

Ecological effects of repeated low-intensity fire

in a mixed eucalypt foothill forest in
south-eastern Australia

summary report (1984–1999)



Research report no. 57

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December 2003

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Foreword

The vegetation, topography and climate found in south-eastern Australia combine to make the region one of the most wildfire-prone areas on earth. It has been estimated that, in the last one hundred years, two-thirds of all of Australia's bushfire-related human deaths have occurred in Victoria, as has more than half of all significant related property losses.

Conversely, over tens of thousands of years, naturally occurring fires have been highly significant in shaping the distribution and composition of much of the region's native flora and fauna. The arrival of humans in this part of the world is also considered to have had a more recent influence on these evolutionary processes.

Dealing with this paradox, the threat posed by fire to life and property and the relationship between fire regimes and biodiversity are arguably the key on-going issues confronting the managers of Victoria's parks and forests.

The severity of a bushfire depends on topography, weather and fuel conditions. Fuel is the only factor over which a land manager can exert some control. The strategic use of prescribed fire (under specified environmental and fire behaviour prescriptions), generally in spring or autumn, is the only practical method of reducing fuels over significant areas.

The strategic management of fuels, using prescribed fire, has been a key component of park and forest management in Victoria since the late 1950s – early 1960s.

In 1984, the then Victorian Forests Commission established in the Wombat State Forest north-west of Melbourne, what was to become one of the longest-running and most comprehensive studies of the ecological impacts of the repeated use of low-intensity prescribed burning yet undertaken. The location of the study, in part of Victoria's extensive mixed eucalypt foothill forests is also very significant, as this is where about half of all major wildfires occur, most of the prescribed burning is carried out and where significant rural-urban interface areas are found.

The study is quantitative, multidisciplinary and statistically based. Its multidisciplinary nature is one of its key features. On the same permanent plots understory flora, invertebrates, birds, bats, reptiles, terrestrial mammals, soil nutrition, tree growth and defect, local climate, fuel dynamics and fire behaviour are being studied, as are their interactions.

In 1992 a major review of the work to date was published, together with a section covering land management implications flowing from the interim results.

With the publication of this subsequent 'Summary Report' and associated 'Management Implications', and nine detailed individual reports (listed below), a major milestone in the project has now been reached.

One of the most significant messages flowing from the study is the demonstration that short-term fire effects research can be misleading, given the longevity of forest ecosystems.

Of further significance is the influence the research has had on the development of Victoria's *Code of Practice for Fire Management on Public Land*. This code, first published in 1995, brought together, in a balanced and unique way, all aspects of wildland fire management into a single document.

The fire effects research has also seen the development of considerably improved techniques for assessing forest fuel hazards. These techniques are now in widespread use across Victoria and interstate and have had a major impact on forest fuel management, fire suppression approaches and firefighter safety.

Similarly, the fire effects study has been a key element in the development of a process for determining ecologically sustainable fire regimes for given areas of park and forest and in the publication of *Guidelines and Procedures for Ecological Burning on Public Land in Victoria*. Again, the approaches in this area that have been developed in Victoria since 1998 have been adopted in a number of other Australian states.

In terms of the future, the current fire study areas and their treatments are being maintained and the research undertaken to date is set to become an important part of ongoing research that is soon to commence as part of the program of the recently established 'National Bushfire Cooperative Research Centre'.

In introducing this 'Summary Report', I would like to acknowledge the very considerable efforts of the scientists and technical officers who have contributed so constructively to this most significant project. I would like to also pay particular tribute to Dr Kevin Tolhurst, of the University of Melbourne, who has so ably kept the project 'on the rails' over the last two decades and who has coped so patiently with many funding and organisational vicissitudes.

In conclusion, the findings summarised here and reported in more detail in the various individual reports provide unique insights, not just for the managers of similar forest types but, in my view, much more widely. The work undertaken since 1984 in the Wombat State Forest is well placed to provide a most useful backdrop to the many future studies still required in other ecosystems. The Wombat work can, in many cases, obviate the need for time-consuming 'before and after' research, allowing new work to focus on operational burning regimes, not on expensive experimental burns.

