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www.publish.csiro.au/journals/ijwf *International Journal of Wildland Fire*, 2006, **15**, 203–211

Relationships between prescribed burning and wildfire occurrence and intensity in pine–hardwood forests in north Mississippi, USA

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Abstract. Using Geographic Information Systems and US Forest Service data, we examined relationships between prescribed burning (from 1979 to 2000) and the incidence, size, and intensity of wildfires (from 1995 to 2000) in a landscape containing formerly fire-suppressed, closed-canopy hardwood and pine–hardwood forests. Results of hazard (failure) analyses did not show an increased likelihood of large, small, or intense wildfires with an increase in the number of years since the last prescribed fire. Wildfires of various sizes and intensities were more likely to occur in years with lower than average precipitation, regardless of when these areas were last burned. Calculations of expected lightning-fire potential based on weather patterns predicted a peak in lightning-started fires in the early to late summer. Lightning fires were rare, however, and wildfire activity was greatest in the spring and fall. We hypothesize that the ineffectiveness of prescribed burning in reducing wildfire hazard and the low incidence of wildfires in the midsummer in north Mississippi are both artifacts of fire suppression in the past, which converted open oak–pine woodlands with persistent pyrogenic surface fuels that accumulated over time to closed-canopy forests that lack such fuels. We suggest that open canopies and grass-based surface fuels must first be restored before prescribed burning will achieve most desirable management goals in this region, including hazard reduction and ecological restoration of natural fire regimes.

Additional keywords: ecological restoration; fire management; fuel management; lightning; oak forests; survival or failure analysis.

Introduction

Throughout much of North America, land managers use prescribed burning to manage fragmented fire-dependent ecosystems. Prescribed fire is an important tool for managing fuels to reduce the ignition, size, and severity of wildfires (Fernandes and Botelho 2003). In addition, several have argued for using prescribed burning to restore fire-dependent ecosystems (Noss 1989; Mutch 1994; Leach and Givnish 1996; Laatsch and Anderson 2000). Many plants and animals in fire-dependent ecosystems survive frequent fires and some even depend on certain types of fires for successful reproduction (Platt 1999). The actions taken to manage fire regimes have a direct impact on society, and the conservation of biodiversity and maintenance of ecosystem function (Mutch 1994).

Rough-reduction burning (i.e. controlled burning of the surface fuels in forests) is currently commonly used to reduce wildfire hazard in the national forests of the south-eastern USA (Wade and Lunsford 1989; Haines *et al.* 2001). In general, repeated burning of the surface fuels appears to reduce the size or severity of wildfires in coastal plain ecosystems of the south-eastern United States (Martin 1988; Wade and Lunsford 1989; Brose and Wade 2002). Containment and suppression of wildfires are easier when fuels and potential flame lengths (fire intensity levels) are reduced. Hence, with proper control of fuel loads, relatively few personnel and minimal equipment are required to directly suppress wildfires and exclude them from sensitive areas (e.g. pine plantations), thereby minimizing costs and potential damages associated with wildfires. However, the likelihood of wildfires varies significantly between pine-dominated and hardwood-dominated ecosystems (Zhai *et al.* 2003). In addition, fire behavior varies greatly between open-canopy forests and closed-canopy forests and between hardwood forests that experience fire at different times of the year (Deeming *et al.* 1977). Policies regarding the use of prescribed burning are often implemented throughout one to several administrative units (e.g. one to several national forests within a state or region; Haines *et al.* 2001). An appropriate use of prescribed fire in one ecosystem, however, may not be appropriate for other ecosystems within the same administrative unit.

Surface fuels in closed-canopy forests are largely composed of leaf litter from deciduous trees or short-needle pines,

 10.1071/WF05068 1049-8001/06/020203

which are not as flammable as grass-based or long-needle-based fuels in pine savannas (Williamson and Black 1981; Platt *et al.* 1991). Furthermore, deciduous leaf litter in forests of the southern USA decomposes rapidly during the growing-season months each year (Deeming *et al.* 1977) and thus may not accumulate from one year to the next years. The effectiveness of prescribed burning in reducing wildfire hazard needs to be examined more closely in closed-canopy upland forests with a significant hardwood component.

