Box-Ironbark Experimental Mosaic Burning Project

Report to the Department of Environment, Land, Water & Planning and Parks Victoria

Acknowledgements

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Cover image: A planned burn in a box-ironbark forest (Andrew Bennett).

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Executive summary

Planned burning is used to reduce bushfire risk, and for managing ecosystems for conservation purposes. In response to the Victorian Bushfires Royal Commission 2009, the State of Victoria committed to an increase in the annual extent of planned burning on public land. To achieve this target large 'landscape mosaic burns' are being conducted. The effect of such burns on plants, animals and ecological processes is poorly understood. The Box-Ironbark Experimental Mosaic Burning Project is a collaborative project between the Department of Environment, Land, Water and Planning, Parks Victoria, and researchers from Deakin and La Trobe Universities. It was established to improve our understanding of the ecological outcomes of planned burns.

Objectives

The aim of the Box-Ironbark Experimental Mosaic Burning Project was to use an experimental approach to assess the ecological outcomes of planned 'landscape mosaic burns' in a large, continuous box-ironbark forest (Heathcote-Rushworth-Graytown Forest). Planned burns were conducted in replicate landscapes to allow comparison of two main aspects of a fire mosaic: a) the extent of burnt vs unburnt vegetation in a landscape; and b) the season of burn (autumn, spring). Key objectives of the study were:

a) to implement planned burns that differ in the extent of burn across a landscape and season of burn;

b) to monitor forest structure, selected habitat components, plant species composition, and selected faunal groups before and after the experimental burns;

c) to evaluate the effects of different burn mosaics (i.e. extent of burn) on these attributes; and

d) to investigate the effect of mosaic burns on selected animal species, including effects on individuals and their behaviour and use of resources.

Methods

Twenty-two 'landscapes' were selected for study, each ~100 ha in size. Six landscapes remained as unburned reference areas, eight were assigned to be burned in autumn 2011 and eight in spring 2011. None of the study landscapes had been burned for more than 30 years prior to the study. Within each study landscape, 12 plots (each 20 x 20 m) were systematically located to assess and monitor ecological attributes.

Assessment of fuel hazard prior to the experimental burns showed that hazard levels in all landscapes were low – moderate (i.e., none were high, very high or extreme). Monitoring of ecological attributes before the burns showed no systematic differences between landscape treatment groups (i.e., for most ecological attributes measured, there were no pre-existing differences between landscapes to be burned in autumn, landscapes to be burned in spring, and reference landscapes).

The autumn burns ranged in extent from 22–51% of the landscape burnt and for spring burns it was 52–89%. Weather conditions (e.g., abnormally high summer rainfall prior to autumn 2011) and other factors meant that the goal of half the burns in each season being 30–50% and half 70–90% was not achieved. The separation of burn outcomes between seasons (i.e., all autumn burns <51%, all spring burns >51%) has greatly limited the ability to compare ecological outcomes between burn seasons.

Results

Habitat components

The effect of planned burns on numerous habitat components was significantly related to the overall extent of burning. The number of logs and stumps, the cover of coarse litter (>1 cm diameter) and average litter depth all decreased with increasing percentage of the landscape burned. For logs, size-class was also important: for any given burn extent, large logs (>20 cm diameter) were consumed at a greater rate than smaller logs. The cover of low and medium vegetation (<1 m height) also declined with increasing burn extent, but cover of vegetation at higher levels (>1 m) was not significantly affected.

Plant species richness and occurrence of individual species

The richness of perennial plants per landscape was greater in post-fire surveys (2012 and 2013) than in the pre-fire survey (2010), but there was no evidence that this was clearly related to burning. This increase between years likely reflects plant germination and growth stimulated by drought-breaking rains experienced in 2010/11. The extent of burn was an important influence on plant species richness at the plot level (20 x 20 m): species richness increased with burn extent up to a burn cover of ~40%, before slowly declining as burn cover reached higher levels. The number of weed species per plot increased with increasing extent of the plot burnt. There was also a seasonal effect: more weed species were recorded in spring-burned landscapes than in unburned landscapes, but this was not true for autumn burnt landscapes.

Of seven individual plant species modelled, two (Gold-dust Wattle *Acacia acinacea* and Daphne Heath *Brachyloma daphnoides*) showed no change in occurrence at the landscape-level associated with burn extent or year of survey. Year of survey, but not burn extent, was an important predictor for four species (Spreading Wattle *Acacia genistifolia*, Drooping Cassinia *Cassinia arcuata*, Gorse Bitter-pea *Daviesia ulicifolia* and Twiggy Bush-pea *Pultenaea largiflorens*): all were detected at a higher proportion of plots in the post-fire surveys compared to the pre-fire survey (likely related to drought-breaking rains in 2010/11). Cranberry Heath *Astroloma humifusum* was the only species that displayed a response to burn extent at the landscape level: occurrence was largely unchanged by a burn extent of up to ~45%, but as burn extent increased beyond 45%, occurrence declined quickly. At the plot level, *A. humifusum* showed the same response of decline with burn extent as it did at the landscape level. Two other species (*A. genistifolia* and *B. daphnoides*) also showed some evidence of decline in occurrence with increasing burn extent at the plot level, although modelled relationships with burn extent were not strong.

Vegetation composition

Vegetation composition (based on all vascular plant species) differed in relation to burn extent categories (i.e., unburned, low <50%, high >50% of landscape burned) when compared between landscapes for 2012 and 2013. The greatest difference was between unburnt and extensively burnt landscapes, with species contributing to this difference being mostly annuals (mainly introduced species) that were more frequent on burned landscapes. When only perennial species were included, differences in vegetation communities between burn treatments were less marked.

Red Ironbark flowering

There was no evidence that burning affected flowering of Red Ironbark *Eucalyptus tricarpa* in 2013 (two years after experimental burns). Variation in the percent of trees flowering was most strongly related to geographic location and forest structure.

Bird species richness and occurrence of individual species

A total of 93 species of terrestrial birds were encountered. The number of bird species detected per landscape was not influenced by the extent to which a landscape was burnt. However, species richness did display temporal variation with fewer species being detected in surveys conducted in 2012 (winter and spring) compared to other survey periods (spring 2010, 2011 and 2013). These differences likely reflect seasonal/temporal variation in environmental conditions and resources (e.g., rainfall, food availability).

Models were constructed for 25 individual species. For nine species, frequency of occurrence at the landscape-level did not change in relation to burn extent or survey period. Survey period (but not burn extent) was an important influence for ten species. The response of these ten species to survey period was variable, and again likely reflects local conditions. Seasonal migratory movements of some species were evident. The occurrence of six species was related to burn extent, but the nature of these relationships varied among species, and across survey periods for a given species. Occurrence of the Fuscous Honeyeater *Lichenostomus fuscus* increased with increasing burn extent in three of the four post-burn survey periods. Other species to increase with burn extent were the Scarlet Robin *Petroica boodang* (in two of four post-burn survey periods), Spotted Pardalote *Pardalotus punctatus* (one of four), and Weebill *Smicrornis brevirostris* (one of four). Two species declined with increasing burn extent: the Grey Fantail *Rhipidura fuliginosa* (one of four) and White-throated Treecreeper *Cormobates leucophaeus* (two of four). Although burn extent was identified as an important influence on these species, change in frequency of occurrence along the burn extent gradient was not pronounced (i.e., burn extent was typically not a strong influence).

Bird species composition

There was no evidence that the overall composition of the bird community was influenced by burning in any of the post-burn survey periods. However, composition did vary through time, irrespective of burning. Analysis of the reference landscapes only, revealed that the composition during the winter 2012 survey was different to all other survey periods (spring 2010, 2011, 2012 and 2013). This was largely driven by five species. The Black-faced Cuckoo-shrike *Coracina novaehollandiae*, Common Bronzewing *Phaps chalcoptera* and Rufous Whistler *Pachycephala rufiventris* were encountered in spring but not in winter, while the reverse was true for the Golden Whistler *Pachycephala pectoralis* and Flame Robin *Petroica phoenicea*. This result largely reflects seasonal migratory movements by these species.

Individual faunal species

Effects of burning were investigated for three animal species: Scarlet Robin *Petroica boodang*, Yellow-footed Antechinus *Antechinus flavipes* and Brush-tailed Phascogale *Phascogale tapoatafa*.

Scarlet Robins survived the immediate effects of burns and marked individuals were observed up to a year post-fire. After the burns, individuals increased the size of their territory (up to 300%), expanding into additional burned forest rather than shifting to unburned areas.

Yellow-footed Antechinus also survived the immediate effects of burning, but longer-term survival is not known. Individuals showed strong selection for large trees (>60 cm diameter) and large logs (>40 cm diameter) as den sites; these are rare in this box-ironbark forest (<1% of trees are >60 cm diameter). Even though burn extents were relatively low in the two landscapes where Yellow-footed Antechinus were studied (covering just 41% and 51% of the landscapes), autumn burns destroyed 31% (16/52) of known den sites.

Post-fire surveys for the Brush-tailed Phascogale recorded the species in 13 of 14 landscapes surveyed. Activity was influenced by burning: phascogales were less likely to be recorded at burnt plots and, at the landscape-scale, activity decreased as the extent of the landscape burnt increased.

Herbivory

An experiment using fenced exclosures was designed to test whether herbivore (e.g., Eastern Grey Kangaroo *Macropus giganteus*, Black Wallaby *Wallabia bicolor*, Goat *Capra hircus*) grazing on vegetation is influenced by burning patterns. Richness of perennial plants in the year 2013 was best predicted by whether a plot was burnt or not (greater richness in unburnt plots), and not related to grazing exclusion. For vegetation structure, in 2013 there was evidence that both grazing exclusion and burning were influential. Greater structural complexity <50 cm above ground occurred in plots that were not burnt and not grazed.

Discussion

This study has documented the effects of planned burning in a box-ironbark forest on a range of ecological attributes in the short-term (up to two years post-fire). Several general conclusions can be drawn at this stage.

1. There is little evidence for any *ecological* benefit from the planned burns, at least in the short term. Longer-term monitoring will provide greater insight as to whether burning enhances germination and persistence of some plant species. In contrast, there is evidence that burning results in depletion of habitat resources for a range of faunal species.
2. Patchy mosaic burning has less ecological impact than extensive burning, with less loss of habitat components (e.g., logs, vegetation cover). If burning is necessary, a patchy burn of approximately 50% cover or less, with unburnt patches scattered throughout the landscape, will reduce fuel levels while having less ecological impact than extensive burning.
3. Frequency of burning is a critical issue, given that even low-cover burns result in loss of faunal habitat components. The interval between burning needs to be sufficient for habitat resources to be replenished. Given the low productivity of box-ironbark forests, it will take many years for resources such as logs and deep litter layers to re-accumulate. It is unlikely that such re-accumulation will occur within the current *minimum* tolerable fire interval for the Box-Ironbark EVC (12 years). Extending the frequency of occurrence of planned burns, within the proposed tolerable fire interval (12 to 150 years), will allow greater opportunity for replenishment of habitat resources.
4. Given the commitment to increased levels of burning on public land annually, it is critical to undertake strategic planning to develop a vision and target for the post-fire, age-class structure of these forests. This requires determining the appropriate mix of forest growth stages, at a regional scale, that will ensure the resilience of this ecosystem and its flora and fauna. A high priority is to determine those areas to be maintained as the ‘long unburnt’ growth stage.
5. The study landscapes had low fuel hazard, even after 30+ years without fire. A risk-based approach to determining strategic locations for planned burns has greater merit, from an ecological perspective, than burning large areas to meet an area-based target.
6. This collaborative project has established a carefully designed infrastructure and baseline data set that provide an excellent foundation for long-term monitoring. We recommend a commitment to regular and systematic monitoring of key forest and habitat attributes.

Introduction

Following the 'Black Saturday' bushfires of 2009, the State of Victoria committed to implementing Recommendation 56 of the Victorian Bushfires Royal Commission (Teague *et al.* 2010), that prescribed burning be increased to an annual rolling target of 5% minimum of public land in the State, and Recommendation 58 to monitor and model the effects of that burning on biodiversity. Consequently, the amount of planned burning being conducted across the State has increased substantially, with much of it taking the form of landscape mosaic burning. Implementing planned burning in a mosaic pattern was considered to potentially achieve increased fire safety by preventing the buildup of fire fuels, whilst also having potential benefits for biodiversity (Victorian Government 2008).

The effect of such landscape mosaic burns and the way in which different properties of a fire mosaic (e.g., the overall extent of burnt vs unburnt, the patchiness of the burn, intensity of burn) affect the status of plants, animals and ecological processes, is poorly known. This is particularly true for Victoria's dry box-ironbark forests (Fig. 1).

The Box-Ironbark Experimental Mosaic Burning Project was established in 2010 as a collaborative project between the Department of Environment, Land, Water and Planning (DELWP), Parks Victoria (PV), and researchers from Deakin and La Trobe Universities. The aim of the project is to use a large-scale, experimental burning protocol to assess the ecological outcomes of planned mosaic burns in an extensive box-ironbark forest, the Heathcote-Rushworth-Graytown Forest.

Box-ironbark forests

In Victoria, box and ironbark forests primarily occur inland of the Great Dividing Range in the 'Goldfields Bioregion' of the State. They are dry sclerophyll forests dominated by eucalypts, with a shrubby understorey (Muir *et al.* 1995; Environment Conservation Council 1997). Seventeen Ecological Vegetation Classes (EVCs) have been described for the terrestrial vegetation of the region (Muir *et al.* 1995), with two of the most widespread EVCs being Heathy Dry Forest and Box-Ironbark Forest.

Disturbance history

Since European settlement, this ecosystem has been highly modified, with clearing of >70% of the former forests (Environment Conservation Council 1997). The remaining forests have experienced major disturbance over the last 150 years associated with gold mining, timber harvesting, and many other land uses (Environment Conservation Council 1997; Lawrence and Bellette 2010). Historical land use has had a marked effect on the ground layer and forest structure in the region (Environment Conservation Council 1997).

Fire regimes

Little is known of historical fire regimes in box-ironbark forests in Victoria (Environment Conservation Council 1997; Tolsma *et al*. 2007a). Prior to European settlement, aboriginal people in northern Victoria employed fire as part of food gathering and land management activities in grassy woodlands (Curr 1883),

Figure 1. Box-ironbark forest in the Heathcote-Rushworth-Graytown Forest (Andrew Bennett).

but their use of fire in dry shrubby box and ironbark forests is not known. Similarly, there is little detailed knowledge of burning patterns in the early years of European settlement. Some anecdotal reports exist, such as that of a Goldfields Commissioner in 1853, describing 'black charred gullies' when visiting the Rushworth area (Lawrence and Bellette 2010).

Since the 1970's (when reliable records began to be kept) there have been relatively few fires (wildfire or planned burns) recorded in Victoria's box-ironbark forests (Fig. 2). Most areas have not been burnt in the last 40–50 years, and areas that have burnt have typically experienced just one fire. The dominant feature in the Heathcote-Rushworth-Graytown Forest is a planned burn along a road running through the middle of the forest to create a strategic fire-break (Fig. 2) (note: no study landscapes in this project contain areas mapped as being previously burnt). The present fire regime is primarily driven by fires caused by human agency rather than ignition by natural causes (e.g., lightning). For example, data for the period 1983/84 to 2002/03 for the Bendigo Fire District (Department of Sustainability and Environment 2003) show that only 7% of fires (133/1849) were attributed to lightning, with the remainder being of human origin (deliberate or accidental) (66%), or of unknown cause (27%). Most (93%) of these fires were ≤5 ha in area, with only five fires >400 ha. The largest during this period was in Jan 1985 at Maryborough, when 50,800 ha of land was burned (Department of Sustainability and Environment 2003; Fig. 2).

Aspects of current fire management

Planned fire has been used mainly to reduce fuel loads and risk of bushfire in forests close to regional cities and towns (e.g., Bendigo, Castlemaine, Rushworth, St. Arnaud) and to create strategic breaks through public land blocks (Department of Sustainability and Environment 2003). Typically, until 2009, approximately 2,000 ha was burned annually in the Bendigo Fire Management Area (Department of Sustainability and Environment 2003). However, the implementation of current government policy means that the annual burn area target is greatly increased, to more than 11,000 ha per annum.

Few ecological studies of fire have been conducted in the box-ironbark ecosystem (but see Meers and Adams 2003; Orscheg 2006), and little is known regarding either the short term or longer term responses of plants and animals to fire regimes (Tolsma *et al.* 2007a). In particular, little is known of the potential ecological effects of implementing planned burning at a more extensive scale, thus shifting the post-fire age structure of these forests at a regional level.

Tolerable fire intervals (TFI) based on plant life history attributes are used as a guide to fire management in different broad vegetation classes in Victoria (Cheal 2010, Noble and Slatyer 1980). The most fire-vulnerable plant species, or Key Fire Response Species (KFRS), are used to indicate the tolerable upper and lower limits of fire frequency at a site. For the Ironbark/Box Ecological Vegetation Division (EVD), the <u>minimum</u> tolerable fire interval is recommended as 12 years

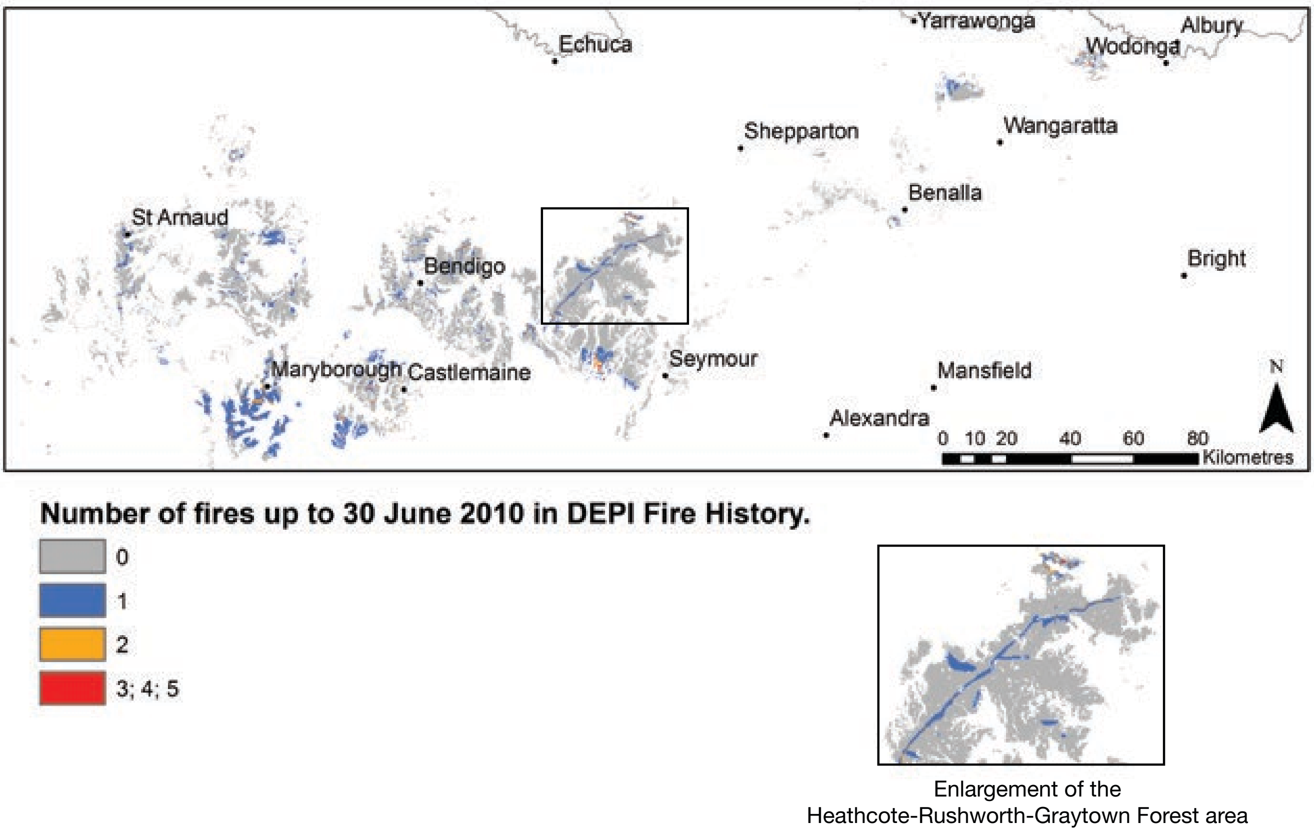

Figure 2. Recorded fires (from ~1970 to 2010) across the entire Box-Ironbark Ecological Vegetation Class in Victoria, including an enlargement of the Heathcote-Rushworth-Graytown Forest area. Source: DELWP FireHat database.

for a low severity fire and 30 years for a high severity fire. The **maximum** tolerable fire interval is recommended as 150 years (Cheal 2010). Key Fire Response Species proposed for the box-ironbark ecosystem include Grey Grass-tree *Xanthorrhoea glauca* subsp. *angustifolia*, Daphne Heath *Brachyloma daphnoides,* Spreading Wattle *Acacia genistifolia*, Showy Parrot-pea *Dillwynia sericea*, Drooping Cassinia *Cassinia arcuata*, Gorse Bitter-pea *Daviesia ulicifolia*, and Twiggy Bush-pea *Pultenaea largiflorens* (Tolsma *et al*. 2007b, Cheal 2010).

Note that *tolerable fire intervals* do not take into account the needs of faunal species or their habitat requirements, and how these are affected by burning. Vegetation growth stages (seral stages) (Cheal 2010) are used to plan for the availability of faunal habitat attributes across space and time. The current approach to fire management by DELWP gives emphasis to creating landscape mosaics of differing post-fire, vegetation age-classes.

Landscape mosaics are relevant at two scales:

a) within a single large planned burn, it is proposed that burning in a patchy mosaic will achieve the goal of reducing fuel loads while also having less impact on ecological values (and potentially having benefits in the longer term). This is the scale of investigation in this study.

b) at a regional scale, the goal is to maintain a mosaic of differing 'growth stages', representing stands of vegetation in distinct stages of maturity following fire (Cheal 2010). Maintaining a diverse assemblage of growth stages across a landscape/region is thought to enhance the persistence and resilience of a diverse assemblage of plants and animals. Six growth stages, in relation to fire, have been documented for the Ironbark/Box EVD (Cheal 2010):
   - Renewal – 0–1 year after fire
   - Juvenility – 1–3 years after fire
   - Adolescence – 3–13 years after fire
   - Vigorous maturity – 13–25 years after fire
   - Stasis – 25–140 years after fire
   - Established – 140+ years after fire.

This study

The overall aim of this project was to use an experimental approach to assess the ecological outcomes of planned 'landscape mosaic burns' in a large, continuous box-ironbark forest (Heathcote-Rushworth-Graytown Forest). A distinctive feature of this study is replication of study units at the *landscape-scale*; that is, the study is based on replicate *landscapes* that receive different burn treatments, such that different 'fire mosaics' can be compared. The study design aimed to compare two main aspects of a fire mosaic: a) the extent of burnt vs unburnt vegetation in a landscape; and b) the season of burn (autumn, spring).

Objectives

Key objectives of this study were:

a) to implement planned burns that differ in the extent of burn across a landscape and season of burn;

b) to monitor forest structure, selected habitat components, plant species composition, and selected faunal groups before and after the experimental burns;

c) to evaluate the effects of different burn mosaics (i.e., extent of burn) on these attributes; and

d) to investigate the effect of mosaic burns on selected animal species, including effects on individuals and their behaviour and use of resources.

This report summarises the study design, monitoring approach, the experimental planned burns and their cover, and the effects of the planned burns on habitat structural components, plant species composition, eucalypt flowering, woodland birds, small mammals and the ecology of selected species within the first two years post-fire. Our focus here is to present a relatively concise overview of the methods and key results.

Study design and methods

Study design

The study design incorporated two main aspects of planned burning (Bennett *et al.* 2012):

a) *Extent and pattern of burn (and hence the type of fire mosaic created).*
   Two treatments were selected: a patchy mosaic burn in which ~30–50% of the landscape is burned, and an extensive burn of ~70–90%. The latter is typical of past fuel reduction burns in the region, whereas a patchy mosaic burn may be more suitable as an ecological burn.

b) *The season of the burn.*
   Two seasons were included, autumn and spring. Currently most planned burning is carried out in autumn.

