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Fire ecology on Black Mountain

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Abstract. Early research on the ecological effects of fire on vascular plant species in dry sclerophyll forest undertaken in and around Black Mountain in the 1970s has informed subsequent fire management both on Black Mountain and in other dry sclerophyll forest communities on the NSW tablelands. However, despite a very sound understanding of the effects of fire on plant species and their short-term responses, the long-term dynamics associated with different fire regimes is an ongoing research area, particularly with a recent refocus on planned burns for fuel management on Black Mountain. Long-term ecological monitoring is an essential component of sound fire management practice, so given the proximity of Black Mountain to Canberra's academic institutions and an engaged and interested community, the mountain remains a fertile ground for ongoing fire research.

1. Introduction

Fire is an important recurrent disturbance agent in most Australian ecosystems and influences vegetation dynamics via effects on community composition, plant species richness and vegetation structure (Fox and Fox 1986; Morrison et al. 1995; Bradstock et al. 1997; Morrison 2002; Spencer and Baxter 2006; Gosper et al. 2011). Although any single fire event can have an effect, it is the accumulation and pattern of fire events over time that determine vegetation dynamics in fire prone ecosystems—the 'fire regime'. Gill (1975) defined four components of a fire regime: fire frequency, fire intensity, fire season and fire type and it is the cumulative effects of these factors operating at a point over time, interacting with underlying environmental factors, that affect vegetation composition, richness and structure. Also fundamental to an understanding of fire effects is knowledge of how individual organisms respond to a fire event. In the case of plants, there are two major response types: a) death and recovery from protected seed banks (e.g. soil; canopy) or b) vegetative recovery via sprouting from aerial or subterranean protected buds (e.g. epicormics; lignotubers; bulbs) (Gill 1975; Gill 1981).

Because animals are mobile, their response to fire is somewhat different compared to sedentary organisms such as plants. Animals can, depending on fire spread and fire intensity, either move out of the way of a fire front (e.g. birds and some macropods) or in the case of burrowing organisms, survive in burrows (e.g. wombats). Arboreal mammals and small ground dwelling mammals are more limited in their ability to avoid fire so must either persist *in situ* or, if local populations are killed by fire, must recolonise back into burnt areas post-fire if local populations are to persist (Catling and Newsome 1981; Posamentier et al. 1981; Catling et al. 2001; Lindenmayer et al. 2009, 2010, 2013). In the case of Black Mountain, there has been significant research undertaken on plant responses to fire, but much less research on the responses of animals.

2. Fire history of Black Mountain

Black Mountain has experienced many unplanned ('wildfire') and planned ('prescribed burn') fire events over the past 50 years. Fig. 1 shows the fire history of Black Mountain from 1966–2015. While much of the central steep area around Black Mountain summit and some of the lower north-west flanks had no recorded fire activity from 1966–2000, approximately one third of Black Mountain on the north- and south-eastern flanks was burnt by unplanned fires in four separate events: 1983, 1985, 1991 and 1994. Additionally, two small unplanned fires occurred in 1993 and 1999 on the southern and western flanks respectively. A significant proportion of the areas burnt in the north-east had two unplanned fires close together: 1991 and 1994 (3 years apart), and one smaller area had three unplanned fires close together: 1983, 1991 and 1994 (8 years apart, then 3 years apart). During the 1966–2000 period, a number of planned burns were undertaken, the larger being in the north-western areas of Black Mountain in 1978, 1979, 1990 and 1999. Smaller areas were burnt by planned fire in 1977, 1985 and 2000.

The Canberra fires of 2003 burnt significant areas of the ACT and the eastern edge of the fire front burnt up to cork oak plantation immediately to the south-east of Black Mountain, but did not burn into Black Mountain. Post-2003 fire management has seen a marked increase in the number of planned burns in the ACT generally, and on Black Mountain in particular. The areas targeted for planned burns post-2003 are occurring in many of the areas burnt by both planned and unplanned fires up to 2000; but while the total recorded area burnt pre-2003 occurred over 23 years, the same total area has been burnt post-2003 in 12 years, all from planned fires.

