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Patchiness of prescribed burns in dry sclerophyll eucalypt forests in South-eastern Australia

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Abstract

Prescribed burning is commonly used in eucalypt forests to reduce fire risk and minimize damage to people and property in the event of a wildfire. The dry sclerophyll forests of south-eastern Australia are naturally fire-prone. Little is known about the heterogeneity of prescribed fires in these forests. This paper reports on the spatial variability of repeated low intensity fires under two burning regimes, in both logged and unlogged forests, for a 17-year period from 1988 to 2005. Prescribed burns were extremely patchy at both the coupe and the plot scale for all treatments. Burns implemented soon after logging covered significantly greater areas than the standard prescribed burns. On the coupe scale, the extent of the burn was influenced by the average aspect of the coupe and the percent of the coupe burnt in the last fire. At the smaller plot scale, the extent of the burn was influenced by the neighbourhood burn patterns, the distance the plot was from the nearest drainage line, the time since the last fire and the percentage of the plot that was burnt in the last fire. Under operational conditions, sites on ridges are likely to burn approximately every 11 years and sites in gullies approximately every 20 years. Patchy burns achieved in this study will provide refuges for fire sensitive species and newly burnt areas for colonizing species and are therefore likely to have significantly lower ecological impacts than homogenous burns.

Keywords: Biodiversity; Conservation; Forestry; Fuel-reduction; Prescribed fire

1. Introduction

Prescribed burning is used widely as a forest management tool in eucalypt forests to reduce the risk and intensity of future wildfires by reducing or removing forest fuels through burning under cool conditions (Morrison et al., 1996; Bradstock et al., 1998; Fernandes and Botelho, 2003). Prescribed burning can also encompass ecological burns, post-logging burns and strategic burns. Ideally, these fires occur at low intensities so that they do not burn the lower forest canopy and re-growth, but are hot enough to maintain a spread through the leaf litter, small woody debris and standing fuel (e.g. grasses, understorey shrubs and dry bark on trees) (McArthur, 1966; Cheney et al., 1992). Within commercial forests, frequent low intensity burns are associated with significantly lower costs than occasional but more destructive wildfires. Forest managers may attempt to reduce litter as often as possible to protect the resource. However, altered fire regimes are considered to result in significant ecological change (Whelan, 2002) particularly the effects of increased fire frequency on biodiversity (Trainor and Woinarski, 1994; Bradstock et al., 1997). The challenge to land managers, therefore, is to develop fire management strategies that help control wildfires while maintaining ecological diversity (Gill, 2001).

Frequent fire is listed as a key threatening process for native flora and fauna (DEC, 2005) due to the increased likelihood of local population extinctions. At the site or local population level, there is a well established theoretical and empirical basis for understanding the role of frequent fire in altering the abundance of species (Keith, 1996). At the landscape level, however, the evidence is less clear and much of the supporting data are derived from chronosequence studies of repeated unplanned, high intensity fire. Keith (1996) identified fire regimes involving frequent fire and those that involve little

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vertical penetration of heat as being associated with multiple mechanisms of plant decline and extinction. Regular prescribed burning has both of these characteristics and obligate seeders (*sensu* Gill, 1977) are recognised as a particular fire-sensitive group of species. Bradstock et al. (1996) note that fire regime prescriptions are often based on models of the population biology of fire-prone obligate seeders and that these models do not account for the spatial heterogeneity of either plant populations or fire regimes. They recognise that knowledge of spatial variability (patchiness) of fire regimes at appropriate scales is critical to understanding the ecological impact of fires.

Landscapes often include refuge sites which remain unburnt, or burn at much lower frequency or intensity than surrounding areas. These sites may be less fire-prone because they are wetter (e.g. gullies and sheltered aspects) or because they support intrinsically less flammable vegetation (e.g. forest patches in fynbos; van Wilgen et al., 1990), or both. That is, such sites are not only less frequently burnt, but they also present contrasting environments to the surrounding landscape matrix. Due to contrasting environments, such sites typically support vegetation that differs from surrounding vegetation. These areas are refuges in the sense that, at the landscape level, they support species for which the broader habitat is unsuitable. However, they have limited capacity to act as refuges for fire-sensitive species which cannot persist in the refuge environment and for which the burnt environment is generally suitable, except for the adverse fire regime. With regard to the latter situation, Gill and Bradstock (1995) described the notion of ‘‘fire-shadows’’ which they suggest may act as refuges for fire sensitive species. Fire-shadow refuges are areas within an otherwise burnt landscape that are burnt at lower frequency than surrounding areas, due to changes in fire behaviour associated with particular features of environmental variation. For example, a creek may disrupt the movement of fire to the extent that parts of the downward creek bank remain unburnt. This notion is especially relevant to obligate seeders in a frequently burnt landscape.