The characteristics of fire-suppressed, closed-canopy forests that preclude effective control of hazardous fuel loads may also prevent effective ecological restoration. Not unlike longleaf pine savannas of the coastal plain of southeastern USA, upland oak and oak–pine communities of the interior plains and highlands of this region are fire-dependent ecosystems (Van Lear and Waldrop 1989; Foti and Glenn 1990; Batek *et al.* 1999; Brewer 2001). Prescribed burning during the lightning season is thought by some to aid restoration efforts. These fires have been shown to have numerous ecological benefits, including more effective control of fire-sensitive invasive species (Ferguson 1961; Glitzenstein *et al.* 1995; Barnes and Van Lear 1998; Brose *et al.* 1999; Drewa *et al.* 2002) and increased reproduction of some fire-dependent plant species (Parrott 1967; Streng *et al.* 1993; Brewer and Platt 1994). Nevertheless, most studies of current and past fire regimes in oak and oak–pine communities in the Midwest and the interior southern USA indicate that wildfires are more commonly started by humans than by lightning (Martin 1989; Guyette and Cutter 1991; Batek *et al.* 1999). Furthermore, the rapid decomposition of surface fuels from deciduous trees in closed-canopy forests after leaf flush in the spring reduces fuels during the peak lightning season (Deeming *et al.* 1977), thereby potentially reducing the incidence of lightning fires in the summer. There is increasing doubt about whether burning alone (regardless of season) is sufficient to restore oak and oak–pine ecosystems that have experienced a long history of fire suppression (Arthur *et al.* 1998; Brose *et al.* 1999; Laatsch and Anderson 2000). Regardless of how common (or uncommon) lightning fires are now or were in the past, given the potential for lightning-season fires to maintain biodiversity and restore natural regeneration of oaks and pines, the prospects for creating conditions that would favor effective lightning-season burning in these forests need to be investigated.

To date, no one has attempted to determine the peak lightning-fire season in north Mississippi. Recent records of lightning-fire activity in highly fragmented and fire-suppressed landscapes (e.g. those in north Mississippi) are probably not good indicators of the frequency of wildfires during the lightning season before active fire suppression, owing to a current lack of fire conductivity and altered fuel structures (Leach and Givnish 1996). Nevertheless, the coincidence of high lightning-strike frequencies and long rain-free intervals can be a reliable indicator of seasonal variation in lightning fire potential, provided that conditions of natural fire conductivity and fuel loads are restored (Robbins and Myers 1992; Howe 1994; Ruffner and Abrams 1998).

The objectives of the present study were three-fold. First, we examined the incidence, size, and intensity of wildfires in Holly Springs National Forest between 1995 and 2000. Second, we examined relationships between wildfire incidence and precipitation totals for each month between 1995 and 2000 to determine whether wildfires of given size or intensity were more likely to occur in drier-than-average months. Third, we calculated expected seasonal patterns of lightning-fire potential using monthly averages of lightning strike frequencies and monthly variation in the length of rain-free intervals.

Our observations of significant decomposition of leaf litter during the summer months in oak-dominated forests in north Mississippi (see also hardwood forest fuel models of Deeming *et al.* [1977]) led us to generate the following three hypotheses: (1) the incidence of small or large or more intense wildfires does not increase with increasing time since the most recent prescribed fire; (2) increases in the incidence of wildfires in a given month or year, irrespective of size or intensity, are largely the result of lower-than-average rainfall amounts; and (3) seasonal patterns of wildfire incidence do not match those predicted by our calculations of seasonal variation in lightning potential and rainfall patterns. By testing these three hypotheses, we show that the ineffectiveness of prescribed burning in reducing the size and intensity of wildfires and the lack of wildfires during the lightning-season are likely both artifacts of fire suppression in the past, which has produced the closed-canopy forests we see today.

Methods

Background information on Holly Springs National Forest

Our analyses focused on wildfire patterns in the Main Unit of Holly Springs National Forest (hereafter, HSNF) in north-central Mississippi, USA. The forests that dominated most of the upland areas in this region before they were cleared in the 1800s and 1900s were open, fire-maintained woodlands dominated by black oak (*Quercus velutina*), black jack oak (*Q. marilandica*), post oak (*Q. stellata*), and shortleaf pine (*Pinus echinata*; Hilgard 1860, Nutt 1805 in Jennings 1947; Brewer 2001). Southern red oak (*Quercus falcata*), white oak (*Q. alba*), and hickories (*Carya* spp.) also occurred in these areas, but at lower frequencies (Hilgard 1860; Brewer 2001). *Andropogon* and *Schizachyrium* spp. were apparently common in the groundcover of these forests (Nutt 1805 in Jennings 1947).

