration Area, Arkansas. Units are divided based on the number of prescribed fire treatments received (0 treatments, $n = 1$; 1 treatment, $n = 3$; 2 treatments, $n = 7$; 3 treatments, $n = 5$).

Fine Scale

Fine-scale vegetation measurements were variable both within and among years making general patterns difficult to discern (Table 1). Two general patterns were evident. First, the point estimates for horizontal understory cover (0–1 m), understory height, number of small shrubs, number of large shrubs, and percent grass cover during both the one to three and four to six years since being burned were generally greater in magnitude than during both the no burn and zero years since being burned. Second, the exceptions to the fire treatment pattern included horizontal understory cover (0 to 1 m), percent woody ground cover, and percent forb ground cover where there was an underlying pattern in the no burn locations of a decrease in cover across years. Finally, of the ground covers, the most consistently low values, both within and among years, was for grass cover.

Table 1. Means (SD) of vegetation variables by time-since-prescribed-fire from 2011 to 2013 collected on the White Rock Ecosystem Restoration Area, Arkansas.

Variable	Time since prescribed fire	Year 2011	2012	2013
Percent horizontal understory cover (0-1m)	No burn record	60 (29)	40 (25)	28 (33)
	0 yrs since burn	57 (28)	52 (36)	58 (32)
	1–3 yrs since burn	71 (32)	63 (36)	75 (30)
	4–6 yrs since burn	64 (24)	72 (28)	52 (27)
Understory height (m)	No burn record	0.87 (0.53)	0.58 (0.43)	0.68 (0.63)
	0 yrs since burn	0.85 (0.45)	0.68 (0.55)	0.85 (0.41)
	1–3 yrs since burn	1.17 (0.62)	1.1 (0.7)	1.27 (0.61)
	4–6 yrs since burn	1.08 (0.47)	1.31 (0.58)	0.8 (0.44)
No. small shrubs (≤5 cm)	No burn record	18 (14)	18 (12)	15 (12)
	0 yrs since burn	13 (11)	11 (13)	10 (11)
	1–3 yrs since burn	30 (27)	23 (16)	31 (25)
	4–6 yrs since burn	25 (13)	31 (20)	13 (9)
No. large shrubs	No burn record	4 (3)	6 (3)	4 (5)
	0 yrs since burn	3 (4)	1 (2)	2 (4)
	1–3 yrs since burn	7 (9)	3 (5)	10 (18)
	4–6 yrs since burn	6 (8)	7 (9)	3 (3)
No. medium trees	No burn record	3 (3)	3 (2)	8 (5)
	0 yrs since burn	4 (4)	3 (2)	3 (3)
	1–3 yrs since burn	3 (3)	3 (3)	3 (3)
	4–6 yrs since burn	5 (4)	3 (2)	3 (3)
Percent woody ground cover	No burn record	28 (28)	3 (6)	8 (10)
	0 yrs since burn	40 (29)	8 (16)	7 (14)
	1–3 yrs since burn	14 (23)	17 (27)	14 (26)
	4–6 yrs since burn	4 (10)	7 (20)	14 (24)
Percent grass ground cover	No burn record	2 (4)	2 (3)	0 (0)
	0 yrs since burn	0 (0)	0 (1)	0 (0)
	1–3 yrs since burn	2 (6)	4 (13)	4 (10)
	4–6 yrs since burn	9 (14)	0 (0)	3 (8)
Percent forb ground cover	No burn record	4 (19)	<1 (1)	0 (0)
	0 yrs since burn	11 (17)	10 (16)	17 (21)
	1–3 yrs since burn	13 (21)	12 (25)	14 (24)
	4–6 yrs since burn	12 (18)	8 (15)	3 (5)

Table 2. Model averaged parameter estimates for explanatory management variables (≤ 2 ΔAIC) from generalized linear models of each vegetation structure variable collected on the White Rock Ecosystem Restoration Area of the Ozark-St. Francis National Forest from 2011 to 2013.