Even within a relatively homogeneous landscape, low intensity fires are anecdotally regarded as patchy. Unburnt patches have the potential to act as refuges for fire-sensitive species between successive fires. In contrast to other types of refugia described above, which are more closely related to permanent site characteristics, local-scale unburnt patches are unlikely to be persistent in the long term. However, they may be more effective refuges for species which cannot persist in less fire-prone parts of the landscape due to competitive effects from other species or unsuitable habitat. Local extinctions which are predicted for frequent, low intensity burning on the basis of plant population demography may be avoided to the extent that unburnt patches are effective as refugia. Bradstock et al. (1996, 1998) used spatially explicit simulation to explore plant population viability and extinction in relation to spatial heterogeneity of fire regimes. However, studies of these effects are otherwise very limited and quantitative, empirical data on patchiness within a fire boundary are lacking.

The dry sclerophyll forests of South-eastern Australia are among the most fire-prone forest communities in the world (Cheney, 1976). Thirteen major wildfires were recorded in the Eden region, South-eastern New South Wales, between 1865 and 1980 (Lunney and Moon, 1989). The Eden Burning Study Area (EBSA) was established in 1986 to compare the ecological impacts and management implications of three fire regimes (no burning, burning at nominal two year frequency and burning at nominal four year frequency), in both logged and unlogged dry sclerophyll forests in the Eden Forest Management Area. This study has been conducted at an operational scale to ensure that the results are relevant to commercial forest activities.

This paper reports on the factors that influence spatial variability of repeated low intensity fires under two burning regimes for a 17 year period from 1988 to 2005. These results are then used to discuss the potential for frequent burning to influence the long term survival of plant populations.

2. Materials and methods

2.1. Study area

The EBSA is located in the Yambulla State Forest, 29 km south west of Eden, New South Wales (37°14′S, 149°38′E). The EBSA covers 1080 ha of forest that is primarily Timbillica Dry Shrub Forest (Keith and Bedward, 1999) with smaller patches of other shrub and grass forest types. The major overstorey species are *Eucalyptus sieberi*, *E. consideniana*, *E. agglomerata* and *E. muelleriana* on the ridges, with *E. cypellocarpa* and *E. obliqua* locally dominant in lower lying areas. The most common understorey species are *Allocasuarina littoralis*, shrubs *Daviesia buxifolia*, *Epacris impressa, Acacia terminalis*, *A. longifolia* and *Platysace lanceolata*, and herbs *Gonocarpus teucrioides*, *Lomandra multiflora* and *Pteridium esculentum* (Binns and Bridges, 2003). The EBSA lies on Devonian adamellite parent material and ranges in elevation from 180 to 440 m above sea level. It is moderately undulating and broadly homogeneous with respect to geology and climate.

The EBSA was established in 1986 at which time there were no records of planned logging or evidence of unplanned logging in the area (Binns and Bridges, 2003). Fire history in the recent past is well documented. The most recent wildfire in the study area occurred in January 1973, albeit at a low intensity throughout the study area. A low intensity prescribed burn was attempted in April 1979, however, it is estimated to have burnt only 2% of the area. In April 1981 an attempt was made to burn all compartments in the EBSA using aerial ignition. The intensity of this fire was low and burnt only about 10–15% of the area. No attempts were made to burn the area between 1981 and 1988 (Binns and Bridges, 2003).

2.2. Experimental design

The EBSA is comprised of 18 experiment coupes in a randomized block design (Fig. 1). Coupes in the EBSA range from approximately 8 to 56 ha with a mean size of 32 ha (Binns and Bridges, 2003), which is consistent with standard