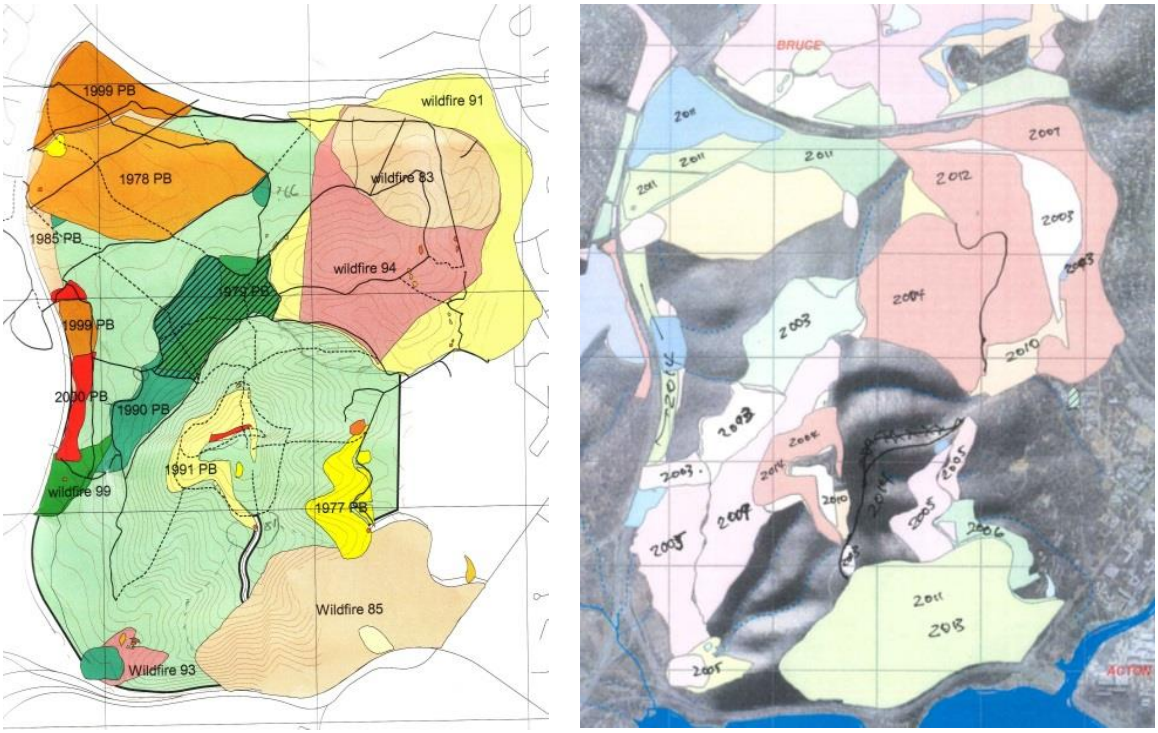

Fig. 1. a) left, fire history 1966–2000. b) right, fire history 2000–2015.

It is important to document fire history in order to plan for both fuel management and conservation management. The concept of the fire regime is fundamental in such planning for framing and understanding the ecological effects of fire. Recording fire history enables variables such as time since last fire, inter-fire intervals and fire frequency to be determined and used in ecological fire management (Kitchin 2008). Additionally, information on the effects of fire intensity—as measured post-fire, by fire severity—can further assist in determining management actions. For example, the cumulative effects of two successive low intensity fires will not necessarily be the same as the cumulative effects of two high intensity fires (Doherty 2011).

3. Fire, flora and vegetation structure

The Bruce Ridge area immediately to the north of Black Mountain contains similar dry sclerophyll vegetation types to those found on Black Mountain itself, both areas being dominated in the canopy by *Eucalyptus rossii*, *E. mannifera* and *E. macrorhyncha*, with occasional *E. polyanthemos* and *E. dives* (see Doherty 2018). Bruce Ridge was the focus of early work on fire behaviour and the response of dry sclerophyll forest communities to fire. CSIRO Division of Forest Research conducted 210 experimental burns in this area between 1964 and 1970 to study the behaviour of planned fires under controlled conditions. This work formed the basis for initial descriptions of fire behaviour relating fuel loads to rate of spread, fire intensity and flame height in dry eucalypt forests (McArthur 1967; Luke and McArthur 1978) and also fuel moisture, fuel availability and fire characteristics (Ashcroft 1967).

