Effects of Prescribed Fire on Shinnery Oak (*Quercus havardii*) Plant Communities in Western Oklahoma

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Abstract

Changes in structural and compositional attributes of shinnery oak (*Quercus havardii* Rydb.) plant communities have occurred in the twentieth century. These changes may in part relate to altered fire regimes. Our objective was to document effects of prescribed fire in fall (October), winter (February), and spring (April) on plant composition. Three study sites were located in western Oklahoma; each contained 12, 60 × 30-m plots that were designated, within site, to be seasonally burned, annually burned, or left unburned. Growing season canopy cover for herbaceous and woody species was estimated in 1997–1998 (post-treatment). At one year post-fire, burning in any season reduced shrub cover, and spring burns reduced cover most. Winter and annual fires increased cover of rhizomatous tallgrasses, whereas burning in any season decreased little bluestem cover. Perennial forbs increased with fall and winter fire. Shrub stem density increased with fire in any season. Communities returned rapidly to pre-burn composition with increasing time since fire. Fire effects on herbaceous vegetation appear to be manifested through increases in bare ground and reduction of overstory shrub dominance. Prescribed fire can be used as a tool in restoration efforts to increase or maintain within and between community plant diversity. Our data suggest that some plant species may require or benefit from fire in specific seasons. Additional research is needed to determine the long-term effects of repeated fire over time.

Key words: plant community restoration, shrub ecology, *Quercus*.

Introduction

Shinnery oak (*Quercus havardii* Rydb.)-dominated communities cover approximately 2 million ha in western Oklahoma, northern Texas, and southeastern New Mexico (Peterson & Boyd 1998). Cultivation and herbicidal control efforts have reduced the spatial extent of the shinnery range (Peterson & Boyd 1998) and fragmented its remaining distribution (Dhillion et al. 1994). This loss of shinnery habitat is compounded by lack of spread of existing communities (McIlvain 1954; Gross & Dick-Peddie 1979) and the apparent lack of regeneration after successive years of cultivation. Historically, shinnery oak was reported as growing to heights of about 50 cm (Marcy 1854) and tallgrasses formed the overstory of the plant communities (Osborne 1942). In contrast, shinnery oak may comprise up to 80% (Dhillion et al. 1994) of the canopy cover of communities on the present-day landscape, tallgrasses have apparently decreased in abundance, and, in western Oklahoma, oak stems may approach or exceed 1 m in height (Fig. 1).

The apparent changes in structure of shinnery oak communities during the twentieth century may relate to concomitant alterations of fire regime and livestock grazing. Limited research on fire effects in shinnery communities suggests that fire in spring may increase abundance of forbs and grasses (McIlvain & Shoop 1965; McIlvain & Armstrong 1966), top kill a high percentage of shinnery oak stems, and induce vigorous oak resprouting (Slosser et al. 1985). Additionally, fire can increase the amount of bare ground, which has been shown to correlate positively with herbaceous seedling density (Dhillion et al. 1994) and recruitment (Holland 1994).

The pre-European fire return interval for shinnery oak communities is not known. However, in Oklahoma, the fire return interval for neighboring tallgrass prairie has been estimated at 5 to 10 years (Wright & Bailey 1982). Similarities in herbaceous (i.e., fine fuel) composition and fuel ignition sources (e.g., lightning, Native Americans) between tallgrass and shinnery oak communities in Oklahoma lead us to believe that fire may have had an influential role in the development of the shinnery oak ecosystem in western Oklahoma.

Management and restoration of remaining shinnery oak communities depends on an adequate knowledge

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Figure 1. Unburned western Oklahoma shinnery oak community photographed in June 1997.

of their ecological structure and function. Currently, little is known of the fire ecology of these communities, and there has been no published work on the effects of growing season or winter fire on shinnery oak communities. Evaluating the role of fire in this system requires assessing the effects of fire timing, frequency, and behavior on plant community dynamics. Boyd (1999) previously explored the influence of fire behavior on plant response in shinnery oak communities. The objective of this research is to evaluate the role of season of fire, time since fire, and annual fire on plant composition of shinnery oak communities in western Oklahoma and to provide baseline information for determining the role of fire in restoration efforts.

