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Intense wildfire in southeastern Arizona: transformation of a Madrean oak–pine forest to oak woodland

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Abstract

Fire exclusion in southwestern USA and northern Mexico has decreased fire frequency and increased fire intensity in many forest types. I examined the effects of intense crown fire in a Madrean oak–pine community in southeastern Arizona, an ecosystem characterized by frequent, moderate-intensity fire during presettlement times. I sampled seedling establishment and sprouting response of crown-killed pines and oaks after the 10,330-ha Rattlesnake wildfire of 1994 and a smaller intense fire. About 90% of the *Quercus hypoleucoides* sprouted from top-killed stems. Sprouting occurred in none of the *Pinus engelmannii* and 13–23% of the *P. leiophylla*, a low level that nevertheless may help maintain this species after intense fire. Seedlings were very rare for all species, especially *P. engelmannii*. Height of pine sprouts and seedlings lagged far behind that for *Q. hypoleucoides*. Results of this case study suggest that anomalous crown fires radically changed community structure and composition for the long-term, from a lower canopy of oaks with large, emergent pines to a more homogeneous oak woodland. Such transformation, if repeated across the landscape, could seriously threaten the maintenance of biological diversity in these ecosystems.

Keywords: Biodiversity; Community composition; Crown fire; Madrean; Oak–pine; Rattlesnake fire; Sprouting

1. Introduction

Fire is a key process in forests and woodlands throughout southwestern North America and northern Mexico. Until the late 1800s, most of these ecosystems were subject to frequent, low intensity fires through which much of the original stand persisted (Covington and Moore, 1994; Fulé and Covington, 1994; Swetnam et al., 1999, 2001). Fire frequency in this region has declined precipitously in the 20th century, largely as a result of active fire suppression and intensive livestock grazing (Bahre, 1991; Covington and Moore, 1994; Fulé and Covington, 1996; Swetnam and Baisan, 1996; Swetnam et al., 2001). Reduced fire frequency has led to build-up of live and dead fuel (Leopold, 1937; Marshall, 1962; Cooper, 1960; Covington and Moore, 1994; Minnich et al., 1995; Fulé et al., 1997; Barton, 1999), which has increased the probability of large, intensive, stand-replacing fires (Covington and Moore, 1994; Fulé and Covington, 1996; Dahms and Geils, 1997; Swetnam and Baisan, 1996; Swetnam et al., 1999, 2001). The last decade has, in fact, seen an irruption of crown fires that have destroyed houses, purportedly damaged ecosystems, and blazed into the headlines (e.g. Ffolliott and Bennett, 1996; Fulé et al., 2000; Swetnam et al., 2001).

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A key question is to what extent the apparent shift from frequent, low intensity fires to infrequent crown fires will lead to irreversible transformations of these plant communities. This case study addresses this question by examining the consequences of the 1994 Rattlesnake fire and an earlier crown fire for tree community composition of a mixed oak–pine Madrean forest in the Chiricahua mountains in the Arizona–Mexico borderlands. With the possible exception of spruce-fir forests at the highest elevations, the Madrean forests and woodlands of this region supported a presettlement fire regime of frequent, low intensity fires (Swetnam et al., 1989, 1992, 2001; Swetnam and Baisan, 1996). In 1994, however, a lightning strike in the Chiricahua mountains grew into one of the largest and most intense crown fires on record in the borderland region. The Rattlesnake fire burned 10,330 ha between 28 June and 24 July, from high elevation spruce-fir forest at around 2400 m elevation down to Madrean oak woodland at 1800 m. During the first 3 weeks of the fire, very dry, warm weather and high winds at times created extreme fire conditions with widespread crowning, torching as high as two to three times the height of trees, and upslope runs approaching 1 km/h. Six million US dollars were spent fighting the fire (Coronado National Forest, USDA Forest Service, unpublished data).