Response variable	Explanatory variable	β	Lower 95% CI	Upper 95% CI
Horizontal cover	Fire severity	0.369	0.244	0.494
Understory height	Fire severity	0.369	0.243	0.495
	Time since fire: year 2013	−0.16	−0.315	−0.008
No. woody stems	Number of prescribed fires	−0.14	−0.275	−0.008
	Year 2012	−0.15	−0.297	−0.002
No. small shrubs	Fire severity	0.415	0.161	0.669
	Fire severity: year 2012	−0.22	−0.394	−0.051
No. large shrubs	Year 2013	−0.29	−0.469	−0.12
	Fire severity	0.233	0.004	0.462
	Fire severity: year 2013	0.502	0.237	0.767

Model averaged parameter estimates indicated visual concealment, understory height, the number of small shrubs and large shrubs increased as fire severity increased (Table 2). Therefore, as the severity of the prescribed fire increased, the ground and understory vegetation responded positively. The number of prescribed fires had a negative effect on woody stem counts suggesting that there was a cumulative effect of fires on killing woody stems. We found no effect of time since fire on any vegetation variable; however, in 2013, we found a negative interaction between horizontal understory cover (0 to 1 m) and time since fire, and a positive interaction between the number of medium trees and time since fire. On average, no vegetation variables responded to the time since a prescribed fire treatment in an orderly way with the two exceptions in 2013 of horizontal understory cover decreasing and the number of medium trees increasing. The only other effect that we found was for large shrubs to be fewer in number in 2013.

Discussion

Returning closed-canopy forests to woodlands and savannas has clear ecological consequences: 1) increased plant and animal species richness and diversity (Brawn 1998), 2) a change in fire ecology (changes in fire response times, fuel types and loads; Knapp et al. 2007), 3) changes in resource use patterns (water use and nutrient cycling; Franklin et al. 2003), and 4) a more resilient system to future disturbances including plant and animal invasions and climate change (Hutchinson et al. 2005). However, we will only realize the positive consequences if restoration efforts prove successful at restoring ecological communities and processes to the landscape. Our findings suggest that these positive consequences may not be achieved without more intensive management practices or longer durations of management implementation.

To revert these closed-canopy forests to their former ecological state at the landscape level requires aggressive management in both time and space to address forest structure, function, and composition (McElhinny et al. 2005). The usual list of management practices to accomplish these restoration efforts includes mechanical and herbicide treatment and prescribed fire. To implement any of these practices is challenging, but of the three approaches, prescribed fire is the least expensive, logistically easiest to implement, and can be manipulated in intensity and spatial extent (Hesseln 2000). The advantages of using prescribed fire are particularly important because public agencies like the U.S. Forest Service are targeting large tracts of closed-canopy forests for restoration. For example, the U.S. Forest Service initiated a Collaborative Forest Landscape Restoration Program (CFLRP) (USFS 2014) whose goals include, among others, to re-establish natural fire regimes in forested landscapes using restoration techniques that achieve ecological and watershed health objectives and makes those ecosystems more resilient. The implementation of the CFLRP at many sites across the United States is long-term (2009–2019) and well-funded ($40 million annually) (USFS 2014). The success of CFLRP and other similar programs elsewhere will depend on sound research and monitoring. Our study area, the WRERA, is a CFLRP project. Our research findings provide both research and monitoring information for the CFLRP and similar hardwood programs to reach their restoration goals. At WRERA, the USFS goal is to develop a woodland and savannah forest that has <60% forest canopy coverage with little midstory and a robust ground cover of grasses and forbs. After 10 years of restoration efforts, our findings do not indicate restoration has achieved these goals nor significant progress towards them on a landscape scale.

Since 2002, the primary tool used to restore WRERA has been early growing-season prescribed fire on a three-to five-year rotation. At the landscape scale in the canopy layer, after eight years of prescribed fire treatments and up to three treatments in a treatment unit, we found that neither percent forest canopy cover nor forest canopy cover heterogeneity approached targeted goals. For both of these vegetation variables, there was movement in the vegetation structure in the direction of the targeted goals, but vegetation structure did not meet those goals in either case. Our finding of no major change in forest canopy cover is consistent with other studies that found the combination of mechanical canopy removals and frequent fire, rather than fire alone, was necessary to produce canopy conditions most similar to woodlands and savannas (Franklin et al. 2003, Hutchinson et al. 2005). However, these supportive studies were at a smaller scale than our study. In natural systems, we found one example where both mechanical and fire disturbance were occurring in concert. Nangendo et al. (2005) documented that the combination of elephant disturbance (mechanical) and fire disturbance maintained and created woodland and savanna conditions in Uganda. It seems that prescribed fire alone is insufficient to convert closed-canopy upland hardwoods back to a functioning woodland and savanna system.