Methods

Study Sites

Study sites were located on the Black Kettle National Grassland in Roger Mills County, Oklahoma (35° 32′ 44″ N, 99° 43′ 39″ W) and the state-owned Packsaddle Wildlife Management Area in Ellis County, Oklahoma (36° 4′ 22″ N, 99° 54′ 5″ W). Sites were chosen subjectively to be representative of shinnery oak communities found on sandy soils within the western Oklahoma region. All sites were lightly grazed by cattle during the growing season before study initiation but were excluded from grazing in 1995 and throughout the course of the study. Before our study, these sites had not burned on a regular basis and had been free of fire for at least 10 years.

Soils were fine sands (Nobscott-Brownfield Association) with no limiting layers in the top 150 cm (USDA 1982). Shinnery oak, a deciduous clonal species, was the dominant shrub with lesser amounts of sand sagebrush (*Artemisia filifolia* Torr.), Oklahoma plum (*Prunus gracilis* Engelm.), fragrant sumac (*Rhus aromatica* Ait.), and netleaf hackberry (*Celtis reticulata* Nutt.). Dominant grasses and forbs included little bluestem (*Schizachyrium scoparium* Nash), indiangrass (*Sorghastrum nutans* Nash), switchgrass (*Panicum virgatum* L.), sand bluestem (*Andropogon gerardii* Hack), sand lovegrass (*Eragrostis trichodes* Nutt.), sideoats grama (*Bouteloua curtipendula* Michx.), western ragweed (*Ambrosia psilostachya* DC.), erect dayflower (*Commelina erecta* L.), and sundrop (*Calylophus berlandieri* Spach). Nomenclature followed that of Barkley (1986). Average annual precipitation was 65.6 cm; growing season (March–August) precipitation averaged 40.6 cm (USDA 1982).

Experimental Design

We divided each of the three study sites (blocks) into 12 60 × 30-m plots. Plots were arranged in a 2 × 6 matrix and separated by 7-m firebreaks. We randomly assigned each of the plots within a site to the following nine treatments: (1) remain unburned, (2) burn fall 1996, (3) burn fall 1997, (4) burn winter 1997, (5) burn winter 1998, (6) burn winter 1997 and 1998, (7) burn spring 1997, (8) burn spring 1998, and (9) burn spring 1997 and 1998 (Table 1). The number of unburned plots was 21 and 6, for the 1997 and 1998 growing seasons, respectively. Pre-treatment data were collected during the growing season in 1996 and treatment response data during the growing season in 1997 and 1998.

Fire Ignition and Behavior

All plots were burned using a strip-headfiring technique (Wright & Bailey 1982). The downwind and flank

Table 1. Year, season of fire, burning date, and sample size for experimental plots in western Oklahoma.

Year	*Season*	*Burning Date*	n
1996	Fall	Oct. 23–24	3
1997	Winter	Feb. 4–5	6
1997	Spring	Apr. 28–29	6
1998	Fall	Oct. 1	3
1998	Winter	Jan. 27–28	6
1998	Winter (annual)	Jan. 27–28	3
1998	Spring	Apr. 30–May 1	6
1998	Spring (annual)	Apr. 30–May 1	2

sides of the plots were ignited and allowed to burn about 5 m into the plot. We then ignited a series of headfires about 10 m upwind from the backfire. All burns were conducted with relative humidity more than 20%, air temperature less than 29°C, and a surface wind speed of less than 16 km/hr. We estimated fire behavior characteristics for all headfires and determined pre-burn fuel loading and fuel consumption from quadrats clipped before and after burning. Fire behavior and fuel characteristics are discussed in Boyd (1999).

Vegetation Sampling

Because of the ignition pattern, the outer 5 m of plots were excluded from vegetation sampling to eliminate differential effects of headfires, backfires, and flankfires. We estimated canopy cover and percent bare ground for each plot, by species, at 30 randomly located points (Daubenmire 1959). At each point, canopy cover of each species influencing a 20 × 50-cm quadrat was categorized as 0 to 5%, 5 to 25%, 25 to 50%, 50 to 75%, 75 to 95%, or 95 to 100%. We averaged mid-point values to obtain an estimate of canopy cover of each species in a plot for a given sampling period. We estimated canopy cover during three sampling periods: 25–31 May, 6–22 June, and 8–17 August.