Gary Morgan AFSM

CHIEF FIRE OFFICER

Department of Sustainability and Environment

2004

The 2003 series of Fire Research Reports developed from the Wombat fire effects study are:

- | No. | Title |
|-----|--|
| 57. | Ecological effects of repeated low-intensity fire in a mixed eucalypt foothill forest in south-eastern Australia - Summary report (1984-1999) - Department of Sustainability and Environment |
| 58. | Effects of repeated low-intensity fire on the understorey of a mixed eucalypt foothill forest in south-eastern Australia - K.G. Tolhurst |
| 59. | Effects of repeated low-intensity fire on fuel dynamics in a mixed eucalypt foothill forest in south-eastern Australia - K.G. Tolhurst & N. Kelly |
| 60. | Effects of repeated low-intensity fire on carbon, nitrogen and phosphorus in the soils of a mixed eucalypt foothill forest in south-eastern Australia - P. Hopmans |
| 61. | Effects of repeated low-intensity fire on the invertebrates of a mixed eucalypt foothill forest in south-eastern Australia - N. Collett & F. Neumann |
| 62. | Effects of repeated low-intensity fire on bird abundance in a mixed eucalypt foothill forest in south-eastern Australia - R. Loyn, R. Cunningham & C. Donnelly |
| 63. | Effects of repeated low-intensity fire on terrestrial mammal populations of a mixed eucalypt foothill forest in south-eastern Australia - M. Irvin, M. Westbrooke & M. Gibson |
| 64. | Effects of repeated low-intensity fire on insectivorous bat populations of a mixed eucalypt foothill forest in south-eastern Australia - M. Irvin, P. Prevett & M. Westbrooke |
| 65. | Effects of repeated low-intensity fire on reptile populations of a mixed eucalypt foothill forest in south-eastern Australia - M. Irvin, M. Westbrooke & M. Gibson |
| 66. | Effects of repeated low-intensity fire on tree growth and bark in a mixed eucalypt foothill forest in south-eastern Australia - K. Chatto, T. Bell & J. Kellas |

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Executive summary

By the early 1980s, and despite almost two decades of ecological research spanning the 1960s and '70s, no significant advance had been made in understanding the effects of repeated fuel reduction burning on the forests and heathlands of Victoria. The principal reason for this was a lack of statistically sound fire experiments. Research had focused primarily on a single fire event—usually with no pre-fire data—or on a small component of the forest ecosystem, such as the flora or fauna, or even on just a few species within these components. These problems were recognised and, in 1984, following the fourth Fire Ecology Symposium held at Monash University, the Forests Commission decided to establish a comprehensive fire ecology experiment to address these issues: the Wombat Fire Effects Study. This was the first, and remains the only, multidisciplinary study to investigate the ecological impacts of repeated low-intensity prescribed burning in mixed eucalypt foothill forests.

The Wombat Fire Effects Study was established in five areas in the Wombat State Forest north-west of Melbourne. These areas were called Fire Effects Study Areas (FESAs). Each FESA was subdivided into five Treatment Areas: Control, short-rotation spring, short-rotation autumn, long-rotation spring, long-rotation autumn. The aim of the study was to determine the impacts of repeated low-intensity prescribed fire on fauna, flora, soils, fuels, tree growth and defect development in foothill forests. Various methodologies were used to investigate the ecological impacts of fire on flora, bats, invertebrates, reptiles, terrestrial mammals, birds, fuels, bark thickness of trees and soil chemistry in each FESA.

The following is a summary of some of the major findings of this study.

Forest fuel dynamics

- Surface fine fuels in this forest have an average steady state level of 16 tonnes per hectare (t/ha), but seasonal variations may result in the fuel loads ranging from 9 t/ha to 26 t/ha.
- Surface fine fuel reaccumulates to within 90% of the long-unburnt state within four years of either spring or autumn burning at the fire intensities studied. The impact of prescribed burning on surface fuels is therefore relatively short-lived. (While the study reported here focused on surface fuels, other studies, which have focused on the overall fuel complex, have shown that the effects of prescribed burning may last 15–25 years.)
- The rate of reaccumulation of surface fine fuel is not significantly affected by the season of burn or the burn frequency.

Vegetation

- Over a 14-year period, no plant species were either lost or gained as a result of up to four successive spring fires or three successive autumn fires. The relative cover/abundance increased for about 30% and decreased for about 20% of the species.
- Short-rotation spring burning favoured Austral Bracken (*Pteridium esculentum*), herbs, geophytes and Poa (*Poa sieberiana*), and disadvantaged Forest Wire-grass (*Tetrarrhena juncea*), rushes, shrubs, small shrubs, legumes, trees, small trees and climbers. Long-rotation spring burning favoured legumes and, to some extent, Poa.
- Short-rotation autumn burning favoured only herbs, but disadvantaged Bracken, Wire-grass, Poa, rushes, legumes, small shrubs, trees and geophytes. Long-rotation autumn burning favoured Poa, rushes, legumes and trees by providing a regeneration opportunity followed by a growth period. Bracken, shrubs and small shrubs were the only plant groups that seemed to be disadvantaged by long-rotation autumn burning.
- There were subtle changes in the forest understorey in the absence of fire. These changes were only small on a year-to-year basis, but amount to significant changes over a period of a decade or more.