Twenty-two study landscapes were selected in the Rushworth-Heathcote-Graytown forest block (Fig. 3). These encompassed both state forest and national park. Each landscape was ~100 ha, bounded by roads or tracks, and dominated by vegetation typical of the box-ironbark Ecological Vegetation Class (Muir *et al*. 1995). The study landscapes were separated by >0.5 km (typically much greater) and were >200 m from the nearest forest–farmland boundary (Fig. 3). Areas recently logged or burned were avoided. Forests mapped as being ecologically important due to a higher density of large old trees also were avoided.

The fire history of these landscapes is not known, other than that none of them have burned for more than 30 years and probably much longer, based on fire history mapping for the Bendigo Fire Region (Fig. 2).

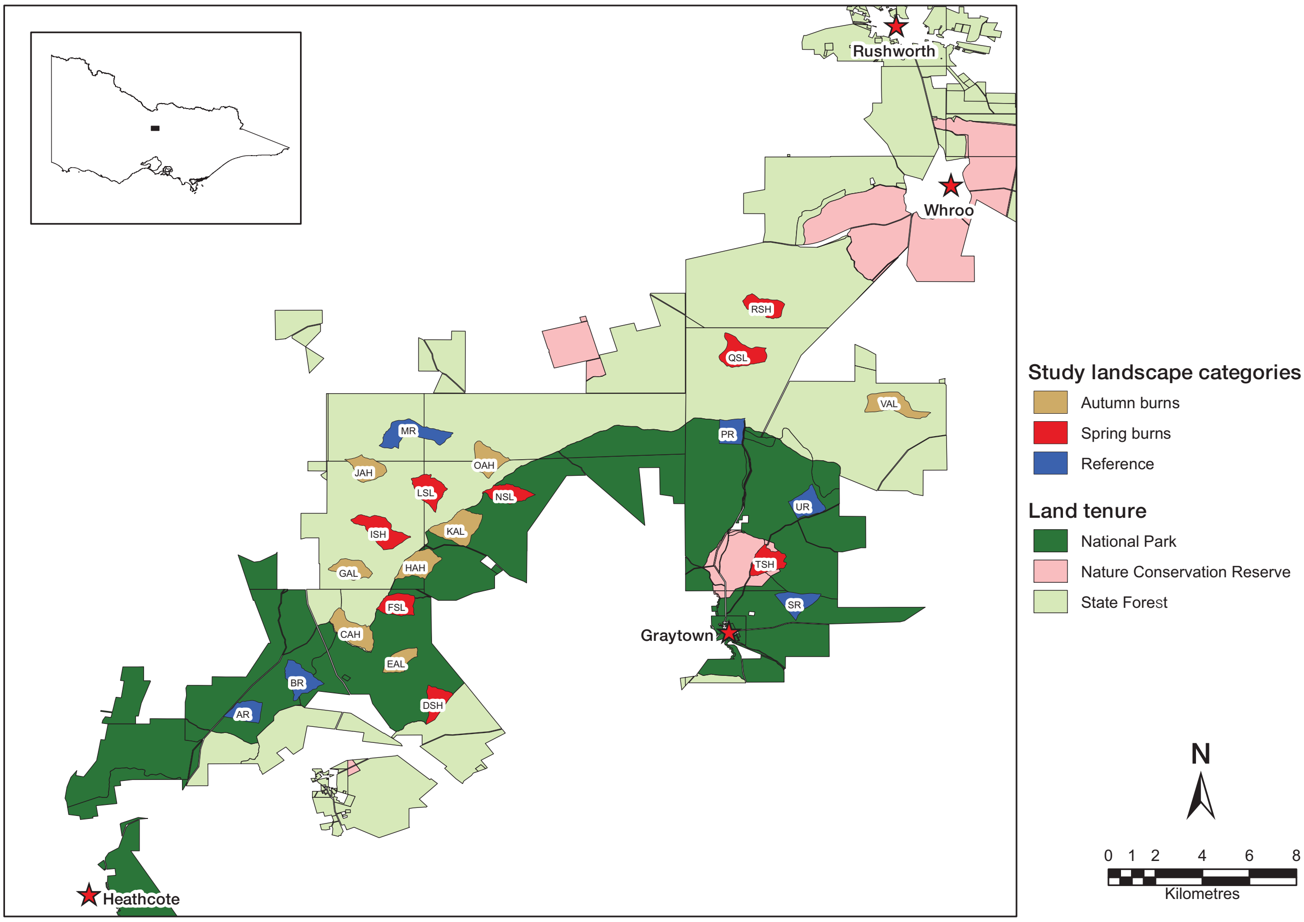

Figure 3. The Heathcote-Rushworth-Graytown forest block, showing the location of 22 study landscapes and the treatment (reference, autumn burn, spring burn) assigned to each.

Landscapes will be incorporated into DELWP's Monitoring Sites Database and available via Biodiversity Interactive Map. Plot labels: A–V signifies geographic position from Heathcote to Rushworth, R = reference landscape, A = autumn burn landscape, S = spring burn landscape, L = designated low cover burn (30–50%), and H = designated high cover burn (70–90%). Thus, landscape BR is close to Heathcote and is a reference landscape, while RSH is closer to Rushworth and was designated to receive a high cover spring burn.

The study design (Fig. 4) was based on eight of the 22 landscapes being burned in autumn and eight in spring, with six left as unburned 'reference' areas. In each season (autumn and spring), four of the eight landscapes were assigned to be low-cover burns (i.e., ~30–50% burn cover) and four as high-cover burns (i.e., ~70–90%). Treatments (season, burn cover, reference) were assigned randomly to sets of landscapes, with the exception of three reference landscapes which were assigned *a priori* due to a lack of suitably sized blocks bounded on all sides by roads/tracks.

Monitoring

Monitoring was designed to investigate how a range of ecological processes and attributes within the forest were affected by the experimental planned burns. Within each study landscape, 12 plots (each 20 x 20 m) were systematically located to assess and monitor vegetation and habitat structural components. Potential plot locations were initially marked on maps such that they sampled all quarters of a study landscape and were separated by at least 100 m. Sites were then located in the field (based on GPS locations) and a permanent site marker (steel picket in north-west corner) was established. In several instances where site locations fell on or immediately adjacent to internal tracks, they were displaced by up to 50 m. This approach of locating replicate plots within landscapes means that the effects of burning could be evaluated at both the site-level (i.e., individual plots), and at the landscape-scale by collating data across all plots in a study landscape.

Data were collected for the following categories of environmental attributes, by making quantitative surveys before and then at intervals after the experimental burns. Monitoring was also undertaken in reference (unburned) landscapes at the same time.

a) habitat structural components (trees, logs, stumps, litter, vegetation cover)

b) plant species composition

c) woodland birds

d) small mammals

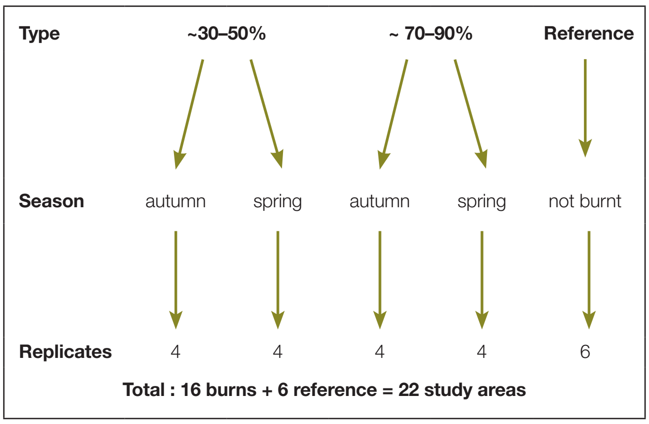

Figure 4. Study design for the Box-Ironbark Experimental Mosaic Burning Project showing the stratification of 22 study landscapes among treatments.

Trees, stumps and logs: all trees, stumps and logs in each 20 x 20 m plot were counted. The species of each tree was recorded and the size (cm diameter at breast height) of each stem was measured. A log was defined as fallen timber at ground-level >1 m in length and >3 cm in diameter, while a stump (Fig. 5) was defined as the cut (or broken) base of a eucalypt <1.5 m tall (anything >1.5 m tall was classified as a dead tree). Each log and stump was allocated to a size-class (≥3<10, ≥10<20, ≥20<40, ≥40<60, and ≥60 cm diameter). Additionally, stumps were recorded as being either solid or containing a hollow.

Substrate, litter depth and above ground vegetation structure: in each 20 x 20 m plot two 20-m transects were established. Measurements were made at 1-m intervals along these transects (n = 40 points in total) using a 2-m tall structure pole, held vertically. The substrate (material at ground-level directly under the pole: bare ground, rock, moss/lichen, fine litter (≤1 cm diameter), coarse litter (>1 cm diameter), log (>1 m long, >3 cm diameter), stump (<1.5 m tall), vegetation) was recorded. Where the substrate was litter, the pole was pushed through the litter to the earth underneath and a measure of litter depth obtained. Above-ground vegetation touching the pole was also recorded. The life-form (grass, herb, sedge-like plant, shrub, eucalypt, dead material) of such vegetation was noted, as was the height at which it touched the pole (in height-classes: <0.5, ≥0.5<1.0, and ≥1.0≤2.0 m).

Floristics: plant species surveys were conducted at a subset of landscapes/plots. Within 15 landscapes (six autumn burns, six spring burns, three Reference) plants were documented in eight 20 x 20 m plots. In a single plot, five 1 x 1 m quadrats (total 1 x 1 m quadrats across 15 landscapes = 600) were evenly spaced and all vascular plants in these quadrats were identified and recorded. Species present in the larger 20 x 20 m plot but not recorded in the 1 x 1 m quadrats were also noted.

Birds: bird surveys were conducted in all 22 landscapes at each monitoring plot (i.e., 264 survey plots in total). Surveys consisted of a five minute point count at each plot during which time all species seen or heard within a radius of 40 m and 80 m, respectively, were recorded. Species were also

Figure 5. Measuring a stump in a 20 x 20 m study plot (Greg Holland).

recorded incidentally within the landscape while moving between survey points. In each survey round, half of the plots in a landscape (n = 6) were surveyed in the morning (generally prior to 11 am) and the other half later in the day on a separate day. Thus, each survey round involved observations over two different days. On the subsequent survey round the timing of plots was switched between morning and later in the day. Three separate survey rounds were completed prior to the experimental burns (in October–March 2010/11) and further rounds post-burning.

Mammals: mammal surveys were initially conducted via live-trapping. Collapsible aluminium live-traps (Elliott traps) were set along 625-m transect lines (traps positioned every 25 m; 25 traps per transect line) in four Reference and all eight autumn burn treatment landscapes (two transects were established in each landscape; total traps per landscape = 50). Bait consisting of peanut butter, rolled oats, golden syrup and linseed oil was placed inside traps as an attractant. Due to low capture rates, surveys employing remote digital infrared cameras were also conducted (Fig. 6). A single camera (ScoutGuard DTC-530) was positioned at five of the 20 x 20 m plots per landscape, within each of the 22 landscapes. Cameras were strapped to the trunk of a tree such that they faced down towards the ground. At ground-level, a short length of PVC pipe containing the same bait as was used in live-traps was fixed in position to act as a lure. Camera traps were left in the field for a total of 21 nights.

Other project components

In addition to monitoring at specific study plots, other studies were undertaken. An outline of the relevance, methods and results for each of these is provided in the 'Results' section below.

a) Change in structural features with time since fire (broader study in Bendigo Region)

b) Effects of fire on the flowering patterns of Red Ironbark *Eucalyptus tricarpa*

c) Effects of fire on the Scarlet Robin *Petroica boodang*

d) Effects of fire on shelter and den use by the Yellow-footed Antechinus *Antechinus flavipes*

e) Effects of planned burns on the distribution of the Brush-tailed Phascogale *Phascogale tapoatafa*

f) Interaction between herbivore grazing/browsing and fire

Experimental burns

Experimental burns were conducted by staff from the Department of Environment, Land, Water and Planning, and Parks Victoria (Fig. 7). Burns were performed according to standard protocols for planned burning, while aiming to achieve specified burn coverage targets (i.e., 'low': 30 – 50%, and 'high': 70 – 90% of the landscape). Eight autumn burns were conducted between 26 February and 8 April 2011, while the eight spring burns were conducted between 18 October and 5 December 2011.

Prior to the burns, tracks around the perimeter of each study landscape were cleared as fuel breaks. Preparation also included a team of people raking (by hand) litter away from the base of larger trees, to reduce the risk of large old trees being burned. This was a standard practice in preparation for planned burns as of 2010–2011.

Reports on burn timing, procedures and weather conditions; and a mapped estimate of the proportion of each landscape that experienced different fire severities, were prepared by DELWP staff (Galvin and Medlyn 2011; Welsby and Medlyn 2011).

To quantify the burn cover and degree of patchiness in each landscape, a systematic assessment was made (see Appendix 1 for details). A series of transects was established in each study landscape, running from edge to edge, with the longest edges generally used for transect start/end points. Within a given landscape, transects were parallel and spaced 150–200 m apart (mean = 164 ± 3 m [1 SE]) (see Fig. 8).

Figure 6. Example of a record of Common Brushtail Possum *Trichosurus vulpecula* obtained from a remote camera.

Figure 7. Scene of operations for one of the spring burns, 2011 (Andrew Bennett).

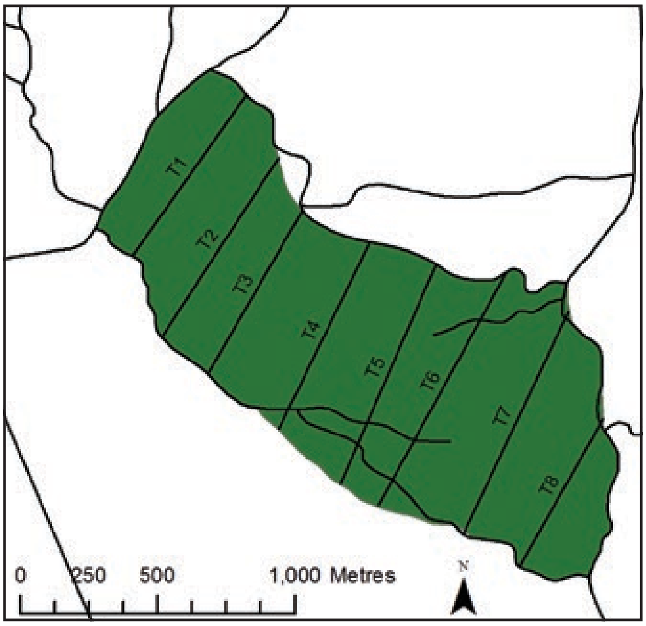

Figure 8. Landscape CAH showing eight transects used to assess burn cover and patchiness.

Transects were assessed on foot with the aid of a hand-held GPS unit, and all burn boundaries (i.e., boundaries between unburnt/burnt ground) were recorded. This process allowed the total length of each transect that was burnt to be determined, and to be expressed as a percentage of the transect. By averaging across all transects in a given landscape, a measure of the burn coverage for the landscape as a whole was calculated. This process focused only on the presence of unburnt and burnt patches at ground level: it did not consider measures of burn 'severity'. In addition to burn coverage, the number and width (i.e., size) of unburnt/burnt patches were determined.

Rainfall

Rainfall data were obtained from the Bureau of Meteorology. The long-term average annual rainfall for the weather station nearest to study landscapes (Heathcote) is 575 mm. The region experienced a prolonged period of below-average rainfall leading up to the start of this study (from 2001–2009; Fig. 9). However, drought-breaking rains well above the long-term annual average were received at the beginning of this project (2010 when pre-burn monitoring was conducted, and 2011 when experimental burns were carried out; Fig. 9). This high rainfall influenced various aspects of this project, including the ability to conduct planned burns and the post-fire response of various ecological attributes (see following sections).

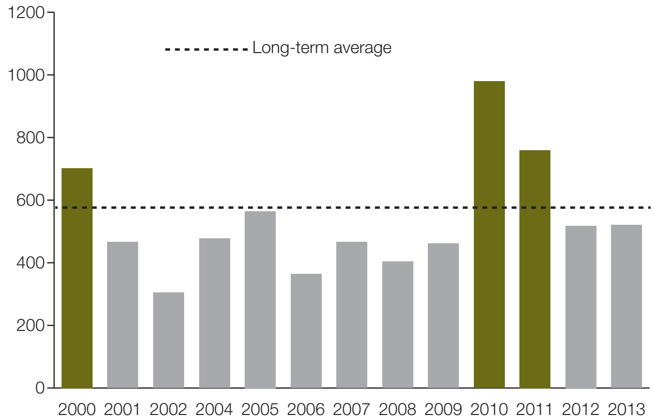

Figure 9. Rainfall data for the Heathcote weather station for the years 2000 – 2013 (note: complete data for the year 2003 was not available). The broken horizontal line indicates the long-term average annual rainfall. Grey bars indicate years in which below average rainfall was received, while green bars indicate years in which above average rainfall was received.

Results

Assessment of fuel hazard prior to burning

A standard fuel hazard assessment was carried out in each landscape by staff of the Department of Environment, Land, Water and Planning prior to the experimental burns. Each landscape was assessed at 15 points and an overall fuel hazard assigned for each point. The overall fuel hazard for all study landscapes was low. In total, 80.3% of all points assessed were assigned a 'low' hazard rating, 17.8% were 'moderate' and only 1.8% were 'high or very high' (Fig. 10). No points were 'extreme'. These data show that even after at least 30 years without fire, the level of fuel hazard in these forests is uniformly low.

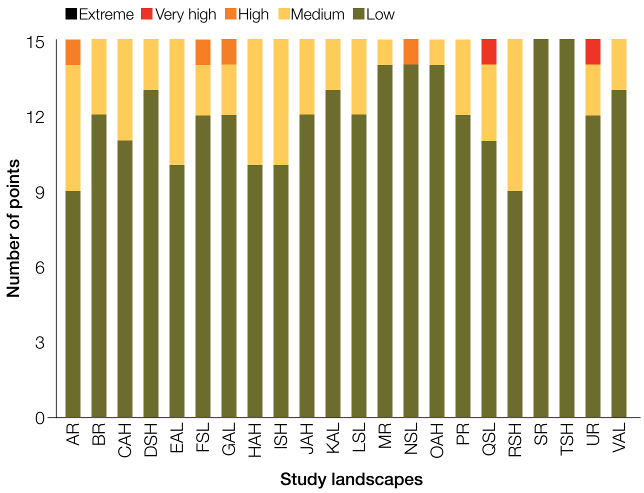

Figure 10. Outcomes of an assessment of the fuel hazard for the 22 study landscapes. Each landscape was assessed at 15 points and an overall fuel hazard assigned for each point.

Experimental burns and burn cover achieved

Sixteen of the 22 study landscapes were burned in 2011, the remaining six serving as unburned 'reference' landscapes. Estimates of burn coverage ranged from 22% to 89% across the 16 landscapes (Table 1). For the eight autumn burns, this range was 22% to 51% (Fig. 11); while for the spring burns it was 52% to 89%. Thus, the extent of burning was lower in all autumn burns than spring burns, irrespective of burn coverage targets. In the summer prior to autumn 2011 the study region experienced rainfall that was well above average (Fig. 9). This created difficult conditions for planned burning operations and restricted the extent of autumn burns (Galvin and Medlyn 2011).

These results highlight the difficulty associated with meeting specific burn targets, particularly when low burn coverage (e.g., 30 – 50%) is required. They also underscore the role that weather conditions play in determining burn outcomes. The complete separation of burn outcomes between seasons (i.e., all autumn burns <51%, all spring burns >51%) limits the ability to make comparisons of ecological outcomes between burn coverage and season.

Table 1. Summary of experimental burns: details for each of the burns conducted, including season of burn, burn coverage target, actual burn coverage achieved, and extent of different burn severity classes.

Study block	Season of burn	Burn coverage target (%)	Actual burn coverage (%)	Burn severity (%)[1]				
				Unburnt	Low	Medium	High	V. High
CAH	Autumn	70–90	39.1	10.7	58.6	30.2	0.5	0.0
EAL	Autumn	30–50	22.2	30.3	64.3	5.4	0.0	0.0
GAL	Autumn	30–50	24.5	24.5	65.3	9.6	0.6	0.0
HAH	Autumn	70–90	45.8	2.1	56.7	33.4	2.9	4.8
JAH	Autumn	70–90	40.5	3.1	62.3	33.5	1.3	0.0
KAL	Autumn	30–50	31.8	5.4	70.0	24.6	0.0	0.0
OAH	Autumn	70–90	50.8	0.0	36.6	62.0	1.5	0.0
VAL	Autumn	30–50	32.8	15.6	47.2	36.6	0.6	0.0
DSH	Spring	70–90	67.0	0.0	39.5	58.3	1.9	0.3
FSL	Spring	30–50	62.5	1.6	58.4	40.1	0.0	0.0
ISH	Spring	70–90	89.3	0.0	1.9	10.4	18.1	69.6
LSL	Spring	30–50	61.0	3.7	57.0	36.7	2.5	0.0
NSL	Spring	30–50	64.2	1.6	67.0	31.4	0.0	0.0
QSL	Spring	30–50	51.7	0.7	62.0	36.9	0.5	0.0
RSH	Spring	70–90	75.1	0.6	10.1	23.4	32.5	33.4
TSH	Spring	70–90	83.2	0.0	2.2	2.9	32.5	62.7

1 Burn severity estimates were obtained from Galvin and Medlyn 2011 (autumn burns) and Welsby and Medlyn 2011 (spring burns).

Figure 11. An area of patchy burnt and unburnt vegetation after an autumn burn (Andrew Bennett).

Comparison of environmental attributes before experimental burns

A comparison of environmental attributes collected *before* the experimental burns were performed is important for two reasons: a) as a baseline with which post-fire data can be compared; and b) to allow any pre-existing differences in attributes across treatment groups to be identified.

Overall, there were few differences between landscape treatment groups before experimental burns were conducted (see Appendix 2). Only two datasets (tree stem density and average tree stem size) showed evidence of pre-existing differences between treatment groups. This gives confidence that there is a sound basis for undertaking the experimental work and that any observed post-fire differences between treatment groups will not be the result of pre-existing differences.

Tree species

A total of 6,007 individual trees (and 8,955 stems) were encountered. Seven tree species were recorded: Red Ironbark *Eucalyptus tricarpa*, Grey Box *E. microcarpa*, Long-leaved Box *E. goniocalyx*, Red Box *E. polyanthemos*, Yellow Box *E. melliodora*, Red Stringybark *E. macrorhyncha* and Green Mallee *E. viridis*. Red Ironbark was the dominant tree species across all three landscape treatment groups, accounting for ≥30% of trees on average (Fig. 12). Dead standing trees accounted for 6–9% of the total trees recorded. An ordination (non-metric multidimensional scaling) of tree species composition between landscapes assigned to different treatment groups (reference, autumn burn, spring burn) showed a general overlap among treatments (Fig. 13); and a comparison by ANOSIM (analysis of similarity) showed no significant difference between treatments ($R = 0.08$, $P = 0.12$).

Typically, there was a high density (mean = 848 stems per ha) of small trees (mean = 14 cm diameter). There was a significant difference in mean stem density across the three landscape treatment groups: reference landscapes had higher density than either the autumn or spring treatment groups. There also was a marginally significant difference in mean stem size across treatment groups: reference landscapes had a lower mean stem size than both the autumn and spring treatment groups.

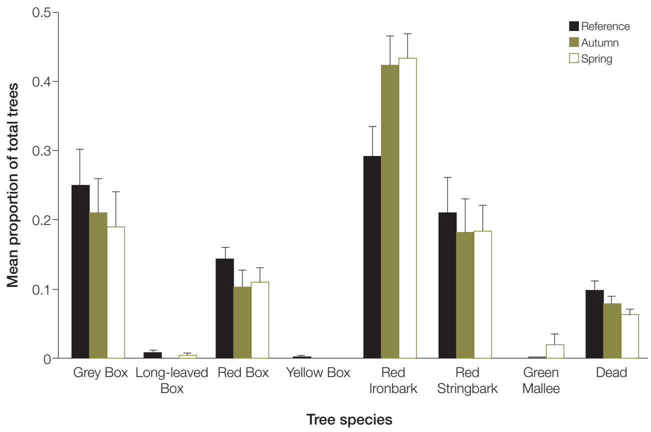

Figure 12. The mean proportion (± 1 s.e.) of total trees accounted for by individual tree species in landscapes allocated to each treatment group.