We created summary variables to represent the sum of all canopy cover values for a given vegetation class in a given plot and year (Table 2). Average seasonal canopy cover values for vegetation classes were calculated by averaging canopy cover values by plot, class, and year (West & Reese 1996). Our purpose was to combine species that respond similarly to environmental perturbation and reduce data to a meaningful level for analysis and presentation. Annual and perennial forbs may respond positively to fire (McIlvain & Armstrong 1966), but because annual forbs may be more sensitive to other environmental factors (Bazzaz & Morse 1991), they were grouped separately. Legumes (woody and non-woody) were grouped because they often respond positively to fire (Towne & Knapp 1996) and because of their ability to fix nitrogen in the nitrogen dynamic post-fire environment (Pyne et al. 1996). Rhizomatous C_4 tallgrasses were grouped because of their similar reproductive strategy and their generally positive response to fire (Towne & Owensby 1984). Little bluestem was classified by itself because it was the dominant grass species in unburned plots. Additionally, the bunchgrass growth form of little bluestem differed from other dominant grasses, which were mainly rhizomatous, and little bluestem often declines after fire (Towne & Owensby 1984; Ewing & Engle 1988). All remaining perennial grasses, predominantly bunchgrasses, were grouped together. Dominant species in this grouping included sideoats grama, sand lovegrass, and sand dropseed (*Sporobolus cryptandrus* Torr.). All other shrub species were grouped and represent the most abundant vegetation class. The only sedge species encountered (*Cyperus schweinitzii* Torr.) was classified by itself.

We estimated post-fire shrub stem density in September 1997 and 1998 by counting the number of aboveground stems present in 10 randomly located 0.50-m^2 quadrats per plot. We defined stems as shrubs that had a unique aboveground base. Values for individual species were summed to determine total shrub stem density for purposes of statistical analysis.

Table 2. Shinnery oak community vegetation classes and dominant species.

Vegetation Class	*Dominant Species*
Annual forbs	*Conyza canadensis* L., *Monarda punctata* L., *Pyropapus carolinianus* Walt.
Perennial forbs	*Ambrosia psilostachya*, *Calylophus berlanderii*, *Commelina erecta*
Legumes	*Amorpha canescens* Pursh., *Desmodium sessilifolium* Torr., *Lespedeza stuevei* Nutt.
Little bluestem	*Schizachyrium scoparium*
Tallgrasses	*Andropogon gerardii*, *Panicum virgatum*, *Sorghastrum nutans*
Other grasses	*Bouteloua curtipendula*, *Eragrostis trichodes*, *Sporobolus cryptandrus*
Sedges	*Cyperus schweinitzii*
Shrubs	*Artemisia filifolia*, *Prunus gracilis*, *Quercus havardii*

Nomenclature follows Barkley (1986).

Statistical Analysis

We assessed treatment effects on plant canopy cover at the community and individual vegetation class scales. Changes in community composition were evaluated using multivariate analysis of covariance (Stroup & Stubbendieck 1983; SAS Institute Inc. 1988; Fuhlendorf & Smeins 1998) with canopy cover of vegetation classes as the dependent variables and season of fire, time since fire, or annual fire as main effects. Pretreatment canopy cover values were used as covariables in the model. Treatment significance was evaluated using the p value associated with the Wilks' lambda (Johnson & Wichern 1992) test statistic for the treatment variable effect. To test for differences in vegetation class canopy cover values between years, we used the above multivariate model with response period year (1997 and 1998) as the independent variable; this analysis included unburned plots only. Because of a significant year effect ($p < 0.01$), we analyzed 1997 and 1998 data separately.

We also evaluated treatment effects on individual vegetation classes using analysis of covariance with vegetation class as the dependent variable; season of fire, time since fire, or annual fire as the independent variable; and pretreatment vegetation class score as the covariate (SAS Institute Inc. 1988). Treatment effects on shrub stem density were analyzed using univariate analysis of variance with density, by year, as the dependent variable and season of fire, time since fire or annual fire as the independent variable (SAS Institute Inc. 1988). When significant model and treatment variable effects were found, we used multiple comparisons (least square difference, $\alpha = 0.10$) to detect differences between treatment means. Model and treatment effects were considered significant at $p = 0.10$.

To increase specificity of our univariate and multivariate analysis of variance models, we used response data measured only during the first year post-fire for the season-of-fire and annual-fire models. In the time since fire model, unburned and annually burned plots were excluded, and only data from the 1998 growing season were used because data two years post-fire were not available in 1997. Unburned plots were also excluded from the annual-fire model, and only data from the 1998 growing season were used because data from successive fires were not available in 1997. One plot slated to be burned annually did not have sufficient fuel for ignition in successive years and was excluded from the annual-fire analysis.

We evaluated the effects of bare ground on canopy cover for individual vegetation classes using Pearson correlation analysis (SAS Institute Inc. 1988). For this analysis, we combined all response data and analyzed data for burned plots separately from unburned plots to assess any fire-induced alterations in the influence of bare ground on vegetation class abundance. All p values less than 0.01 are reported as "$p < 0.01$" and values equal to or greater than 0.01 are reported exactly.