The upland landscape of HSNF looks very different today. Rather than being open woodlands, most mature upland forests in HSNF have relatively closed canopies. Black jack oak is rare and now occurs mainly at forest edges on xeric ridges. White oak and sweetgum (*Liquidambar styraciflua*), which historically were much more common in floodplains in this region, are now common in the overstory of mature forests. Black gum (*Nyssa sylvatica*), hickories, and sweetgum are more common in the midstory of these forests than they were historically. In contrast to the early 1800s, the upland oaks and shortleaf pines are very rare in the midstory and sapling layers of these forests. Warm-season grasses (e.g. *Andropogon* spp.) are all but absent from the groundcover of these forests. Fuel conditions of most areas could best be described by the National Fire Danger Rating Systems fuel models P (closed-canopy pine or pine–hardwood forest), E (closed-canopy hardwood forest after leaf fall), and R (closed-canopy hardwood forest after leaf flush and during the summer).

In 1979, the staff of HSNF began a program of prescribed burning after ~30 years of active fire suppression and fire exclusion. The primary objectives of the fire program were to reduce the incidence and severity of wildfires and to improve wildlife forage and habitat conditions. Up to ~25 000 ha (62 000 acres) is subjected to prescribed burning in the Main Unit of HSNF in any given year. The average frequency of prescribed burning varied among locations in the ranger district from no prescribed fires in some areas to four fires in other areas between 1979 and 2000. The district is currently attempting to reach a target prescribed fire-rotation of most upland areas of 4 years (B. Oswalt, personal communication).

Relationships between wildfires, prescribed burning, and precipitation in HSNF

To determine relationships between wildfire incidence, size, and intensity, and prescribed fire history and precipitation, we compiled and analyzed data on wildfires from 1995 to 2000 provided to us by the HSNF district office in Oxford, Mississippi. The analyses we could do were limited by the type of data available to us. The district office provided us with ArcView shape files of wildfires from 1995 to 1999, but it lacked electronic records of wildfires before 1995. The ArcView shape files contained points (not polygons) of each named wildfire. We obtained information on the size (area), location, and date of detection of each wildfire from the associated attributes table. We also obtained paper records of wildfires in 2000 to get these same attributes. The district office also provided us with shape files showing the location of prescribed burning blocks from 1979 to 1999. We received a paper map of prescribed fire blocks for 2000.

To examine the relationship between wildfire incidence from 1995 to 2000 and the history of prescribed burning in the vicinity of each wildfire, we overlaid wildfire points on maps of the prescribed fire polygons. Because the district office did not have any information on the shape of each wildfire, we had to make some assumptions about the occurrence of overlap between wildfires and prior prescribed fires. We assumed that the shape of the fire was a circle (except when roads were located near the point of origin; see below) and then simply determined whether there was any overlap between the wildfire and one or more prescribed fire blocks (or previous wildfires). When a wildfire point occurred within a burn block, we assumed that the wildfire was at least partially contained within that burn block. We also looked for the occurrence of roads and creeks between the point location of the wildfire and the burn block. When these were present, we conservatively assumed no overlap between the wildfire and the burn block. In most cases, wildfires that we concluded overlapped with prescribed burn blocks were started within and were fully contained within these same blocks, leaving no doubt of overlap. We also assumed that wildfires did not overlap with prescribed fires conducted earlier in the same year. Because we lacked records of wildfires before 1995, we could not quantify overlap between wildfires before and after 1995. We assumed there was no overlap between these wildfires, which we argue is not unreasonable, given that most wildfires were small (<1 ha) and very few wildfires that occurred after 1995 overlapped with one another.

We used Kaplan–Meier product-limit tests to analyze the relationship between wildfire incidence and the number of years since an overlapping prescribed fire occurred. We used the same test to analyze the relationships between wildfire size (i.e. area) and years since the most recent fire, and the incidence of intense wildfires and years since the most recent fire (see below for a definition of intense wildfire). In all three analyses, we treated wildfire occurrence as a nominal 'event' variable and years since the most recent prescribed fire as a continuous time variable. In the first analysis, the event variable was represented by one of two states – a wildfire occurred *v.* a wildfire did not occur. In the second case, we treated wildfire size as a nominal event variable with two states – a wildfire larger than the median size (1.21 ha or 3 acres) occurred *v.* a smaller wildfire occurred. In the third analysis, we treated wildfire intensity as a nominal event variable with two states – a wildfire with an intensity level of 3 or higher did or did not occur. Fire intensity levels were taken from the wildfire reports and are based on typical or potential flame length on the fire line. They are intended to provide an assessment of the difficulty of fire line control. A fire intensity level of 3 or higher corresponds to potential flame lengths greater than 1.22 m (4 feet) and fireline intensities of greater than 381 kilojoules m^{-1} s^{-1} (110 BTUs ft^{-1} s^{-1}). Fires of this intensity generally are considered beyond the limit for control by direct methods, and thus machine methods or indirect control are typically required (Deeming *et al.* 1977).