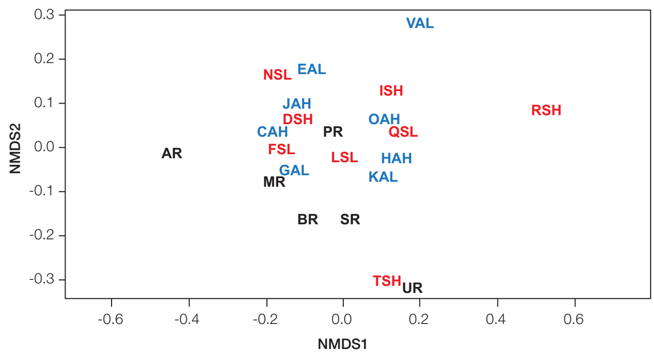

Figure 13. Ordination plot (NMDS) of the similarity of tree species composition in reference (black), autumn burn (blue), and spring burn (red) landscapes (Stress = 0.11).

Stumps

A total of 3,698 individual stumps were recorded from the 264 20 × 20 m plots surveyed. Most stumps were small (<20 cm diameter). Stumps in the 20–40 cm size-class accounted for between 22–25% of all stumps; large stumps (≥60 cm diameter) were rarely encountered, with only about 2% of all stumps being in this size-class in each landscape group. There was no significant difference in mean density of stumps or the proportion of stumps in different size-classes across treatment groups.

Logs

A total of 11,065 individual logs (>3 cm diameter) were recorded, with a mean density across all landscapes of 1,048 logs per ha (± 56 s.e.). In all treatment groups, 80–90% of the total number of logs was in the smallest size-class (3–10 cm diameter at mid point). Logs larger than 40 cm diameter were rare; and only eight logs ≥60 cm diameter were recorded. There was no significant difference between treatment groups in log density or in the composition of log size-classes.

Substrate

Fine litter (≤1 cm diameter) was the dominant substrate, accounting for >71% of records in all three landscape treatment groups. All other substrate types (bare ground, coarse litter, rock, moss/lichen, vegetation, log, stump) were far less common: bare ground was the only other substrate to account for >10% of records in any treatment group (autumn and spring burns). There was no difference in substrate composition between treatment groups.

Litter depth

Mean litter depth (both fine and coarse litter categories combined) averaged 2.1 cm (± 0.1 s.e.) across all landscapes. There was no evidence that mean litter depths varied across the treatment groups. When the two litter categories (fine and coarse) were considered separately, fine litter typically attained lower depth profiles (mean = 1.9 cm ± 0.1 s.e.) than coarse litter (mean = 4.3 cm ± 0.2 s.e.), but again there was no significant difference across treatment groups for either category.

Understorey vegetation structure

Six vegetation life-forms were recorded when assessing vegetation structure: grass, herb, sedge-like plant, shrub, eucalypt and dead material (any non-living vegetation). Dead material accounted for most structure records (30–35% of observations) across treatment groups, reflecting the sparse nature of understorey vegetation (Fig. 14). Herbs and shrubs each accounted for about 20% of records, while grasses, sedge-like plants and eucalypts were less common. There was no difference in life-form composition across treatment groups.

In each treatment group, more than 80% of records were from the lowest height category (<0.5 m). The two higher categories (>0.5≤1.0 m and >1.0≤2.0 m) accounted for less than 10% of records in all groups. There were no pre-existing differences between treatment groups relating to height categories of above-ground vegetation.

Figure 14. Box-ironbark forests typically have a sparse understorey vegetation structure. The start of the study coincided with the end of a decade of drought during which ground layer vegetation became increasingly depleted (Andrew Bennett).

Plant species composition

A total of 165 plant species was recorded, with 45 to 87 (mean = 67.9 ± 2.8 s.e.) species per landscape. There was no significant difference in the mean number of plant species detected across the three landscape treatment groups. Twenty species were recorded in all 15 landscapes surveyed, including Cranberry Heath *Astroloma humifusum*, Drooping Cassinia *Cassinia arcuata*, Red Ironbark *E. tricarpa*, and Wattle Mat Rush *Lomandra filiformis*. Almost a quarter of the total species detected was found only in a single landscape. There were no consistent differences in plant community composition between the three landscape treatment groups.

Birds

A total of 76 species of birds was recorded, with a mean of 35.1 (± 0.8 s.e.) species per landscape (range 29–41 species), during three survey rounds prior to burning. Thirteen species were recorded in all 22 landscapes, including the Grey Fantail *Rhipidura fuliginosa*, Scarlet Robin *Petroica boodang* and Rufous Whistler *Pachycephala rufiventris*. By contrast, 17 species were recorded in just one landscape, including the Jacky Winter *Microeca leucophaea*, Western Gerygone *Gerygone fusca* and White-bellied Cuckoo-shrike *Coracina papuensis*. There was no significant difference in the mean number of species detected per landscape, and likewise no difference in the species composition of the bird assemblages between treatments (Fig. 15).

Mammals

Live-trapping for mammals (total effort = 2,800 trap nights) resulted in just 13 individual small mammals being captured: 12 Yellow-footed Antechinus *Antechinus flavipes* and one Brush-tailed Phascogale *Phascogale tapoatafa*. Due to this low capture rate, surveys employing remote digital infrared cameras were also conducted. A total of five cameras was deployed in each of the 22 landscapes for 21 nights. This effort resulted in eight mammal species being recorded

(3 – 7 species per landscape, mean = 4.1 ± 0.2 s.e.). There was no significant difference in the mean number of mammal species detected across the three treatment groups. The Yellow-footed Antechinus was detected in 16 landscapes, followed by the Common Brushtail Possum *Trichosurus vulpecula* (15) and Brush-tailed Phascogale (14) (Fig. 16). Two introduced species were detected: the Red Fox *Vulpes vulpes* was found in 12 landscapes and Goats *Capra hircus* were found in three. The composition of mammal communities (based on presence/absence of species per landscape) did not differ across treatment groups.

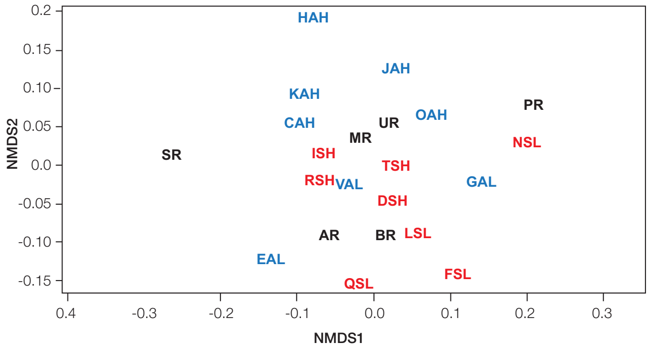

Figure 15. Nonmetric multidimensional scaling plot of the similarity of bird community composition in reference (black), autumn burn (blue), and spring burn (red) landscapes (stress = 0.23). The high degree of overlap of landscapes allocated to different treatments indicates similarity of bird communities.

Figure 16. Brush-tailed Phascogale *Phascogale tapoatafa* recorded on a remote digital camera.

Effects of planned burns on habitat structural components

Components of habitat structure such as logs, stumps, litter, dead standing trees and vertical understorey complexity are critical resources in Australia's dry eucalypt forests. First, they provide habitat (including shelter and den sites, foraging substrates) for a diverse range of organisms (Lindenmayer *et al.* 2006). Second, logs, stumps and litter help to reduce water flow and trap nutrients in an otherwise dry and nutrient poor forest system (Tongway and Ludwig 1997). Owing to past land-uses and disturbances, such habitat structural components are scarce in Victoria's Box-Ironbark forests (Environment Conservation Council 1997). Further, replacement rates are low due to the low productivity and slow growth rates of these forests.

We investigated the effects of experimental planned burns on logs, stumps, litter, dead standing trees and understorey complexity in the Heathcote-Rushworth-Graytown Forest. Logs, stumps and dead trees were counted and allocated to a size-class in 12 of 20 x 20 m plots in 22 study landscapes (6 Reference, 8 autumn burn, 8 spring burn). Litter was divided into two categories (fine: ≤1 cm diameter; coarse >1 cm diameter) and its cover and depth assessed at 40 points in each 20 x 20 m plot. Understorey vegetation complexity was assessed using a 2 m ranging pole at the same 40 points and was recorded in three categories: low (≤50 cm), medium (>50≤100 cm), and high (>1≤2 m). Surveys were conducted pre-fire and within three months post-fire. Post-fire changes in structural components were analysed in relation to the extent to which landscapes were burnt using generalised linear models.

Logs

Logs were grouped into three size-classes (small ≤10 cm, medium >10≤20 cm, large >20 cm diameter) for analysis. The proportional change in logs from pre- to post-fire surveys (i.e., change in logs from pre- to post-fire expressed as a proportion of pre-fire counts) was influenced by both the extent to which a landscape was burnt and log size. For all size-classes, increasing burn cover resulted in a corresponding increase in negative proportional change (i.e., loss of logs) (Fig. 17). However, logs in the medium and large size-classes were lost in greater proportions than those in the small size-class. In other words, for any given burn extent value, a greater proportion of medium and large logs were consumed than small logs (Fig. 18).

Figure 17. A burning log in one of the autumn burn landscapes (Greg Holland).

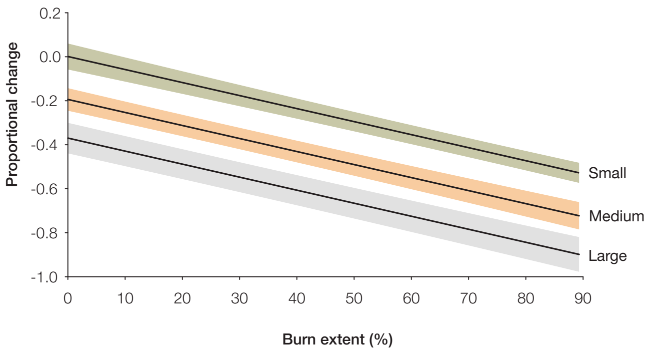

Figure 18. Predicted proportional change of logs in three size-classes (Small: ≤10 cm, Medium: >10≤20 cm, Large: >20 cm diameter) as a function of landscape-level burn extent. Shaded areas represent ± 1 std. error for predicted values.

Stumps

Stumps were grouped into two size-classes (≤20 cm, >20 cm diameter) for analysis, and only stumps containing hollows were considered (small, solid stumps typically associated with coppicing were excluded). The proportional change in stumps from pre- to post-fire was influenced by burn extent but not by the size of stumps (i.e., burn extent resulted in similar proportional change for both size-classes). Stumps were found in fewer numbers post-fire, with the proportional loss increasing as burn extent also increased (Figs. 19 and 20).

Dead trees

Dead trees were grouped into two size-classes (≤10 cm, >10 cm diameter at breast height) for analysis. The proportional change in dead trees from pre- to post-fire was not found to be related to burn extent for either size-class.

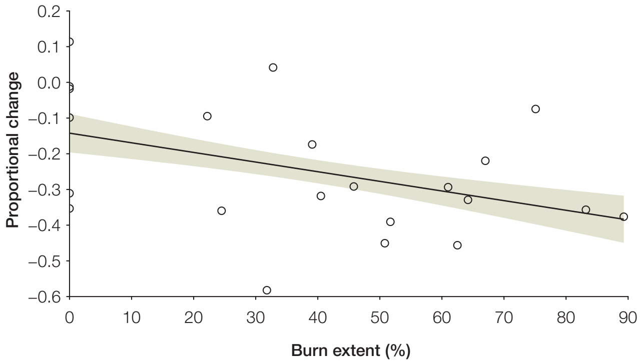

Figure 19. Predicted proportional change of stumps as a function of landscape-level burn extent. The shaded area represents ± 1 std. error for predicted values.

Figure 20. Two examples of stumps being burned during experimental planned burns (Andrew Bennett).

Litter cover

Litter cover (fine and coarse) was expressed as the number of times litter was recorded as a substrate per landscape (max. = 480). We found no evidence that fine litter (≤1 cm diameter) cover was influenced by the extent of experimental burns. Proportional change in coarse litter (>1 cm diameter) cover was influenced by burn extent in a non-linear fashion. Proportional change was quite variable across landscapes where burn extent remained under 50%. This resulted in predicted values showing little change. However, as burn extent increased above 50%, a decline in coarse litter cover was observed (Fig. 21).

Litter depth

Litter depth (fine and coarse) was measured wherever litter was recorded as a substrate. The depth of both fine and coarse litter was influenced by experimental burns (Fig. 22), with increasing burn extent resulting in greater proportional loss of depth. Figure 23 shows a plot of predicted values for fine litter (the corresponding plot of predicted values for coarse litter is very similar).

Understorey vegetation complexity

The interaction between the extent to which a landscape was burnt and the height class in which understorey vegetation complexity was measured was an important influence on the proportional change in complexity from pre- to post-fire surveys. This means that change in understorey complexity was related to burn extent, but the nature of the relationship differed for different height categories. Complexity in both the low (≤50 cm) and medium (>50≤100 cm) height categories declined at similar rates as burn extent increased (Fig. 24). However, complexity in the high (>1≤2 m) category did not show any response to burn extent. It is important to note that, despite declining as a function of burn extent, proportional change in the low and medium categories was often predicted to be positive (i.e., the model predicted an overall increase in complexity from pre- to post-fire surveys) (Fig. 24). This likely reflects the above-average rainfall in 2010/11 triggering growth of non-woody plants in between pre- and post-fire surveys.

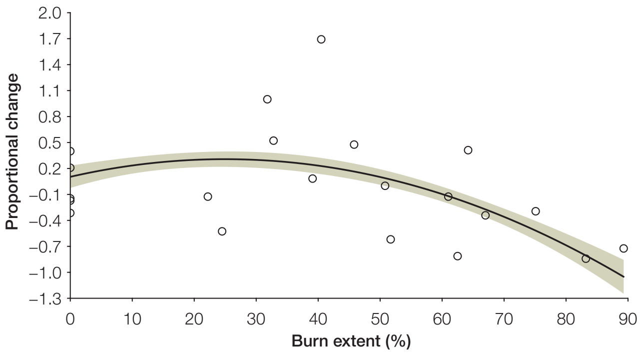

Figure 21. Predicted proportional change of coarse litter (>1cm diameter) cover as a function of landscape-level burn extent. The shaded area represents ± 1 std. error for predicted values.

Figure 22. Litter burning during an experimental planned burn (Greg Holland).

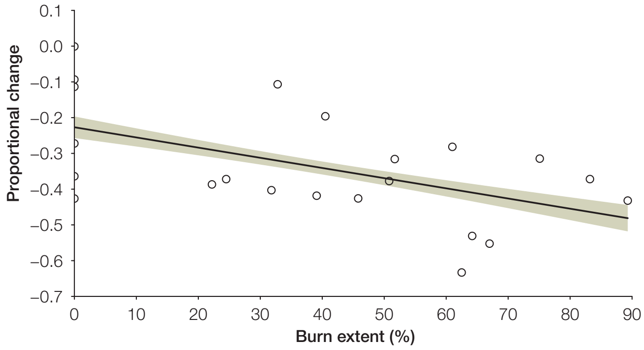

Figure 23. Predicted proportional change of fine litter (≤1 cm diameter) depth as a function of burn extent. The shaded area represents ± 1 std. error for predicted values.

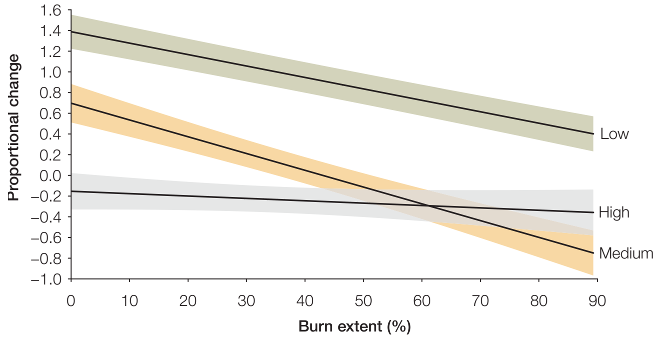

Figure 24. Predicted proportional change of understorey complexity in three height categories (Low: ≤50 cm, Medium: >50≤100 cm, High: >1≤2 m) as a function of landscape-level burn extent. Shaded areas represent ± 1 std. error for predicted values.

Longer term effects of planned burns on habitat structure

This project was undertaken in 2009 as a separate Honours study at Deakin University (Segrave 2009) to investigate longer term effects of planned burning on habitat structure in box-ironbark forests. It differs from the experimental burn study in that it used a chronosequence approach, in which planned burns of different ages were sampled post-fire to represent a gradient in time-since-fire. Study areas were in two broad locations: Bendigo and Rushworth-Heathcote-Graytown Forest (Fig. 25). A total of 22 planned burn areas were selected from fire history maps (DELWP).

Design

Two replicate sites ('burn sites') were placed within each burn area, to account for the possible patchiness of the planned burn. To reduce the potential influence of historical differences in land use (e.g. logging history) between burn areas, each burn site was paired with a 'reference site' located in directly-adjacent forest (usually within ~100 m) that had remained unburnt since at least 1983. Care was taken to ensure reference and burn sites were comparable in vegetation type, the composition and age structure of dominant tree species, and past forest thinning. A vegetation assessment was conducted at each burn and reference site along a randomly oriented 100 m transect line.

The fire-age of all burn sites was determined from fire history maps, and also for two reference sites that burnt in 1981 and 1983. These sites had been burnt in planned burns. Fire-age was unknown for all other reference sites. Therefore, time-since-fire estimates were allocated randomly within a time-frame based on the expert opinion of local fire managers. All Bendigo reference sites were considered likely to have burnt during the 1960s and 1970s; while for Heathcote-Rushworth-Graytown the most recent fire was likely to have occurred anywhere between 1940 and 1976 (when the local manager arrived in the region). Based on this information, fire-year was randomly assigned to reference sites from within these ranges: 1960–1979 for Bendigo sites; 1940–1975 for Heathcote-Rushworth-Graytown sites.

Results

Time-since-fire relationships were modelled for a suite of variables for the full 69-year chronosequence and also for the 28-year chronosequence for sites of known age. Figure 26 shows the predicted post-fire changes in habitat attributes that showed significant change. In general, stronger relationships were found with the shorter 28-year chronosequence, indicating that time-since-fire is a stronger influence over this period; but overall the amount of variation explained was low.

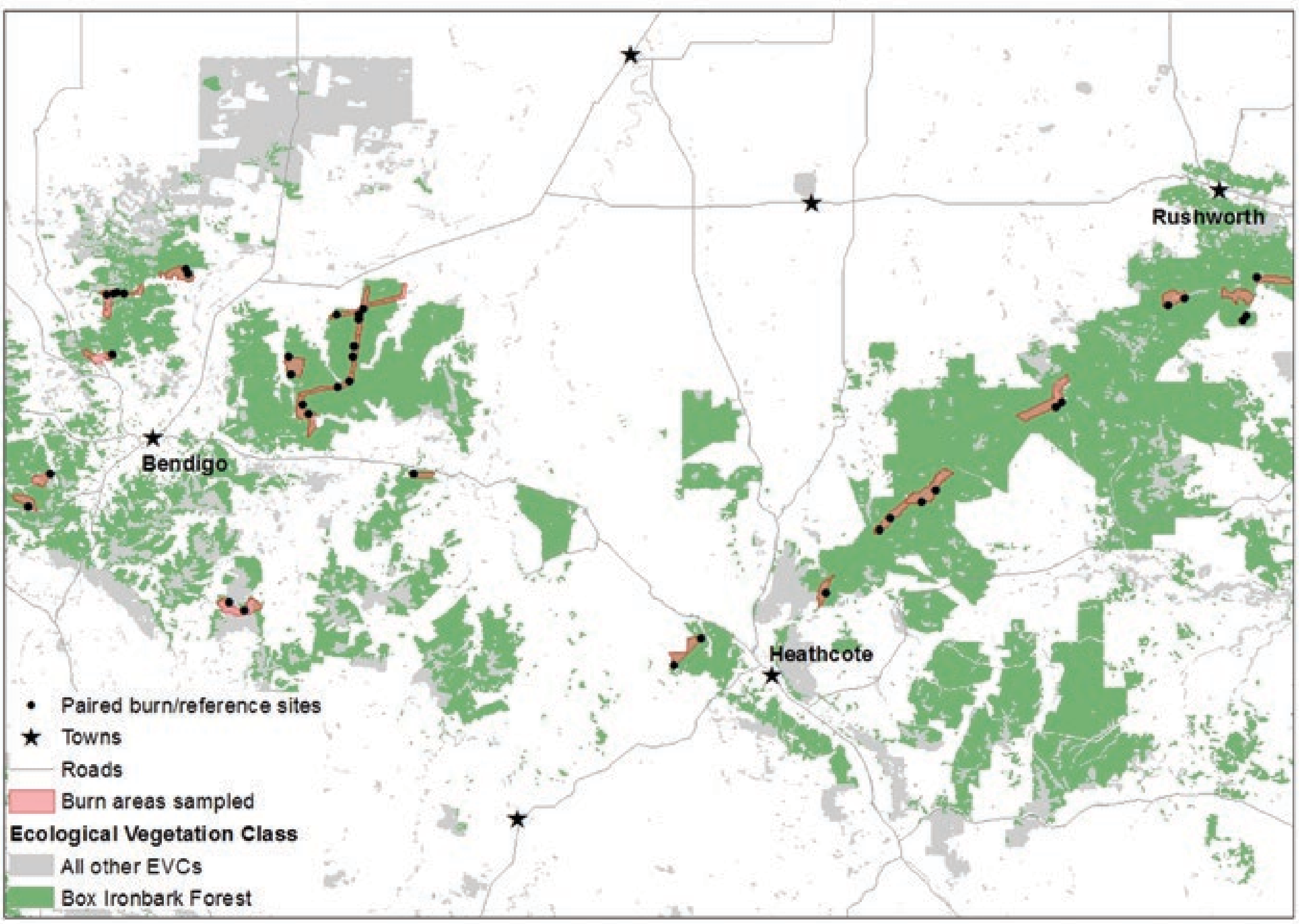

Figure 25. Location of paired burn and reference sites within Box-Ironbark Forest in the Bendigo and Heathcote-Rushworth-Graytown Forest regions.

The cover of bare ground was the only variable to decrease with a time-since-fire greater than 28 years. Other variables, including fine litter cover, low vegetation cover (<20 cm, 20–50 cm, <50 cm, <1 m), small dead trees and small logs all increased with time-since-fire (Fig. 26). For variables showing significant change over both chronosequences, the only variable to exhibit different responses between the time-frames of examination was small dead trees, which increased in number over the first 28 years post-fire but decreased when considering the full 69-year, post-fire period.

Several limitations need to be noted for this study. First, there is likely to have been substantial variation among these planned burns in terms of their intensity and burn cover. Second, it is not possible to account for other activities that may have occurred in these areas after fire (e.g., firewood collection). Third, the changes in vegetation structure post-fire have also been affected by at least a decade of drought (2000–2009) which also has substantially affected vegetation growth and structure.