Results

Season of fire influenced the community composition of vegetation classes in 1997 and 1998 ($p < 0.01$; Table 3). Univariate analysis (Table 4) revealed that shrubs decreased in cover after fire in any season in both 1997 and 1998 ($p < 0.01$). Spring burns decreased shrub cover more than any other burn season; a decrease of over 50% relative to unburned plots (Fig. 2). Little bluestem cover decreased with fire in any season in 1997 and 1998 ($p < 0.01$). Cover of tallgrasses was influenced by season of fire in 1997 ($p < 0.01$) and was higher for winter-burned plots than unburned. Tallgrass cover was unaffected by burning season in 1998 ($p = 0.2742$). Other grasses were unaffected by season of fire in 1997 ($p = 0.97$) or 1998 (p = 0.37). Annual forb abundance was influenced by burning season in both 1997 ($p = 0.03$) and 1998 ($p < 0.01$), with fall and winter burns producing highest abundance. Season of fire influenced perennial forb abundance in both 1997 and 1998 ($p < 0.01$), and cover was greatest in winter and fall-burned plots. Legumes were not affected by burning season (1997, $p = 0.94$; 1998, $p = 0.34$). Sedges were influenced by burning season in 1997 and 1998 ($p < 0.01$), generally increased with burning treatment, and were most abundant in spring-burned plots.

Several forb and legume species were limited in occurrence to only one or two burning seasons. For instance, toad flax (*Linaria canadense* L.) was only found in unburned plots, blue false indigo (*Baptisia australis* L.) in fall and spring-burned plots, sleepy daisy (*Aphanostephus ridellii* T. & G.) in winter-burned plots, and purple

Table 3. Multivariate analysis of variance p values for the effect of season of fire (unburned, fall, winter, or spring), time since fire (1 vs. 2 years), and annual fire (annual vs. single-event fire) on vegetation class composition of experimental plots in western Oklahoma.

Year	*Treatment Variable*	n	p *Value*[a]
1997 (1 year post-fire)	Season of fire	36	0.0001
1998 (1 year post-fire)	Season of fire	21	0.0001
1998 (1 and 2 years post-fire)[b]	Time since fire	25	0.0094
1998 (1 year post-fire)[c]	Annual fire	20	0.0702

[a] p value is for the Wilks' lambda test statistic.
[b] Unburned and annually burned plots excluded. Analysis includes only data from the 1998 growing season because data 2 years post-fire were not available in 1997.
[c] Unburned plots excluded. Analysis includes only data from the 1998 growing season, because data from successive fires were not available in 1997.

Table 4. Mean canopy cover scores and standard errors for vegetation classes by year and fire treatment for experimental plots in western Oklahoma.