Two pine species (Apache pine, *Pinus engelmannii*, and Chihuahua pine, *P. leiophylla*) and several oak species (mainly silverleaf oak, *Quercus hypoleucoides*) dominate the lower elevation communities examined in this study. These pines and oaks contrast sharply in their response to fire (see Barton, 1995, 1999; Fulé and Covington, 1998). The oaks tend to suffer top-kill, but sprout prolifically in response to the moderate-intensity fires typical of these ecosystems, whereas the pines exhibit a high degree of resistance to such fires but only modest sprouting capacity (restricted to Chihuahua pine). Fulé and Covington (1998) and Barton (1999) predicted that, as a result of these generic contrasts, high intensity fires resulting in massive crown-kill could lead to the long-term transformation of Madrean oak–pine forests to oak woodlands. I address this hypothesis by examining the degree to which pines and oaks replaced themselves after crown-kill by post-fire sprouting and seedling establishment.

2. Study site

This study was carried out in the Chiricahua mountains, which are located in southeastern Arizona in Cochise county (31°52′N, 109°15′W), and are considered part of the Sierra Madre ‘archipelago’ (DeBano et al., 1995). The mountains extend southeast to northwest for about 80 km and rise from about 1100–3000 m altitude. The climate is semi-arid, with two wet seasons, one July–September, when more than 50% of total precipitation falls, and the second December–March. A pronounced dry season usually occurs between the final winter storms in March or April and the onset of the rainy season in July (Sellers et al., 1985).

This study focused on oak–pine forest, classified as Mexican oak–pine woodland subtype #3 by Brown (1982), a community type that covers major canyon bottoms and sideslopes. Study sites for the Rattlesnake fire were in Cave Creek Canyon on the east side of the mountains, from an altitude of about 1800–2000 m (Fig. 1). I also sampled at a second fire site, the 23-ha Methodist fire, which burned from moderate to high intensity in 1983 between 1800 and 1850 m in Pine Canyon on the west side of the mountains (Fig. 1). Most of the tree biomass in these two areas consisted of Chihuahua pine, Apache pine, and silverleaf oak, but also included Arizona madrone (*Arbutus arizonica*), Arizona white oak (*Q. arizonica*), and other less common tree species (see Barton, 1994). Nearby fire history studies suggest that moderate-intensity fires occurred every 5–15 years in this community type in the Chiricahua mountains until the 1880s. At that point, fire exclusion and increased livestock grazing drastically reduced fire occurrence (Swetnam et al., 1989, 1992, 2001; Swetnam and Baisan, 1996). Presettlement fires generally occurred at the beginning of the summer rainy season, sparked by lightning from storms producing little rainfall (Swetnam et al., 2001).

3. Methods

Sampling was carried out in areas that met the following criteria: (1) slope and aspect relatively homogeneous; (2) vegetation dominated by silverleaf oak, Chihuahua pine, and Apache pine; and (3) trees