Invertebrates

- Neumann and Tolhurst (1991) found that no long-term changes in activity or abundance among invertebrates occurred following a single low-intensity prescribed burn in spring or autumn; Collembola and Diptera populations recovered to pre-burn levels within one year, and earthworms within three years.
- Three low-intensity prescribed spring burns within eight years had minimal impact on litter arthropods in foothill forest, although the effect of short-term rotational burns on the abundance of the Coleoptera and Diptera is less clear due to the significant change in activity levels also observed in the Control Treatment Areas.

Bats

- Bats range at least 1-5 km while foraging. Therefore, to study bat ecology on a landscape scale, a much larger area than the FESA Treatment Areas is necessary.
- The information gained from this research program was insufficient to predict the effects of repeated fuel reduction burning on bats. However, to date, the data suggests no effect in the differing Treatment Areas.

Reptiles

- None of the reptile species studied was favoured by a particular given burning treatment.
- Unburnt microhabitats (particularly logs, deep beds of leaf litter and areas frequently missed by low-intensity fire, such as gullies) provide important refuges and food, shelter and oviposition sites in the post-fire period. While some reptiles able to survive fire utilise alternative microhabitats, the rapid recovery of preferred microhabitat components is of major importance in the post-fire survival and recovery of reptiles.

Terrestrial mammals

- No particular burning treatment favoured either of the species studied, but habitat preferences were observed. Although *Antechinus agilis* (Brown Antechinus) and *Rattus fuscipes* (Bush Rat) have different habitat preferences, the survival and recovery of both species depend largely upon retention of unburnt habitat patches.
- Due to the timing of breeding and juvenile recruitment of *A. agilis*, autumn burns were found to have a greater impact on the population than spring burns. Population of *A. agilis* required at least 16 to 24 months to recover to pre-fire levels after autumn burns.
- Populations of *A. agilis* were significantly higher two to three years following spring and autumn burns than in long-unburnt areas.
- Populations of *R. fuscipes* took three breeding seasons (36 months) to recover when more than half of its preferred habitat was burnt during spring, but there was no recovery during the same period when the entire habitat was burnt.
- A single spring fire produced a greater impact than a single autumn fire on *R. fuscipes* populations in the first 12 months post-fire.

Birds

- The effects of burning in spring or autumn are quite similar (on subsequent bird abundance compared with unburnt areas) and there is no clear case for burning in one season and not the other. However, the beneficial effects of fire appear to be somewhat greater with autumn burns (probably because they are more intense) and the detrimental effects somewhat greater with spring burns (probably because birds are nesting at that time).
- Some birds respond positively to fire, and some species may depend on it in this forest type—e.g. White-winged Chough (*Corcorax melanorhamphos*), Spotted Quail-thrush (*Cinclosom punctatum*), Blue-winged Parrot (*Neophema chrysostoma*) and Red-browed Finch (*Neochmia temporalis*). Fuel reduction burning serves to provide ephemeral patches of bare-ground habitat at the landscape scale, though it does not mimic the patterns expected under a regime of occasional extensive wildfires.

- Birds may also need access to unburnt vegetation within their home ranges, especially in the immediate aftermath of a fire as noted for mammals by Newsome et al. (1975), Catling and Newsome (1981), Humphries (1994), Tolhurst (1996a) and Friend (1993). The present study provides no information on the ability of birds to persist in areas subject to such treatment.

Soils: carbon, nitrogen and phosphorus

- There was a significant decline in carbon and nitrogen in surface soils due to repeated low-intensity fires at three-yearly intervals. Furthermore, there was some evidence of a change in the quality of organic matter. However, there was little, if any, change in carbon and nitrogen from less-frequent fires (ten-year intervals), indicating that this strategy can be expected to maintain soil organic matter in the long term.
- Observed changes in extractable phosphorus were not attributable to fire.
- In the long term, low-intensity fires occurring in these foothill forests at less than a ten-year frequency can be expected to lead to a decline in soil organic matter and soil fertility.