			Vegetation class							
Year	*Treatment Variable*	n	*Shrubs*	*Little Bluestem*	*Other Grasses*	*Tallgrasses*	*Annual Forbs*	*Perennial Forbs*	*Legumes*	*Sedges*
			% Canopy Cover							
1997 (1 year post-fire)	Season of fire									
	Unburned	21	74.4 ± 2.0 a[a]	41.9 ± 2.6 a	17.7 ± 1.3	21.7 ± 2.3 b	1.7 ± 0.3 b	9.0 ± 1.0 c	0.2 ± 0.1	0.11 ± 0.04 c
	Fall	3	56.5 ± 10.2 b	22.6 ± 9.0 b	19.2 ± 5.6	29.1 ± 6.3 ab	1.2 ± 0.4 b	18.1 ± 2.8 a	0.3 ± 0.3	0.43 ± 0.35 b
	Winter	6	60.3 ± 4.7 b	24.7 ± 4.3 b	16.4 ± 2.8	32.5 ± 7.0 a	3.3 ± 1.3 a	13.9 ± 2.7 b	0.4 ± 0.4	0.27 ± 0.14 cb
	Spring	6	30.4 ± 4.2 c	24.8 ± 4.7 b	16.2 ± 2.3	29.5 ± 9.9 ab	1.1 ± 0.5 b	11.5 ± 2.3 cb	0.5 ± 0.3	0.79 ± 0.33 a
1998 (1 year post-fire)	Season of fire									
	Unburned	6	73.8 ± 6.0 a	46.3 ± 4.8 a	8.0 ± 1.5	17.5 ± 3.2	0.7 ± 0.3 b	3.4 ± 0.6 b	0.2 ± 0.2	0.01 ± 0.01 c
	Fall	3	60.5 ± 9.2 b	18.9 ± 7.0 c	7.5 ± 1.5	20.3 ± 7.3	5.1 ± 1.8 a	8.1 ± 0.5 a	0.4 ± 0.3	0.19 ± 0.11 b
	Winter	6	58.1 ± 1.5 b	30.5 ± 3.5 b	12.4 ± 2.2	26.0 ± 6.7	0.7 ± 0.3 b	7.8 ± 1.6 a	0.3 ± 0.1	0.19 ± 0.10 b
	Spring	6	31.9 ± 2.5 c	20.2 ± 3.1 c	8.4 ± 1.7	20.8 ± 4.7	1.4 ± 0.4 b	4.1 ± 1.1 b	0.1 ± 0.1	0.32 ± 0.14 a
1998 (1 and 2 years post-fire)	Time since fire[b]									
	One year	15	48.1 ± 4.0 b	24.1 ± 2.6 b	9.8 ± 1.2 b	22.8 ± 3.4	1.9 ± 0.6	6.4 ± 0.9	0.3 ± 0.1	0.24 ± 0.07 a
	Two years	10	66.2 ± 4.1 a	43.9 ± 5.7 a	17.8 ± 2.0 a	23.4 ± 4.4	4.0 ± 1.6	7.2 ± 1.0	0.2 ± 0.2	0.03 ± 0.02 b
1998 (1 year post-fire)	Annual fire[c]									
	Single event fire	15	48.1 ± 4.0	24.1 ± 2.6	9.8 ± 1.2	22.8 ± 3.4 b	1.9 ± 0.6	6.4 ± 0.9	0.3 ± 0.1	0.24 ± 0.07
	Annual fire	5	45.6 ± 9.2	30.1 ± 3.3	14.1 ± 1.6	38.9 ± 8.1 a	3.4 ± 1.5	7.6 ± 1.0	0.7 ± 0.5	0.07 ± 0.04

[a] Means within a year and treatment variable with no letters or without different letters are not significantly different (LSD) at $\alpha = 0.10$.
[b] Unburned and annually burned plots excluded. Analysis includes only data from the 1998 growing season because data 2 years post-fire were not available in 1997.
[c] Unburned plots excluded. Analysis includes only data from the 1998 growing season because data from successive fires were not available in 1997.

A

B

Figure 2. (A) Western Oklahoma shinnery oak research plot in August 1997, 4 months after spring prescribed fire. Perennial grasses dominate the post-burn community. The measuring board is 120 cm in height. (B) Close-up of shinnery oak in same plot, August 1997. Oak re-sprouts are 30 to 40 cm tall.

coneflower (*Echinacea angustifolia* DC.) in fall- and winter-burned plots.

Time since fire influenced ($p < 0.01$) community vegetation class composition in burned plots (Table 3). Univariate analysis (Table 4) indicates that shrub abundance increased with time since fire ($p = 0.01$); in fact, abundance of shrubs in plots 2 years post-fire was comparable with unburned plots in 1998. Cover of little bluestem ($p < 0.01$) and other grasses ($p < 0.01$) increased with time since fire, and the year 2 post-fire mean for little bluestem was similar to 1998 unburned plots. Sedge abundance decreased with increasing time since fire ($p = 0.01$). Tallgrass abundance was affected by time since fire ($p = 0.01$); however, the mean separation test revealed no difference between 1 and 2 years post-fire. Annual forbs ($p = 0.17$), perennial forbs ($p = 0.12$), and legumes ($p = 0.74$) were unaffected by time since fire.

Annual fire also influenced community vegetation class composition ($p = 0.07$; Table 3). Univariate analysis revealed that shrubs ($p = 0.93$), little bluestem ($p = 0.24$), other grasses ($p = 0.28$), legumes ($p = 0.27$), and sedges ($p = 0.18$) were not influenced by annual fire, relative to plots burned only once. Tallgrasses ($p < 0.01$) increased in annually burned plots and were the only vegetation class in which abundance differed between single event and annually burned plots. The model p values for annual ($p = 0.04$) and perennial ($p = 0.02$)

Table 5. Correlation coefficients of vegetation class canopy cover and percent bare ground for experimental plots in western Oklahoma.