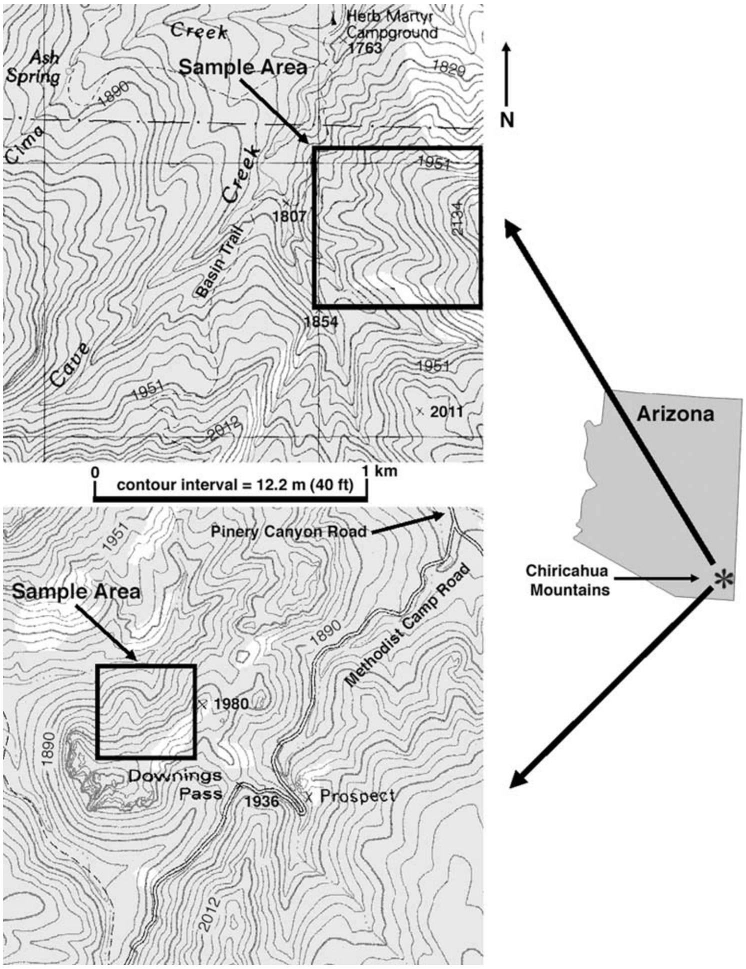

Fig. 1. Maps of the two study sites: the Rattlesnake fire area, in Cave Creek Canyon on the east side of the Chiricahua mountains (top), and the Methodist fire area, in Pine Canyon on the west side of the mountains (bottom).

exhibiting 100% crown-kill (i.e. complete mortality or top-kill). Extensive areas met these criteria in the lower portions of the Rattlesnake fire. Twelve randomly-selected $50\,m \times 5\,m$ belt transects were sampled in April 1999 in such areas in Cave Creek Canyon near the lower parts of the Basin Trail that runs from Herb Martyr Campground to Pine Park. One continuous large area met the sampling criteria in the Methodist fire. Four randomly-selected $50\,m \times 5\,m$ belt transects were sampled there in March 1993. In contrast to the higher elevations of the Rattlesnake fire, both fires burned patchily in lower elevation Madrean communities, resulting in a wide range of fire intensities (Barton, personal observation). It is important to emphasize that this study focuses specifically on lower elevation Madrean communities that were subject to stand-replacing, high-intensity crown fire.

Within each belt transect, I sampled the following: species identity (based on distinguishing traits not modified by fire) of all crown-killed stems at least 1.4 m in height; whether or not each crown-killed stem sprouted after the fire; diameter at breast height (DBH) (at 1.4 m height); and the number of seedlings. In the Rattlesnake fire only, I also sampled the height (to the nearest 5 cm) of the tallest sprout for each top-killed tree and the height of seedlings. Live stems <1 m in height were assumed to be seedlings if no top-killed tree of the same species was within 0.5 m. This approach was verified for several seedlings by digging up entire root systems. For Chihuahua pine, however, the origin of several small stems was problematical because of this species' habit of continual sprouting and dying-back, even in small, young stems. Problematical stems were arbitrarily coded as seedlings. Silverleaf oak often grows in sprout clumps of two to five stems. Such clones were counted as one individual (a genet), although all group individuals were measured and combined for basal area calculations. Some stems had lost part or all of their bark since the fire. To add the diameter attributable to missing bark, bark was coded as either completely missing or half-missing. DBH was then added according to this determination using species-specific DBH–bark thickness relationships (Barton, unpublished data).

The retrospective sampling design used in this study allowed the estimation of pre-fire basal area and abundance, as well as response and recovery from fire from sprouts and seedlings. This protocol works well for damaged or killed stems that persist after the fire, but obviously cannot include pre-fire stems entirely or mostly consumed by the fires. As a result, the sampling design probably is accurate for larger stems, but may underestimate mortality in smaller stems. This potential bias should have only trivial effects on estimates of changes in pines versus oaks from before to after the fires.