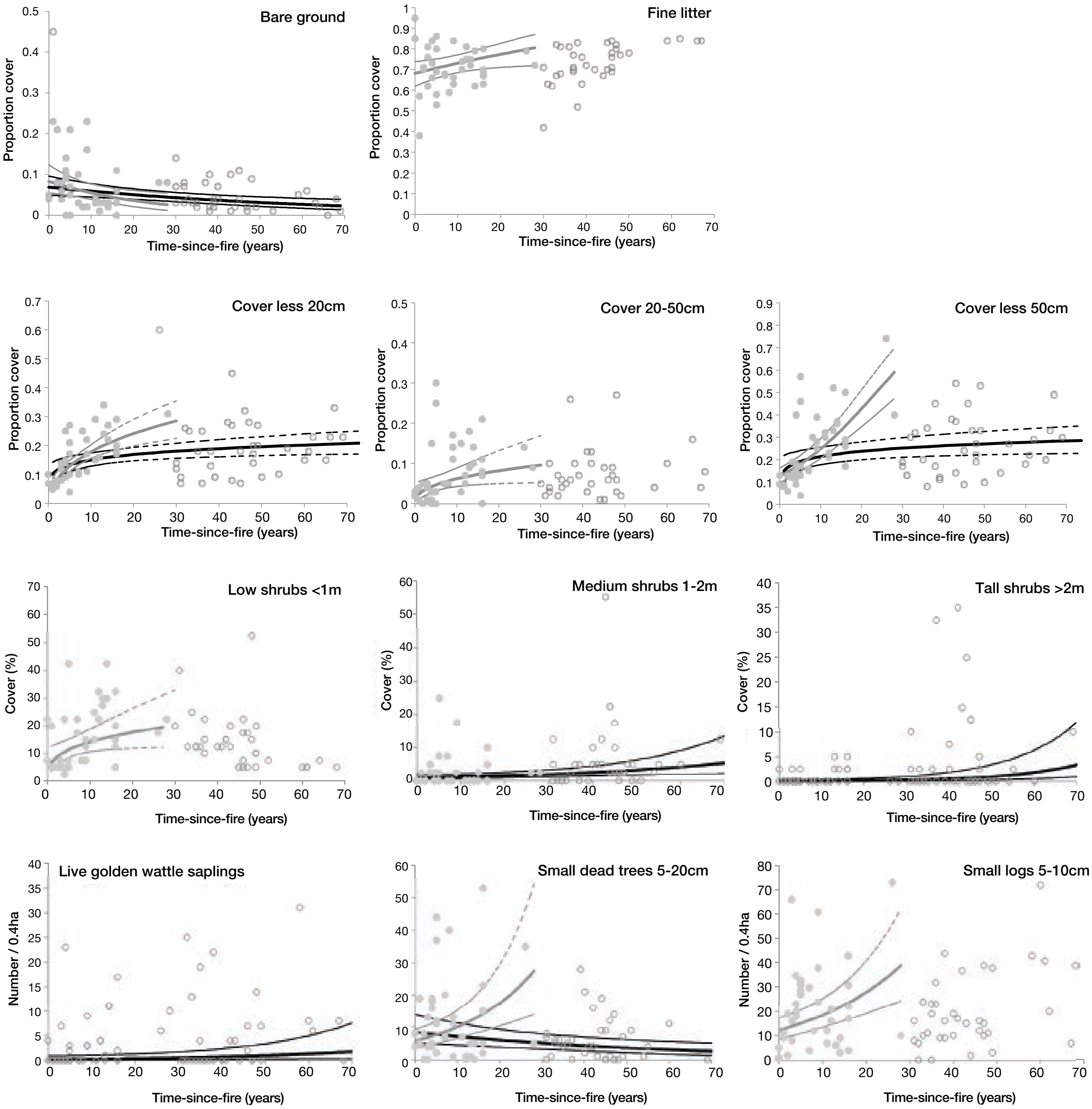

Figure 26 Predicted relationships between time-since-fire and habitat attributes: black lines show relationships across the 69-year chronosequence (from averaged models); relationships across the 28-year chronosequence are shown in grey. Only significant relationships are plotted. Dashed lines show the 95% confidence intervals of predictions; circles show the raw data (solid = known fire-ages; hollow = randomly assigned fire-ages based on expert opinion). The time-since-fire variable included in the best (i.e., lowest AIC_c) univariate model for each response variable has been used to plot raw data.

Effects of planned burns on plant species richness, composition, and selected plant species

The response of vegetation to fire is a critical process that drives post-fire forest dynamics. Recovery of vegetation following fire determines resource availability for fauna (Fox 1982) and plays an important role in stabilising soils and retaining nutrients. Individual plant species respond in differing ways to burning according to their inherent life history attributes. For example, some species can regenerate from existing root stock while others are obligate seed regenerators. Some can attain reproductive maturity quickly post-fire while others mature more slowly. Such attributes are used to identify 'Key Fire Response Species' — species that are sensitive to fire and for which the use of fire must be carefully managed to ensure their ongoing persistence (Cheal 2010) (Fig. 27). While the likely response to fire of some species is well known, many remain poorly studied. There is also a need to better understand the response of vegetation communities to burns of differing mosaic patterns.

Figure 27. Daphne Heath *Brachyloma daphnoides*, considered a 'Key Fire Response Species' within the Ironbark/Box Ecological Vegetation Division (Andrew Bennett).

Design

Floristic surveys were conducted at eight survey plots in 15 study landscapes (3 Reference, 6 autumn burn, 6 spring burn; Fig. 28). At each survey plot, all vascular plant species occurring in five evenly-spaced 1 x 1 m quadrats were recorded. Surveys of all landscapes were conducted in spring (September – December) pre-fire (2010) and on two occasions post-fire (2012 and 2013). The response of plants to burning was investigated using a combination of generalised linear models (landscape-level analyses) and generalised linear mixed models (plot-level analyses). Response variables consisted of the following:

a) species richness (landscape-level): number of unique perennial species detected across all eight plots in a landscape

b) species richness (plot-level): number of unique perennial species detected at a given survey plot (across all five 1 x 1 m quadrats)

c) individual species (landscape-level): the number of plots in which a species was detected per landscape (expressed as a proportion of total plots)

d) individual species (plot-level): presence or absence of a species at a given survey plot (across all five 1 x 1 m quadrats)

e) weed species richness (plot-level): number of unique introduced species (all vascular plants) detected at a given survey plot (across all five 1 x 1 m quadrats)

Seven species were chosen for analysis of the occurrence of individual species: Gold-dust Wattle *Acacia acinacea*, Spreading Wattle *Acacia genistifolia* , Cranberry Heath *Astroloma humifusum*, Daphne Heath *Brachyloma daphnoides*, Drooping Cassinia *Cassinia arcuata,* Gorse Bitter-pea *Daviesia ulicifolia* and Twiggy Bush-pea *Pultenaea largiflorens*. With the exception of Gold-dust Wattle, all these species are deemed to be of some utility as Key Fire Response Species in Box-Ironbark forests (Tolsma *et al.* 2007b). For landscape-level analyses, survey year (categorical) and the extent to which each landscape was burnt (%) were included as explanatory variables. At the plot-level, explanatory variables included survey year (categorical), the treatment assigned to a landscape (Reference, autumn burn, spring burn), and the extent to which a plot was burnt.

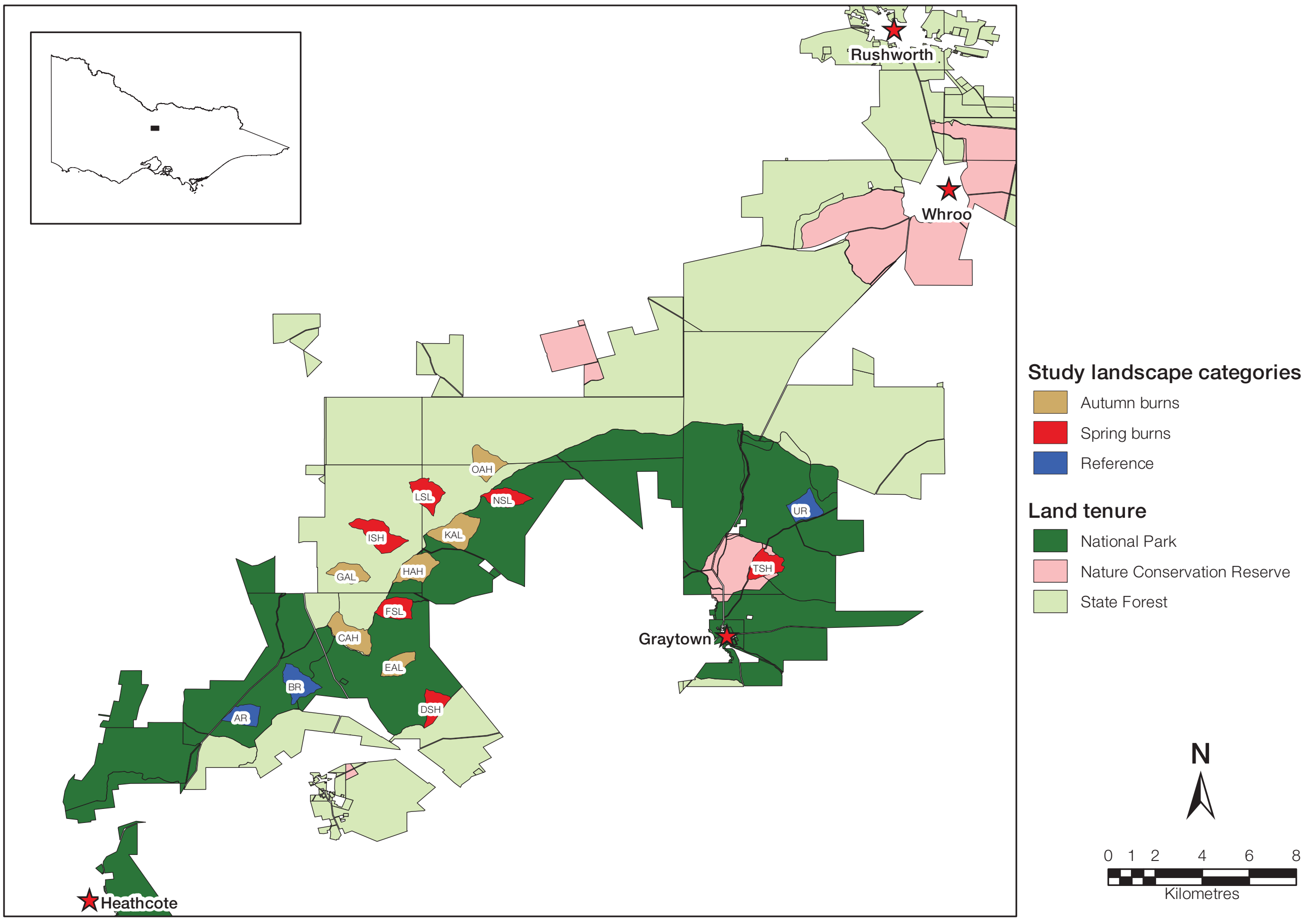

Figure 28. The Heathcote-Rushworth-Graytown forest block, showing the location of 15 study landscapes used for floristic surveys, including the treatment (reference, autumn burn, spring burn) assigned to each.

Plot labels: A-U signifies geographic position from Heathcote to Rushworth, R = reference landscape, A = autumn burn landscape, S = spring burn landscape, L = designated low cover burn (30-50%), and H = designated high cover burn (70-90%). Thus, landscape BR is close to Heathcote and is a reference landscape, while TSH is closer to Rushworth and was designated to receive a high cover spring burn.

Species richness

Species richness (landscape-level)

The year in which surveys were conducted was the predominant influence on species richness at the landscape-level. Species richness was found to differ in both of the post-burn surveys (2012 and 2013) compared to the pre-burn (2010) survey. Predictions from the model indicate that landscape-level species richness was higher in the two post-burn surveys (Fig. 29), irrespective of landscape treatment (reference, autumn burn, spring burn). The extent to which a landscape was burnt was of no predictive value. Note that there were above average rains in 2010 and 2011, after a decade of drought (2000–2009). The increase in species richness in 2012 and 2013, regardless of the fire treatment, is consistent with there being a marked increase in plant germination and growth in response to this rainfall.

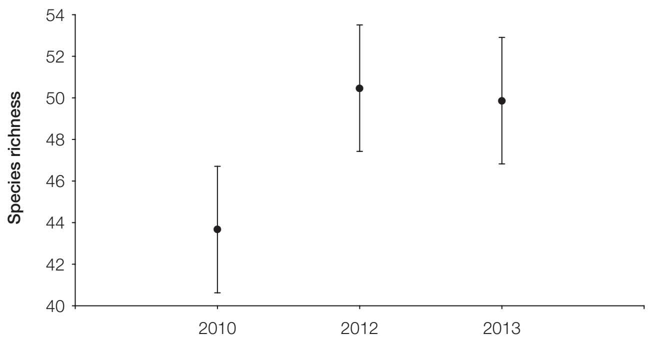

Figure 29. Predicted landscape-level species richness of perennial plants as a function of survey year. Error bars depict 95% confidence intervals around predicted values.

Species richness (plot-level)

At the plot-level, species richness was again predominantly influenced by the year in which surveys were conducted. As was found at the landscape-level, the model predicted species richness to be higher in the two post-burn surveys compared to the pre-burn survey. The extent to which a landscape was burnt was also important at the plot-level, although its influence was not strong (Fig. 30). There is evidence of a slight increase in species richness as burn extent increases up to about 40% of a plot, before species richness then declines slightly as burn extent continues to increase up to 100% (Fig. 30). However, predicted richness values vary by only 2–3 species along the burn extent gradient.

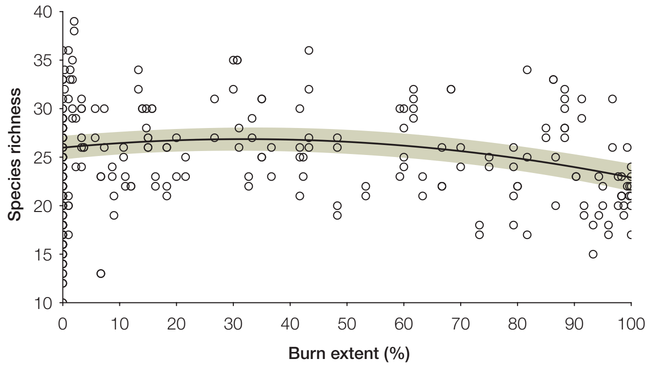

Figure 30. Predicted plot-level species richness as a function of burn extent. The shaded region represents the 95% confidence interval for predicted values.

Individual species

Individual species (landscape-level)

Of the seven individual species modelled, two (Gold-dust Wattle and Daphne Heath) showed no change in occurrence at the landscape-level associated with burn extent or year of survey. Year of survey was an important predictor of occurrence for four species (Spreading Wattle [Fig. 31], Drooping Cassinia, Gorse Bitter-pea and Twiggy Bush-pea). As was found for species richness, all four species were detected at a higher proportion of plots in the post-fire surveys compared to the pre-fire survey. Cranberry Heath was the only species that displayed a response to burn extent. The model predicts that the occurrence of Cranberry Heath is largely unchanged by a burn extent of up to ~45%. However, as burn extent increases beyond 45%, occurrence declines quickly (Fig. 32).

Figure 31. Spreading Wattle *Acacia genistifolia* was one of four species found in a higher proportion of plots per landscape in the two post-fire surveys. This likely reflects increased plant germination and growth in response to above-average rainfall received in late 2010/early 2011 (Greg Holland).

Individual species (plot level)

The occurrence of three species (Gold-dust Wattle, Drooping Cassinia and Twiggy Bush-pea) at the plot-level was not associated with any explanatory variables. Survey year was an important influence on Gorse Bitter-pea and Spreading Wattle, with probability of occurrence for both being higher in 2012 and 2013 compared to 2010. Burn extent (for the plot) was an important influence on both Spreading Wattle and Daphne Heath, with both species declining in occurrence as burn extent increased. However, confidence intervals around predicted values were wide for both species, indicating that the relationship with burn extent is not strong (Fig. 33). As was the case at the landscape-level, burn extent was also found to influence Cranberry Heath at the plot-level. Predictions from the plot-level model reveal a similar response by this species to burn extent at the two scales of investigation: probability of occurrence remains relatively stable up to a burn extent of 45%, before declining rapidly as burn extent continues to increase.

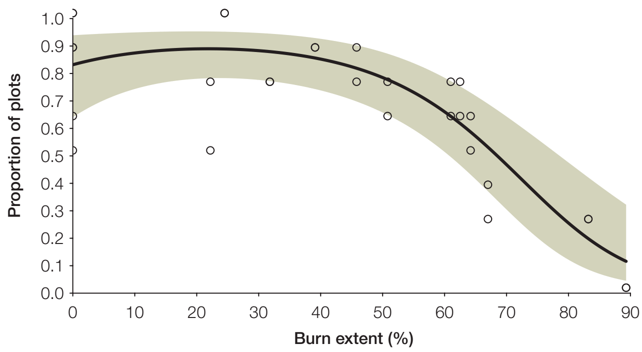

Figure 32. Predicted landscape-level occurrence (proportion of plots) of Cranberry Heath *Astroloma humifusum* as a function of burn extent (for survey year 2012; results were essentially identical for survey year 2013). The shaded region represents the 95% confidence interval for predicted values.

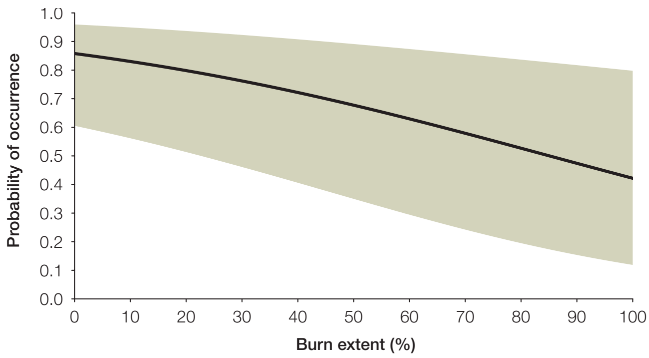

Figure 33. Predicted probability of occurrence of Daphne Heath *Brachyloma daphnoides* at the plot-level as a function of burn extent. The shaded region represents the 95% confidence interval for predicted values.

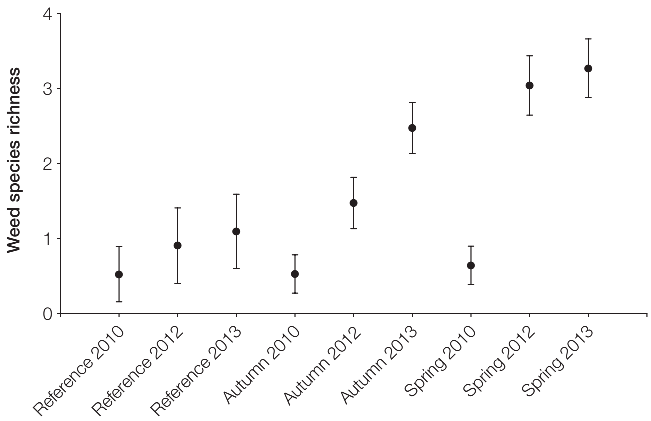

Figure 34. Predicted weed species richness as a function of landscape treatment group and survey year. Error bars depict ± 1 std. error for predicted values.

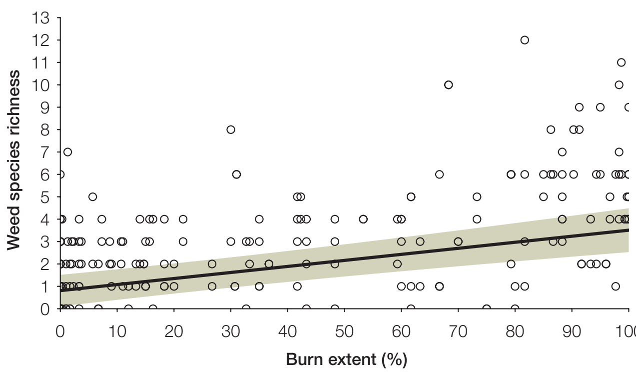

Figure 35. Predicted plot-level weed species richness as a function of burn extent (for the plot). The shaded region represents the 95% confidence interval for predicted values.

Weed species richness

Weed species richness (plot-level)

The most frequently occurring weed species were grasses and daisies: Hair Grass *Aira* sp., Flatweed *Hypochaeris radicata*, Smooth Cat's-ear *Hypochaeris glabra*, Fescue *Vulpia* sp. and Common Sow-thistle *Sonchus oleraceus*. With the exception of Flatweed, these species are all annuals. The interaction between survey year and the treatment assigned to a landscape was an important predictor of the number of weed species recorded. This indicates that there is a difference between treatment groups, but that this difference depends on the year of survey. All treatment groups had similar numbers of weed species pre-fire in 2010 (Fig. 34). The number of weed species recorded was higher post-fire in 2012 and 2013 compared to 2010 in all three landscape treatment groups (Fig. 34). Variation across years in autumn burn landscapes was not found to differ from that in reference landscapes. However, the increase in weed species in 2012 and 2013 in spring burn landscapes was found to be greater than that in reference landscapes (Fig. 34). Burn extent was also an important predictor of weed species richness, with the number of weed species increasing as burn extent also increased (Fig. 35).

General observations

A total of 43 species was recorded only on burnt plots in post-fire surveys (i.e., they were not recorded pre-fire or on unburnt plots post-fire). However, 72% (31) of these species had ≤5 records. This scarcity precludes formal analyses and makes it difficult to draw conclusions regarding associations with burning. No species were completely eliminated from plots following fire.

Rare and threatened species

Three species were identified that are currently listed within Victoria under the *Flora and Fauna Guarantee Act 1988*. Small-flower Mat-rush *Lomandra micrantha* subsp. *tuberculata*, Scented Bush-pea *Pultenaea graveolens* and Grey Grass Tree *Xanthorrhoea glauca* subsp. *angustifolia* are listed as 'rare', 'vulnerable' and 'endangered', respectively, in the Victorian advisory list (Department of Environment and Primary Industries 2014).

Plant species composition

Analyses were performed at both the landscape- and plot-level, to test for differences in plant species composition in relation to extent of burn. These analyses were performed: a) with all plant species surveyed; and b) with only perennial species (also excluding grasses and orchids), for the years 2012 and 2013. Each year was analysed separately to control for year-to-year environmental variation (e.g., rainfall). Results for 2013 have been presented: results for 2012 were very similar for all analyses. Analyses included analysis of similarity (ANOSIM) and non-metric multi-dimensional scaling (MDS) ordination.

Plant species composition (landscape-level)

At the landscape-level, there was a significant effect of burn extent (i.e., unburnt, low <50%, high >50%) on species composition in 2012 and 2013, when considering all plant species (Table 2). The greatest difference was between unburnt reference landscapes and landscapes with high burn extent (Fig. 36). When the analyses were done with perennial species only, there was no significant difference in species composition in relation to extent of burn at the landscape-level in either year, 2012 or 2013. These analyses were based on presence/absence data for plant species.

We were not able to investigate the effect of season of burn on plant species composition because all landscapes with high burn extent were burnt in spring, and all landscapes with low burn extent were burnt in autumn.

A SIMPER (similarity percentages) analysis was performed to determine which species were contributing most to differences between unburnt reference landscapes and those with high burn extent. The species contributing most to the dissimilarity were largely annuals and found only, or much more frequently, on landscapes following fire including Cudweed *Helichrysum luteoalbum*, Cotton Fireweed *Senecio ? quadridentatus,* and the introduced Fleabane *Conyza ? bonariensis* and Sowthistle *Sonchus oleraceus.* Of the 20 species accounting for 21.5% of the dissimilarity, only Dusky Caladenia *Caladenia fuscata* was identified as occurring more frequently on unburnt reference than highly burnt landscapes.

Table 2. Analysis of Similarity (ANOSIM) of plant species composition at the landscape-level showing pairwise comparison between burn categories (unburnt, low<50%, high>50%) for all species in 2013.

Pairwise comparison	R value	P value
Unburnt, low	0.303	0.071
Unburnt, high	0.44	0.017
Low, high	0.252	0.038

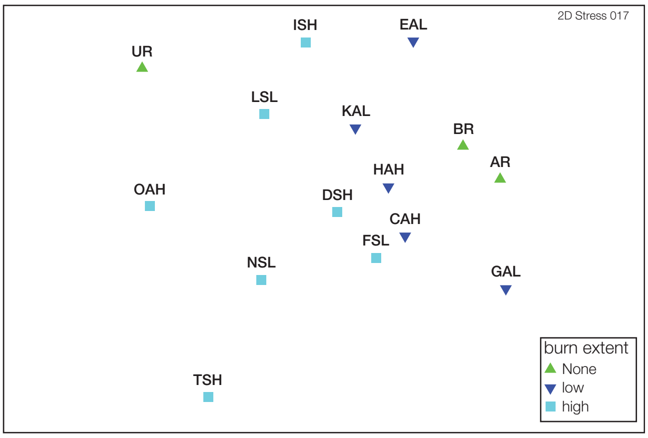

Figure 36. An MDS ordination plot showing the relationship among landscapes of different burn extent categories (unburnt, low <50%, high >50%) based on plant species composition (presence/absence data) in 2013. Landscapes with similar composition are closer together in the ordination space. Here there is a clustering of landscapes based on their burn extent.