			Vegetation Class							
Treatment	*n*	*Average Bare Ground (%)*	*Shrubs*	*Little Bluestem*	*Other Grasses*	*Tallgrasses*	*Annual Forbs*	*Perennial Forbs*	*Legumes*	*Sedges*
			Correlation Coefficient							
Unburned	27	6.6 ± 0.8	−0.398*	0.431*	0.110	0.421*	0.702*	−0.173	0.493*	−0.001
Burned	45	48.6 ± 2.0	−0.158	−0.298*	−0.256**	0.233	0.047	0.030	0.225	−0.029

Data are combined across years post-fire (1 or 2 years) and year of data collection (1997 or 1998).
* $p < 0.05$.
** $p > 0.05$ and ≤ 0.10.

forbs were significant; however, mean separation tests revealed no differences ($p > 0.10$) due to annual fire.

Correlation analysis indicated that bare ground was associated negatively with shrub cover and associated positively with cover of little bluestem, tallgrasses, annual forbs, and legumes in unburned plots (Table 5). In burned plots, bare ground was associated negatively with cover of little bluestem and other grasses. Shrub stem density was influenced by season of fire in both 1997 and 1998 ($p < 0.01$; Fig. 3). Burning in any season increased the density of shrub stems, and spring-burned plots had lower stem density than fall or winter burns in 1998. Model p values for annual fire ($p = 0.08$) and time since fire ($p = 0.10$) were significant, but mean separation tests revealed no differences in density means.

Discussion

Our results are somewhat unique because late-growing season (fall) burns did not reduce shrub cover more than other burning seasons; other oak species are greatly reduced by growing-season fire (Fergusson 1961; Glitzenstein et al. 1995). This may relate to the fact that our burns were near the end of the growing season (October), after the period of peak carbohydrate storage in the dominant shrub, shinnery oak (Boo & Pettit 1975). The relatively strong reduction in shrub canopy cover after spring fire may also relate to carbohydrate storage. Boo and Pettit (1975) reported that the low point in the carbohydrate cycle of shinnery oak was during the first 2 weeks of May, when leaf expansion was 50–75%. Our spring burns corresponded to this time period. At the time of our spring burns, leaf expansion of the two dominant shrub species (shinnery oak and Oklahoma plum) was about 50%. McIlvain and Armstrong (1966) noted a similar response to spring burning of shinnery communities in Ellis County, Oklahoma. The positive association of time since fire with shrub abundance is indicative of adaptations of the dominant shrub, shinnery oak, to disturbance. Top kill of shinnery oak stems approached 100% for all burns, but shinnery oak reproduces vegetatively from rhizomes (Muller 1951) and resprouts vigorously in response to fire (Slosser et al. 1985), thus enhancing its persistence in the community after fire disturbance. In our study, burning in any season nearly doubled shrub stem density compared with

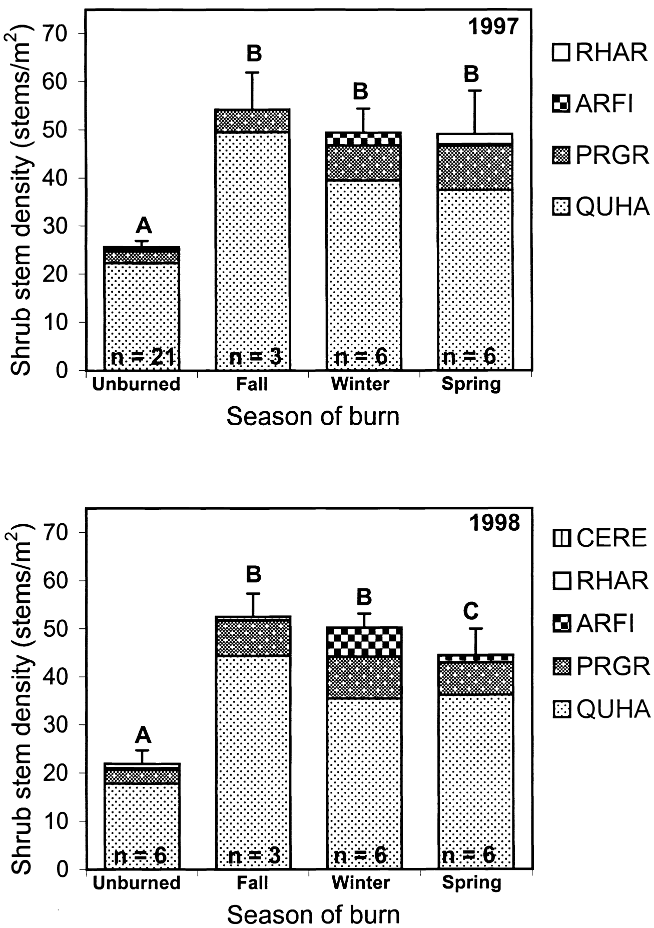

Figure 3. Shrub stem density (stems/m^2) by year and season of fire for experimental plots in western Oklahoma. Data were collected for plots 1 year post-fire in 1997 and 1998. Bars within year without a common letter are significantly different (least square difference) at $\alpha = 0.10$. Shrubs include shinnery oak (QUHA), Oklahoma plum (PRGR), sand sagebrush (ARFI), fragrant sumac (RHAR), and netleaf hackberry (CERE).