Tree growth, mortality and bark thickness

- Bark loss due to burning was shown to depend significantly on burning treatment. Both season and frequency of fires were important in the loss and recovery of bark after fire. Bark loss was found to be greater as a result of autumn burning than spring burning. This pattern was found to be strongly related to Soil Dryness Index and poorly related to fire intensity.
- Tree mortality rates during the study were relatively low and not attributable to any one particular treatment. There was considerable variation between FESAs, with fewer deaths recorded for areas with relatively 'cooler' burns (e.g. Blakeville) and more deaths at Treatment Areas with higher-intensity burns that often resulted in crown scorch (e.g. Kangaroo Creek).

Concluding comments

The Wombat Fire Effects Study has demonstrated that some of the conclusions drawn from short-term studies or space-for-time¹ studies can be misleading. Given the longevity of a forest ecosystem, studies undertaken in a period of less than ten years may erroneously attribute trends or variations to unlinked causes, or overlook underlying trends altogether. Longitudinal studies, such as this one, provide a better insight into the dynamics of a forest ecosystem.

This report is a summary of the results found to date which incorporate data presented in Tolhurst and Flinn (1992). Since 1992, internal progress reports have been written, several papers published in peer-reviewed journals and research findings reported at a number of different forums. A complete list of publications related to the Wombat Fire Effects Study is provided in Appendix A.

The future

Although no formal measurements have been taken or data collected since 1999, it is expected that the five FESAs and their treatments will be maintained indefinitely. Four of the five weather stations have been maintained and will continue to be maintained as long as possible. Continued application of the treatments to the FESAs is currently included in the Department of Sustainability and Environment's Fire Protection Plan for this area. The whole study will become an important part of the research program of the National Bushfire Cooperative Research Centre as one of the nationally significant long-term fire ecology research studies in Australia.

¹ Studies made at a single point in time at a number of different locations with different histories that are combined to represent the succession of changes over time at the one location.

Management implications

These management implications are framed in the context of the results from experimental low-intensity prescribed burning in the mixed eucalypt foothill forests of west-central Victoria. They are framed on the basis that the maintenance of biological diversity and the rapid recovery of environmental elements following the use of prescribed fire are desired management aims. Experimental fire intensities were generally less than 500 kW/m with little canopy scorch. The size of the treatment areas averaged around 10 to 15 ha. The results of this study come from the 14-year period, 1985 to 1999.

Ecological implications

Frequency

Main findings

- Three- to five-year fire frequencies have not resulted in any losses of fauna or flora species, but there is a significant change in the relative abundances of species.
- Rapid recovery of invertebrate, bird, reptile and terrestrial mammal populations after burning is dependent on there being unburnt patches scattered throughout the burnt area. At least 40% unburnt in more structurally complex vegetation, such as gullies, and at least 10% in unburnt patches in structurally simpler slope vegetation are indicated.
- Soil carbon and nitrogen and the quality of soil organic matter significantly decrease with burning frequencies of three to five years. The data indicate that extractable phosphorus may also decline with these high burning frequencies, but it may take several rotations before the decline becomes statistically significant.
- Surface litter cover had reached about 90% of the long-term levels within two to four years after burning. Biological decomposition processes were fully functioning again three to four years after burning.

Implications

Burning on a three- to five-year cycle over a long period of time will most likely result in irreversible changes to the structure, fertility and relative abundance of fauna and flora species. Two or three low-intensity fires in quick succession are unlikely to cause any irreversible compositional or structural changes in forest such as was studied here, provided they are given an extended period of time over which to recover after such a series of fires.

Implication 1

Short-term (less than five years) rotational burning over long periods of time is ecologically undesirable. Prescribed burning strategies, where short-term rotational burning is planned, should be reviewed and alternatives considered.

In the short term, recovery will be much more rapid where most of the overstorey cover and scattered unburnt patches are retained across the burnt area, than in areas which are burnt more severely.

Implication 2

Low-intensity prescribed fires in foothill forests should generally aim to result in less than 30% canopy scorch and to retain scattered unburnt patches across slopes, combining to be at least 10% of the slope area. At least 40% of gully vegetation should remain unburnt unless this does not achieve a specific burn objective.

Burning on a 10-year cycle or longer allows for full recovery of the fauna and flora and gives an opportunity for an increase in structural habitat diversity. Full understorey structural diversity takes longer than 20 years to develop.