Plant species composition (plot-level)

Analyses were also performed at the plot-level (20 x 20 m plots) to test for differences in species composition with burn extent, again with 'all species' and with 'only perennial species', for 2012 and 2013. Here, burn extent on the plot was grouped into four categories (i.e., unburnt, low <30%, medium 30–70%, and high >70%).

A significant difference in plant species composition was found between different categories of burn extent based on the data set with all plant species, for both 2012 and 2013. The strongest difference was between plots that were unburnt and those with high burn extent (Table 3). However, when the analysis was limited to perennial species only, burn extent was found to explain less of the variation in perennial species composition between plots (i.e., low R values) (Table 4).

A SIMPER (similarity percentages) analysis was performed to determine the perennial species contributing most to differences in species composition between plots with high burn extent and unburnt plots. Of the ten species contributing most to the dissimilarity, Cranberry Heath *Astroloma humifusum*, Gold-dust Wattle *Acacia acinacea*, Chocolate Lily *Arthropodium strictum*, Spreading Wattle *Acacia genistifolia*, and Trailing Speedwell *Veronica plebeia* occurred in higher frequency on unburnt plots. Small-flower Mat-rush *Lomandra micrantha* subsp *tuberculata*, Twiggy Bush-pea *Pultenaea largiflorens*, Daphne Heath *Brachyloma daphnoides,* Sticky Everlasting *Xerochrysum viscosum* and the introduced Flatweed **Hypochaeris radicata* were in higher frequency on burnt plots.

Table 3. Analysis of Similarity (ANOSIM) of species composition (all plant species, 2013) at the plot-level showing comparisons between burn categories (unburnt, low<30%, medium 30–70%, high>70%).

Pairwise comparison	R value	P value
unburnt, med	0.099	0.003
unburnt, low	0.088	0.002
unburnt, high	0.292	0.001
med, low	0.034	0.142
med, high	0.063	0.022
low, high	0.189	0.001

Table 4. Analysis of Similarity (ANOSIM) of species composition (perennial species only, 2013) at the plot-level showing comparisons between burn categories (unburnt, low<30%, medium 30–70%, high>70%).

Pairwise comparison	R value	P value
unburnt, med	-0.001	0.453
unburnt, low	0.056	0.028
unburnt, high	0.123	0.001
med, low	0.033	0.124
med, high	0.072	0.009
low, high	0.135	0.001

Plant species composition – year of survey

To test for differences in plant species composition in relation to the year of survey, analyses were performed using data from the reference (unburnt) landscapes only, for the years 2010, 2011, 2012 and 2013. As no reference landscapes were burnt, any differences between years can be attributed to variation in environmental conditions.

Analyses at the plot-level showed that there were differences between years for some, but not all, comparisons (Table 5). The strongest difference was between 2010 and 2011. The study region experienced above-average rains in 2010 and 2011, which stimulated plant growth across all study landscapes. These observations highlight the value of having 'reference' (control) landscapes in the study, and show that changes in species composition are not solely due to burning patterns.

Table 5. Analysis of Similarity (ANOSIM) of species composition (all species) at the plot-level showing comparisons between years.

Pairwise comparison	R value	P value
2010, 2011	0.295	0.001
2010, 2012	0.044	0.080
2010, 2013	0.151	0.001
2011, 2012	0.294	0.001
2011, 2013	0.238	0.001
2012, 2013	0.060	0.055

Effects of planned burning on flowering of Red Ironbark *Eucalyptus tricarpa*

Red Ironbark *Eucalyptus tricarpa* (Fig. 37) is a dominant tree species in the study landscapes and region. It flowers in winter (typically May – September) (Keatley *et al*. 2004) and provides an important nectar source for a wide range of faunal species, particularly birds such as honeyeaters, wattlebirds, friarbirds, lorikeets and the endangered Swift Parrot (Mac Nally and McGoldrick 1997).

Figure 37. Red Ironbark *Eucalyptus tricarpa* in flower (Tim Jansen).

Experimental design

We studied the effect of experimental planned burns on the flowering patterns of Red Ironbark trees in July–August 2013, in the second season after burns were conducted in 2011. Flowering was assessed for every Red Ironbark tree in each of the twelve 20 x 20 m plots in 16 study landscapes (6 reference, 5 autumn burn, 5 spring burn; Fig. 38). Flowering was recorded as present if there were fresh or dead flowers, or buds, on a tree. Variables representing flowering were then modelled (using generalised linear models and generalised linear mixed models) in relation to several predictors:

a) burn extent – the percentage of the plot or landscape that was burned

b) geographic location in the forest (northing)

c) basal area of trees (m^2) per study plot (or averaged across plots in the landscape)

d) tree size (diameter) – for analysis of flowering of individual trees

e) topography.

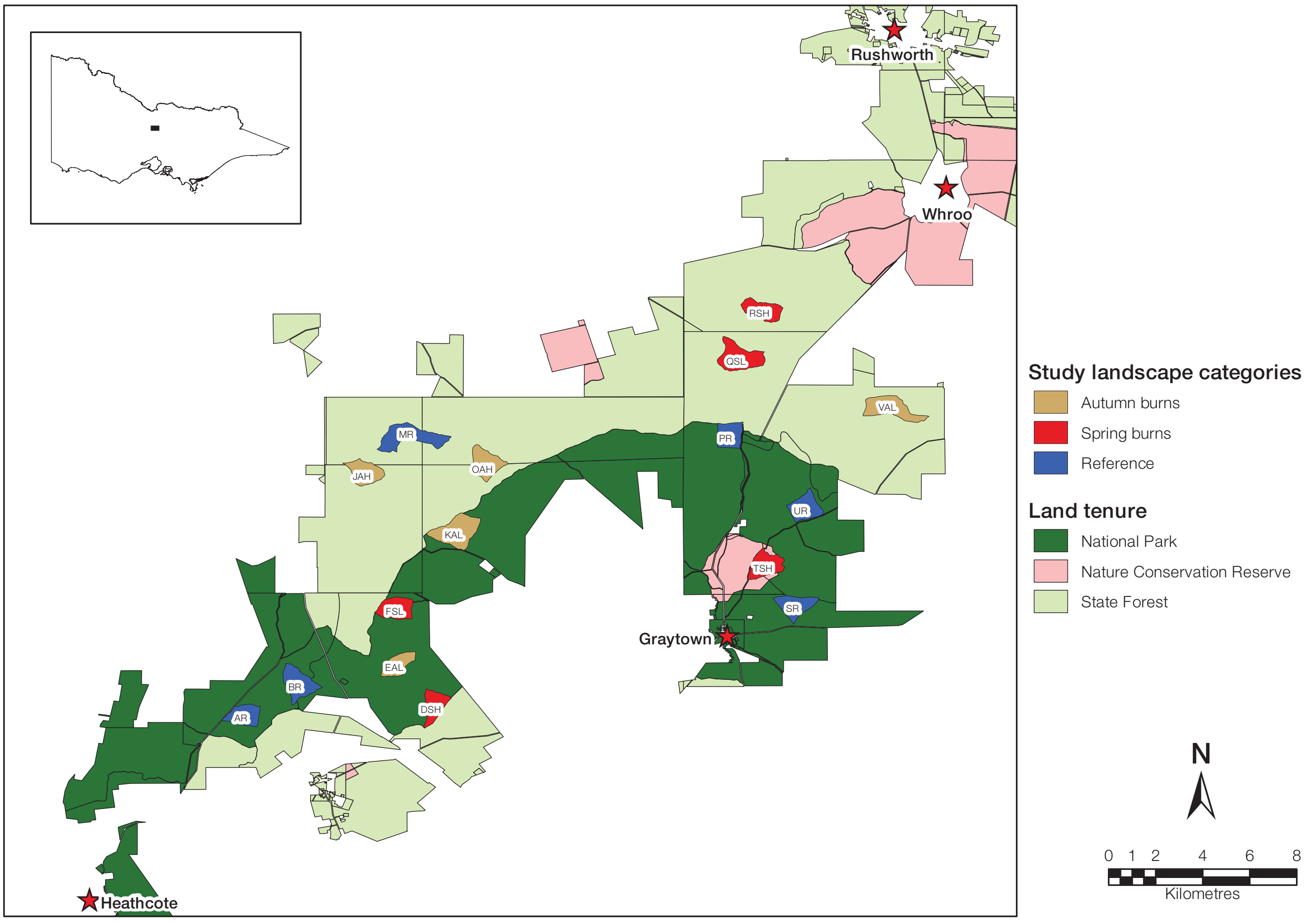

Figure 38. The Heathcote-Rushworth-Graytown forest block, showing the location of 16 study landscapes used for surveys of flowering Red Ironbark *Eucalyptus tricarpa*, including the treatment (reference, autumn burn, spring burn) assigned to each.

Plot labels: A–V signifies geographic position from Heathcote to Rushworth, R = reference landscape, A = autumn burn landscape, S = spring burn landscape, L = designated low cover burn (30–50%), and H = designated high cover burn (70–90%). Thus, landscape BR is close to Heathcote and is a reference landscape, while RSH is closer to Rushworth and was designated to receive a high cover spring burn.

There was a low frequency of flowering of Red Ironbark in the study area in 2013. Of 1,596 trees surveyed, flowering was recorded for 205 (12.8%). Flowering was recorded in all 16 landscapes but there was marked variation between them. In the landscape with greatest flowering, 44% of trees were flowering, whilst in the landscape with least flowering it was just 1%. At the plot-level, flowering was noted in 97 of 192 study plots (50.5%), with the proportion flowering in each plot ranging from >80% to <5% of trees.

Results

Landscape-scale

The *proportion of plots per landscape* (i.e., x/12) at which flowering was recorded was best predicted by northing, tree basal area and topography, with northing having the strongest effect. For the *overall proportion of trees flowering* (per landscape), important predictors were northing and basal area of trees. Flowering of Red Ironbarks was greatest in landscapes in the north of the Heathcote-Rushworth-Graytown Forest, and in those which had higher tree basal area (Fig. 39).

The burn extent within a landscape did not influence flowering.

Plot-scale

The presence or absence of flowering per 20 x 20 m plot was best predicted by tree basal area and northing; while the proportion of trees flowering (per plot) had only one important predictor, northing. The burn extent within a study plot did not influence flowering, nor did topography. Flowering was most likely for plots in the north of the forest study area, and with greater tree basal area.

Tree-scale

At the tree-scale, trees were more likely to flower if they were in the north of the study area, and flowering was more likely to occur as tree size increases (Fig. 40).

Overall

We found no evidence that flowering of Red Ironbarks in 2013 was influenced (positively or negatively) by the experimental burns in either autumn or spring 2011.

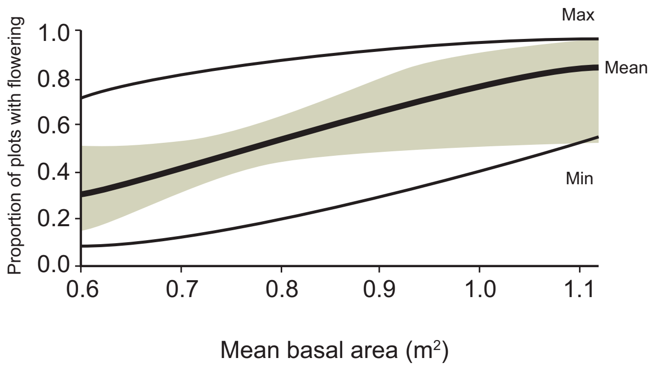

Figure 39. Predicted proportion of plots in a landscape with flowering as a function of basal area. The model is fitted for three values of northing (maximum, mean and minimum values for northing coordinates). The shaded area represents the 95% confidence interval for predicted values with northing at its mean value.

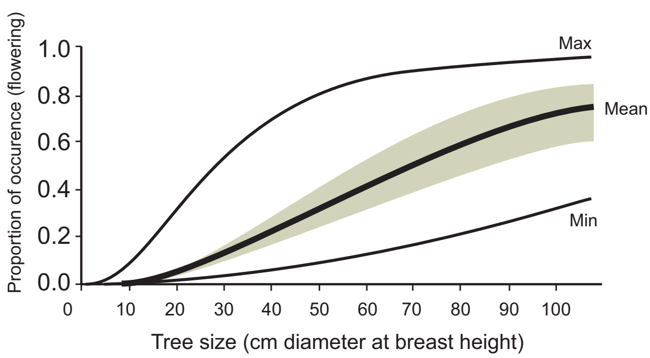

Figure 40. Predicted probability of an individual Red Ironbark flowering as a function of tree size. The model is fitted for three values of northing (maximum, mean and minimum). The shaded area represents the 95% confidence interval for predicted values with northing at its mean value.

Effects of planned burns on the bird community

Victoria's box-ironbark region provides important habitat for more than 180 species of terrestrial birds (Tzaros 2005). The location of the box-ironbark system results in a diverse and distinctive bird community, including species typical of dry woodlands, as well as species associated with the wetter forests of the Great Dividing Range and the drier plains country of northern Victoria. A unique feature of box-ironbark forests that enhances their habitat value for birds is the provision of eucalypt blossoms. Dominant tree species (e.g., Red Ironbark and Grey Box) flower during the cooler months, providing an important food source (nectar and invertebrates attracted to nectar) throughout autumn and winter. In good flowering years, the number of birds in box-ironbark forests increases dramatically during winter as seasonal migrants move into the region to exploit floral resources. This includes species of conservation concern such as the Swift Parrot *Lathamus discolor.* Box-ironbark forests also provide critical habitat for a wide range of 'woodland' species (i.e., species that are largely or wholly restricted to dry forests and woodlands). A suite of such species are recognised as being in decline (e.g., Crested Bellbird *Oreoica gutturalis*, Speckled Warbler *Pyrrholaemus sagittatus*, Hooded Robin *Melanodryas cucullata*).

Given the importance of box-ironbark forests as habitat for a diverse and unique assemblage of bird species, it is important to understand how burning of such forests influences the avian community. We addressed this in the Heathcote-Rushworth-Graytown forest by conducting bird surveys at each monitoring plot in all 22 study landscapes (i.e., 264 survey plots in total). All species seen or heard during a five minute point count within a radius of 40 m and 80 m, respectively, were recorded. Surveys were conducted in five distinct 'periods': (1) pre-burn (spring 2010 – summer 2011); (2) post-burn 1 (spring 2011); (3) post-burn 2 (winter 2012); (4) post-burn 3 (spring 2012); and (5) post-burn 4 (spring 2013). During the pre-burn period, each site was surveyed on three separate occasions, while each site was surveyed on two separate occasions during each of the four post-burn survey periods (resulting in each site being surveyed a total of 11 times across the five survey periods). The influence of burning on birds was investigated using generalised linear models, generalised linear mixed models, analysis of similarity (ANOSIM) and non-metric multi-dimensional scaling (MDS) ordination.

Bird species richness (landscape-level)

A total of 93 terrestrial bird species was recorded across the five survey periods. The mean number of species detected per landscape per survey period was 27.7 (± 0.4 s.e.) (range: 19 – 39 species). There was no evidence that burning influenced the number of species encountered in the study landscapes: there was no relationship between species richness at the landscape-level and the extent to which a landscape was burnt. However, bird species richness did vary through time. Fewer species were recorded in the two surveys conducted in 2012 (winter 2012 and spring 2012) than in the other three survey periods (Fig. 41). This was not related to burning since trends were observed across all landscape treatment types (i.e., including unburnt reference landscapes). Instead, differences in the number of species detected through time are likely to be related to broad environmental factors such as rainfall, eucalypt flowering patterns and food availability. The reduced number of species detected in winter 2012 may also reflect the time of year that these surveys were conducted (all other survey periods were conducted in spring/summer

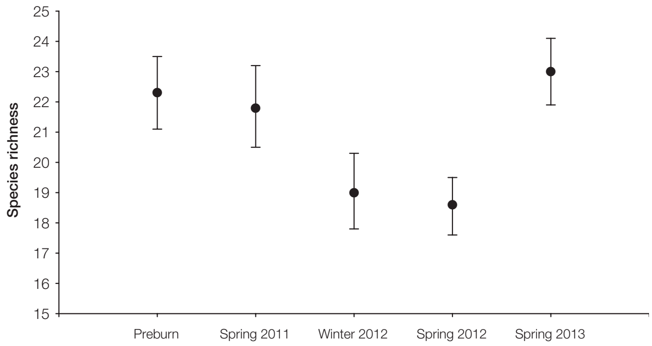

Figure 41. Predicted richness of bird species as a function of survey period (one 'pre-burn' survey period and surveys conducted at four separate 'post-burn' periods). Error bars represent 95% confidence intervals for predicted values.

Individual bird species (landscape-level)

Of the 93 terrestrial bird species recorded, sufficient data were obtained for 25 to allow analysis of the occurrence of individual species. Statistical models were constructed to relate the frequency of occurrence (number of survey plots per landscape where a species was encountered) of each species per landscape per survey round (n = 11 survey rounds) to: (a) survey period (pre-burn and four post-burn survey periods: spring 2011, winter 2012, spring 2012, and spring 2013); and (b) burn extent (%). The frequency of occurrence of nine of the 25 species (36%; Australian Magpie, Crimson Rosella, Yellow-tufted Honeyeater, Brown-headed Honeyeater, Red-capped Robin, Grey Currawong, Varied Sittella, Laughing Kookaburra, Yellow Thornbill) was not related to either survey period or burn extent. Survey period (but not burn extent) influenced the frequency of occurrence of ten species (40%), but the response of these species was different in each survey period (Fig. 42). The Rufous Whistler and Striated Thornbill were encountered more frequently in spring 2011 compared to the pre-burn period, while the opposite was true for the Red Wattlebird and Sulphur-crested Cockatoo. Winter 2012 had the greatest influence on the frequency of occurrence of species, with a total of six species being recorded at different rates in this period compared with pre-burn surveys (Fig. 42). The Buff-rumped Thornbill, Striated Thornbill and White-eared

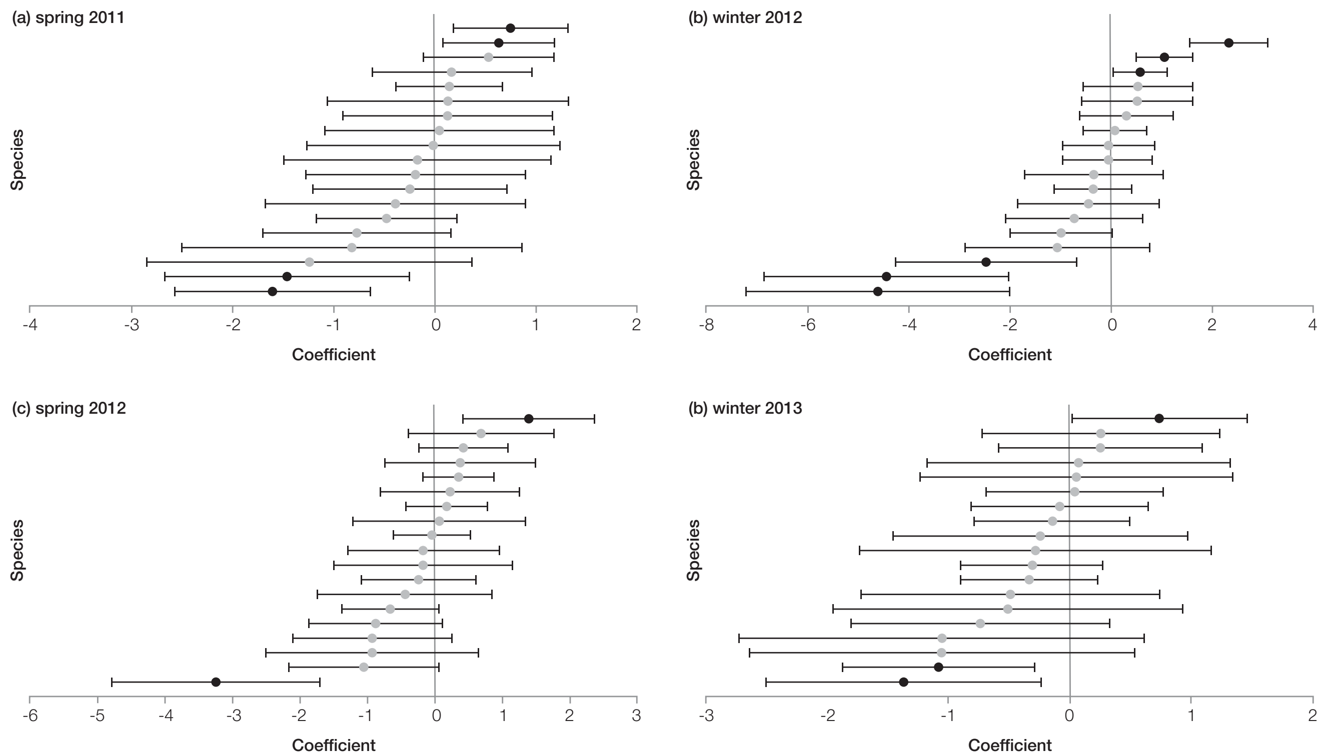

Figure 42. Results from statistical models for individual bird species investigating differences in frequency of occurrence across five survey periods (note: the pre-burn survey period was set as the baseline to which other survey periods were compared and therefore is not shown here). Circles represent linear model coefficients and horizontal bars indicate 95% confidence intervals for each coefficient.

Real differences between survey periods in (a) to (d) compared with the pre-burn survey period occur where 95% confidence intervals do not overlap zero (indicated by black circles). For example, in spring 2011, two species were encountered more frequently than in the pre-burn surveys (positive coefficients whose 95% confidence intervals do not overlap zero), while two species were encountered less frequently than in pre-burn surveys (negative coefficients whose 95% confidence intervals do not overlap zero).

Honeyeater were more commonly encountered in winter 2012, while the Rufous Whistler, Red Wattlebird and Striated Pardalote were encountered less frequently. Spring 2012 was important for just two species: the Brown Treecreeper was encountered more frequently, and the Red Wattlebird less frequently, in this period compared to pre-burn surveys. Finally, spring 2013 saw increased detections of the White-winged Chough and fewer detections of the Striated Pardalote and Grey Shrike-thrush. These relationships between the frequency of occurrence of bird species and survey period likely reflect changing environmental conditions through time (e.g., rainfall, food resources) and seasonal movements of some species.

The frequency of occurrence of six species (24%) was influenced by the extent to which a landscape was burnt. Occurrence of the Fuscous Honeyeater *Lichenostomus fuscus* was greater in landscapes burnt more extensively, during the spring 2011, winter 2012, and spring 2012 survey periods (Fig. 43). A different pattern was observed for the Grey Fantail *Rhipidura fuliginosa*, with frequency of occurrence for this species declining gradually with increasing burn extent in winter 2012 only (Fig. 44). The Scarlet Robin *Petroica boodang* was encountered more frequently with increasing burn extent, in both spring 2011 and winter 2012 (Fig. 45). A similar increase was observed for the Spotted Pardalote *Pardalotus punctatus*, but only in spring 2011 (Fig. 46). The Weebill *Smicrornis brevirostris* showed a relatively strong response to burn extent in winter 2012 only, being encountered more frequently in this survey period in landscapes burnt more extensively (Fig. 47). Finally, the White-throated Treecreeper *Cormobates leucophaeus* declined in occurrence in spring 2012 and spring 2013 with increasing landscape-level burn extent (Fig. 48).

These relationships between the occurrence of individual species and burn extent must be interpreted cautiously. First, the majority of species (76%) for which models were constructed displayed no relationship with burn extent. Second, with the exception of the Weebill and White-throated Treecreeper, changes in the proportion of plots in which species were encountered as a function of burn extent were not pronounced. Thus, although models suggest that burn extent is influential, its overall effect on frequency of occurrence is not strong for most species. Further evidence of this is provided by the variance explained by models: for all six species the variance explained ranged from 2.4–22.6 %, indicating that much variance in the frequency of occurrence of species remained unexplained. Third, the influence of burn extent varied across the four post-burn survey periods for all six species. There were no consistent patterns, suggesting that any influence of burn extent is dependent on year-to-year variations in prevailing environmental conditions and how these interact with the unique resource requirements of different species.