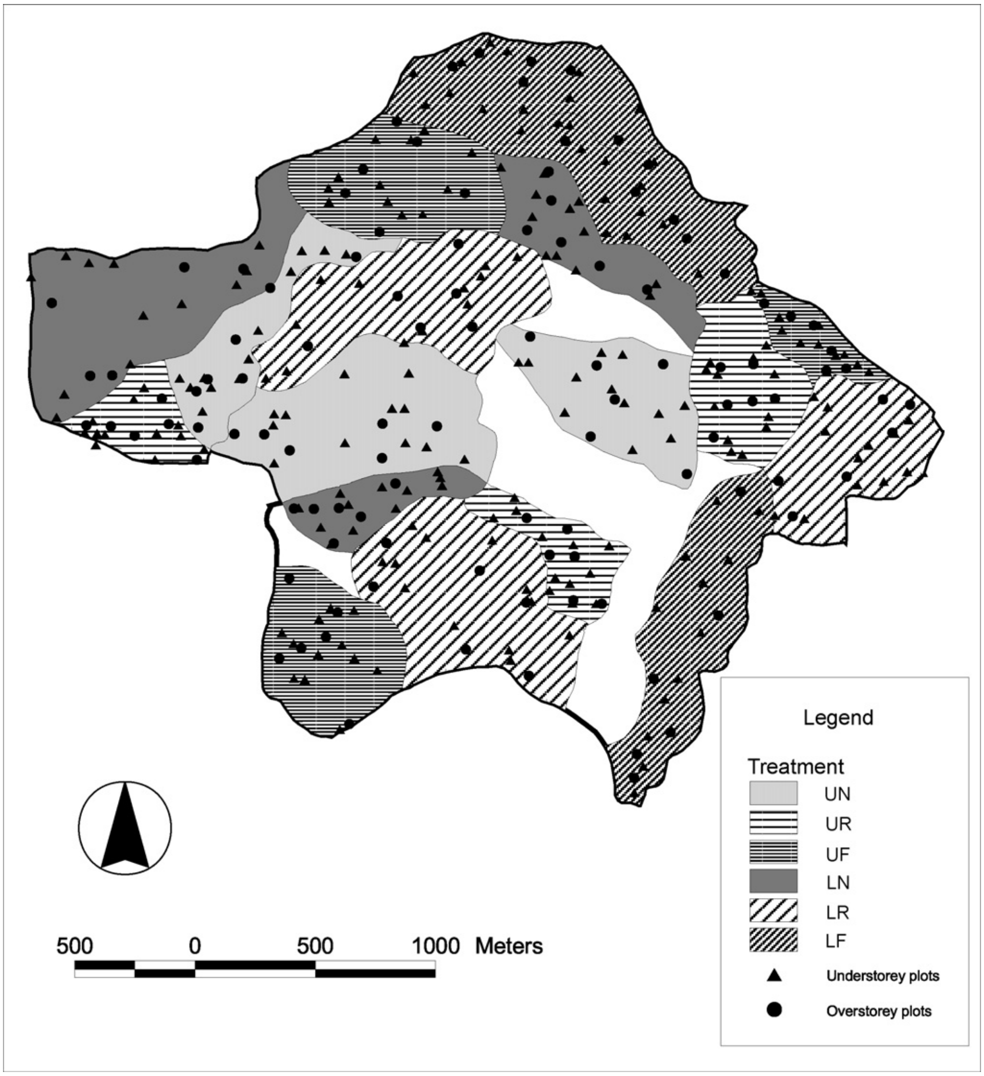

Fig. 1. Experimental layout of the Eden Burning Study Area. Treatment codes: LF, logged frequent burn; LR, logged routine burn; UF, unlogged frequent burn; UR, unlogged routine burn.

operations in this region. Three replicate coupes were randomly allocated to one of the following six treatments:

- (UR) unlogged, routine burn (first burn in 1992, then at 4-year intervals);
- (UF) unlogged, frequent burn (first burn in 1990, then at 2-year intervals);
- (UN) unlogged, not burnt;
- (LF) logged, frequent burn (post-log burn in 1988, then at 2-year intervals);
- (LR) logged, routine burn (post-log burn in 1988, then in 2001, then at 4-year intervals); and
- (LN) logged, not burnt.

For the purposes of this paper, we are only concerned with those coupes that experienced fire (i.e. UR, UF, LF, LR).

Logging in the study area, depending on treatment allocation, occurred between 7 November 1987 and 1 April 1988 according to the planning procedures and logging practices current in the Eden Region. Logging in these areas was an integrated operation extracting both saw logs and pulp wood. These logging operations retained a proportion of mature trees for seed trees, existing and potential fauna habitat, future saw logs and visual amenity. The combined tree-retention prescriptions retained approximately 30% of the original overstorey basal area in the net logging area (Binns and Bridges, 2003).

Burning treatments were based on those prescribed by the Fire Management Policy for the Eden Region (SFNSW, 1982). The routine burning treatment was considered as the control for this experiment, based upon the maximum burning frequency expected for unlogged stands at the commencement of the study. By contrast, frequent burning was a purely experimental treatment which aimed to produce a high frequency (the greatest possible at a coupe scale) of low intensity prescribed burns within both unlogged and logged coupes.

Ground crews used drip torches to ignite spot fires or line fires in various patterns according to fuel and weather conditions. Boundary ignition lines were along forest roads and secondary tracks with additional internal lines included where appropriate and safe. This approach is consistent with operational conditions and therefore the results from the study are directly applicable to management. Aerial ignition of spot fires from a helicopter supplemented the ground ignition in 1992 and 1996 (Binns and Bridges, 2003).

All burning was carried out in the autumn of the programmed year during periods when prevailing weather conditions were considered most suitable. During the burns, temperatures ranged from 16 to 25 °C, relative humidity between 44% and 76%, and wind speeds were below 5 km/h. The Keetch–Byram drought index (KBDI) (calculated from the Merimbula airport Bureau of Meteorology weather station) was used to account for climatic variation between the burns. KBDI is the number of mm of rainfall required to saturate the soil while accounting for daily temperature (Keetch and Byram, 1968). The values for KBDI range from 0 to 200 mm but, in this study, were relatively low for all burns with values between 5 and 64. Conditions were unfavorable for burning between 1996 and 2001. Attempts to ignite coupes in 1996 were largely unsuccessful and have been removed from the analysis (see below). Autumn rainfall patterns in both 1998 and 2000 meant that burns were not attempted in these years (Binns and Bridges, 2003).

The occurrence of fire in the survey plots was assessed visually within 8 weeks of burning for each treatment. Fire was assessed at two sets of plots that were not in overlapping locations. The nature of the measurements varied as these plots were also used to assess either the responses of the understorey (small quadrats) or overstorey (large plots) vegetation to the experimental treatments. In total, 18 plots were present in each coupe and these were comprised of 12 understorey measurement sites and 6 overstorey measurement sites (Fig. 1). At each of the understorey sites, the occurrence of fire was assessed in eight quadrats 0.25 m^2 within a 5.64 m radius (100 m^2) of the reference point. At the overstorey sites the occurrence of fire was assessed in ten 4 m^2 circular quadrats within a 20 m radius (0.13 ha) of the centre point.