An average interfire period of at least 10 years is needed to allow soil nutrients (nitrogen, carbon and extractable phosphorus) to replenish after burning. Burning more frequently than every 10 years will result in a gradual decline in soil fertility and hence site productivity.

Long-unburnt areas do not provide a full range of habitats. The level of food and shelter in long-unburnt areas can be significantly less than in younger understoreys. Therefore there is no optimum age-class for biological diversity or sustainability, but there is a need for a functionally connected patchwork of age-classes across the forest.

Implication 3

There needs to be a range of age-classes across a forest area in order to meet all biodiversity needs. Inter-fire periods of less than 10 years should be avoided in foothill forests to maintain forest fertility and structural habitat values.

Season

Main findings

- Spring burning favoured such plants as Bracken, herbs, geophytes and Poa, while autumn burning favoured plants like the herbs, legumes, trees, rushes and Poa. Herbs and Poa were therefore favoured by burning in either season. Generally speaking, spring burning favoured resprouting plants whilst autumn burning tended to favour the reseeding plants. This is a function of the growing conditions at the time of the burns and subsequent conditions during the recovery period.
- Decomposer invertebrates such as springtails (Collembola), fly larvae (Diptera) and earthworms were affected more by burning in spring, when they were most active, than burning in autumn. However, all populations had recovered within three years of a fire.
- Autumn burning had a greater impact on Bush Rat (*Rattus fuscipes*) population levels than spring burning because of the greater impact of fire on the species' gully habitat. Autumn burning also had a greater impact on Brown Antechinus (*Antechinus agilis*) populations than spring burning because of the impact that fires had on juveniles.
- The effect of burning in spring or autumn on birds was quite similar with no clear case for burning in one season and not in the other. In the longer term there may be some beneficial effect of autumn burning because of greater shrub development and, potentially, there may be short-term impacts of spring burning on breeding success; but these differences were not significant in this study.
- Nutrient losses were generally similar from both spring and autumn burning. Fire frequency had a more important impact than fire season. In the longer term, burning in autumn may have the advantage of promoting greater legume germination and hence nitrogen fixation.
- Bark loss from tree boles was greater with autumn burning than spring burning due to the drier burning conditions. This may reduce habitat for invertebrates and bats using tree boles, but this was not detected in this study, probably due to the incomplete consumption of bark and access to unburnt trees close to the burning treatments.

Implications

There is no clear indication that prescribed burning should occur in spring or in autumn. Neither season is without its advantages and disadvantages. What is clearer, is that regularly burning in the same season would result in a tendency to push the ecosystem in a particular direction. Across a forest, it would seem prudent and ecologically sensible for fires to occur across a range of seasons ranging from spring to summer to autumn.

Implication 4

Any particular area should be burnt under a range of seasonal conditions to maximise biodiversity within a specific area, and across the forest.

Intensity

Main findings

The range of intensities in this study was deliberately kept to an average of less than 500 kW/m. The range of potential fire intensities for a forest such as the one studied here is probably up to about 70 000 kW/m and so the results here represent the bottom 1% of the potential intensity range.

Implications

The rate of recovery observed in this study was, in large part, a result of the low-intensity nature of the fires used. Higher-intensity fires would tend to remove a much greater proportion of the canopy, a much greater part of the bark on the tree boles, would remove more of the woody debris from the forest floor, induce greater soil heating and plant death and cause higher fauna mortality.

Implication 5

The recovery of fauna, flora and soils after low-intensity fires is likely to be much more rapid than after a higher-intensity fire. Low-intensity fires are more subtle in their effect on forest structure and it takes a sequence of low-intensity fires to potentially change the structure of the forest rather than a single fire event.

Fire size and patchiness

The fire treatment areas in this study varied from about 4 ha to about 35 ha and averaged 10–15 ha. Therefore, whilst the areas are more than just experimental plots, they are relatively small compared with the average size of a fuel reduction burn in Victorian foothill forests. Some of the results of this study are affected by the fact that the burnt areas adjoined a large unburnt area in a forest of some 30 000 ha. However, at a small scale, the results of this study are very valuable.

Main findings

- Patches as small as 1 m², when they are scattered across the burnt area, were found to provide significant habitat for birds, reptiles, invertebrates and terrestrial mammals.
- Unburnt patches were, more often than not, associated with discontinuities on the forest floor such as fallen logs, stumps, mounds, holes and rocks, with fallen logs being the most important.
- More unburnt patches were found in damp gullies than on the drier slopes and ridges.
- Where unburnt patches were present in sufficient quantity, the recovery of the fauna was markedly faster than where the burn coverage was almost complete.