unburned plots (Fig. 3). The negative correlation of shrubs with bare ground in unburned plots (Table 5) reflects the fact that bare ground decreases with increasing shrub cover (i.e., increased oak leaf litter deposition), not that bare ground is a controlling factor of shrub abundance.

The response of grasses and sedges to fire was characterized by an increase in sedges with burning in any season and an increase in the ratio of tallgrasses to little bluestem canopy cover with burning in any season. Other authors have noted like responses of little bluestem to fire (e.g., Towne & Owensby 1984; Ewing & Engle 1988), which may relate to the non-rhizomatous growth form of little bluestem, which makes its basal growing points more susceptible to fire damage than rhizomatous co-dominants. Cover of little bluestem and other perennial grasses increased, whereas that of rhizomatous tallgrasses remained unchanged with increasing time since fire (Table 4). This suggests that frequent fire may be necessary for the maintenance of the rhizomatus tallgrass component of shinnery communities. In support of this generalization, canopy cover of rhizomatus tallgrasses increased in annually burned plots relative to plots burned only once (Table 4). Sedge abundance increased after seasonal fire (Table 4), suggesting that fire treatment is beneficial to the abundance of this species. The positive association of sedges with the sampling year is related to a relatively wetter spring growing period in 1997 (Fig. 4); sedges are cool-season species and actively grow during spring (Kindscher & Wells 1995). Coppedge et al. (1998) found a similar response of sedges and rushes to spring precipitation in a tallgrass prairie system.

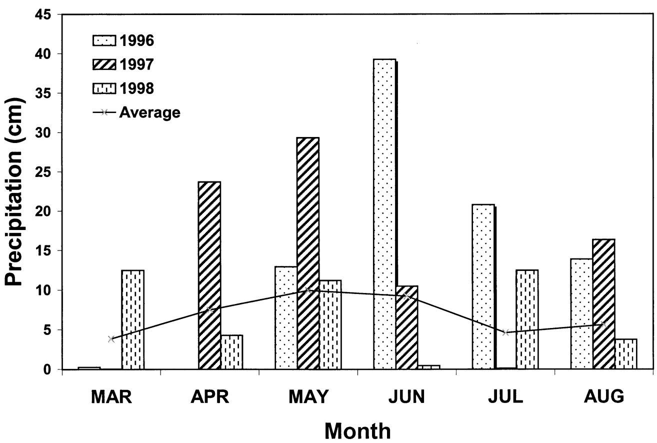

Figure 4. Growing season precipitation by month during the study period shown with long-term average precipitation data from USDA (1982). Total growing season (March–August) precipitation was 87.1 cm in 1996, 79.9 cm in 1997, and 44.6 cm in 1998, compared with a long-term average of 40.6 cm.

In general, cover of annual and perennial forbs (Table 4) increased with burning treatment in fall and winter relative to unburned plots. Western ragweed showed particular affinity for burned plots and often dominated the post-burn forb community. The lack of a spring fire effect on forb canopy cover may relate to the timing of burning relative to plant morphological development. Plots burned in fall were burned before growth initiation of cool and warm season forbs, winter burns coincided with the growing period of some-cool season forb species, whereas spring burns coincided with active growth periods for cool and warm season forbs. In burned plots, annual variation in the canopy cover of annual and perennial forbs (Table 4) was associated with variable growing season precipitation (Fig. 4). Perennial forb abundance was highest with increased growing season precipitation (1997), whereas in fall and spring-burned plots, annual forbs were most abundant with decreased growing season precipitation (1998) (perhaps due to decreased competition with perennial forbs). Increases in annual forbs associated with burning also may relate to increased availability of germination sites (i.e., increased bare ground; Trabaud 1987) and decreased phytochemical inhibition from perennial woody species or litter (Menges & Kimmich 1996). Phytochemical inhibition of annual or perennial forb species has not been reported in shinnery oak communities, but Matizha and Dahl (1991) reported strong reductions in shoot growth of weeping lovegrass (*Eragrostis curvula* Schrad) with application of leaf extract from shinnery oak.