We conducted two separate product-limit tests of wildfire intensity. In one case, we included all burn blocks. In the other, we included only those burn blocks in which at least one wildfire had occurred. Hence, the first analysis examined the relationship between the likelihood of an intense wildfire occurring and time since the last prescribed fire.

The second analysis examined changes in the relative likelihoods of intense and non-intense wildfire with time since the last prescribed fire. To examine the impact of precipitation, we calculated the average departure of precipitation totals from the long-term average for each month in which a wildfire occurred. We then used a one-sample, one-tailed t-test to test the hypothesis that the average of precipitation totals for months in which wildfires occurred was lower than the long-term average. We used a two-sample, one-tailed t-test to determine if months with larger wildfires were drier than months that only had smaller wildfires. Likewise, we used a t-test to determine if more intense fires were associated with drier months.

Because prescribed burns were conducted in many different years, the Kaplan–Meier test is appropriate to use for datasets that contain 'censored' observations (i.e. burn blocks that did not experience wildfires before the end of the study period). In the current application, the likelihood of wildfire (or large or intense wildfires) was estimated by the wildfire hazard function, which was calculated by taking the negative log of the cumulative proportion of burn blocks that did not experience a wildfire of a given type during a specified time interval. We calculated hazard functions and associated 95% confidence intervals from the survivorship functions provided by Statistix 8 (Analytical Software, Tallahassee, FL, USA), which were based on the methods of Lee (1992) and Simon and Lee (1982), respectively. We then examined the hazard function for significant departures from a positive linear relationship between wildfire hazard and time since the most recent prescribed burning. A linear relationship between these two variables indicated a constant wildfire probability, whereas an increase in the slope of the relationship between the wildfire hazard function and time since the most recent prescribed fire indicated an increasing likelihood of the wildfire event.

In addition to using product-limit tests to examine the relationship between wildfire size and time since prescribed fire, we used chi-square tests of independence between the occurrence of large wildfires and the occurrence of overlapping prescribed fires within the last 3 years. We examined 65 wildfires (i.e. all wildfires that occurred within prescribed-burning blocks), 12 of which were greater than 12.1 ha (30 acres). We tested the hypothesis that wildfires larger than 12.1 ha were more likely to occur in areas that had not been burned by prescribed fires in the last 3 years.

In addition to product-limit tests, we used logistic regression to examine the relationship between the probability of an intense wildfire occurring and time since the last prescribed fire. To increase sample size, we included wildfires that had occurred in areas that had not been prescribed burned since 1979. We simply assumed that these areas were prescribed burned in 1978 and then calculated time since the last prescribed fire accordingly. This increased the number of observations from 65 to 144.

Prescribed burning may reduce the incidence of ignitions. In theory, the more area that is burned across a region in a given year, the less area with flammable fuels will be left to ignite. This relationship assumes, however, that enough land is being burned in a given year to substantially reduce opportunities for wildfire ignition and that fuel reduction efforts are not being overridden by weather events (e.g. severe droughts). We examined this possibility using Spearman rank correlations of the total area burned by wildfires in a given year with the total area burned by prescribed fires in that year (from 1995 to 2000). We also examined the relationship between area burned by wildfires in a year and 2-, 3-, and 4-year moving averages of area burned by prescribed fires. Negative correlations would indicate that the more area burned by prescribed fires in a given year (and previous years), the less area will be burned by wildfires in that year.

Inferring the peak season of lightning-fire probability in north Mississippi

To identify the lightning-fire season in oak–shortleaf pine forests in north Mississippi, we analyzed seasonal patterns of lightning and precipitation in or near HSNF. By taking the product of mean lightning-strike density per month over a 10-year period and mean rain-free interval per month over a 30-year period, we inferred what the peak season of lightning-started fires would be in this region, under conditions of natural fire conductivity and fuel loads. We purchased and compiled lightning data from Global Atmospherics (Tucson, AZ, USA) for the years 1992 to 2001 in and around the Main Unit of HSNF. We calculated rain-free intervals using a spreadsheet from daily precipitation data (from 1972 to 2000 in Oxford, Mississippi) obtained from the National Climatic Data Center (Asheville, NC, USA).

Measuring the peak season of wildfire probability

Lightning and precipitation patterns do not necessarily provide reliable predictors of current seasonal patterns of wildfire activity for a variety of reasons. We examined seasonal patterns of wildfires by tabulating wildfires detected within the Holly Springs Ranger District between 1995 and 2000 and then sorting them by month of occurrence.