2.3. Statistical analysis

We assessed the factors that influence how an area will burn at two scales: the coupe level and the plot level. All analyses were conducted using SAS v9.13 (SAS Institute Incorporated, USA). As the study was based on repeated measures of sites, we used a generalized estimating equations (GEE's) approach with a summarized binomial response, i.e. number of successes over the number of attempts (Venables and Ripley, 1994). GEE's are an extension of generalized linear models that account for the serial correlation in longitudinal data, such as time series. At the coupe level the response was the number of plots (understorey and overstorey) that were partially or completely burnt in each fire over the number of plots assessed. A plot was considered burnt if two or more quadrats within a plot were partially or completely burnt. At the plot level the response was the number of quadrats within a plot that were burnt during each fire over the number of quadrats assessed. Models were built manually by testing individual factors and sets of factors and only including factors significant at the $p = 0.05$ level.

A range of topographic variables, treatment variables and site history variables were used in the analysis. There were differences in the datasets as the coupe data were averaged across the area whereas the plot data were derived for an individual point. A summary of the variables used in the analyses is presented in Table 1. Data for the topographic variables were extracted from Forests NSW data layers using ArcView GIS (ESRI, USA). These included elevation, aspect, soil wetness index, solar radiation, landscape roughness (degree of undulation) index and topographic position (see NSW, National Parks and Wildlife Service, 1998 for the formulas for calculating these data). Due to the circular nature of aspect, (i.e. there is little difference between 10° and 350°) we converted aspect values as relative to north resulting in values from 0° to 180°. Low aspect to north values indicate sites that are northerly facing whereas higher aspect to north values indicate sites with a more southerly aspect. Treatment variables included in the analysis incorporated the number of burns at a plot or coupe during the course of the study, the time since the last burn, the

Table 1
Variables used in the analysis

Variable	Coupe analysis	Plot analysis
Aspect relative to north	Average for the coupe	Point value
Soil wetness	Average for the coupe	Point value
Solar radiation	Average for the coupe	Point value
Roughness (degree of undulation)	Standard deviation of the elevations within the coupe	Standard deviation of the elevations within 250 m
Coupe area (ha)	Single value	N/A
Topographic position	N/A	Point value
Keetch–Byram drought index (KBDI)	Value for day of burn	Value for day of burn
Distance to drainage line	(a) % of coupe within 30 m of a drainage line, (b) % of coupe within 50 m of a drainage line	Distance to the nearest drainage line
Treatment	Categorical value (four levels)	Categorical value (four levels)
Fire treatment	Two levels: routine (0) and frequent (1)	Two levels: routine (0) and frequent (1)
Log treatment	Two levels: unlogged (0) and logged (1)	Two levels: unlogged (0) and logged (1)
Number of burns	Number of burns in coupe since 1987	Number of burns at the site since 1987
Time since last fire	Time since ignition of coupe	Time since last fire passed over the site
Percentage of coupe burnt in the last fire	Percentage value	N/A

percentage of the plot or coupe burnt during the last fire and the extent of logging disturbance. The KBDI was used to account for weather at the time of the burns.

Correlations between the variables were tested using the Pearson correlation coefficient. If correlated factors were identified, factors were entered in the model individually and together to test for confounding effects and multi-collinearity using the methods described by Chatterjee et al. (2000). If multi-collinearity occurs between variables, the estimate of the co-efficient will change significantly when one of the affected variables is added to the model. Models containing multi-collinearity were discarded (Chatterjee et al., 2000).

A spatially lagged response variable (SLRV) was used to account for spatial autocorrelation between the sites in the plot analysis (Haining, 2003). To form such a variable, the sum of the weighted response within a neighborhood was calculated for each site. The inverse of the distance between two plots was used as the weighting factor. In the EBSA, fires ignited within one coupe never affected adjacent coupes, therefore, the neighborhood considered was limited to only those plots within the coupe in question. Spatial autocorrelation was not calculated for the coupe level analysis as fires in one coupe did not have any impact upon fires in other coupes.

3. Results

There were no plots that burnt on every occasion when the coupe in which they were situated was ignited (Table 2). The percentage of plots that never burnt was generally low (14.8% LF, 16.7% UF and 24.1% UR), however, in the LR treatment, 50% of plots remained unburnt even after three fires were attempted (Table 2). The median number of burns per plot was 2 in both the LF and UF treatments, 1 in the UR treatment and 0 in the LR treatment. The percentage of the coupe burnt ranged from 6% to 90% with an overall mean of 40% of coupes being burnt (Fig. 2). When a plot was burnt, the area burnt ranged from 10% through to 100% with a mean of 65%.(Fig. 3) Fires attempted during the 1996 period burnt very limited areas with only one coupe being burnt. For this reason, results from this period have been excluded from further analysis for both the coupe and the plot analysis. The post-logging burn (1988) had significantly greater coverage than other burns ($Z = 4.66$, $p < 0.0001$). Since the fuel structure post-logging is very different to that of unlogged areas, the first post-logging burn was excluded from the analysis of prescribed burns.