Our observed limited changes in canopy responses after as many as three prescribed fire treatments are consistent with the hypothesized (20+ years) duration of prescribed fire treatment by Hartman and Heumann (2003) in restoration of the Missouri Ozarks. Baker (1994) simulated the length of time to restore forest structure after fire suppression in northern Minnesota predicting a duration of 50 to 75 years. In the Missouri Ozarks, Shifley et al. (2006) found that even under the most intensive mechanical harvest regime used on public lands, changes in forest structure could take 75 to 120 years. The WRERA, with only the use of early growing season prescribed fire, is likely most similar to the moderate mechanical management scenarios simulated by Shifley et al. (2006) that required >100 years to produce overall shifts in forest structure. Admittedly these simulations were based on timber harvest, but a periodic early growing season prescribed fire regime and its impact on tree mortality could be considered analogous to moderate or low intensity uneven-aged harvest scenarios. In addition, variation in site characteristics such as geological substrate could further slow or result in variable changes to sites managed for woodland and savanna restoration in the Boston Mountains and Ozark Highlands (Foti 2004). Taking into account all of these factors, restoration of the woodland and savanna structure to the WRERA using prescribed fire alone might take from 25 years on ideal sites to more than 100 years on sites less suitable for restoration.

Forest canopy cover, or the resulting amount of light reaching the forest floor, is a major driver of understory vegetation dynamics; therefore, heterogeneity of forest canopy cover can indicate variability in understory vegetation structure and composition (Jennings et al.1999, Platt et al. 2006). Forest canopy cover heterogeneity for WRERA indicated that the landscape had become more homogenous since the implementation of restoration efforts. If prescribed fire alone was having the desired restoration effects, we would have expected the opposite of the observed outcome. The decrease in heterogeneity is likely due to a lack of canopy reductions from mechanical efforts and low intensity prescribed fires. Prescribed fires are implemented on the WRERA using aerial ignition, a method that has been found to be less intense than a typical head fire or a natural fire and results in few if any canopy openings (Johansen 1987, Price et al. 2007). Note though that the effects of aerial ignition techniques on landscape patch dynamics are under-studied and likely site-specific.

A lack of heterogeneity in forest canopy cover is cause for concern because it could be leading to a loss of forest biodiversity. Simberloff (1997) and Baumberger et al. (2012) considered structural heterogeneity in vegetation communities as a major factor affecting biodiversity of those communities. Whitlock et al. (2010) found that fire played a major role in shaping vegetation community heterogeneity at landscape scales. Our observed reduction in forest-canopy cover heterogeneity could result in less understory community heterogeneity and biodiversity. This hypothesized loss of biodiversity in the presence of fire finds support in African woodland communities where similar long-term fire regimes had a unifying effect on vegetation communities (Nangendo et al. 2005). Also Platt et al. (2006) found that understory biodiversity was linearly related to the amount of light transmission through the canopy layer to the forest floor in pine woodland and savanna ecosystems of the Gulf Coastal Plain. Our findings suggest under the current fire regime, landscape heterogeneity is declining and likely reducing the biodiversity of these forest communities. The reduction in landscape heterogeneity is concerning and will require a different management approach. Other researchers grappling with this same issue have recommended varying the fire regimes to resemble more closely natural fire regimes or taking an adaptive management approach to determine fire regimes to benefit the conservation of biodiversity and overall community health (Nangendo et al. 2005, Fisher et al. 2009, Lashley et al. 2014).