The area in which these burns were undertaken was subsequently the focus of work by Davis et al. (1975) investigating plant responses to fire including whether species resprouted or were fire-killed and whether there were changes in community composition and vegetation structure in relation to fire frequency and fire intensity. The overall finding was that although composition remained unchanged under a variety of fire regimes, density (total no. individuals/total area quadrats), abundance (total no. individuals of a species/area quadrats occupied by the species) and frequency (no. occupied quadrats/total no. quadrats) did vary significantly, with abundance and frequency generally increasing, particularly after moderate or high intensity fire. Vegetation structure was also altered with reduced cover of shrubs and grasses, but as the dominant understorey species were resprouters, this was a short-term effect.

Additional concurrent work on Bruce Ridge tracked plant responses after experimental burns in late January – early February 1973 in three areas with different times since fire: A (burnt 9 years previously in 1964); B (burnt 12 years previously in 1961); and C (burnt 33 years previously in 1940) (Purdie and Slatyer 1976). Despite differences in time since fire, plots responded in similar ways after the 1973 burns and the floristic composition of the sites did not change. Essentially, the plant species that were recorded pre-fire were also recorded post-fire, corresponding to an 'initial floristic composition' model of secondary succession (Egler 1954). This can be understood by the fact that the majority of plant species recorded in the community resprouted via vegetative means after fire, with 50–75% of plants commencing regrowth within two months of the burns (Purdie 1977a). Hence, not only is the composition unchanged, the same individuals are present pre- and post-burn. Only three species were killed by fire and responded via germination from soil stored seed: *Acacia genistifolia*, *Dillwynia phylicoides* and *Pimelea linifolia* (Purdie 1977b). The post-fire dynamic did result in some shifts in understorey dominance two years post-burn, compared to pre-burn, but the apparent changes in species dominance primarily reflected recovery rates based on whether the dominant understorey species resprouted or were killed. For example, *Rytidosperma pallidum* and *Daviesia mimosoides* resprouted vigorously post-burn to establish cover quickly. In contrast, *Acacia genistifolia* and *Dillwynia phylicoides*, although numerically more abundant post-fire, were establishing as seedlings so their cover was initially low. Although many seedlings across both resprouter and fire-killed species germinated, there was a high mortality rate in the first year post-burn.

Purdie (unpublished) re-surveyed the Bruce Ridge plots in 1982, nine years after the experimental burns, and again in 2006, 33 years after the experimental burn at site A and five years after a wild fire burnt sites B and C. Species richness at each site remained similar up to four decades post-fire. However, the density of the three dominant shrub species dropped significantly 10–20 years post-burn, initially from the deaths of *Dillwynia phylicoides* plants (10–15 year longevity) and then from deaths of *Acacia genistifolia* (15–20 year longevity) and to a lesser extent *Daviesia mimosoides* plants (20–25 year longevity). All three species exhibited a steady although low level of seed germination and seedling survival in unburnt areas over the same period, while the rhizomatous resprouter species *Dianella revoluta*, *Lomandra filiformis*, *L. longifolia* and *L. multiflora* had a sustained increase in the number of shoots in both burnt and unburnt areas over that time. The 2001 wildfire appeared to result in greater mortality of mature plants of *Daviesia mimosoides* (a resprouter) compared with the 1973 burns. Recovery of its population by seed germination and regrowth was very low by 2006, as was the recovery of the *Dillwynia phylicoides* and *Acacia genistifolia* populations from seed germination. This may reflect a combination of either low seed germination and/or low seedling and resprout survival due to the drought condition that prevailed after the fire. The data suggested that repeat fires at 10–15 year intervals would maintain a dense low shrub stratum of the above three species, assuming a burn intensity sufficient to trigger mass seed germination (i.e. a high intensity burn) and the post-fire absence of prolonged drought.