Table 2
Number of times individual plots were burnt for a total of 54 plots per treatment

Treatment	Frequent burning		Routine burning	
	LF	UF	LR	UR
Not burnt	8 (14.8%)	9 (16.7%)	27 (50%)	13 (24.1%)
1 burn	7 (13%)	10 (18.5%)	22 (40.7%)	22 (40.7%)
2 burns	18 (33.3%)	16 (29.6%)	5 (9.3%)	18 (33.3%)
3 burns	15 (27.8%)	10 (18.5%)	0 (0%)	1 (1.9%)
4 burns	4 (7.4%)	8 (14.8%)	NA	0 (0%)
5 burns	2 (3.7%)	1 (1.9%)	NA	NA
6 burns	0 (0%)	0 (0%)	NA	NA
7 burns	0 (0%)	0 (0%)	NA	NA
8 burns	0 (0%)	NA	NA	NA

Percentage values presented in parenthesis. Treatment codes as for Fig. 2.

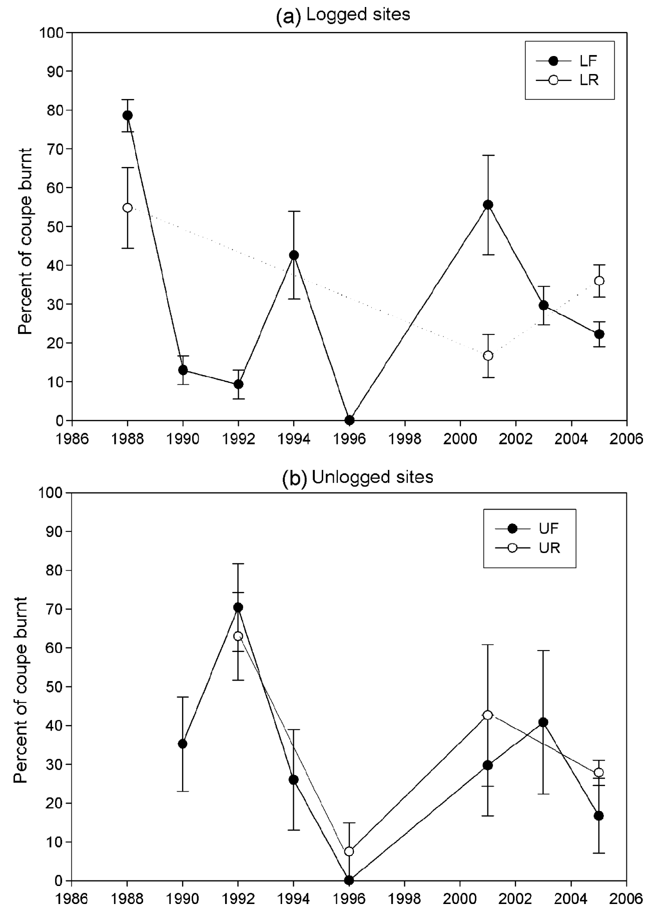

Fig. 2. Percentage of coupes that were burnt during the study period. Values presented as means ± 1S.E. Treatment codes as per Fig. 1. NB: the logging event occurred in 1987–1988.

Correlations between the environmental variables are presented in Table 3a for the coupe variables and Table 3b for the plot variables. Solar radiation was negatively correlated with the aspect relative to north ($p < 0.0001$) for both the coupe and plot data, that is, north-facing slopes had higher levels of solar radiation. Not surprisingly, there was also a strong correlation between the percentage of the coupe within 30 m of a drainage line and the percentage of the coupe within 50 m of a drainage line ($p < 0.0001$). Slope was also correlated with roughness within 250 m at the plot scale. All other correlations were less than 0.8.

3.1. Statistical modeling

Table 4 presents the final model for the coupe analysis. In this model burn coverage had a negative relationship with the mean aspect relative to north ($p = 0.02$), that is, northward facing coupes are more likely to burn than more southerly facing coupes. The model also found a significant negative relationship between the proportion of the coupe burnt in the last fire and the burn coverage in the subsequent fire ($p < 0.01$).