4. Results

Before the 1994 Rattlesnake fire, the 12 sample sites were dominated in terms of both basal area and number of stems by silverleaf oak, Chihuahua pine, and Apache pine (Table 1). Silverleaf oak accounted for nearly half of the stems and each pine species a little more than 20% in all plots combined. The pines, which attain much larger girths than silverleaf oak, made up more than 80% of the basal area, whereas silverleaf oak accounted for about 11%. Pre-fire species composition was similar for the Methodist fire site, although relative abundance and basal area was higher for Apache pine and silverleaf oak and lower for Chihuahua pine (Table 1).

After the Rattlesnake fire, degree of sprouting from crown-killed stems differed strongly among the three main species: silverleaf oak was very successful ($\bar{x} = 95.8\% \pm 1.2$, $N = 12$), Chihuahua pine exhibited a low level of sprouting ($\bar{x} = 30.9\% \pm 6.8$, $N = 12$), and Apache pine did not sprout. Sprouting in silverleaf oak was significantly higher than that for Chihuahua pine (arcsine transformed data: $t_{22} = -11.0$, $P = 0.0001$). Table 2 shows these data for these and other species for all plots combined for each site. Differences in degree of sprouting in the Methodist fire were very similar to those for the Rattlesnake fire (Table 2).

Only 70 seedlings (233.3 per ha) were found in the 12 plots combined for the Rattlesnake fire study area

Table 1
Mean (±S.E.) abundance and basal area per ha of tree species before fire for the Rattlesnake fire (12 plots) and the Methodist fire (4 plots)

Species	Rattlesnake fire				Methodist fire			
	Abundance (ha)		Basal area (m^2/ha)		Abundance (ha)		Basal area (m^2/ha)	
	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.
Silverleaf oak	1147[a]	64.6	8.1	0.9	600[a]	84.9	10.4	1.3
Arizona white oak	37[a]	24.4	0.2	0.1	30[a]	19.5	0.1	0.7
Chihuahua pine	547	119.4	27.1	5.9	150	28.3	3.9	0.8
Apache pine	510	68.1	32.1	6.0	400	43.2	16.0	2.0
Arizona madrone	67	19.3	2.9	1.1	0	0	0	0
Other[b]	207	42.0	1.3	0.3	150	44.3	2.1	0.6
Total	2513[a]	158.0	71.8	2.7	1330[a]	59.7	32.6	1.8

[a] Clonal groups (usually 2–5) counted as single individuals.
[b] Includes Douglas-fir (*Pseudotsuga menziesii*), border pinyon (*Pinus discolor*), alligator juniper (*Juniperus deppeana*), net-leaf oak (*Quercus rugosa*), and trees unidentified because of extreme charring from fire.

Table 2
Response of tree species after crown-kill, 4.5 years after the 1994 Rattlesnake fire (0.3 ha sampled) and 10 years after the 1983 Methodist fire (0.1 ha sampled) for all plots combined in each site[a]

Species	Rattlesnake fire				Methodist fire			
	Pre-fire	Post-fire			Pre-fire	Post-fire		
	Stems	Sprouting	%	Seedlings	Stems	Sprouting	%	Seedlings
Silverleaf oak	344	309	89.8	19	60	56	93.3	3
Arizona white oak	11	11	100.0	26	3	2	66.7	0
Chihuahua pine	164	38	23.2	18	15	2	13.3	4
Apache pine	153	0	0.0	5	40	0	0.0	3
Arizona madrone	20	16	80.0	1	0	0	0.0	0
Other[b]	62	0	0.0	1	3	1	33.0	1
Total	754	374	49.6	70	121	61	50.4	11

[a] Data are pre-fire number of individuals, number of these individuals that sprouted after crown-kill, percentage sprouting, and number of seedlings counted after the fire. Mean, variability, and statistical tests are provided in the text for the major species.

[b] Includes Douglas-fir (*P. menziesii*), border pinyon (*P. discolor*), alligator juniper (*J. deppeana*), net-leaf oak (*Q. rugosa*), and trees unidentified because of extreme charring from fire.