Results

Relationships between wildfires, prescribed burning, and precipitation

Statistical analyses of wildfire hazard between 1995 and 2000 did not show a simple positive relationship between the incidence of wildfires and the number of years since the last prescribed fire. Wildfire hazard initially increased between 1 and 3 years post fire and then decreased and remained relatively constant thereafter (Fig. 1; note that little confidence can be placed in the estimate of wildfire hazard at the right tail, owing to a low number of observations). This

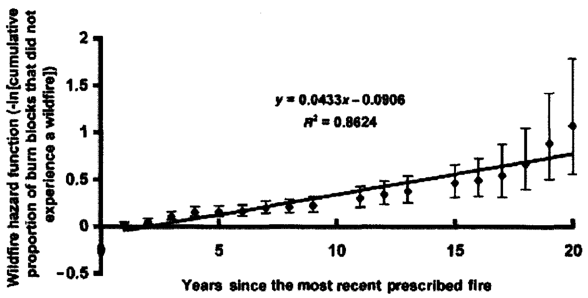

Fig. 1. Occurrence of wildfires as a function of the number of years since the most recent prescribed fire. Slope of the relationship is proportional to the average annual likelihood of wildfire occurrence. A linear response indicates no change in the likelihood of occurrence of wildfires of any size or intensity over time. Error bars are 95% confidence intervals. Proportions are corrected for censored observations. Initial number of burn blocks at risk = 235.

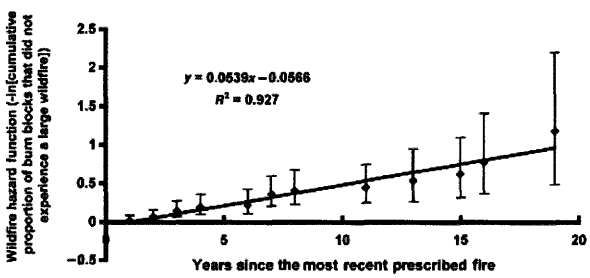

Fig. 2. Occurrence of larger than median (1.21 ha) wildfires as a function of the number of years since the most recent prescribed fire. Slope of the relationship is proportional to the average annual likelihood of occurrence of larger than normal fires. A linear response indicates no change in the likelihood of large wildfires over time. Error bars are 95% confidence intervals. Proportions are corrected for censored observations. Initial number of burn blocks at risk = 65.

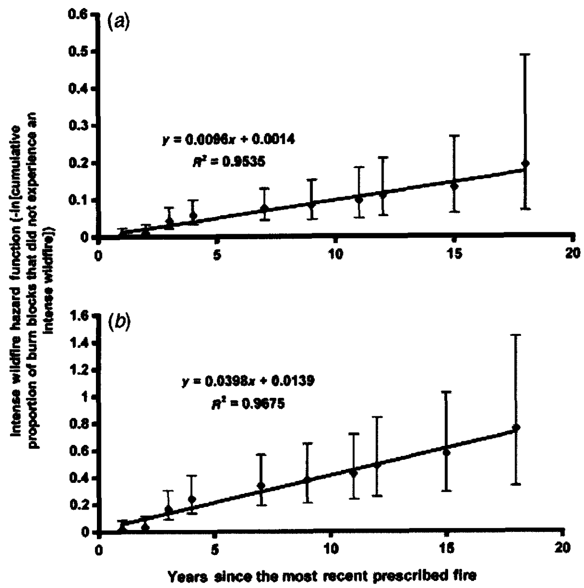

Fig. 3. Occurrence of intense wildfires (Fire Intensity Level of 3 or higher) as a function of the number of years since the most recent prescribed fire. (*a*) All blocks burned, and (*b*) burn blocks that experienced a wildfire. Slopes of the relationships are proportional to the annual average likelihood of occurrence of intense wildfires. A linear response indicates no change in the likelihood of intense wildfires over time. Error bars are 95% confidence intervals. Proportions are corrected for censored observations.

relationship resulted from a peak in wildfire activity in blocks that had been burned 3 years earlier. This peak was coincidental, resulting from the fact that a relatively large number of blocks received prescribed fires in 1997, and 2000 was the most active year for wildfires, owing to lower than normal precipitation. Hence, if one were to remove the 2000 wildfires from consideration, there would essentially be no relationship between wildfire hazard and time since the last prescribed fire.