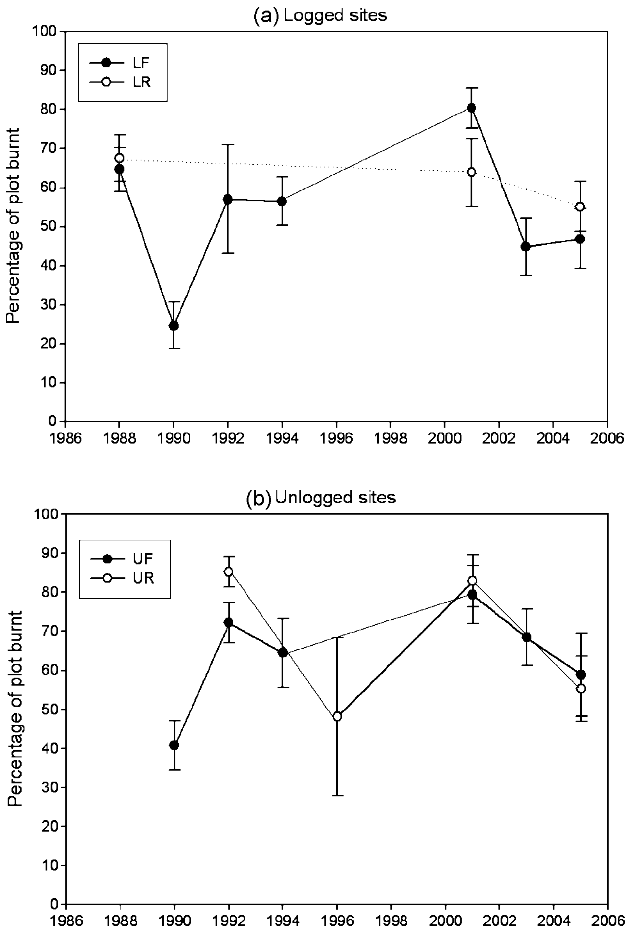

Fig. 3. Percentage of plots that were burnt during the study period. Values presented as means ± 1S.E. NB: all zero values have been removed. Treatment codes as for Fig. 1.

Table 5 presents the final model for the plot analysis. This model contained a strong relationship with the spatially lagged response variable (SLRV), indicating the strong neighbourhood influence of burns. Put simply, a plot is more likely to burn if the surrounding plots are also burning. The probability of a plot burning increased with distance from the nearest drainage line. Plots on ridge tops are therefore more likely to burn than plots in gullies or drainage lines. Burn history at a plot had a significant effect on plot burning patterns. The probability that a plot would burn decreased in relation to the percentage of the plot burnt in the last fire, however this effect decreased over time.

The final model for the plot analysis for the post-logging burn (1988) only is presented in Table 6. In this burn, the probability of a plot burning was directly related to the percentage of the plot that was affected by logging. As with the prescribed burn model, a strong neighbourhood effect was seen with the SLRV. Environmental variables were also significant in the final model, with the probability of a point burning increasing in relation to solar radiation and decreasing in relation to the soil wetness index. The drier plots were therefore more likely to burn than the shaded or wetter gully plots. No attempts were made to model the post-logging burn at the coupe scale as there were only six replicates.

Predictions from the plot models are presented in Fig. 4. These predictions are based on plots that experienced a burn coverage of 80% at time 0. Ridge sites were defined as 300 m from a drainage line with gully sites as those 30 m from a drainage line. These values were considered representative of the study site. Based on these predictions, ridge sites were unlikely to burn again until approximately 11 years after the previous burn. In contrast, gully sites may not burn for

Table 3
Correlations in the data set: (a) coupe variables, (b) plot variables

	Aspect relative to north	% of coupe within 30 m of a drainage line	% of coupe within 50 m of a drainage line	Roughness	Solar radiation	Wetness	Coupe area
(a) Coupe variables							
Mean aspect relative to north	1.000	−0.559	−0.470	0.143	**−0.908**	0.018	0.238
% of coupe within 30 m of a drainage line	−0.559	1.000	**0.983**	−0.559	0.669	0.647	−0.327
% of coupe within 50 m of a drainage line	−0.470	**0.983**	1.000	−0.596	0.613	0.667	−0.376
Roughness	0.143	−0.559	−0.596	1.000	−0.225	−0.434	0.675
Solar radiation	**−0.908**	0.669	0.613	−0.225	1.000	0.284	−0.203
Soil wetness	0.018	0.647	0.667	−0.434	0.284	1.000	−0.110
Coupe area	0.238	−0.327	−0.376	0.675	−0.203	−0.110	1.000

	Aspect to north	Wetness	Solar radiation	Slope	Distance to nearest drainage line	Topographic position	Roughness within 250 m
(b) Plot variables							
Aspect to north	1	0.156	**−0.818**	0.074	0.148	−0.080	0.167
Soil wetness	0.156	1	−0.048	−0.234	−0.483	−0.639	−0.247
Solar radiation	**−0.818**	−0.048	1	−0.475	−0.208	0.102	−0.492
Slope	0.074	−0.234	−0.475	1	0.168	−0.130	**0.817**
Distance to nearest drainage line	0.148	−0.483	−0.208	0.168	1	0.491	0.373
Topographic position	−0.080	−0.639	0.101	−0.130	0.491	1	−0.017
Roughness within 250 m	0.167	−0.247	−0.492	**0.817**	0.373	−0.017	1

Correlations greater than 0.8 appear in bold.

Table 4
Parameter estimates for the GEE model depicting the relationship between the extent of the coupe that was burnt and environmental and treatment variables

Parameter	Estimate	S.E.	Lower	Upper	*Z*-stat	*p*-Value
Intercept	0.6708	0.3327	0.0188	1.3228	2.02	0.0438
Aspect relative to north	−0.0097	0.0041	−0.0177	−0.0017	−2.38	0.0172
Percentage coverage of last burn	−0.0124	0.004	−0.0202	−0.0046	−3.11	0.0019

The post-logging burn and the 1996 burn were excluded from the analysis.