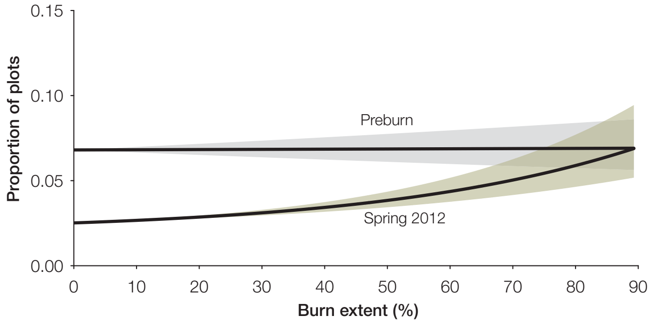

Figure 43. Predicted landscape-level occurrence (proportion of plots) of Fuscous Honeyeater *Lichenostomus fuscus* as a function of burn extent. Shaded regions represents the 95% confidence intervals for predicted values. No relationship with burn extent is expected in the pre-burn survey period since landscapes had not yet been burnt. Predictions are shown for spring 2012; results were similar for spring 2011 and winter 2012. There was no relationship between frequency of occurrence and burn extent in spring 2013.

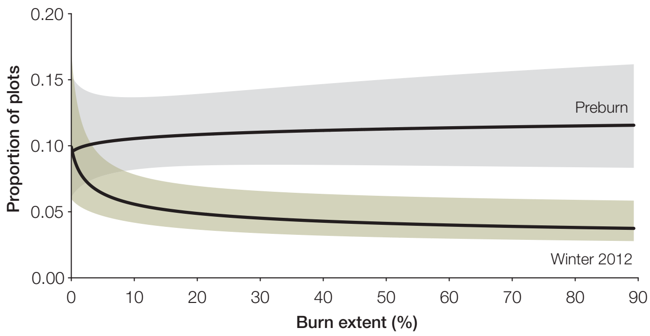

Figure 44. Predicted landscape-level occurrence (proportion of plots) of Grey Fantail *Rhipidura fuliginosa* as a function of burn extent. Shaded regions represents the 95% confidence intervals for predicted values. No relationship with burn extent is expected in the pre-burn survey period since landscapes had not yet been burnt. Predictions are shown for winter 2012. There was no relationship between frequency of occurrence and burn extent for spring 2011, spring 2012 or spring 2013.

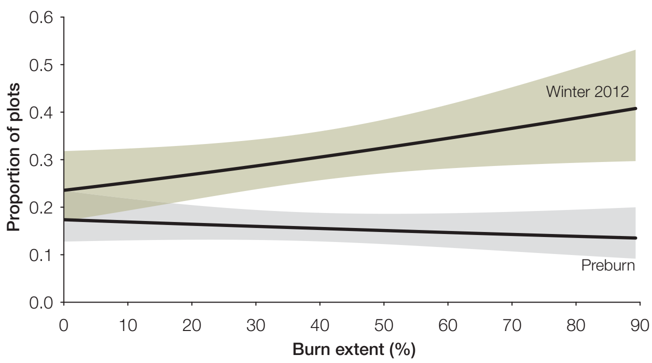

Figure 45. Predicted landscape-level occurrence (proportion of plots) of Scarlet Robin *Petroica boodang* as a function of burn extent. Shaded regions represents the 95% confidence intervals for predicted values. No relationship with burn extent is expected in the pre-burn survey period since landscapes had not yet been burnt. Predictions are shown for winter 2012; results were similar for spring 2011. There was no relationship between frequency of occurrence and burn extent, for spring 2012 and spring 2013.

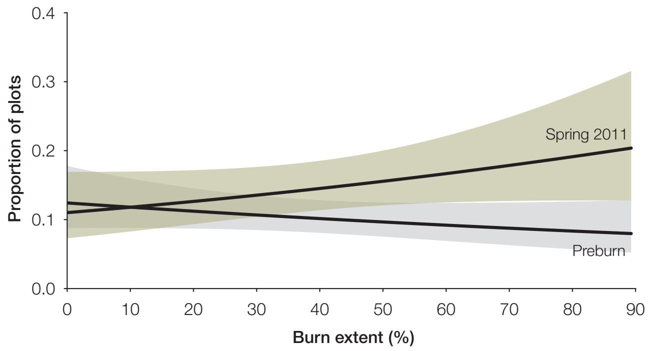

Figure 46. Predicted landscape-level occurrence (proportion of plots) of Spotted Pardalote *Pardalotus punctatus* as a function of burn extent. Shaded regions represents the 95% confidence intervals for predicted values. No relationship with burn extent is expected in the pre-burn survey period since landscapes had not yet been burnt. Predictions are shown for spring 2011. There was no relationship between frequency of occurrence and burn extent for winter 2012, spring 2012 or spring 2013.

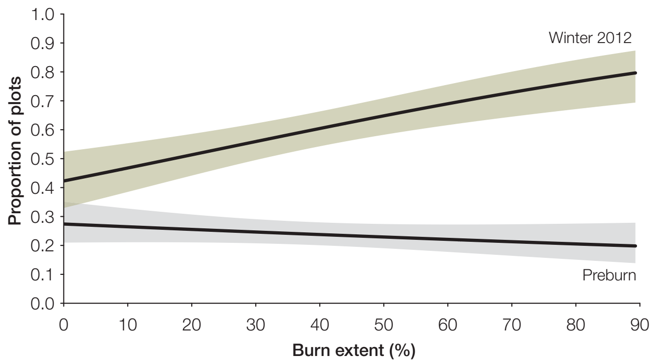

Figure 47. Predicted landscape-level occurrence (proportion of plots) of Weebill *Smicrornis brevirostris* as a function of burn extent. Shaded regions represents the 95% confidence intervals for predicted values. No relationship with burn extent is expected in the pre-burn survey period since landscapes had not yet been burnt. Predictions are shown for winter 2012. There was no relationship between frequency of occurrence and burn extent for spring 2011, spring 2012 or spring 2013.

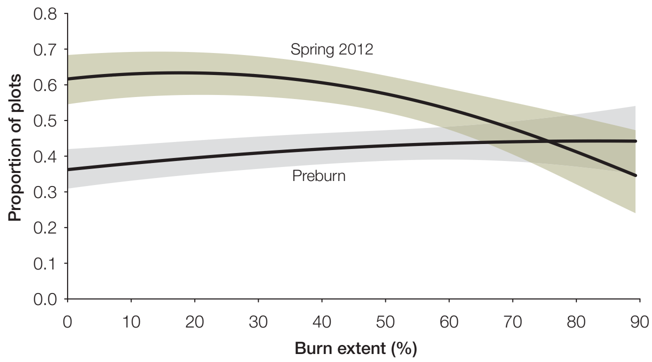

Figure 48. Predicted landscape-level occurrence (proportion of plots) of White-throated Treecreeper *Cormobates leucophaeus* as a function of burn extent. Shaded regions represents the 95% confidence intervals for predicted values. No relationship with burn extent is expected in the pre-burn survey period since landscapes had not yet been burnt. Predictions are shown for spring 2012; results were similar for spring 2013. There was no relationship between frequency of occurrence and burn extent for spring 2011 or winter 2012.

Bird species composition (landscape-level)

The influence of burn extent on bird species composition at the landscape-level was also assessed. Analyses were performed separately for each post-burn survey period (spring 2011, winter 2012, spring 2012, and spring 2013). Each survey period was analysed separately to control for year to year environmental variation (e.g., rainfall). Data from the two bird survey rounds conducted per period were combined for analysis, and the community composition data consisted of the presence or absence of bird species (n = 93) in each landscape.

Analysis of Similarity (ANOSIM) revealed that burn extent (i.e., unburnt, low <50%, high >50%) did not influence bird species composition in any of the post-burn survey periods (Table 6). This was confirmed graphically via ordination: there was a high degree of overlap amongst landscapes from different burn extent categories, indicating that the composition of bird communities was similar irrespective of burn extent (Fig. 49).

Table 6. Analysis of Similarity (ANOSIM) of bird species composition at the landscape-level in relation to burn extent (unburnt, low<50%, high>50%) for each post-burn survey period.

Survey period	R value	P value
Spring 2011	0.08	0.12
Winter 2012	-0.04	0.69
Spring 2012	0.02	0.37
Spring 2013	0.05	0.23

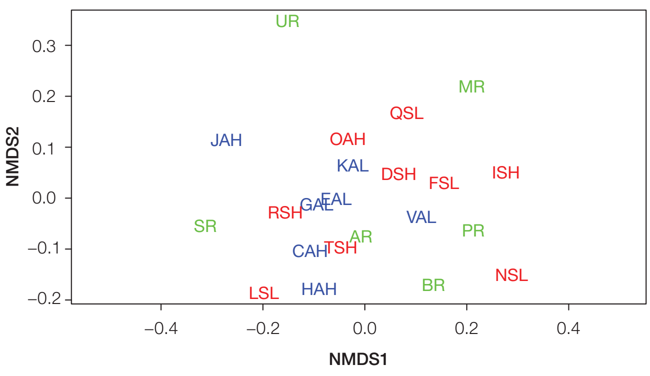

Figure 49. An MDS ordination plot showing the relationship among landscapes of different burn extent categories (unburnt (green), low <50% (blue), high >50% (red)) based on bird species composition (presence/absence data) in spring 2011 (stress = 0.19).

Landscapes with similar composition are closer together in the ordination space. Here there is a high degree of overlap amongst burn categories. Note: a similar lack of separation across burn categories was also observed in the other post-burn survey periods (winter 2012, spring 2012, spring 2013).

To test for differences in bird species composition in relation to the survey period, analyses were performed using data from the reference (unburnt) landscapes only, for the pre-burn (spring 2010) and post-burn (spring 2011, winter 2012, spring 2012, spring 2013) survey periods. As none of the reference landscapes were burnt, any differences between periods can be attributed to variation in environmental conditions. Analysis of Similarity (ANOSIM) revealed that community composition differed across survey periods (R = 0.41, P = 0.001). Ordination revealed a high degree of overlap across the four survey periods conducted in spring (spring 2010, 2011, 2012, 2013), indicating that community composition was generally similar in these periods (Fig. 50). However, the winter 2012 survey period separated from all of the spring surveys, suggesting that the composition of the bird community was different at this time (Fig. 50).

A SIMPER (similarity percentages) analysis was performed to determine which bird species contributed most to differences between survey periods in reference landscapes. The species contributing most to the dissimilarity could be divided into two broad groups: (1) those largely or completely absent during winter surveys but generally present in spring surveys; and (2) those largely absent from spring surveys but present in winter surveys. Species not detected in winter but regularly detected in spring included the Black-faced Cuckoo-shrike *Coracina novaehollandiae*, Common Bronzewing *Phaps chalcoptera* and Rufous Whistler *Pachycephala rufiventris*. Conversely, those detected in winter but generally absent during spring surveys included the Golden Whistler *Pachycephala pectoralis* and Flame Robin *Petroica phoenicea*. Differences in the occurrence of these species between spring and winter surveys likely reflect seasonal migratory movements in and out of the box-ironbark system.

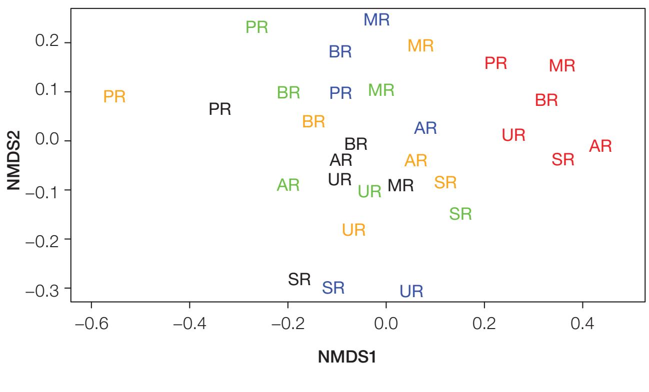

Figure 50. An MDS ordination plot showing the relationship among reference landscapes surveyed at different times (spring 2010 (black), spring 2011 (blue), winter 2012 (red), spring 2012 (green), spring 2013 (orange)) based on bird species composition (presence/absence data) (stress = 0.22).

Landscapes with similar composition are closer together in the ordination space. Here the winter 2012 survey (red landscapes) clearly separates from all others.

The impact of spring burning on the home range, foraging behaviour and habitat use of breeding Scarlet Robins *Petroica boodang*

The Scarlet Robin *Petroica boodang* (Fig. 51) was chosen as a focal species of study, to look more closely at the impact of burning on individual birds during the breeding season. The Scarlet Robin is a widespread species, known to forage commonly on the ground, mostly by perch-hunting and pouncing. Habitat features used by this species, such as ground litter layer and woody debris, are likely to be affected by planned burns.

Methods

Radio-tracking was used to investigate behavioural responses during burning and changes in home range, habitat use and foraging behaviour after burning.

Prior to spring burning, 25 Scarlet Robins were caught in landscapes scheduled for high burn coverage (DSH, ISH, RSH, TSH) and two reference landscapes (AR, BR). Each was fitted with a unique combination of coloured leg bands and a radio transmitter attached to tail feathers (Fig. 51). Tagged birds were followed before and after burning to collect records on location, habitat use, foraging and nesting activity. The lower burn coverage outcome in DSH provided the opportunity to compare low and high-cover burns.

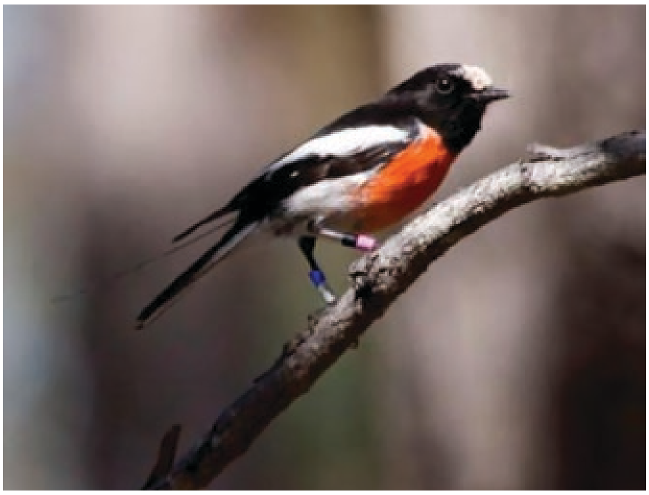

Figure 51. Scarlet Robin *Petroica boodang* fitted with radio transmitter and coloured leg bands (Ash Herrod).

Results

Immediate response (days)

It was possible to track some of the Scarlet Robins during the actual burns (Fig. 52). Birds evaded the burns in different ways, depending on the extent, intensity and coverage of the burn. In the low-cover burn landscape (DSH), the birds moved within the canopy, as low intensity meant low flame height. During the high-cover burn (TSH), the birds retreated into adjacent unburnt forest. All 12 Scarlet Robins, with working transmitters during burning, were found to be alive and in or near to their pre-burn home range when radio-tracking resumed two days after the fires.

Figure 52. Radio-tracking Scarlet Robins *Petroica boodang* during a planned burn (Andrew Bennett).

Longer term response (weeks) – home range

Fine-scale burn mapping (20 m x 20 m grid) was completed to assess the extent and severity of burn coverage within the home ranges of tracked birds. Scarlet Robins proved to be quite resilient to high-cover burning of their habitat, remaining in areas of >90% burn cover and canopy scorch. However, there was a significant increase in the area used by Scarlet Robins after their habitat was subjected to burning for both burn cover levels (ranging from 50–300%) (Fig. 53).

Only one robin shifted from its pre-burn area (i.e., <1% overlap), but the new home range used was immediately adjacent and shared a border with its previous home range (Fig. 54a). Despite a number of the robins having ventured into unburnt forest areas during the burn, post-burn home range expansion was into equally burnt habitat (Fig. 54). There was a strong correlation between burn cover and severity in the before and after home range areas in low-cover and high-cover burns, refuting the prediction that robins would move into better 'less burnt' habitat.

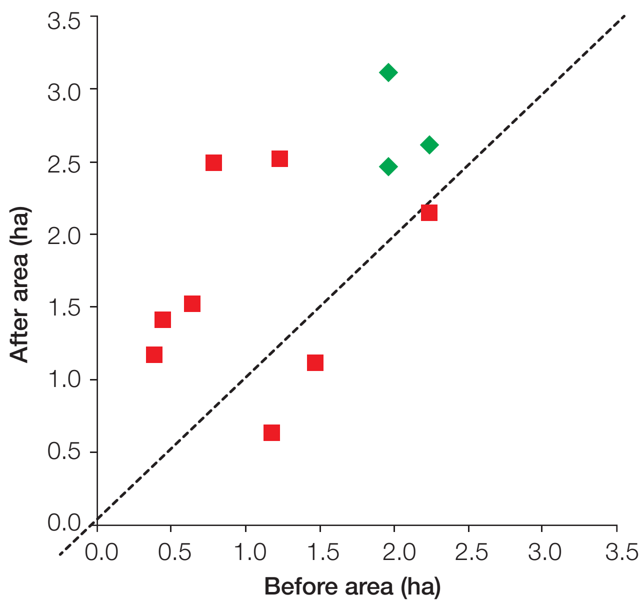

Figure 53. Area used before and after burning by 11 Scarlet Robins *Petroica boodang* (red square = high-cover burns, green diamond = low-cover burns). The dotted line indicates where home ranges that had not changed in area would have fallen.

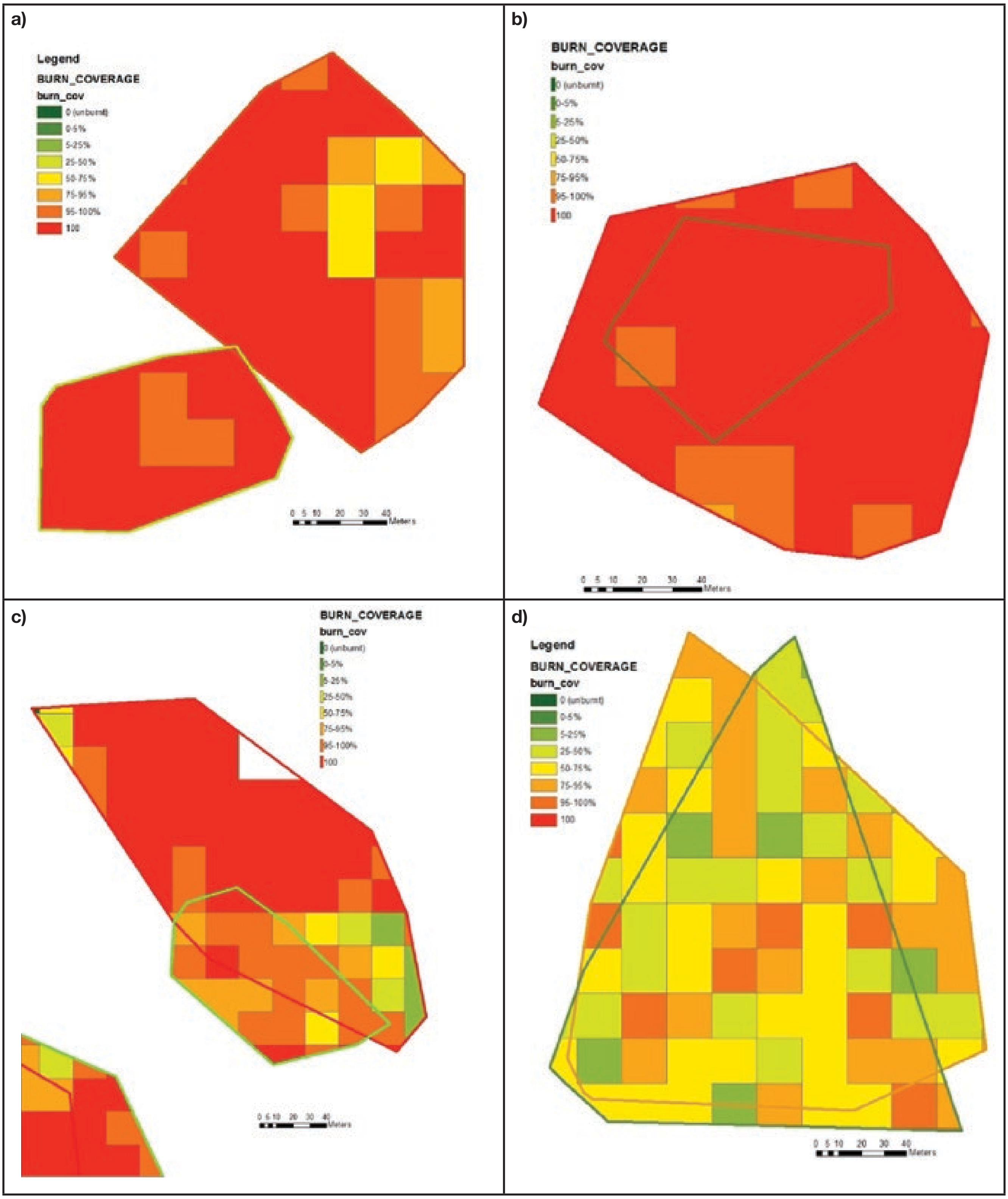

Figure 54. Before (green outline) and after (red outline) home range areas and burn coverage for four Scarlet Robins *Petroica boodang*. High-cover burns: a) <1% overlap. b) 100% overlap. c) 55% overlap. Low-cover burn: d) 88% overlap.

At the time of capture, there were eight pairs with juveniles (although one was a Horsfield's Bronze-Cuckoo, *Chalcites basalis*) and two active nests. There were four known pairs with active nests when burning occurred. They continued tending to nests immediately after burning. However, the ultimate success of these nests is unknown. In the post-burn habitat, four pairs constructed new nests in high-cover burn units, and one pair made two further nesting attempts in landscape DSH.

Most of the tagged robins were observed at least once more after their tracking period had finished and 75% were found within the same mapped locations 2–3 months later in early 2012 (Fig. 55). One year after burning (spring 2012), almost 60% of tagged robins were resighted in or near to home ranges used in the previous year (Fig. 55). Evidence of breeding in spring 2012 was observed for pairs across all landscapes.

This study enabled a confident assessment of survival during burns and, through radio-tracking, revealed an increase in home range area used as an immediate impact on Scarlet Robins, an outcome likely missed in broad-scale community surveys. Low severity fire and small fire extent left unburnt refuges inside and outside the burn area. This is important when devising fire operations plans and burn plans with fauna in mind. Under a different fire regime (e.g., larger extent, increased severity, faster rate of spread) the outcome may be different. However, subsequent cover/patchiness of burns was not important as the Scarlet Robins resettled in high burn cover areas rather than unburnt patches. The species' territorial behaviour could influence this response, including strong site fidelity and competition with conspecifics, as well as their adaptability to the post-burn habitat (e.g., available food supply). It should not be expected that all fauna can recover so quickly, and further autoecological studies on a range of fauna, and different fire regimes, are needed.

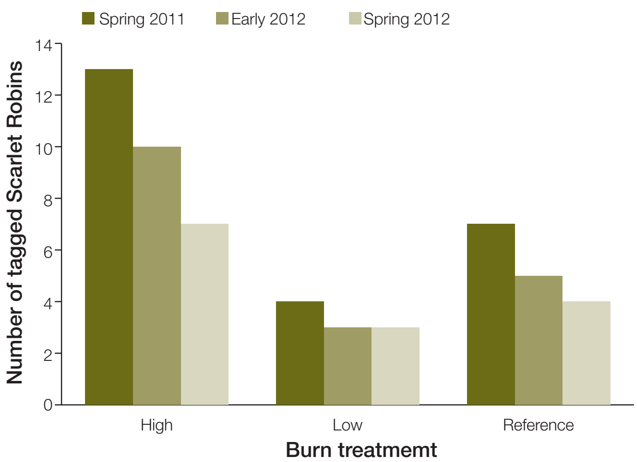

Figure 55. Numbers of male Scarlet Robins *Petroica boodang* initially tagged (spring 2011) and resighted over subsequent year.

Effects of planned burns on den sites of the Yellow-footed Antechinus *Antechinus flavipes*

The Yellow-footed Antechinus *Antechinus flavipes* (Fig. 56) is a small (~30–60 g), carnivorous marsupial that characteristically occurs in dry forests. Individuals are active mainly at night and during the day occupy a den or nest site. All males die after the annual breeding season in July. Females care for young in a nest, making den/nest sites a critical habitat component for this species.