Beneath the canopy, no long-term metrics were available through LANDFIRE but we did collect vegetation measurements to examine the short-term effects (zero to six years since fire) of prescribed fire. We found the number of small and large shrubs, and average understory vegetation height were in general more abundant/robust after one to six years post-prescribed fire and all showed increases with fire severity. We also found that the number of prescribed fires was having a negative effect on the number of medium trees. Therefore, beneath the canopy and after 11 years of prescribed fire, the structure of the midstory vegetation (number of medium trees) was moving in the direction of the targeted goals, but there remained much variation in those measurements across the study area. However, the increases in number of shrubs replacing medium trees in the midstory are likely slowing or preventing the establishment of the desired herbaceous understory. Our results differ from the findings of other researchers in the Central Hardwoods region of North America (Hartman and Heumann 2003, Hutchinson et al. 2005) who found larger changes in hardwood forest structure at the mid- and understory levels in the Central Hardwoods region. However, these studies observed these changes in response to fire at much smaller scales than the landscape scale fire implemented on WRERA. Therefore our differences in mid- and understory response could be a result of varying responses to underlying environmental conditions across the landscape. These factors and the resulting variability more clearly reflect the natural behavior of fire on the landscape scale before human control.

Fine scale: Understory

While we found a small increase in forb cover after prescribed fire treatments, we observed no change in grass cover and a decrease in the coverage of woody vegetation that included woody vines. Either our observed small increase in forb cover and no increase in grass cover suggest those communities were not present in sufficient numbers before disturbance to benefit from the effects of fire or another limiting factor such as forest canopy cover is inhibiting their response (Platt et al. 2006). This lack of response may be a result of prescribed fire treatments alone not creating canopy gaps allowing enough sunlight to the forest floor to initiate a response by forbs and grasses. Other studies have found that more severe disturbances, such as mechanical canopy removals or more frequent or intense fire, are necessary to open the canopy and result in a response by the understory (Jennings et al.1999, Franklin et al. 2003, Hutchinson et al. 2005). Our observation of no major changes in forest canopy cover at the landscape level and increased small shrub counts supports the idea that light reaching the understory is limiting the herbaceous vegetation response (Jennings et al. 1999). Another potential explanation is the lack of an existing seed bank of understory species to be stimulated by prescribed fire treatments. Franklin et al. (2003) found that vegetation responses after prescribed fire treatments were dependent on the previous forest composition. Since there was an absence of a significant herbaceous component in the understory at our study site, due to fire suppression, Franklin et al.'s (2003) conclusion could explain the limited response of herbaceous ground cover. Rokich et al. (2002) also documented this same relationship between the success of restoration and the existing herbaceous community in banksia woodlands in Western Australia. In that case, broadcast seeding of woodland species was necessary to improve restoration effectiveness and achieve desired plant communities.

In 2012, we unexpectedly observed a decrease in horizontal understory cover and woody and herbaceous ground cover compared to in 2011 which we believe resulted from an extreme drought in summer of 2011 (index—D4 out of 5, U.S. Drought Monitor 2014). Vegetation response differs over a moisture gradient but few have documented the impact of severe drought on the vegetation response to prescribed fire treatments (Anning et al. 2014, Harmon et al. 1984, Hollingsworth et al. 2013). We observed the most severe drought effects immediately following prescribed fire treatments and in unburned units compared to units treated one to six years before sampling. We also documented a drought year effect on small shrubs and a delayed effect on large shrubs and medium trees. We observed this effect through the relationship between the drought year and fire severity indicating drought influences the response of vegetation to fire (Table 2). This relationship is important to consider in the future when assessing the short-term success of landscape restoration programs especially under changing climatic conditions.

Management Implications

We found the use of prescribed fire only for landscape-scale restoration of woodland and savanna ecosystems in the WRERA has not achieved desired vegetation goals at either the fine- or landscape-scale. Prescribed fire did increase advanced oak regeneration, which is one of the management goals. Without greater reductions in forest canopy cover and increases in forest-canopy cover heterogeneity, advanced regeneration will be limited in success, and woodland and savanna conditions will not return soon or to the extent desired. Managers may need to implement other restoration activities to create a less dense canopy such as, but not limited to, mechanical removals and herbicide treatments in combination with prescribed fire. However, the success or plausibility of such treatments at this landscape scale may be difficult or implausible. In either event, continued monitoring of forest conditions by mangers will be necessary to determine if management activities begin to create woodland and savanna conditions. Our findings demonstrate that it is important for managers to examine ecosystem restoration at the scale implemented and determine what scale may be most appropriate for accomplishing their objectives. In addition, studies such as ours should examine fine scale responses to restoration to understand the underlying mechanisms behind landscape-scale ecosystem restoration. Based on our results we emphasize to managers the importance of considering the scale of management in any ecosystem where the goal is to restore historical conditions and conserve species.

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