As with the incidence of wildfires, we did not find a positive relationship between the incidence of large wildfires and the number years since the last prescribed fire. The wildfire hazard function associated with larger than median wildfires showed a linear relationship with time since the last prescribed fire, indicating that the likelihood of occurrence of a large wildfire did not vary significantly with time since the last wildfire (Fig. 2). There was no association between the occurrence of wildfires greater than 12.1 ha (30 acres) and prescribed burning within the previous 3 years ($\chi^2 = 0.01$, $P = 0.9$). We lacked sufficient numbers of wildfires to statistically examine relationships between the largest wildfires and prescribed burning. Nevertheless, between 1995 and 2000, five fires >121 ha (300 acres) were recorded in all areas of the ranger district (including those that had not been burned since 1979). Of these five, three occurred in areas that had not been prescribed burned in the previous 3 years, one occurred in a compartment that had been burned 2 years earlier, and one wildfire overlapped with a compartment that had been burned 3 years earlier and with one that had been burned 8 years earlier. When you consider that, in any given year, a greater portion of the HSNF is occupied by land that has not been burned in the last 3 years, large wildfires were no less likely to occur in areas that had been recently burned by prescribed fires.

The likelihood of intense wildfires occurring in a year did not increase consistently with increasing time since the last prescribed fire. The incidence of intense wildfires largely reflected the incidence of all wildfires, with a greater than expected incidence of intense wildfires 3 years after the most recent prescribed fires (although this effect was not statistically significant; Fig. 3). There was a large number of intense wildfires in 2000, which, as mentioned above, occurred 3 years after numerous prescribed fires were conducted. Logistic regression showed no relationship between the likelihood of an intense wildfire occurring and time since the last prescribed fire. The parameter estimate for time since fire was only −0.016 (±0.024 s.e.; $\chi^2 = 0.45$, $P = 0.503$; $n = 144$).

Dry years (such as 2000) resulted in more wildfires, regardless of their size or intensity. Wildfires in HSNF occurred in drier than normal years between 1995 and 2000 (on average, 3.73 cm below the monthly average; one-sample $t_{64} = -6.87$, $P < 0.0001$). Months with larger than median wildfires tended to be drier than months with smaller wildfires, but this effect was not statistically significant (3.92 *v.* 2.51 below normal, respectively, two-sample $t_{df=27} = 1.12$, one-tailed $P = 0.14$). Months with more intense fires were not drier than months with less intense fires (3.07 *v.* 3.43 below normal, respectively, two-sample $t_{df=43} = -0.29$, two-tailed $P = 0.78$).

We did not find a negative relationship between area burned by prescribed fires in a given year and area burned by wildfires in the same year ($r_s = 0.26$, $P = 0.564$, $n = 6$), nor was there a negative relationship between area burned by wildfires in a given year and 2, 3, or 4-year moving averages of area burned by prescribed fires ($r_s = 0.14$, 0.43, and 0.83, respectively).

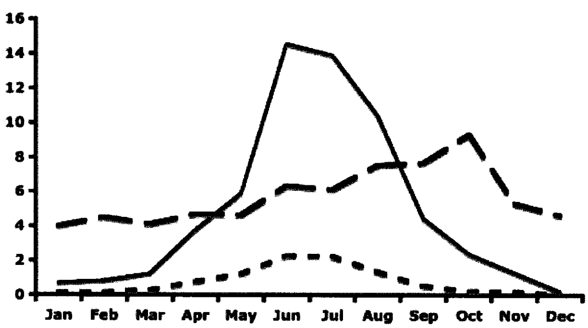

Fig. 4. Seasonal patterns of lightning strike density (short dashes; expressed as mean strikes per year per km^2 [1992–2001]), mean rain-free interval (long dashes; expressed in days per month per year, 1972 to 2001 in Oxford, Mississippi) and expected lightning fire potential (solid line; the product of the two).

Seasonal patterns of wildfire frequency and relationships to lightning-fire potential

High lightning strike densities coincided with relatively long rain-free intervals in the summer. We therefore suggest that, under conditions of natural fire conductivity and fuel loads, lightning-fire frequencies would be expected to remain low through the winter months and early spring, and then increase dramatically in June, July, and August (Fig. 4). Nevertheless, lightning patterns, even when combined with precipitation patterns, did not provide reliable predictors of seasonal patterns of wildfire activity from 1995 to 2000. Relatively few wildfires occurred in June or July during this time period (Table 1). Of all the wildfire records we examined, we found only one record of a lightning-started fire (which occurred in July 2000). The greatest numbers of wildfires occurred in March and in September, and the greatest area burned by wildfires occurred in April, August, and October (Table 1). The relatively high frequency of fires that were larger than the median in August and October (combined with the fact that none of the fires observed in these 2 months were started by lightning) suggests that lower than average precipitation amounts during these months in 2000 increased the likelihood of large wildfires (Table 1; Fig. 4). Other factors such as abundant leaf litter and frequent ignitions by humans contributed to the high wildfire activity during March and April.

Discussion

Is prescribed burning reducing wildfire hazard in Holly Springs National Forest?