Individuals were trapped and fitted with radio-transmitters to study their movements, home range and use of diurnal den sites prior to, and then after, planned burns. In autumn, six males (mean weight 51 g) and four females (31 g) were tracked before the planned burns. From these animals, five males and two females were tracked again after the burns. An additional female (not tracked pre-fire) was also tracked after the burn. For the spring burn landscapes, extensive trapping (~3,400 trap nights) resulted in the capture of just two individual females. They were fitted with transmitters and tracked both before and after burning. Animals were located once per day during daytime to obtain a diurnal den site location, and up to three times per night to determine home range size/use. Locations (fixes) during daylight were assumed to represent den sites. For each diurnal location, the type of habitat component (live tree, dead tree, log, stump, grass tree *Xanthorrhoea glauca* subsp. *angustifolia*) was recorded. For those in trees, the tree species *(E. tricarpa, E. microcarpa, E. macrorhyncha* and *E. polyanthemos*), and tree stem diameter class at ~1.3 m height were recorded. The fate of habitat components used as den sites prior to the planned burn was checked after burning: these were noted as being completely burnt, partially burnt or unburnt.

Figure 56. Yellow-footed Antechinus *Antechinus flavipes* (Russell Jones).

Surviving the burns

All individuals survived the immediate effect of the planned burns. In autumn, these were in two landscapes that had 41% and 51% burn cover, while the burn cover in the two spring landscapes was 75% and 89%. In spring, one of the two females tracked died from predation shortly after the fire (in the landscape with 89% burn cover).

Den sites

In autumn, individuals most commonly used trees as den sites (45% of daily den site use), followed by logs (38%), stumps (11%) and grass trees (6%). Trees included both living (78% of daily den site use) and dead (22%) trees, with the living trees mainly being *E. microcarpa* and *E. tricarpa.* During spring, in the breeding season, both females used only a single den/nest in a large tree.

Animals were highly selective of the tree species used as den sites compared with those available in the study landscapes (Fig. 57). Dead trees were used about six times more frequently than suggested by their availability. The most common tree species (*E. tricarpa*, 60% of trees), was used disproportionately less than its availability (Fig. 57a). *A. flavipes* strongly favoured large trees as den sites (χ^2=234.09, DF=2, p<0.001), being most commonly found in trees >60 cm diameter (Fig. 57b). Such large trees were scarce in the landscape. Of 337 trees measured in the 20 x 20 m study plots in these same landscapes (12 plots per landscape), none were >60 cm (there were a small number of trees >60 cm but they were not common or widespread enough to be encountered in 20 x 20 m study plots). Despite their rarity, trees >60 cm diameter comprised 53% (39 out of 73) of daily den sites selected in trees, or 24% (39 out of 160) of daily den site selection overall. In total, 82% (60 out of 73) of daily den tree use or 70% (23 out of 33) of unique den trees selected by *A. flavipes* were larger than 40 cm diameter, yet only 5% of available trees (as measured in 20 x 20 m study plots) were in this size category.

Antechinus flavipes also selected the largest size-class of logs as den sites. The largest size class (>40 cm diameter) were used over six times more frequently than suggested by their availability in the forest (as indicated by counts in 20 x 20 m study plots) (Fig. 57c).

The animals tracked before the autumn burns used an average of 6.4 (range: 3–10) unique den sites each. Seven of the eight individuals tracked before the autumn burns (with a home range within the fire area) lost at least one of their den sites due to fire. Of 52 unique den sites used before the burns, 31% (16 of 52) were no longer present following burning. This included 17% of den sites in trees (4 of 23), 48% of logs (10 of 21), 20% of stumps (1 of 5) and 33% of grass trees (1 of 3). This level of loss occurred in two landscapes in which just 41% and 51% of their area was burned.

The den sites of the two female animals tracked prior to spring burns were still present following the fires. Both females used a single den/nest tree each (both trees >60 cm diameter) on consecutive nights and continued to use these same trees following the fires. The nest sites of both females were at a height of 2 m or greater.

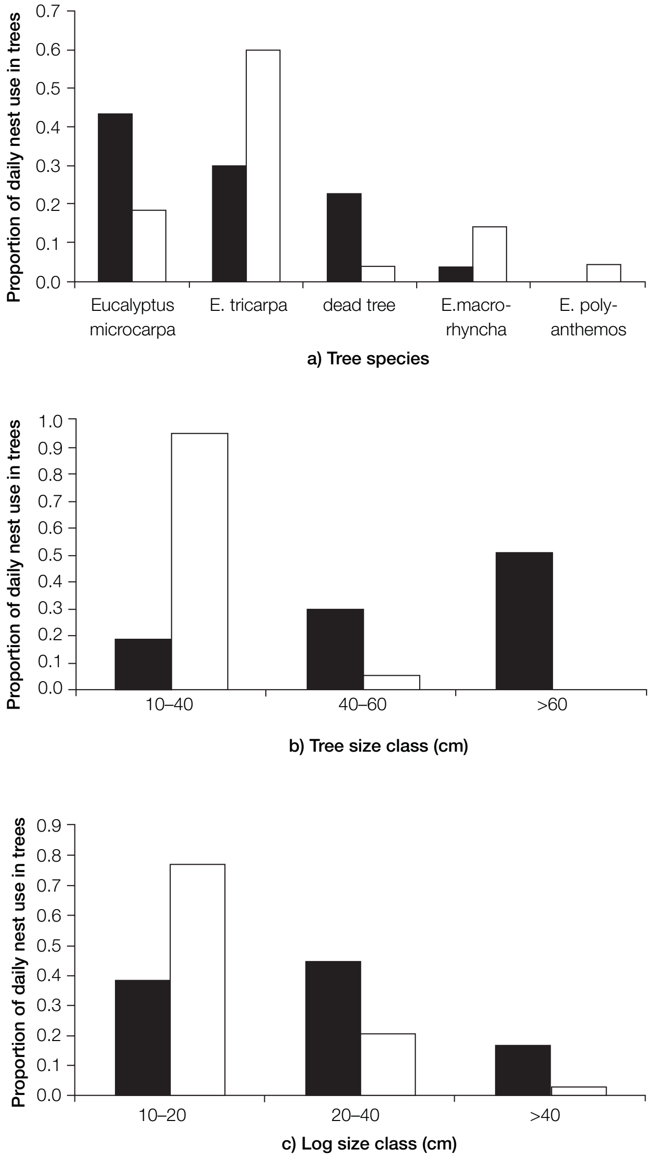

Figure 57. Selection of den sites by the Yellow-footed Antechinus *Antechinus flavipes* in comparison to availability in the landscape. Proportional daily use of den sites pre-fire (black) compared to their availability in the landscape (white) for a) tree species, b) tree size-classes and c) log size-classes.

Effect of planned burning on the distribution of the Brush-tailed Phascogale *Phascogale tapoatafa*

The Brush-tailed Phascogale *Phascogale tapoatafa* (Fig. 58) is a carnivorous marsupial (110–310 g body weight), typically associated with dry forests inland of the Great Dividing Range. It is listed as a 'vulnerable' species in Victoria (Department of Sustainability and Environment 2013), and in recent years (2000–2010), monitoring data have revealed a population decline across its range in north-central Victoria (Holland *et al.* 2012). Phascogales occur at low density: females occupy a home range of about 50 ha, whilst male home ranges are larger (~100 ha). All males die at the end of the annual winter breeding season.

Figure 58. Brush-tailed Phascogale *Phascogale tapoatafa* (Russell Jones).

Methods

The distribution and occurrence of the Brush-tailed Phascogale was studied in 14 of the 22 study landscapes: five subject to planned burns in autumn, five in spring, and four reference landscapes (Fig. 59). In each landscape, five monitoring points were used as locations for remote camera surveys (total = 70 camera survey points). Monitoring points were chosen such that they were spread evenly across the landscape, with at least 150 m between them. ScoutGuard (DTC-530) infrared digital cameras were mounted horizontally (such that they faced downwards) at a height of 150 cm on trees (Fig. 60). Bait holders were placed directly below cameras and, where possible, in close proximity to logs and stumps. We conducted surveys at all sites in summer (February 2013) and winter (June–July 2013). These periods represent the post-breeding season (February) when adult female and both male and female juveniles are active throughout the forest; and the onset of the breeding season (June–July), when both adult males and females are active, respectively. Cameras recorded data for an average of 32 nights per survey (range 24–25 nights in summer survey, 37–39 nights in winter survey). Generalised linear mixed models were employed to investigate the influence of burning on the relative occurrence of Brush-tailed Phascogales.

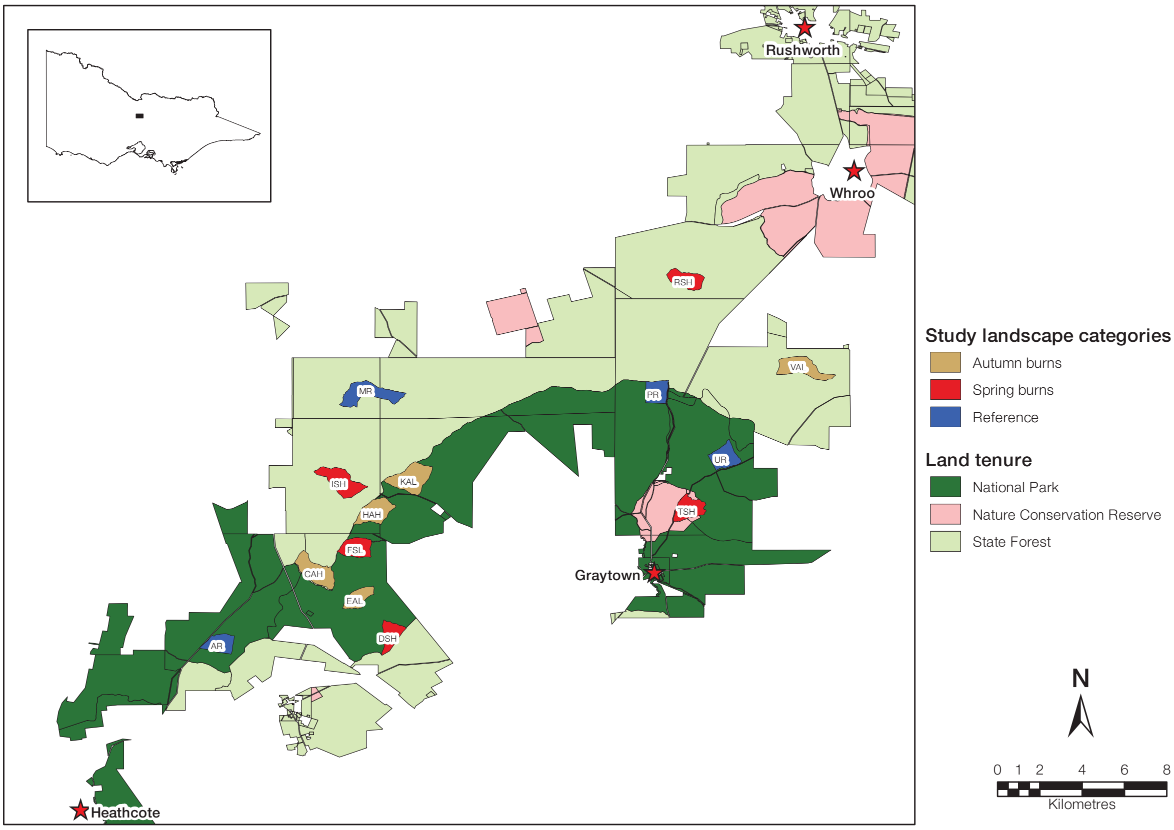

Figure 59. The Heathcote-Rushworth-Graytown forest block, showing the location of 14 study landscapes used for surveys of the Brush-tailed Phascogale *Phascogale tapoatafa*, including the treatment (reference, autumn burn, spring burn) assigned to each.

Plot labels: A–V signifies geographic position from Heathcote to Rushworth, R = reference landscape, A = autumn burn landscape, S = spring burn landscape, L = designated low cover burn (30–50%), and H = designated high cover burn (70–90%). Thus, landscape AR is close to Heathcote and is a reference landscape, while RSH is closer to Rushworth and was designated to receive a high cover spring burn.

Overall (across both survey rounds), Brush-tailed Phascogales were detected in 13 of the 14 landscapes (9 in summer, 11 in winter). The single landscape in which the species was not detected had the highest burn extent (89% of landscape) of all landscapes burned in this study. Phascogales were detected at 25 of 70 camera survey sites (14 in summer, 19 in winter).

Figure 60. Infrared digital camera mounted horizontally (facing downwards) to tree with a bait holder in the field of view at ground-level (Rebecca Peisley).

Factors influencing the relative occurrence of Brush-tailed Phascogales

Landscape-level

At the landscape-level, the relative occurrence of the Brush-tailed Phascogale (number of nights on which phascogales were detected as a proportion of total camera nights) was influenced by three variables: a) the extent to which a landscape was burnt (% cover); b) heterogeneity in tree density; and c) density of logs >20 cm diameter. Burn extent had the strongest influence. Phascogale activity declined as an increasing amount of the landscape was burnt (Fig. 61).

Discussion

There was a strong relationship between the number of logs, stumps, and depth of leaf litter and the extent/cover of burns. With increasing burn cover in a landscape, the number of logs and stumps declined, as did the average depth of litter. Loss of these key habitat resources may be the mechanism driving the response of the Brush-tailed Phascogale to burning. Brush-tailed Phascogales are predominantly carnivorous, with most of their diet consisting of invertebrates (e.g., spiders, beetles, ants, orthopterans, cockroaches, earwigs, ticks, dipterans, moth larvae, psyllids and centipedes [Scarff *et al*. 1998]). They are also scansorial, foraging both in trees and at ground-level. Features such as logs, stumps and the litter layer support many invertebrate species and provide foraging substrates for the Brush-tailed Phascogale. The potential loss of food (invertebrates) and associated foraging opportunities may reduce the occurrence of phascogales. Loss of den sites also may contribute to phascogale decline in burnt landscapes. Brush-tailed Phascogales use similar den sites to the Yellow-footed Antechinus (tree hollows, hollow logs). Greater than 30% of identified den sites of Yellow-footed Antechinus were lost as a result of burning (see previous section), and it is likely that phascogales would have suffered a similar loss of den sites.

Planned burns had a negative influence on the Brush-tailed Phascogale at the landscape-level, even though the planned burns were relatively mild. Phascogales generally were not detected at highly burnt sites, but occurred at sites that were only slightly burnt or not burnt. This explains the landscape-level result: avoidance of burnt areas means they are less active at the larger scale of whole landscapes. Two important implications arise from this work:

1. The effect of burning on the Brush-tailed Phascogale is reduced by patchy burns that create a mosaic of burnt and unburnt vegetation at the landscape-scale. A patchy burn (e.g., 30–40% burn cover) will render some areas less suitable (burnt areas) but will retain some suitable unburnt areas.
2. Burning will have a long-term effect on this species, depending on the time required for key resources (logs, litter, hollows) to be replenished. The time required for such recovery is not known.

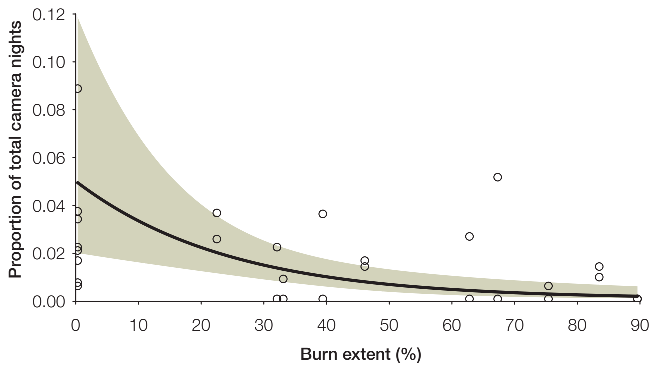

Figure 61. Predicted relative occurrence (proportion of total camera nights detected) of the Brush-tailed Phascogale *Phascogale tapoatafa* at the landscape-level as a function of burn extent.

Interactions between planned burning and grazing by herbivores

The effects of fire and vertebrate herbivore grazing may interact to affect plant communities and vegetation structure. The Black Wallaby *Wallabia bicolor* and Eastern Grey Kangaroo *Macropus giganteus* are native herbivores that are common and widespread throughout the study area. Feral goats *Capra hircus* are also reasonably common. Browsing and grazing by such herbivores may influence plant regeneration and growth throughout the forest, and this pressure may increase after fire because areas of regeneration (palatable young growth) are concentrated and easily targeted by animals. Eastern Grey Kangaroos have been shown to negatively affect post-fire regeneration within burns in a similar forest, with local loss of plant species (Meers and Adams 2003). Different patterns of mosaic burning may attract different levels of grazing pressure and hence modify plant communities to different levels. For example, burned areas in low burn (30–50%) mosaics may experience greater grazing pressure and vegetation change than burned areas in higher coverage (70–90%) mosaics.

Methods

We used small (3 x 3 m) herbivore exclosures (Fig. 62) to measure potential interactions between burning and herbivore grazing. Two exclosures were constructed in each landscape (total = 44) shortly after burning occurred. In each landscape, one exclosure was positioned in a burned area while the other was positioned in an unburned area (except for reference landscapes, where both exclosures were in unburned areas). Within each exclosure, four 1 x 1 m quadrats were established where vascular plant species and vegetation structure were assessed at regular post-fire intervals. A further four 1 x 1 m quadrats were positioned immediately adjacent to each exclosure (within 10 m) to allow comparison of fenced and unfenced (but otherwise similar) plots.

For species richness of perennial plants, data collected in spring 2013 was used for analysis. For vegetation structure, data collected on three separate occasions were analysed: (1) immediately after exclosure construction (winter 2011/summer 2011/12); (2) spring 2012; and (3) spring 2013. Generalised linear mixed models were used to investigate changes in species richness of perennial plants and vegetation structure in relation to:

a) fenced v unfenced plots
b) burned v unburned plots
c) the extent to which a landscape was burned (%)
d) the interaction between fenced/unfenced and burned/unburned (to test the hypothesis that grazing pressure may differ between burned and unburned locations)
e) the interaction between fenced/unfenced and extent of burning (to test the hypothesis that grazing pressure may vary according to the mosaic burn achieved)

Figure 62. A 3 x 3 m grazing exclosure in a study landscape (Greg Holland).

Species richness (perennials)

The number of perennial plant species recorded at a given location (exclosure or associated unfenced quadrats) was best predicted by whether or not that location had been burned: plots in unburned areas were predicted to have more species than those in burned areas (Fig. 63). Neither the presence of an exclosure nor the extent to which a landscape was burned were important. The interaction terms were also of no predictive value, meaning there was no evidence to support the hypothesis that grazing pressure varies with differing mosaic patterns.

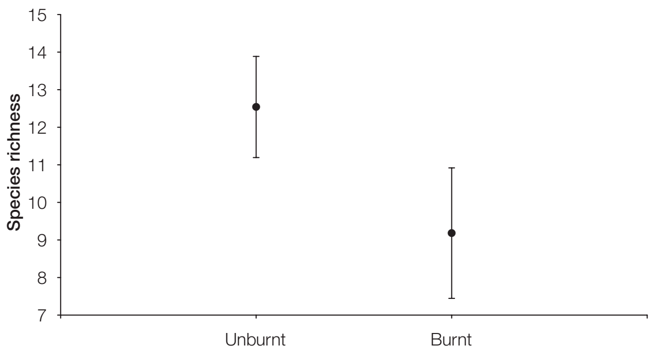

Figure 63. Predicted richness of perennial plant species in herbivore exclosures (and associated unfenced quadrats) according to whether the location had been burned. Error bars represent 95% confidence intervals for predicted values.

Vegetation structure

Data were analysed separately for each of the three surveys conducted. In each survey period, structure data were also analysed separately for two height categories: (1) ≤50 cm; and (2) >50 cm. In all three survey periods, the proportion of monitoring points at a given location (exclosure or associated unfenced plots) where vertical structure was recorded was best predicted by whether or not that location had been burned. Quadrats in unburned areas were predicted to have more complex structure than those in burned areas (Fig. 64). This was the same for both vegetation height categories assessed. In the first two survey periods, the burnt/unburnt status of a location was the only influential variable identified. However, in the third survey period (spring 2013), whether or not a location was fenced became an important predictor of the structural complexity of the ≤50 cm height category: plots within exclosures were predicted to have greater structural complexity than adjacent unfenced plots (Fig. 65). This indicates that grazing may be an important process, with structural complexity increasing through time after the construction of exclosures. Since interaction terms were of no predictive value, this grazing pressure appears to be uniform across the study area and not strongly influenced by differing mosaic burn patterns or burn extent.

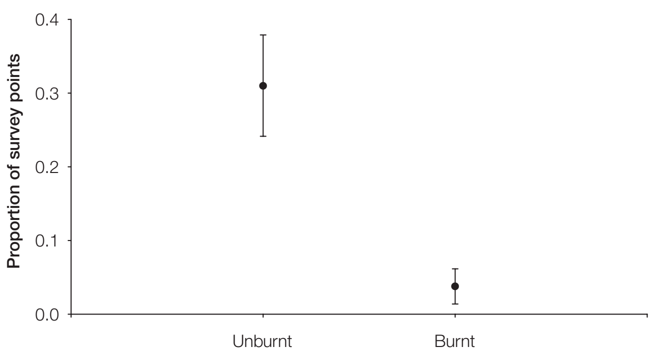

Figure 64. Predicted proportion of survey points in herbivore exclosures (and associated unfenced quadrats) containing vegetation structural complexity ≤50 cm in height according to whether the location had been burned. Predictions were generated from data collected in the first survey period after the construction of exclosures. Similar results were obtained for the other two survey periods, and also for structural complexity >50 cm in height in all three survey periods. Error bars represent 95% confidence intervals for predicted values.

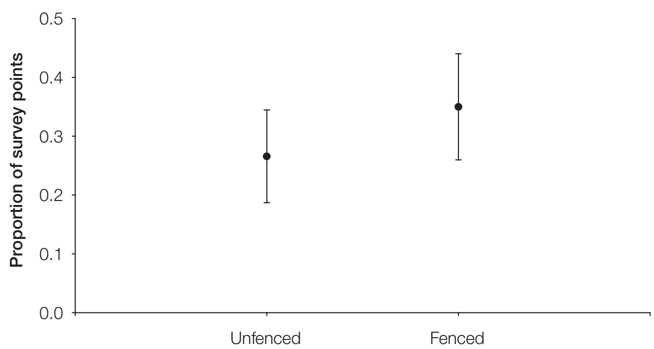

Figure 65. Predicted proportion of survey points in herbivore exclosures (and associated unfenced quadrats) containing vegetation structural complexity ≤50 cm in height according to whether survey plots were fenced. Results relate to the third (latest) survey round (spring 2013) only. Error bars represent 95% confidence intervals for predicted values.

Implications

Study design – strengths and limitations

Experimental approach

There are several important features of this project. First, a strength is that it has been undertaken as a 'management experiment' in which specific treatments have been applied in a planned way in collaboration with land managers (DELWP, PV). Notably, in addition to different fire treatments (extent of burn, season of burn), six landscapes were retained as reference landscapes in which no treatment occurred. This proved to be particularly important given the marked change in climatic conditions during the study, from the end of a decade of drought to flooding rains in 2010–11. The change in these conditions affected all landscapes, both burnt and unburnt, particularly in relation to increased understorey plant growth. By having unburnt reference landscapes, we were able to account for changes not associated with burning effects.

Landscape-scale

Second, the study has been undertaken at the 'landscape-scale', with replicate landscapes as the unit of study. This type of approach is essential in order to make comparison between different 'whole fire mosaics'. Such conclusions are not possible from studying a series of individual sites. In this case, the approach has allowed a clear demonstration that the extent of burn in a landscape has significant influence on habitat resources, vegetation structure and some fauna (e.g., phascogales) (see below).

Duration

A limitation of this project is that it allows evaluation only of short term effects of different burn treatments (up to two years post fire). This is an inherent limitation of all 'longitudinal' studies, compared with those that take a 'space for time substitution' approach and sample different areas of different age post-fire. The latter approach was not possible in this situation because: a) the fire history is poorly known; and b) there are not comparable fire mosaics from historic burning. However, if monitoring is sustained, a longitudinal approach is particularly powerful because it can track changes through time from a known baseline.