Although highly applicable to the management of flora on Black Mountain and also in Aranda Bushland, the work undertaken on Bruce Ridge in the 1970s marked both the beginning and the end of formal fire research in the area. However, post-2003, with an increase in the number and area of planned burns, there has been a revival of interest in fire ecology on Black Mountain. Doherty (unpublished) established plots prior to a planned burn on the lower western slopes of Black Mountain that was last burnt by a planned burn in 1990. The community is similar to that studied by Davis et al. and Purdie, with a canopy of *E. rossii – E. macrorhyncha*, and occasional *E. polyanthemos*. Four 20 m × 20 m plots were established: two upslope in an area previously burnt in 1990 and two down slope in a long-unburnt area. The two upslope plots were the burnt in September 2003, with the two down slope plots remaining unburnt. All vascular plant species were recorded on each plot pre-fire using a 1–6 cover abundance scale and remeasured opportunistically post-fire, with the last sample in late 2017. Similar to earlier findings, overall composition is little affected by burns of low to moderate frequency i.e. greater than 10 years apart. This is also reflected in terms of vascular plant species richness, where there is no difference over time between plots burnt in 1990 and 2003 and plots not burnt since 1966 (Fig. 2).

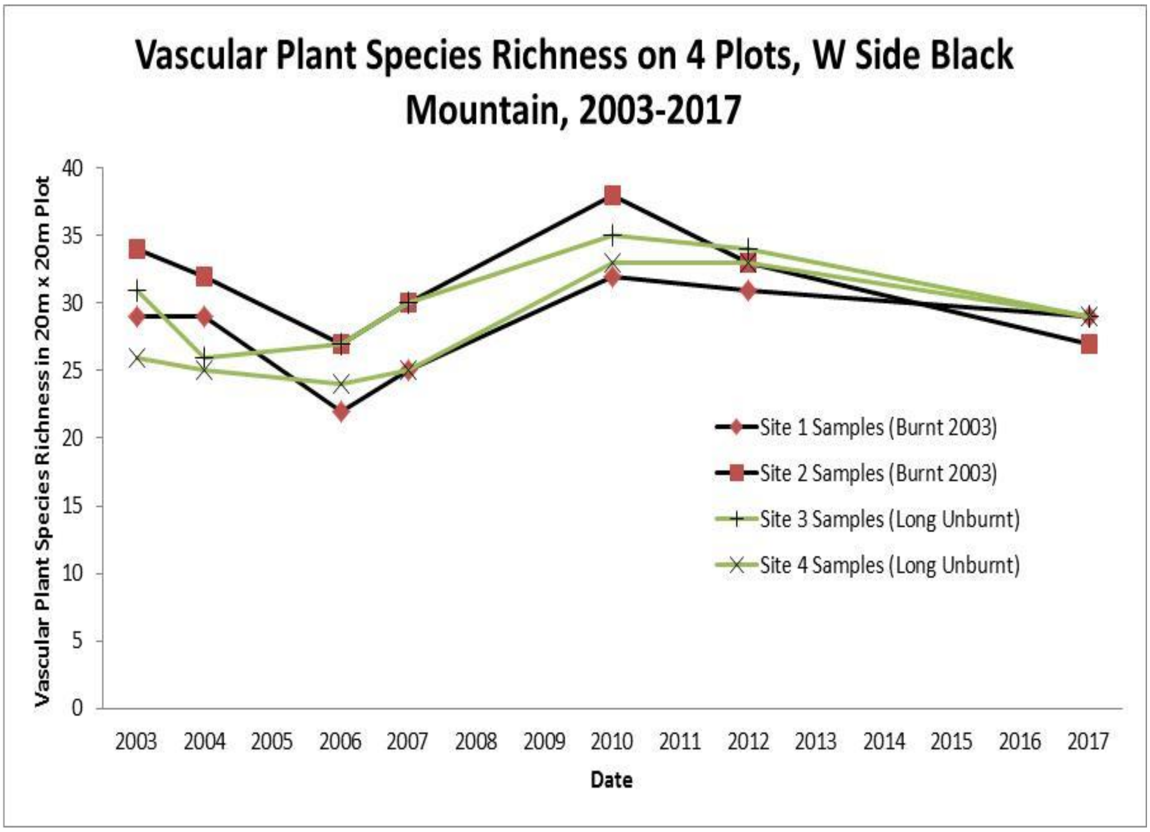

Fig. 2. Vascular plant species richness in burnt and unburnt plots, 2003–2017.