Prescribed burning in HSNF did not significantly reduce the incidence, the size, or the intensity of wildfires from 1995 to

Table 1. Total number of wildfires per month in the Holly Springs Ranger District (Main Unit) from January 1995 to December 2000 (including those that occurred in areas that had not received any previous prescribed burning)

Month of occurrence	No. wildfires ≤ 1.21 ha		No. wildfires > 1.21 ha		Total no. wildfires	
	n	%[A]	*n*	%[A]	*n*	%[A]
January	1	0.002	1	0.071	2	0.073
February	10	0.047	5	0.75	15	0.797
March	19	0.1	14	1.077	33	1.18
April	10	0.037	9	1.91	19	1.95
May	2	0.01	1	0.139	3	0.149
June	4	0.009	1	0.065	5	0.074
July	1	0.005	6	0.14	7	0.145
August	6	0.006	2	1.94	8	1.94
September	21	0.072	4	0.419	25	0.491
October	15	0.053	5	2.29	20	2.35
November	3	0.013	10	0.535	13	0.548
December	2	0.005	3	0.097	5	0.102
Total	94	0.359	61	9.44	155	9.79

[A] Percentage of area currently subjected to prescribed burning on a 4-year rotation within the Main Unit of Holly Springs National Forest.

2000. We cannot say with certainty why controlled burning was not effective in this regard, but we offer a possible explanation. The oak–pine–hickory–gum forests that currently dominate the forested portions of the upland landscape of this national forest historically experienced prolonged fire suppression and now have closed tree canopies and a suppressed herbaceous layer that contributes little to the flammability of surface fuels. Most flammable fuels in these forests are therefore composed primarily of short needles from pines and leaf litter from deciduous trees that falls during the previous winter. Most litter (especially that of the deciduous trees) decomposes each year. Flammable surface fuels therefore may not accumulate sufficiently in these forests beyond the first year after a fire for prescribed burning to greatly affect the incidence of wildfires. In addition, fire suppression has not led to a significant accumulation of ladder fuels. Most of the plant species that have benefited from fire suppression in forests in north Mississippi are deciduous trees and vines that have few flammable leaves in the crown during seasonal peaks in wildfire activity in March (e.g. sweetgum, black gum, muscadine [*Vitis rotundifolia*]). The understory of these closed-canopy forests is too shady to permit the development of hotter-burning fuels such as warm-season grasses and dense shrubs (as seen in pine savannas; Wade and Lunsford 1989). In contrast, prescribed burning might reduce damage from future wildfires if it is combined with effective restoration of the persistent surface fuels that existed prior to prolonged fire suppression (as has been found in pine–grassland communities in the south-eastern USA; Wade and Lunsford 1989). Ironically, restoring the oak–pine–bluestem community in this region might increase the need for prescribed burning as a hazard reduction tool.

Our results differ from those of Martin (1988), but we caution that the results of the two studies may not be directly comparable. Martin found that large wildfires (i.e. those 121 ha [300 acres] or larger) that occurred in national forests of the south-eastern USA in 1985 were significantly less common in areas that had received a prescribed fire between 1983 and 1985. His assessment of wildfire size, however, pooled results from nearly all national forests in the region and did not examine hardwood or closed-canopy oak–pine forests and pine savannas (and thus different fuel models) separately. We maintain that one cannot assume that the effectiveness of prescribed burning in longleaf pine–wiregrass–palmetto savannas in Florida (which can indeed be significant; Brose and Wade 2002) applies in any meaningful way to closed-canopy white oak–hickory–gum forests in north Mississippi. Our argument is consistent with Fernandes and Botelho's (2003) conclusion that the effectiveness of prescribed burning in a given system cannot be generalized to other systems. Furthermore, some of Martin's conclusions about the effectiveness of prescribed burning in reducing large wildfires in the south-eastern USA were somewhat misleading. He noted that, 'It is significant that 91.5% of the acres burned in large wildfires [in 1985] occurred in areas that had not been burned by prescribed fire in the previous 3-year period.' However, it was also true that the 10 large wildfires that had occurred in assessment areas that had not received prescribed fire from 1983 to 1985 represented only 0.52% of these areas. Large wildfires burned 0.36% of the assessment areas that had received prescribed fire between 1983 and 1985. Because this percentage was based on only two wildfires, the difference between 0.52% and 0.36% might not have been statistically significant. Furthermore, we are given no indication of how much of the treated area received prescribed fire in 1985 (which could have reduced ignitions of wildfires of all sizes in 1985). We therefore cannot conclude with much confidence from Martin's analysis that prescribed burning effectively reduced the size of wildfires beyond the first year of the treatment.