Implementing planned burns in box-ironbark forests

Box-ironbark forests occur in dry environments on nutrient-poor soils, and have a low productivity (Environment Conservation Council 1997). This is reflected in the generally sparse and shrubby understorey, and low levels of leaf litter accumulation on the ground. As demonstrated by fuel hazard assessments prior to burning, the overall hazard levels of all of these landscapes was primarily 'low' (see Fig. 10), despite the forests not having been burnt for at least 30 years. The sustained drought (~2000–2009) contributed to the limited vegetation, but it is clear that these forests have inherently low fuel levels and do not constitute a high fire risk under normal circumstances. Bushfires of natural origin can and do occur, but these are infrequent (Department of Sustainability and Environment 2003; Tolsma *et al.* 2007a). However, anthropogenic ignition is now the main source of fires and under extreme fire weather conditions, as demonstrated by the Black Saturday fires, almost any forest situation can burn.

The experimental burns were carried out by staff from DELWP and PV under standard operating procedures for planned burns, with several exceptions. First, the burns were planned to achieve a specified level of burn cover across the landscape, either 'low' cover (30–50%) or 'high' cover (70–90%). In practice, this proved difficult to achieve. All autumn burns were less than 51% burn cover (see Table 1), essentially low cover patchy burn mosaics. This was largely due to unseasonally heavy rain in summer 2010/11, and a moist and sparse litter layer on the ground (Galvin and Medlyn 2011). In spring, all burns were greater than 51% burn cover. Two important implications are:

a) this outcome confounds the study design, making it very difficult to distinguish the effects of the season of burn from the extent of burn in a landscape.

b) from an operational perspective, for landscape mosaic burning to be used effectively as a management tool, it is essential that operational staff develop expertise in being able to deliver specified burn outcomes, within a reasonable range of variation (e.g., ± 10%).

A second exception is that, prior to burning, an effort was made to reduce the likelihood of large old trees being burnt by raking around their bases to remove flammable litter (Fig. 66). This was a standard practice being implemented for (most) planned burns in the Bendigo region in 2009–2010, but is less feasible when the target area for planned burning has been greatly increased.

Figure 66. Example of raking around a tree to reduce the risk of burning (autumn burn landscape) (Pat McCarthy).

Impacts of planned mosaic burns on fauna and habitat resources

Fauna

The immediate effects of planned burns on animals were demonstrated for two species, the Scarlet Robin and Yellow-footed Antechinus, by monitoring marked individuals. All individuals survived these mosaic burns. Scarlet Robins tracked during the fire either moved outside the burn area (high cover fire) and later returned, or remained within the burn area in unburnt forest patches (low cover burn). Yellow-footed Antechinus are presumed to have avoided the fire by remaining within a den site in a tree hollow or log. They were not tracked during the actual burn, so their movements are not known.

The greatest effects for faunal species (positive or negative) arise from the way in which they respond to changes to habitat resources. Alterations to critical resources (e.g., den sites, foraging substrates) may result in changes to behaviour, displacement, reduced abundance and changes in the composition of communities. Individual Scarlet Robins were largely found in the same areas post-fire as pre-fire, but expanded the size of their territories (up to 300% of original size) following burns. This suggests that some measure of habitat quality for this species declined as a result of burning, requiring a larger area to be occupied in order for resources to be met. The Scarlet Robin is insectivorous, and spends much time foraging on the ground. Loss of logs, stumps and litter (both cover and depth) is likely to have a negative impact as such resources represent foraging substrates for the birds and habitat for their invertebrate prey. It is likely that Scarlet Robins need to forage over wider areas in the immediate post-fire environment to meet dietary needs.

Yellow-footed Antechinus favoured large trees and logs as den sites. Habitat surveys found such resources to be extremely rare. While no individuals were directly killed by planned burns, a high proportion (31%) of den sites identified pre-fire were destroyed in planned burns that covered ~50% of the landscape. With preferred den sites being so scarce, Yellow-footed Antechinus may be restricted by den site options post-fire. This may result in animals using fewer den sites or using sub-optimal sites, with potential for longer-term consequences for the species arising from increased predation rates and lower reproductive success.

The occurrence and activity of the Brush-tailed Phascogale was lower at sites with high burn cover, and a negative relationship with burn extent was observed at the landscape-scale. Planned burns clearly had a negative effect on this species. Phascogales are scansorial, foraging both above ground in trees and at ground-level. Loss of logs, stumps and litter, and a corresponding loss of invertebrate food supplies, may render highly burnt sites unsuitable habitat for Brush-tailed Phascogales. Phascogales also have similar den site requirements to the Yellow-footed Antechinus. Therefore, loss of den sites as a consequence of burning will also negatively affect this species.

Surveys of the terrestrial bird community indicated that burning was not a strong influence. The extent to which a landscape was burnt was related to the frequency of occurrence of just six of 25 species modelled. These species responded in different ways to burning (some positive, some negative), and responses were often inconsistent across different post-burn survey periods. Further, changes in the frequency of occurrence of species as a function of burn extent were generally not pronounced. The timing of surveys was a more consistent influence on the bird community. In particular, the survey conducted in winter produced different results to those conducted in spring, reflecting variations in conditions and seasonal migratory movements of species.

Habitat structure

The planned burns conducted in this study had a clear impact on habitat structural features, which in turn affects the fauna dependent on such features. A key finding is that the degree of change in structural features was often proportional to the extent of burn in the landscape. This means that as burn extent increases, the likely impact on fauna will also increase. There was also a disproportionate effect on some resources. For example, for any given burn extent, a greater proportion of large logs were lost compared with small logs. Cover of coarse litter also declined as a function of burn extent, more so than fine litter. Consequently, planned burns are particularly likely to affect species that depend on such resources.

The critical issue is the time required for habitat resources to develop after fire, and how this relates to the interval between fires. Box-Ironbark forests typically have low productivity, resulting in trees growing at slow rates (Environment Conservation Council 1997). Replacement of features such as large trees (and associated hollows), large logs, and even coarse litter, is a slow process. This means that planned burns continue to affect fauna for many years into the future; and that further repeat burns in the same area will likely lead to an incremental depletion of these resources through time.

Impacts of planned mosaic burns on plant species

This project did not identify a consistently strong response by plants to burning. Of the seven individual species analysed, just one (Cranberry Heath) displayed a relationship with extent of burn, with this species declining rapidly as burn extent increased beyond 45% (at both the landscape- and plot-level). A slight shift in the composition of plant communities was observed post-fire, with landscapes with a burn extent exceeding 50% having more annual species (including introduced species) compared to unburnt reference landscapes. At the scale of individual survey plots, the number of weed species recorded was higher in landscapes burnt in spring compared to reference landscapes. This suggests that the timing of burns may be an important consideration for minimising post-burn invasion by introduced species. Weed species increased as burn extent increased. A total of 43 plant species were found only post-fire on burnt plots suggesting that these species may benefit from burning – but most were scarce. Longer-term monitoring is required to determine the ability of these species to persist through time. It is also possible that at least some of these species were present pre-fire but are cryptic and difficult to detect.

A complicating factor in identifying relationships between plants and burning was the above-average rainfall at the beginning of the project (2010 and 2011). The strongest observed trends were that the total number of perennial species, the occurrence of analysed individual species, and the richness of weed species all increased in post-fire surveys irrespective of treatment (i.e., increases were observed in reference landscapes as well as in landscapes that were burnt). The most likely explanation is that high rainfall in 2010/11 triggered both plant germination and growth, resulting in more species being detected in post-fire surveys. These results indicate that rainfall was a stronger driver of plant responses than were the burns conducted for this project. However, it is important to note that the rainfall received in 2010/11 came on the back of a decade of drought conditions. Interactions between plant responses, rainfall and fire may be different under different environmental conditions. The role of fire in shaping plant responses is also likely to change if areas are subjected to frequent repeat burning, rather than the single fire events investigated here. The acknowledgement of the importance of rainfall in driving observed results highlights the critical role that reference landscapes (i.e., untreated areas) have to play in studies such as this. Without the reference landscapes to provide a comparative baseline, post-fire increases in perennial plant species richness may have been erroneously attributed to the effects of fire.

Effects of planned mosaic burns on other ecological processes

Eucalypt flowering

There was no evidence that burning had either a positive or negative influence on flowering of Red Ironbark, at two years after planned burns. Red Ironbark is an important nectar-producing species in these forests, and year-to-year variation in its flowering pattern has a large influence on bird communities. In winter months, in years of heavy flowering, there is a high density of species such as Red Wattlebird, Musk Lorikeet, and various honeyeaters, including winter migrants such as Yellow-faced Honeyeater and White-naped Honeyeater. In years when flowering is sparse or fails, there is a mass exodus of nectarivores from the forest.

Grazing and fire

In the short-term (two years post-fire) we did not find evidence for a strong interactive effect between burning and grazing by herbivores. In the grazing exclosure study, whether or not a site was burnt was a strong influence on both the richness of perennial plant species and vertical vegetation structure. However, these results applied equally to plots within exclosures and those that were not fenced. A grazing effect was beginning to emerge in the final survey round (2013), when greater structural complexity of vegetation ≤50 cm was recorded inside exclosures compared with unfenced plots. This pattern is likely to become stronger in future years as plants within exclosures continue to grow in the absence of grazing. These results suggest that grazing pressure is relatively uniform across the study area and not strongly influenced by differing mosaic burn patterns or burn extent. However, longer-term monitoring will provide a clearer understanding of grazing effects. An increased abundance of feral Goats in this forest is of concern. Mobs of goats were rarely seen in past years (e.g., prior to 2009, A Bennett, pers obs), but were regularly observed during this study (2010–13).

Soils and run off from exposed surfaces

We were not able to undertake formal study of the effect of planned burns on soil properties and soil dynamics. From visual observations it was clear that there was substantial downslope movement of soils exposed after the burns. This resulted in accumulation of fine silt and charcoal behind obstructions such as logs or stumps, in depressions (associated with old mullock heaps), or along shallow drainage lines. Given the skeletal nature of soils in this vegetation type, further depletion of topsoils is of concern. The amount of soil loss will depend on:

a) the time required for sufficient litter and bryophyte cover to establish to slow and prevent run-off

b) the magnitude of rain events (heavy rain will sweep litter and soil; light rain less so)

c) the patchiness of the burn (a patchy burn with many unburnt patches has greater capacity to reduce run-off)

d) the inter-fire interval (with soil loss each time burning occurs, a short inter-fire interval will result in a greater incremental and cumulative effect).

Drought and fire

The study commenced at the end of a decade of drought when ground layer and shrub vegetation in the forest was severely depleted. Heavy rain in 2010 and 2011 triggered widespread growth of ground-layer plants, new growth of shrubs and new growth of eucalypt foliage throughout the forest. Untangling the interacting effects of rainfall and fire on vegetation is important and requires long-term data collection.

Implications for fire management in box-ironbark forests

Objectives of burning

There was little evidence of an ecological benefit from the planned burns undertaken in this study, at least in the short term (two years post-fire). The most likely benefit would arise from germination and growth of plant species that may have decreased in abundance with time-since-fire, but remained present due to a soil seed store. The proposed maximum tolerable fire interval for these forests of 150 years (Cheal 2010) implies that there is little need for frequent fire in this ecosystem.

For four 'key fire response species' for which there were sufficient data (*Acacia genistifolia, Cassinia arcuata, Daviesia ulicifolia, Pultenea largiflorens*), there was evidence of increased occurrence at the landscape-scale in 2012 and 2013 compared with pre-fire occurrence in 2010; but no evidence of any increase associated with the extent of burning in the landscape. The increased abundance in 2012/13 is likely related to above-average rainfall in these years. However, there was also a number of species (43) which occurred post-fire in burnt plots but not unburnt plots, which may potentially have benefited from burning — but most had very few records (too few to formally analyse). Further long-term monitoring is required to clarify the potential benefits to the plant community of these burns. Targeted monitoring of species with few records could clarify their long-term fire response.

In contrast, there was substantial evidence of burning having either little or no effect (e.g., mortality of animals, flowering of Red Ironbark), or a detrimental effect (e.g., depletion of habitat resources, increased numbers of weed species, reduced distribution of Brush-tailed Phascogale), at least in the short term but likely with longer term consequences.

Patchy mosaic burning is less detrimental than extensive burning

There was clear evidence that, at the landscape-scale, the impact of burning on many attributes was related to the extent of the landscape that was burned. As more of the landscape was burned there was, for example, an increased loss of logs and stumps, reduced depth of litter, and increased numbers of weed species. Thus, a patchy burn will have less impact than an extensive burn, but still offer potential benefits to reducing fuel loads.

Where planned burns must be undertaken and options exist around the level of burn coverage, we recommend that a patchy burn of <50% cover is appropriate. Burns of lower coverage will minimise impacts on biodiversity values.

The frequency of burning is critical

Frequency of burning has a critical influence on the long-term effects of fire management practices. Tolerable fire intervals (TFI) (Cheal 2010) are based on estimates of the requirements of plant species, but do not take into account the responses of animal species or their habitat requirements (Clarke 2008). The latter also need to be considered in determining appropriate intervals. Some species and resources will be little affected by burning, but others are sensitive.

The key issue is whether the interval between fires is sufficient for habitat resources to be replenished. If not, then each successive fire will deplete resources further, with cumulative effects on the biota. Limited evidence indicates that post-fire changes extend over decades. Other relevant points arising from this study include:

a) fuel hazard levels in all study landscapes were low, even after 30+ years without fire (and probably 60 years or more). The level of fuel hazard does not warrant frequent burning (e.g., at or around the proposed minimum tolerable fire interval of 12 years)
b) there was disproportionately greater depletion of larger logs (>20 cm diameter), which are those which will be slowest to accumulate over time
c) almost one third of den sites used by the Yellow-footed Antechinus were burnt in a single patchy burn
d) the increased occurrence of weed species and increased run-off of soil after fire are undesirable outcomes that should be minimised.

Outcomes from this study indicate that for ecological purposes, a minimum tolerable fire interval of 30 years, as designated for high severity bushfire (Cheal 2010), is a more suitable frequency than the currently recommended 12 years for planned burns. Note that this is the minimum TFI, not the required TFI, with an expectation that the actual TFI will vary between the minimum and maximum (i.e., 30 to 150 years).

A regional strategy for a resilient forest ecosystem in relation to fire

Bushfire and planned burning set in train changes to forests that last for decades. Consequently, strategic planning requires a target or goal for what the post-fire age-structure of forests should be like, at the regional scale. For example, a commitment to burning 5% of public land per year would create a potential scenario such that after 20 years these forests will primarily be between 0 and 20 years post-fire.

We recommend that an important priority for fire management is to determine a desirable age-structure for box-ironbark forests across the region. This requires, for example, determining the appropriate mix of forest growth stages (as per Cheal 2010) that will ensure a resilient ecosystem. This task will best be undertaken in conjunction with assessment of fire risk in the region. Areas of high fire risk in proximity to towns, cities and human assets may be designated to be burned more frequently, whereas those of low risk can be designated for infrequent burning or fire exclusion. In particular, a high priority is to determine those areas that can be maintained as examples of 'long unburnt' growth stages.

Long term monitoring

This project has allowed investigation of the short-term (maximum two years after fire) effects of planned burns on plants, animals and ecosystem processes, with a number of clear results being obtained. However, fire-related changes to the forest and its biota will take more than two years to become fully apparent in many cases, and post-fire recovery is a slow process that may take several decades. Consequently, it is imperative that longer-term monitoring is performed to fully appreciate the ecological role of fire in box-ironbark forests.

This project has established a carefully designed infrastructure across the Heathcote-Graytown-Rushworth Forest that provides an excellent foundation for long-term monitoring. Twenty-two landscapes have been identified and 264 monitoring points established. Forty-four grazing exclosures have been constructed. Continued monitoring would value-add immensely to the work already undertaken by tracking changes through time from a known baseline. Results could be used to continually inform management decisions in an adaptive management framework, and to provide the community with evidence-based information regarding the ecological effects of planned burns.

We recommend a commitment to regular and systematic monitoring of the following key attributes.

- **Habitat structure (logs, stumps and litter):** components of habitat structure such as logs, stumps and litter are critically important as habitat for fauna and for moisture and nutrient retention within the forest. These features will likely take many years to re-accumulate following fire in relatively unproductive box-ironbark forests. We recommend that logs, stumps and litter continue to be monitored every **two to three years**.
- **Understorey vegetation structure:** given the relatively slow growth rates of plants in dry box-ironbark forests, understorey vegetation structure will also be slow to develop following fire. We recommend that understorey vegetation structure continue to be monitored every **two to three years.**
- **Floristics:** the response of vegetation communities and individual plant species following fire is a key ecological process. For example, maximum and minimum tolerable fire intervals are currently based on plant responses. We recommend that plant communities continue to be monitored every **two years**.
- **Grazing exclosures:** interactions between grazing and fire are likely to take several years to become apparent given the slow growth rate of plants in box-ironbark forests, and will also be influenced heavily by local rainfall. We recommend that understorey vegetation structure within grazing exclosures continue to be monitored every **one year**, and plant communities (floristics) continue to be monitored every **two years**.
- **Bird community:** fire is likely to influence animal communities for many years via changes to vegetation and habitat resources. We recommend that bird communities continue to be monitored every **two years**.
- **Brush-tailed Phascogale:** the Brush-tailed Phascogale was found to be negatively influenced by planned burns in the short-term. Ongoing monitoring is critical to assess the long-term impacts of burning on this threatened species. We recommend that surveys for this species continue to be conducted every **two years.**

Given that infrastructure is already in place, future monitoring of the above attributes should be conducted in the same landscapes as used initially (i.e., all 22 landscapes for habitat structure, understorey vegetation structure, grazing exclosures and bird community surveys; 15 landscapes for floristic surveys; 14 landscapes for Brush-tailed Phascogale surveys). This would maintain the integrity of the original study design, thereby maximising the likelihood of detecting differences between landscape treatment groups through time. If this is not feasible, a subset of landscapes could be carefully selected to reduce survey effort without excessively compromising the project design. For example, selection of 16 landscapes would still allow for a total of six autumn burn landscapes, six spring burn, and four unburnt reference landscapes. Floristic surveys and surveys for the Brush-tailed Phascogale could continue to be conducted in the same landscapes used originally (15 and 14, respectively). Given the effort to construct grazing exclosures and the relative speed with which they can be surveyed, exclosures should continue to be monitored in all 22 landscapes.

References

Bennett, A.F., Holland, G.J., Flanagan, A., Kelly, S. and Clarke, M.F. (2012). Fire and its interaction with ecological processes in Box-Ironbark forests. *Proceedings of the Royal Society of Victoria* **124**, 72–78.

Cheal, D. (2010). *Growth Stages and Tolerable Fire Intervals for Victoria's Native Vegetation Data Sets. Fire and Adaptive Management Report No. 84*. Department of Sustainability and Environment, East Melbourne.

Clarke, M. F. (2008). Catering for the needs of fauna in fire management: science or just wishful thinking? *Wildlife Research* **35**, 385–394.

Curr, E.M. (1883). *Recollections of Squatting in Victoria*. Melbourne University Press (second edition 1965), Melbourne.

Department of Environment and Primary Industries (2014). Advisory List of Rare or Threatened Plants in Victoria – 2014. Victorian Department of Environment and Primary Industries, Melbourne, Victoria.

Department of Sustainability and Environment (2003). North West Region, Bendigo Fire District: Fire Protection Plan. Department of Sustainability and Environment, Melbourne, Victoria.

Department of Sustainability and Environment (2013). Advisory List of Threatened Vertebrate Fauna in Victoria – 2013. Victorian Department of Sustainability and Environment, Melbourne, Victoria.

Environment Conservation Council (1997). *Box-Ironbark Forests and Woodlands Investigation: Resources and Issues Report*. Environment Conservation Council, Melbourne.

Fox, B.J. (1982). Fire and mammalian secondary succession in an Australian coastal heath. *Ecology* **63**, 1332–1341.

Galvin, P. and Medlyn, L. (2011). *Heathcote Mosaic Burn Project: Burn Outcomes & Severity Mapping Report, Autumn 2011.* Department of Sustainability and Environment.

Holland, G.J., Alexander, J.S.A., Johnson, P.G., Arnold, A.H., Halley, M. and Bennett, A.F. (2012). Conservation cornerstones: capitalising on the endeavours of long-term monitoring projects. *Biological Conservation* **145**, 95–101.

Keatley, M.R., Hudson, I.L. and Fletcher, T.D. (2004). Long-term flowering synchrony of box-ironbark eucalypts. *Australian Journal of Botany* **52**, 47–54.

Lawrence, R. E. and Bellette, M. P. (2010). Gold, timber, war and parks: a history of the Rushworth Forest in central Victoria. *Proceedings of the Royal Society of Victoria* **122**, 130–146.

Lindenmayer D.B., Franklin J.F. and Fischer J. (2006). General management principles and a checklist of strategies to guide forest biodiversity conservation. *Biological Conservation* **131**, 433–445.

Mac Nally, R. and McGoldrick, J.M. (1997). Landscape dynamics of bird communities in relation to mass flowering in some eucalypt forests of central Victoria. *Journal of Avian Biology* **28**, 171–183.

Meers, T. and Adams, R. (2003). The impact of grazing by Eastern Grey Kangaroos (*Macropus giganteus*) on vegetation recovery after fire at Reef Hills Regional Park, Victoria. *Ecological Management and Restoration* **4**, 126–132.

Muir, A.M., Edwards, S.A. and Dickins, M.J. (1995). *Description and conservation status of the vegetation of the Box-Ironbark ecosystem in Victoria. Flora and Fauna Technical Report* 136. Department of Conservation and Natural Resources, Victoria.

Noble, I. R. and Slatyer, R. O. (1980). The use of vital attributes to predict successional changes in plant communities subject to recurrent disturbances. *Vegetatio* **43**, 5–21.

Orscheg, C. K. (2006). *An Investigation of Selected Ecological Processes in Ironbark Communities of the Victorian Box-Ironbark System.* PhD Thesis, The University of Melbourne.

Scarff, F. R., Rhind, S. G. and Bradley, J. S. (1998). Diet and foraging behaviour of brush-tailed phascogales (*Phascogale tapoatafa*) in the jarrah forest of south-western Australia. *Wildlife Research* **25**, 511–526.

Segrave, A. (2009). Fuel-reduction burning in the Box-Ironbark region of north-central Victoria: the effects on habitat attributes and fuel hazard. Honours Thesis. Deakin University, Melbourne.

Teague, B., Macleod, R. and Pascoe, S. (2010). *2009 Victorian Bushfires Royal Commission. Final Report*. State of Victoria.

Tolsma, A., Cheal, D. and Brown, G. (2007a). *Ecological burning in Box-Ironbark forests. Literature review*. Report to North Central Catchment Management Authority. Department of Sustainability and Environment, Melbourne.

Tolsma, A., Cheal, D. and Brown, G. (2007b). *Ecological burning in Box-Ironbark forests. Management strategy*. Report to North Central Catchment Management Authority. Department of Sustainability and Environment, Melbourne.

Tongway D. J. and Ludwig J. A. (1997). The conservation of water and nutrients within landscapes. In: *Landscape Ecology, Function and Management: Principles from Australia's Rangelands* (eds J. A. Ludwig, D. J. Tongway, D. Freudenberger, J. Noble and K. C. Hodgkinson) pp. 13–22. CSIRO Publishing, Melbourne.

Tzaros, C. (2005). Wildlife of the Box-Ironbark Country. CSIRO Publishing.

Victorian Government (2008). Living With Fire: Victoria's Bushfire Strategy. Government of Victoria, Melbourne.

Welsby, E. and Medlyn, L. (2011). *Heathcote Mosaic Burn Project: Burn Outcomes and Severity Mapping Report, Spring 2011.* Department of Sustainability and Environment, Melbourne.

Appendices

The documents listed below are available separately.

1. Box-Ironbark Experimental Mosaic Burning Project. Description of the coverage and patchiness of experimental planned burns
2. Box-Ironbark Experimental Mosaic Burning Project. Summary of data collected in pre-fire (baseline) surveys, including assessment of variability across landscape treatment groups

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