However, there are short-term persistent changes in understorey structure as exemplified by the shrub *Dillwynia phylicoides*, highlighting the complexities involved in managing fire-killed plant species with soil seed banks. Based on the abundance of the species pre-fire in 2003, the 1990 burn caused a successful germination event such that by 2003, there was a mature and obvious layer of *D. phylicoides* on plots 1 and 2. Post the 2003 burn though, the species has yet to achieve its pre-fire abundance in a similar period (13 years) post-fire (Fig. 3). Conversely, on plots 3 and 4 that have not had fire since at least 1966, the abundance of *D. phylicoides* was initially, and still remains, low.

Fig. 3. Site 1, lower western slopes of Black Mountain. a) top left, pre-fire 1 September 2003 with layer of *Dillwynia phylicoides* and *Rytidosperma pallidum*. b) top right, post-fire 8 September 2003. c) bottom, post-fire 14 November 2017 with regenerated *R. pallidum* but very sparse *D. phylicoides*. Photos: M Doherty.

It has been shown that there is a critical minimum level of soil temperature that is required to stimulate the germination of Fabaceae seed (Auld and O'Connell 1991). While the 1990 fire is assumed to have achieved these temperatures, the 2003 burn did not, resulting in poor germination. A second fire too soon under such circumstances may further reduce the local population at these sites and if the seed bank is not replenished over the long-term, would potentially lead to local extinction of the species at these sites. Fire response in the Fabaceae is complicated by the fact that although most species appear to have a long-lived soil seed bank, some shrub species such as *D. phylicoides* are fire-killed whereas other significant understorey Fabaceae such as *Daviesia mimosoides* subsp. *mimosoides* are resprouters. Long-term changes in fire frequency such as decrease in interfire interval would be expected to favour the latter species over the former.

Hence, although the majority of plant species in dry sclerophyll forests appear to be relatively insensitive to fire frequency, fire intensity and fire seasonality, some species are sensitive to short interfire intervals or fires of too low an intensity. The fire-killed *Grevillea alpina*, whose non Victorian distribution is confined to Black Mountain, is also vulnerable to fires of too high a frequency and compared to members of the Fabaceae, the soil seed bank of *Grevillea* is not as persistent (Auld et al. 2000). Fire intervals that do not allow for adequate germination and seed set will reduce populations over time. Other fire-killed species vulnerable to short interfire intervals are *Callitris endlicheri* (Black Cypress Pine) and *Pomaderris intermedia* (Golden Pomaderris, Lemon Dogwood). *Callitris endlicheri* has a canopy seed bank with seeds falling gradually over time and with seedling establishment occurring either in the absence of fire, or else after adult plants are killed by high intensity fire. Its populations expand gradually around parent plants. The fire ecology of *P. intermedia* is not known, other than the fact that the species occurs in isolated patches and expands slowly in the absence of fire similar to *C. endlicheri*. It is likely to have a soil seed bank but adult and seed longevity are not known. In all of these cases of fire-killed species, irrespective of the type of seed bank, if a plant is killed by fire and a second planned or unplanned fire occurs before seedlings reach maturity, local extinction will result. In such species, understanding the impact of frequent fire as well as the interaction between fire frequency and fire intensity is of critical importance for long-term management.

