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Multi-century changes in vegetation structure and fuel availability in fire-sensitive eucalypt woodlands

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ABSTRACT

Understanding how communities change with time since fire is critical for identifying appropriate fire return intervals for biodiversity conservation. In infrequently-burnt communities, vegetation structure, habitat features and fuel availability can change over time-scales much longer than can be measured using contemporary remote-sensing approaches, creating challenges for conservation and fire management. To characterize longer-term patterns of vegetation structural change, we measured vegetation cover, ground cover, tree density and stand basal area across a multi-century time-since-fire sequence derived from growth ring-size relationships in fire-sensitive *Eucalyptus salubris* woodlands of south-western Australia. We hypothesized that: (i) vegetation structural components reflecting fuel availability increase with time since fire; (ii) recovery of vegetation structural components with time since fire requires long time-frames; and (iii) vegetation components indicating senescence are more evident in mature than intermediate fire-age classes. All vegetation structural components showed significant differences between time-since-fire classes (termed 'young', 'intermediate' and 'mature'), and to a lesser extent between years of sampling. The two vegetation structural components with the highest covers overall, and hence likely greatest contributors to fuel availability, were vegetation 4–10 m high and ground fuel. These two layers showed non-monotonic changes indicating a peak at intermediate times since fire (~35–150 or 35–250 years; depending on the model used to estimate stand age), conflicting with the common assumption that fuel availability increases with time since fire. Total stand basal area increased rapidly after fire then appeared to stabilize beyond about 100 years, with competition likely mediating density-dependent thinning such that declining plant density offset increasing trunk size. There was little evidence for an increase in standing dead vegetation in mature woodlands such as would suggest significant senescence when long-unburnt. Replacement of mature woodlands with intermediate time-since-fire woodlands with greater cover and connectivity of key fuel layers potentially instigates a self-reinforcing fire regime shift favouring larger and/or more uniform fires. If such changes eventuate, substantial losses in conservation values in *E. salubris* woodlands are likely. Elucidating these changes in vegetation structure and implications for conservation management only became feasible due to the development of methods to estimate the time since fire of vegetation not burnt for hundreds of years.

1. Introduction

Fire is one of the greatest forms of disturbance to terrestrial communities, and plays an important role in shaping vegetation patterns and plant community composition and structure in most seasonally dry landscapes (Bond and van Wilgen, 1996; Bond et al., 2005). As ecosystems recover from fire, changes occur in the composition and structure of vegetation affecting the mass, spatial arrangement and condition of fuels, and availability of habitat for fauna. Understanding how ecosystem structure changes with time since fire is therefore critical for identifying appropriate fire return intervals for biodiversity conservation and in predicting the behaviour of fires (Álvarez et al., 2009; Gosper et al., 2012).

Changes in the characteristics of fuels influences the flammability of ecosystems, with a variety of response forms of flammability to time since fire recorded, including relatively constant flammability with time since fire, a rapid increase to an asymptote, or an initial increase followed by a decline (McCarthy et al., 2001). Despite this, fire management has often assumed monotonic increases in flammability with time since fire premised on quantitative increases in fuel and changes in fuel arrangement and connectivity (Kitzberger et al., 2012). In many fire-prone

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communities, fire management for biodiversity conservation is based on the assumptions that long-unburnt vegetation declines in vigour (senescence; Bond, 1980) and above-ground plant species richness (Egler, 1954), both of which can be alleviated by fire. These assumptions on changes in fuel availability and community diversity coalesce in fire management approaches based on models postulating 'idealised' time since fire age class distributions and maximum and minimum acceptable fire intervals for specific vegetation communities (Fire Ecology Working Group, 2002, 2004).

In ecosystems dominated by slow growing, long-lived plants, changes in vegetation structure, composition and habitat features after fire may occur over decadal to century time-scales (Clarke et al., 2010; Haslem et al., 2011; Gosper et al., 2013, in press). Understanding community response to fire requires time series or space for time studies spanning these temporal scales (Watson et al., 2012; Gosper et al., in press). Yet, there are a variety of technical and logistical challenges in determining the time since fire of vegetation not burnt more recently than the period covered by contemporary sources documenting fire events, such as satellite images, aerial photos or historical records, which often only cover the last 30–60 years. Many chronosequence studies (e.g. Parsons and Gosper, 2011), therefore, have necessarily artificially truncated times since fire for long-unburnt vegetation, with poorly-understood consequences (Clarke et al., 2010; Gosper et al., in press). Generally, there is a poor understanding of patterns and time scales of temporal changes in vegetation structure after fire in many communities largely stemming from this inability to determine the time since fire of long-unburnt vegetation. This may have substantial biodiversity conservation implications, as this lack of knowledge may constrain fire management decisions where there are concerns over recent shifts in fire regimes (Parsons and Gosper, 2011), projected changes in fire regimes in the future (Prober et al., 2012), or potentially incompatible fire management objectives (Haslem et al., 2011).