Table 5
Parameter estimates for the GEE model depicting the relationship between the extent of the plot that was burnt and environmental and treatment variables

Parameter	Estimate	S.E.	Lower	Upper	*Z* stat	*p*-Value
Intercept	−2.8775	0.2601	−3.3874	−2.3677	−11.06	<0.0001
Spatially lagged response variable	0.0609	0.0041	0.0528	0.069	14.78	<0.0001
Distance to drainage line	0.0044	0.0013	0.0019	0.0069	3.48	0.0005
Time since last burn	−0.0329	0.0118	−0.056	−0.0098	−2.79	0.0053
Percentage coverage of last burn	−0.0134	0.0035	−0.0203	−0.0066	−3.85	0.0001
Interaction term: percentage of last burn and time since last burn	0.0019	0.0006	0.0008	0.003	3.42	0.0006

The post-logging burn and the 1996 burn were excluded from the analysis.

Table 6
Parameter estimates for the GEE model depicting the relationship between the extent of the plot that was burnt and environmental and treatment variables for the post-logging burn (1988)

Parameter	Estimate	S.E.	Lower	Upper	Chi-squared	*p*-Value
Intercept	−7.4722	1.8615	−11.1207	−3.8238	16.11	<0.0001
Spatially lagged response variable	−0.0002	0.0001	−0.0005	0.0000	5.36	0.0206
Solar radiation	0.1738	0.0227	0.1292	0.2184	58.41	<0.0001
Soil wetness	−0.0193	0.0096	−0.0382	−0.0004	4.02	0.045
% of plot affected by logging	0.0186	0.0028	0.0131	0.0241	43.77	<0.0001

approximately 20 years after a fire. The neighbourhood values have a strong influence on burn probabilities. Thus, ridge sites may burn as often as once every year, or as little as once every 15 years, depending on whether adjacent sites are also likely to burn. The range for gully sites was approximately 15–30 years in these forests. Neighbourhood burn values may be influenced by factors such as effort in establishing the fire, climatic conditions (e.g. wind) or other unmeasured variables.

4. Discussion

Prescribed burns in this study burnt in a patchy manner at both the plot and coupe scales. These fires were affected by environmental features and the recent fire history. The burn coverage of the coupes ranged from 6% through to 90% with a mean of 40%. Post-logging burns affected a greater area than the fuel-reduction burns on logged and unlogged coupes but

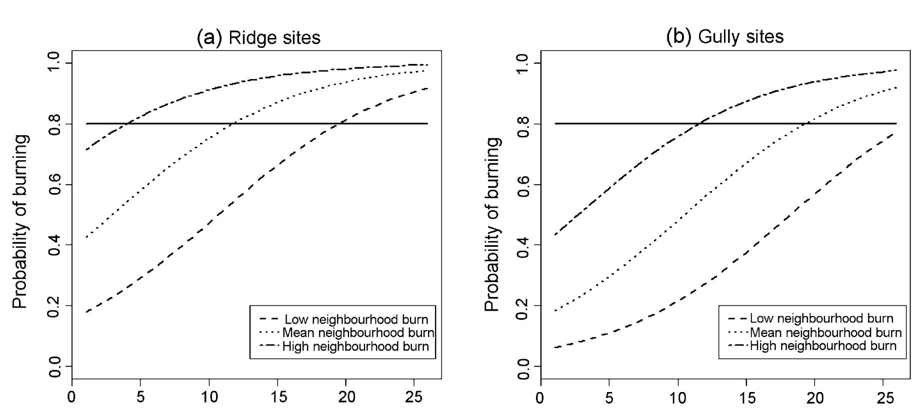

Fig. 4. Predicted capacity of ridge (a) and gully (b) sites to burn in relation to time since previous fire. Solid line represents a burn probability of 0.8 which is considered to be a likely occurrence. Low, medium and high values for the neighbourhood burn are based on values calculated for the spatial lagged response variable.

this was probably a function of the elevated fuel loads resulting from the recent logging event (Bridges, 2005). Results from this study give us greater insight into the potential ecological impacts of repeated low intensity burning regimes. In particular, the data on spatial and temporal heterogeneity of fires allow an investigation of the extent to which patchiness may reduce detrimental impacts of frequent burning previously reported for plant populations.

Serotinous obligate seeders and other obligate seeders with transient seed stores are the plant functional groups considered most likely to suffer local population extinction under a regime of frequent fire (Keith, 1996). In the absence of spatial heterogeneity, models suggest that species will rapidly become locally extinct if the fire interval is less than the juvenile period (Bradstock et al., 1998). The risk of localised extinction of fire-sensitive obligate seeders from otherwise adverse fire regimes may be reduced in patchy burns compared to more spatially homogeneous fires, at least to the extent that such species can survive in unburnt or infrequently burnt patches. Obligate seeder species in the study area with juvenile periods of about 3–5 years, and which may be sensitive to the imposed fire regimes in the absence of refuge patches, include *Hakea sericea*, *Kunzea ambigua* and *Leucopogon microphyllus*.