(Table 2). Seedling numbers were similarly low for silverleaf oak and Chihuahua pine, and were especially low (16.7 per ha) for Apache pine. Seedling densities were even lower for the Methodist fire area (110 per ha; Table 2). Although small seedlings might have been overlooked, the burned area generally exhibited large areas of bare soil, few seedlings of any species, and little continuous grass cover.

Sprouts of silverleaf oak were significantly taller than those of Chihuahua pine ($t_{349} = 6.4, P < 0.0001$; Fig. 2). Seedlings of silverleaf oak were more than twice as tall as those of Chihuahua pine ($t_{35} = 3.5$, $P = 0.001$; Fig. 2). Sprouts of both species were significantly taller than their respective seedlings (silverleaf oak: $t_{33} = 11.8$, $P < 0.0001$; Chihuahua pine: $t_{44} = 6.0$, $P < 0.0001$). Seedling height for Apache pine was especially low ($\bar{x} = 12.6 \pm 6.0$).

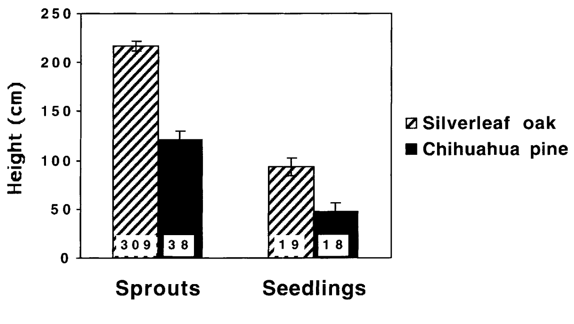

Fig. 2. Mean (±S.E.) height of sprouts and seedlings of silverleaf oak and Chihuahua pine 4.5 years after the Rattlesnake fire. Sample sizes are given at the base of the bars.

5. Discussion

The pre-fire data collected here describe a community with a largely continuous lower canopy of small silverleaf oaks and larger, more widely-spaced emergent pines (also see Barton, 1994, 1999). Stand-replacing crown fire radically changed this structure and composition. By sprouting prolifically from top-killed stems, silverleaf oak has recovered well from the fires, and now dominates these sites in both abundance and basal area. In contrast, pine abundance has plummeted, and current seedling numbers reveal little promise of recovery (Fig. 3). Change was especially marked for Apache pine, which was reduced from a dominant structural element of this community to a rare seedling.

Past studies of Chihuahua pine revealed low levels of sprouting from top-killed trees after moderate-intensity fire, levels dwarfed by the typical high survival percentage of mature trees (Barton, 1995, 1999). This low level of sprouting may, however, take on special importance in high intensity fires such as the Rattlesnake fire, where most mature trees have perished, where establishment conditions may be rigorous, and where seed availability may be restricted

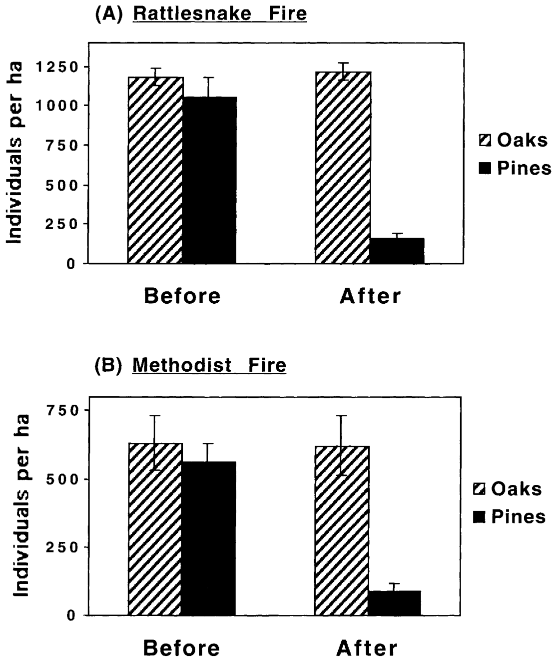

Fig. 3. Mean (±S.E.) for the number of oaks and pines before fire (individual trees) and after fire (sprouted trees plus seedlings) for (a) the 12 plots in the Rattlesnake fire of 1994 and (b) the 4 plots in the Methodist fire of 1983.

by distance to seed sources. Clearly, the 38 sprouts from top-killed Chihuahua pines provide this species with a strong head start in the Rattlesnake fire area in comparison to the non-sprouting Apache pine.