We suggest that the influence of fire on the herbaceous component of shinnery oak communities was strongly related to alterations in availability of bare ground and decreased overstory shrub dominance. Most ground not covered by basal plant material in unburned plots was covered with oak leaf litter to depths of about 8 cm, and little bluestem, tallgrasses, annual forbs, and legumes were correlated positively with increasing bare ground (Table 5). Fire reduced negative effects of leaf litter on herbaceous abundance by increasing availability of bare ground. Tallgrasses, annual forbs, and legumes no longer correlated with bare ground after fire (Table 5). The negative correlations after fire between bare ground and little bluestem and other perennial grasses may relate to competition with rhizomatous tallgrasses in the post-fire environment. Dhillion et al. (1994) reported that density of herbaceous seedlings in a shinnery oak community was correlated positively with increasing bare ground. In undisturbed shinnery oak communities, the shrub component also created a fairly continuous canopy cover, which may act to decrease light availability and microclimate diversity at the ground level. Burning temporarily decreased the shading effects of overstory shrubs, thus creating light gaps in the canopy that may benefit shade intolerant herbaceous species (Nasser & Goetz 1995; Bowles & McBride 1998). Holland (1994) reported that recruitment of herbaceous seedlings decreased with increasing cover of shinnery oak.

Conclusions and Restoration Implications

Prescribed fire in any season can decrease shrub canopy cover, increase cover of forbs, and alter the grass community composition. Winter and annual fire can increase the canopy cover of rhizomatous perennial grasses, whereas little bluestem cover often decreases with fire in any season. Annual and perennial forb cover may increase with burning in any season, but fall fire generally increases forb cover the most. Although vegetation responds differentially to fire in different seasons, other fire-related factors such as increases in bare ground and reduction of shrub canopy cover may influence plant community dynamics regardless of season of burn.

Although the historical literature indicates that shinnery oak was a low-growing species with tallgrasses in the overstory, these same references provide little insight into either the historical fire regime or the heterogeneity of community types across the landscape. Although it is clear that some communities were structured in this way, the lack of reference to other structural forms of shinnery oak communities does not preclude their existence. In that context, the present day shinnery-dominated communities across much of western Oklahoma may not be out of place relative to the structure of pre-European settlement communities. However, their current prevalence at large spatial scales does not agree with historical descriptions and is likely an artifact of management practices, including grazing without deferment and minimizing the occurrence of fire. Reintroduction of fire at the appropriate spatial scales should help increase between-community diversity by reducing overstory oak dominance and increasing cover of rhizomatous tallgrasses and forbs, particularly during wet years. Fire in different seasons can be important to this process, not only due to the differential effects of fire season on the abundance of a given vegetation class, but also because individual species may affiliate with particular fire regimes. Recall that several forb species encountered in our experimental plots were only found in one or two treatments, suggesting that these species may be limited to areas with a particular disturbance regime that creates microclimate conditions necessary for germination and recruitment.

Other authors have stressed the temporary nature of fire-induced changes in shinnery oak communities (e.g., Slosser et al. 1985) and in other shrub-dominated systems (Parsons 1976; Trabaud & Lepart 1980). In fact, our data indicate that shinnery oak communities are very elastic and show signs of rapid return to pre-burn plant composition after fire. What we do not know at this time is the impact of repeated fire over longer time scales (e.g., fire occurrence every 7 years for the next 70 years). There is no evidence that plots burned in our study have been on a reliable fire cycle in the previous century, and thus it is possible that repeated fire over time will produce different results than noted in our study. The effects of plowing may serve as an instructive example. Plowed firebreaks around our experimental plots produced a thick canopy of shinnery oak re-sprouts during the growing seasons after disturbance. However, abandoned old fields in the same area, which were plowed repeatedly until the 1930s, are largely void of shinnery oak, even though dense stands of oak often surround the field. Despite the lack of data on fire effects over time, it can be argued that the immediate results of burning treatment may be important for a variety of short-term management goals, such as increasing abundance of forbs important for certain wildlife species or of perennial grass species important to ground-nesting birds, such as the imperiled Lesser Prairie Chicken (*Tympanúchus pallidicínctus*). Additional research is needed to assess long-term community response to fire and determine the effects of fire treatment at various spatial scales and seasons on landscape level diversity.

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