Unburnt patches may also provide effective refuges for other biota. For example, within a single fire, individuals of many fauna species do not survive direct exposure to fire (e.g. Swanepoel, 1981; Driscoll and Roberts, 1997; Lemckert, 2000). However, individuals that are able to seek shelter in the unburnt patches are potentially able to rapidly recolonise and exploit newly burnt areas (e.g. Kiss and Magnin, 2003). Whether the patchiness of these fires is sufficient to maintain local populations in the long term will depend on life history attributes of the species in question, especially dispersal characteristics, the sensitivity of juveniles to fire, and the distribution of the species in the landscape.

Recent burn history at a site influences subsequent burn patterns. The percentage of a plot that was burnt was influenced by the coverage of the last burn and the time since the last fire, which is likely to be a function of litter accumulation. In dry eucalypt forests, leaf litter has been found to accumulate to pre-burn levels within 3–6 years (Fox et al., 1979; Raison et al., 1983; Birk and Bridges, 1989; Fensham, 1992). This is consistent with the predictions from our models that ridge sites are unlikely to burn within 5 years of the last fire even under high neighbourhood values. Higher soil and litter moisture levels increased this value to more than 10 years at gully sites. Riparian or lower slope areas may thus act as long-term refuges for species that are able to occupy those environments. An example of this is the frog *Mixophyes iteratus* which was reported to remain within 20 m of the streams used for breeding, with 90% of diurnal shelter sites occurring within 5 m of the stream (Lemckert and Brassil, 2000). However, species that occupy areas away from riparian zones are likely to possess adaptations for coping with fire. Examples of this include fauna that burrow beneath the soil surface (Penman et al., 2006) and plant species that re-sprout following fire (Bradstock et al., 1997). Many of the latter, such as *Amperea xipohoclada*, *Lomandra multiflora*, *Persoonia linearis* and *Platysace lanceolata*, are common in the study area.

In this study, no significant relationship was found with the KBDI (a measure of prevailing weather conditions relevant to fire behaviour). Despite this, it is likely that prevailing weather was an important influence on overall fire extent and there are several possible reasons why it was not found to be significant in this study. Relationships with weather conditions may be partly accounted for, and masked by, relationships with the SLRV. The sample size at the coupe level, which excludes the SLRV, may be insufficient to detect the subtle influence of KBDI over the limited range of values experienced in this study. Also, the KBDI may not be well correlated with other important weather attributes within that range, e.g. wind speed or temperature.

Burning treatment was not a significant factor in determining probability of burning. This result suggests that the imposition of alternative fire regimes (either burning every 2 years or every 4 years) does not affect the likelihood of a patch burning in a particular fire. This further implies that, after repeated fires, a greater proportion of an area would be at younger fire age under a 2-year regime compared to a 4-year regime, simply because there are more ignition attempts in the former, but with similar probabilities of ignition at each attempt. The consequence of this is that the extent of refuge patches will be reduced with more frequent burning, at least over the time period of this study. However, over longer time periods, the extent of refuge patches may reach a minimum threshold. These results cannot be extrapolated to fires of greater intensity and do not include any additional effects of unplanned fires (e.g. large-scale wildfires). Comprehensive spatial simulation is required to examine the effects of various burning regimes on the age structure of areas under various regimes, incorporating higher intensity fires and unplanned fires, and exploring the possible existence of longer term thresholds. Predictions from these models would aid our understanding of the effectiveness of refuges and the implications for biodiversity management in dry sclerophyll forests.

Fuel-reduction burning aims to modify fuel characteristics to aid future suppression of unplanned fires. The effectiveness of this strategy is subject to ongoing debate. One of the more contentious aspects is the extent to which fuel loads and forest structure are modified. At the time scale of this study, imposing a more frequent prescribed burning regime is more likely to reduce fuels over a larger proportion of the area, but may reduce the extent and effectiveness of refuge sites. In any case, the gains to wildfire suppression from imposing more frequent burning need to be weighed against the additional costs in potentially adverse conservation outcomes.

5. Conclusion

Prescribed burning in dry sclerophyll forests was found to be patchy at both the plot and the coupe scale. Within the boundary of a low intensity fire there remain unburnt patches which potentially act as refuges for fire-sensitive species. The spatial heterogeneity of these fires is likely to reduce the ecological impacts to the extent that these refuges are effective. Gullies and lower slopes provide more extensive refuges, at least for those species which can occupy such sites. These results suggest that the ecological impacts of high frequency, low intensity fires are likely to be lower than is often predicted based on assumptions of homogeneous landscapes and uniform burning. The demonstrated patchiness of fires means that prescribed burning at frequencies which exceed thresholds derived from considerations of life history attributes of fire-sensitive species do not necessarily imply a high risk of local extinction, as is often assumed. However, frequencies that are high relative to such thresholds do reduce the extent of potential refuges and increase the risk of local extinction.

Analysis of species and community responses to these burning regimes, combined with consideration of distribution of species in the landscape, will allow for a more comprehensive investigation of the impacts of these burning regimes. This would allow determination of an appropriate ecological burning regime for dry sclerophyll forest in South-eastern New South Wales.

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