The current study casts some doubt on the effectiveness of prescribed burning in HSNF to reduce wildfire ignitions. Although, in theory, such an approach could work, we found no evidence of a negative correlation between the amount of Forest Service land burned by prescribed fires in a given year and the amount of land burned by wildfires in that year or 2 to 4 years afterwards. The number of hectares burned by prescribed fires in any year between 1979 and 2000 never exceeded 4000 (~16% of the Main Unit that is on a prescribed fire rotation). It is possible that dramatically increasing the area burned each year (which is currently happening; B. Oswalt, personal communication) could reduce wildfire ignitions, but this hypothesis remains untested. Although our analysis of wildfires was based only on 6 years of data, the data were nonetheless adequate to show that precipitation amounts were the overriding factor in determining the number of wildfires (regardless of their size or intensity) in a given year in this system. Time-series climate models that forecast dry and thus active wildfire years (e.g. Beckage and Platt 2003) may be necessary to use prescribed burning effectively to reduce wildfire ignitions in this system.

What role should prescribed burning play in the management of HSNF?

We want to emphasize that we envision a vitally important role for prescribed burning in fire-dependent ecosystems in HSNF. We certainly do not advocate fire suppression or a total ban on prescribed burning. When used properly, prescribed burning is crucial for maintaining biodiversity and suitable wildlife habitat in fire-dependent oak- and pine-dominated ecosystems (Tester 1989; Haney and Apfelbaum 1990; Masters and Waymire 2001; Glitzenstein *et al.* 2003). Our concerns regarding its current use apply only to prescribed burning in mature, closed-canopy forests and not to its use in open fields, roadsides, or pine–grassland or oak–grassland communities. We suggest that land managers reconsider the use of rough-reduction burning as a general forest-wide strategy for hazard reduction in closed-canopy oak–pine–hickory–gum forests. We advocate a shift in focus from maximizing the area burned in these forests each year to phased restoration of fire as a natural ecosystem process.

Ecological restoration of fire-dependent oak–pine woodlands in north Mississippi will likely require: (1) reduction of mesophytic and floodplain tree species (by mechanical means) that invaded upland woodlands following deforestation and fire suppression in the past; (2) increases in the abundance of warm-season grasses, which produce flammable surface fuels that persist throughout the year and produce hotter fires; (3) repeated prescribed burning (including experiments with burning in the lightning season); and (4) long-term monitoring and adaptive management. Long-term studies of the effects of thinning and burning in shortleaf pine–hardwood forests in the Ouachita Mountains (south-east Oklahoma, USA) provided some encouraging results (Masters and Waymire 2001). These researchers showed that thinning combined with burning increased plant diversity, wildlife forage production (e.g. forage for common species such as deer and declining species such as bobwhite quail), and populations of declining non-game species (including endangered species such as the Red-Cockaded Woodpecker [Masters *et al.* 1996, 1997; Sparks *et al.* 1998; Masters and Waymire 2001; Crandall 2003]). We suggest that some of the lessons learned from these studies apply to HSNF.

Is prescribed burning during the lightning season practical in Holly Springs National Forest?

Factors other than lightning, such as precipitation, seasonal variation in the availability of flammable fuels, and the incidence of human ignition sources, interacted to play a dominant role in the seasonal patterns of wildfires from 1995 to 2000 in HSNF. We argue that the lack of significant accumulation of surface fuels from one year to the next and the low incidence of wildfires in the summer in north Mississippi are both artifacts of fire suppression in the past, which converted open oak–pine woodlands with persistent pyrogenic surface fuels to closed-canopy forests with ephemeral and less pyrogenic surface fuels. Although we would like to see greater use of summer (lightning-season) burning in oak–pine woodlands in north Mississippi, shifting from prescribed burning in cool seasons to mid- to late-thunderstorm seasons will not be practical until herbaceous surface fuels are re-established. We note, however, that lightning-season burning might be useful for reducing fuel build-up and maintaining plant diversity on roadsides and in powerline right-of-ways, where grass-based fuels prevail. Such burning could also facilitate ecological restoration from the edges of closed-canopy forests. Once partial ecological restoration of these ecosystems has occurred, lightning-season burning (and for that matter, frequent cool-season burning) may become a valuable tool for further restoring and maintaining a functional fire-dependent ecosystem.

Acknowledgements

We thank Carl Kilcrease, David Fant, and Bill Oswalt of Holly Springs National Forest for providing us with data on wildfires and prescribed fires, and five anonymous reviewers for their comments on an earlier version of this manuscript. This research was funded by a cooperative agreement between the Southern Research Station (USDA National Forest) and the University of Mississippi (SRS 04-CA-11330127-116).

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