The high levels of silverleaf oak sprouting demonstrate that latent buds of this species have a tremendous capacity to survive high intensity fire. The 89–93% sprouting after top-kill is higher than that found for silverleaf oak in five moderate-intensity fires (36–65%; Barton, 1999) and higher than that for Emory oak (*Q. emoryi*) and Arizona white oak (*Q. arizonica*) in the high-intensity Peak fire of 1988 also in southeastern Arizona (Ffolliott and Bennett, 1996; also see Caprio and Zwolinski, 1995). Higher levels of sprouting for silverleaf oak than for the other two oaks was also found by Barton (1999), although the difference was not as large as found here. These discrepancies may be the result of higher fire intensity in the Peak fire or of higher levels of consumed and therefore unsampled smaller stems in the Rattlesnake and Methodist fires.

The low level of seedling establishment for all tree species may be the result of fire-caused loss of soil organic matter, litter, and nutrients, modified soil thermal regime resulting from loss of cover, or restricted seed availability due to the large size of the fire (Viro, 1974; Campbell et al., 1977; DeBano and Conrad, 1978; Neary et al., 1996; Fulé et al., 2000). As the habitat recovers, establishment conditions may gradually improve and seedling establishment of pines and oaks increase. It is also possible that seed production in the 4 years since the fire have been low for reasons associated with or independent of the fire, and that seed production and seedling establishment will increase in a future year. In fact, many pines of southwestern USA and northern Mexico exhibit a low background level of establishment with rare episodes of extensive seedling establishment. This is usually the result of a combination of masting, previous seed bed preparation by fire, and adequate moisture for several years (White, 1985; Savage et al., 1996; Mast et al., 1999; Barton et al., 2001). Nevertheless, success of pine seedlings establishing in the future may be curtailed by the well-established stand of silverleaf oak. The few pine sprouts and seedlings extant in the plots already must contend with much taller silverleaf oak sprouts and seedlings with proportionally greater access to light and soil resources.

Fires of the intensity and patch size of the Rattlesnake fire are not evident in the extensive fire history record for the Chiricahua mountains or for the region (Swetnam et al., 1989, 1999, 2001; Swetnam and Baisan, 1996). Lower intensity fires typical of Madrean oak–pine forests appear to maintain a balance of pines and oaks over time: fires prevent oak dominance by temporarily favoring the more resistant pines, but the resilient oaks recover rapidly via fast-growing sprouts (Barton, 1995, 1999; Fulé and Covington, 1998). As demonstrated in this case study, intense crown fires have the potential to convert these mixed forests to more homogeneous oak woodlands, with recovery times possibly on the order of centuries rather than decades. It may be a generalization that, in communities of sprouting and obligate seeders (or near obligate, in the case of Chihuahua pine), anomalously large, intense, or frequent fires may lead to long-term shifts in favor of sprouters (e.g. Rebertus et al., 1994; Fulé and

Covington, 1998; Russell-Smith et al., 1998), shifts outside the range of historical reference conditions (e.g. Moore et al., 1999; Swetnam et al., 1999). Given their large size, copious seed production, persistence as coarse woody debris, and long coevolution with other species, pines may well play a keystone role in these communities, and their loss could broadly reverberate among associated species. In the Rattlesnake fire, such damage to Madrean oak–pine communities is minor given the patchy nature of the fire at lower elevations. Wholesale shifts to a regime of infrequent, intense crown fires, however, should be of great concern to ecologists and managers. This threat highlights the need for research identifying strategies for reducing the probabilities of such stand-replacing fires in Madrean ecosystems (see Fulé and Covington, 1998).

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