Some groups of plants may also be sensitive to the season of burn. For example, over 60 species of terrestrial orchids occur on Black Mountain (Purdie 2018a) and patterns of orchid diversity there are complex, with little relationship with fire history *per se* but with some species strongly associated with fire frequency (Seddon unpublished data cited in Mulvaney 2018). There is a variety of flowering times across the full range of orchid species, with some flowering in spring, some in summer and some in autumn (Bullen 2003). Hence, the season of burning is also a factor affecting orchid distribution and abundance As terrestrial orchids lose their leaves and become dormant, they need to re-leaf in the growing season so that they can grow new tubers for the following season. Fire at an inappropriate time of year will not allow this to occur and may cause local extinction with recurrent fire.

Fires of too high an intensity can also be an issue and cause canopy tree loss over time. Too frequent or too intense a fire can lead to the loss of old trees, particularly those with hollows. Whether from planned or unplanned burns, fires may smoulder in hollowed-out trees many days after a burn and eventually cause collapse of the tree.

4. Past and current fire management

Although fuel management burns have been undertaken sporadically on Black Mountain for many years, there has been an increased focus on fuel management on Black Mountain since the 2003 Canberra fires. A state of knowledge review on the ecological effects of fire was undertaken for the ACT after the 2003 fires (England et al. 2004) and this was translated into notional fire thresholds or ecological burning parameters for plant communities based on knowledge at the time. This review incorporated a fire threshold for the *E. rossii* — *E. macrorhyncha* community based on *Davis* et al. (1975) of a 5–8 years minimum interval between fires. Kitchin (2008) further revised this to a 10 year minimum interval between fires in dry sclerophyll forest.

Although such intervals notionally allow vegetation recovery, time since fire alone does not necessarily indicate what the effects of the last known fire were in relation to other factors such as fire intensity, fire patchiness or post-fire conditions such as low rainfall. Hence, long-term monitoring is a fundamental component of fire management. Based on current knowledge of plant response to fire, Doherty and Meyers (2009) proposed a fire effects monitoring protocol for Black Mountain and Aranda Bushland that targeted a selection of both dominant and fire interval dependent understorey species. The focal species were: *Daviesia mimosoides* subsp. *mimosoides* (a dominant understorey resprouting shrub with a long-lived soil seed bank); *Rytidosperma pallidum* (a dominant resprouting understorey grass with a seed bank of unknown longevity); *Dillwynia phylicoides* (a dominant understorey fire-killed shrub with long-lived soil seed bank); *Grevillea alpina* (a common understorey fire-killed shrub with short-lived soil seed bank). A series of six plots, each consisting of six 1 m × 1 m subplots has been established on Black Mountain and in Aranda bushland, with community involvement, so that ongoing monitoring can be undertaken in a simple targeted fashion.

A selection of the Black Mountain plots has also been incorporated into long-term monitoring being undertaken by the Friends of Black Mountain (Beveridge 2018) as part of the community Vegwatch program (Sharp and Gould 2014). A 20 m × 50 m Vegwatch monitoring site on the south-eastern side of Black Mountain was surveyed two weeks prior to a fuel reduction burn in March 2014 and re-surveyed in November the same year. The area appeared to have been long-unburnt prior to the March burn (based on Fig 1a). All *Dodonaea viscosa* subsp. *spatulata* shrubs present in March were killed by fire, but seedlings of the species developed post-burn. All other perennial species present before the burn also recovered from either seed germination and/or vegetative regrowth. The perennial species *Astroticha ledifolia*, *Billardiera scandens*, *Cassinia longifolia*, *Dillwynia phylicoides* and *Hardenbergia violacea* were recorded in the post-fire vegetation in November 2014, but not prior to the burn. While seed of *Cassinia longifolia* may have been blown in after the fire, the other species would have germinated from the soil seed store. The vegetative regrowth of many of the perennial herbacous species had flowered by November 2015 (i.e. 20 months after the burn). Some vegetative regrowth of the woody species *Acacia buxifolia*, *A. gunnii*, *Hardenbergia violacea* and *Monotoca scoparia* had flowering buds or flowers present in April 2018, as did some of the *Dillwynia phyllicoides* and *Dodonaea viscosa* subsp. *spatulata* plants, all of which were derived from post-fire seed germination. This indicates that the primary juvenile period of these fire-killed soil seed bank reliant species is at least four years.