The Great Western Woodlands (GWW) form the world's largest extant Mediterranean-climate woodland, with *Eucalyptus* woodlands occurring in mosaic with mallee, shrublands and salt lakes over ~160,000 km^2 (Watson et al., 2008; Prober et al., 2012). Like most other Mediterranean-climate regions, recurrent fire is a feature of the landscape (Cowling et al., 1996; O'Donnell et al., 2011a). The GWW are unique among Mediterranean-climate regions in the extent of woodland (10–25 m in height) occurring at relatively low rainfall (200–400 mm per annum; Prober et al., 2012). Mature woodlands, because of their open tree canopy structure and patchy distribution of shrubs and litter, have a low probability of burning relative to other vegetation types occurring across the same landscape (O'Donnell et al., 2011a). However, major fires in the region do occur during severe weather, especially when drought conditions follow wet and cool conditions in spring and summer of the preceding year, sustaining large fires which may burn for weeks or months (>100,000 ha; McCaw and Hanstrum, 2003; O'Donnell et al., 2011b). Recent decades have seen a number of large wildfires in the GWW, with fire intervals over this period being much shorter than in the analogous (other than in the degree of landscape fragmentation) adjoining Western Australian wheatbelt (Parsons and Gosper, 2011).

Many of the *Eucalyptus* species that dominate GWW woodland communities are sensitive to fire, being killed by complete canopy scorch. Even in those GWW woodland *Eucalyptus* species in which a proportion of the population resprouts after fire (Yates et al., 1994), growth is slow and hence changes in vegetation structure after fire may occur over protracted periods. Dense seedling recruitment of the dominant trees typically follows fires and significantly alters vegetation structure (Yates et al., 1994). Currently, there is substantial uncertainty regarding temporal changes in woodland structure and the time periods over which these changes occur (Hopkins and Robinson, 1981). Further, once mature woodlands are disturbed by fire, positive feedback between fire and post-fire vegetation structure may render regenerating vegetation more susceptible to further fire than mature woodlands (O'Donnell et al., 2011a); yet for fire management, the typical monotonic increase in fire behaviour rating with time since fire is assumed to apply (DEC, 2010). Recurrent fire in short succession could lead to unfavourable management outcomes including the loss of habitat features of long-unburnt vegetation important for a range of fauna (Watson et al., 2012), decline in carbon stocks (Berry et al., 2010) and decline in the extent of mature woodland vegetation communities which are distinct in floristic composition (Gosper et al., in press).

To inform fire management we aimed to characterize changes in vegetation structure and fuel availability over periods of more than 300 years after fire in *Eucalyptus salubris* (Gimlet) woodlands. We used a chronosequence approach (i.e. substituting space for time), with times since fire determined through a combination of satellite imagery, growth ring counts and growth ring-plant size relationships (Gosper et al., 2013). Based on common assumptions employed in fire management, we hypothesized that (1) vegetation structural components reflecting fuel availability increase with time since fire; (2) recovery of vegetation structural components with time since fire requires long time-frames; and (3) vegetation senescence is more evident in mature than intermediate fire-age classes.

2. Material and methods

2.1. Survey plots

We established 76 plots in *E. salubris* woodlands distributed along the western edge of the Great Western Woodlands, southwestern Australia; mostly near Karroun Hill (30°14′S, 118°30′E); Yellowdine (31°17′S, 119°39′E) and Parker Range (31°47′S, 119°37′E). This area has a semi-arid Mediterranean climate (see Gosper et al., 2013 for more climatic details). The region supports a mosaic of mallee, scrub-heath and woodland, with vegetation type determined locally by edaphic factors, and influenced by historic disturbances. All plots had a dominant crown layer of *E. salubris*, sometimes in association with other eucalypts. *E. salubris* is a non-lignotuberous tree widespread across the GWW (Brooker et al., 2002) that is killed by complete canopy scorch.

Regions chosen for sampling each contained extensive areas last burnt in recent (<10 years) and older (>38, but likely to be less than 60 years; Gosper et al., 2013) fires, and large areas with no evidence of contemporary fire. Plots were distributed in these times since fire across the geographic spread of sampling, with additional plots last burnt between 10 and 38 years ago sampled in the limited localities where such fires had occurred.

Stand age was determined through a combination of Landsat image interpretation, growth ring counts and growth ring-size relationships (see Gosper et al., 2013 for full details). We assume that stand age is equivalent to time since the last fire. There is little doubt over the validity of this assumption in young stands, however, it remains possible that some older stands may have experienced milder (ground) fires that did not result in widespread death of canopy trees. Post-fire observations indicate very high (close to 100%?) mortality when fires pass through *E. salubris* woodlands. Further, *E. salubris* has a very thin protective bark layer (~3 mm in width) compared with co-occurring eucalypts in which some individuals are able to resprout following fire (e.g. bark thickness of ~10 mm in *E. salmonophloia*) (Prober and Macfarlane, 2013), which lends support to our contention that fires that pass through *E. salubris* woodlands lead to extensive mortality of canopy trees.