The impacts of fire on broader landscape function, i.e. the processes that regulate the movement and use of water, topsoil and organic matter (Tongway and Hindley 2004), is also an area than requires long-term monitoring. Sharp (2011) found that overall, 95–99% of Black Mountain was in satisfactory condition as regards to landscape function, but that recently burnt areas, comprising 1–5% of the reserve, were approaching critical condition. Fires which are too frequent and/or not patchy in nature can overly reduce vegetation cover, litter cover and coarse woody debris over time and lead to increased erosion. Sharp recommended that landscape function needs to be monitored in key locations to ensure fires do not occur at a frequency that compromises stability, infiltration and nutrient cycling.

Anecdotal observations of researchers and the local community can also assist with investigating long-term changes in species distribution and abundance and vegetation structure. For example, Purdie (2018b) recollects that during 1972–76 a dense shrub layer was present in many areas of Black Mountain; its reduction today over many parts of the reserve may be attributable to the fuel reduction program. The interplay of fire and vegetation structure is complex in that it appears that a reduction in shrub layers can result from either too frequent fire at one extreme, or from the long absence of fire at the other extreme. Jacobs (1955) notes that *Daviesia mimosoides* subsp. *mimosoides* has a finite lifespan and that in the absence of fire, it senesces over approximately 25 years. Nonetheless, it possesses a long-lived soil seed bank so that even in areas where it is in low abundance or apparently absent currently, a high intensity fire can germinate seed from the soil seed bank and re-initiate a shrub cycle. Such knowledge is invaluable when decisions are made as to fire frequency and fire intensity parameters for planned burns, involving a trade-off between levels of fine fuel accumulation and the persistence, or creation of, an aerial fuel layer (Gould et al. 2007).

While the direct effects of fire are important in assessing impacts on plant persistence, changes to habitat structure or food resources are critical in the persistence of terrestrial vertebrate species. Trappe et al. (2006) used two study sites, one on Black Mountain and one in Aranda Bushland, to study the short-term effects of fire on hypogeous fungi, an important resource for small- to medium-sized mammals. They established ten plot pairs in each site, in burnt and adjacent unburnt vegetation. On the Black Mountain plot in 1999, 18 hypogeous fungal species were recorded in unburnt plots and two species in burnt plots; 23 species were recorded in the unburnt plots and 10 species in the burnt plots in 2000. The overall trend was towards recovery of fruiting body abundance, with differences converging over time.

5. Management implications over the next 50 years

An increased focus on fuel management burns post-2003 has led to an increase in fire frequency in some areas of Black Mountain. Given current knowledge of plant species' responses to fire, there is a need to keep a focus on changes in populations of fire-killed, fire-interval-sensitive plant species via long-term monitoring. Additionally, changes in vegetation structure should also be a focus for monitoring as this component is of critical importance for the maintenance of animal populations and diversity. While plant populations are buffered to some degree by the dynamics of resprouting and the presence of persistent seed banks, animal populations are likely to be more vulnerable to changes in the short-term. Evans (2018) considers that introduced predators combined with a reduction in cover and coarse woody debris from long-term fuel management will impact significantly on the persistence of small native ground dwelling mammals. Similarly, Fennell (2018) has noted that the Black Mountain avifauna is now comprised mainly of common species and notes in particular a dramatic reduction in the amount of decaying wood and litter in areas where hundreds of Painted Button-quail were common, whereas there have only been two sightings of this species since January 2006.

While vegetation is in many ways dynamic, it is usually so over long periods of time rather than over the short-term. Permanent shifts in vegetation composition or structure occurring over short periods of time are more likely to result in adverse conservation outcomes for plant and animal species. Changing fire regimes from anthropogenic causes are the major factor that will determine the composition and structure of vegetation and in turn, the persistence of animal populations, on Black Mountain over the next 50 years. The challenge is to be able to discern between permanent and non-permanent changes in flora and fauna: this requires long-term ongoing research and monitoring.

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