Implications

Implication 6

Weather conditions and ignition patterns need to be selected that allow natural barriers to fire spread to be effective in leaving patches.

Implication 7

The patchiness of fires needs to be mapped and quantified. A technique to do this routinely and consistently needs to be developed.

Implication 8

Fallen timber on the forest floor should be protected. Removal of fallen timber for firewood should be managed to ensure sufficient timber for wildlife habitat.

Implication 9

Burning when seasonal dryness is such that fallen logs are unlikely to actually burn away, should be chosen in preference to when there is a high likelihood of total consumption, e.g. after a prolonged rain-free period.

Implication 10

Lighting patterns should be used that leave the gully vegetation predominantly unburnt.

Implication 11

Burn objectives for a specific burn need to be formulated in the context of the state of the rest of the forest landscape.

Implication 12

Burning prescriptions need to be tailored to achieve the stated burn objectives, but must be broad enough to be achievable for the site of the burn.

Protection Implications

Main findings

- Surface fine fuel reaccumulates to at least 90% of the long-unburnt state in just two to four years after a low-intensity prescribed burn. Once this level is reached, it remains there for at least 60 years after burning.
- There are significant year-to-year variations in fine fuel levels as a result of seasonal conditions affecting both the rate of litterfall and the rate of biological decomposition. This degree of variability is approximately 50% above and below the long-term average.
- There was also a significant difference in surface fine fuel levels between spring and autumn, averaging about 1.5 t/ha. This seasonal difference correlated to the period of rapid litterfall and dry conditions reducing the amount of biological decomposition. It is expected that this difference would be greater in severe drought years.
- There were not only year-to-year variations and long-term trends, but also decadal-scale trends. Within the variation in surface fine fuels observed, the year-to-year variations were found not to be totally random. There was a general increase in surface litter over the period of the study that could have been related to a broader seasonal trend or to forest stand age and development.
- Elevated fuels continue to increase for more than 10 years after burning. Bracken fuels develop first, followed by Wire-grass and then the shrubs. The amount of elevated fuel in the forest is determined by, not only the time since fire, but also the season and intensity of the fires. Spring burning generally encourages Bracken. Autumn burning generally encourages shrub development but tended to reduce Wire-grass.
- Bark on tree boles was reduced more by autumn burning than spring burning because of the seasonally drier conditions shown by the Soil Dryness Index.

Implications

Implication 13

Models of surface and elevated fine fuel accumulation give no more than a broad indication of fuel accumulation trends. Field assessment of fine fuels must be undertaken before critical fire management decisions can be made.

Implication 14

Fine fuel levels in drought years can be significantly higher than in normal or wet years. Planning for fire preparedness should not only be based on expected fire weather, but also on changes in fuel amount and availability (for which 'Drought Index' is a reasonable surrogate).

Whilst the surface fine fuel complex may reaccumulate very quickly, the overall fuel hazard levels are likely to continue to rise for 20 years or more due to accumulations in bark and elevated fuels.

Implication 15

The impact of prescribed burning for hazard reduction must be assessed in terms of the overall fuel hazard levels, not just the surface fine fuel levels.

Autumn burning will generally lead to a greater reduction in fine surface, elevated and bark fuels, but may lead to greater elevated fuels in the longer term. This will be dependant on the type of understorey vegetation present and the time since burning.

Implication 16

Prescribed burning for hazard reduction must be assessed on the basis of the type of species present in the understorey and the fuel hazard outcome required, not just on seasonal and time-since-fire factors.

Implication 17

To achieve ecological management objectives, the interfire period for a particular site as well as the time-since-fire pattern across the landscape need to be variable rather than regular, approaching a more random distribution of fire across the landscape.

Implication 18

The location of Fuel Management Zones 1 and 2 needs to be monitored, and reviewed where necessary, to ensure that the fuel levels required to afford the level of protection desired can be achieved with the resources available and in a manner that is compatible with other stated land management objectives.

Management planning and operational implications

Main findings

- The results of the Wombat Fire Effect Study indicate that there is a wide range of viable fire management options available to meet fire protection and ecological management objectives.
- Strategically based prescribed burning in spring or autumn is viable. The choice of season will depend on what is wanted to be achieved and what the operational constraints will allow.
- Two or three low-intensity fires at frequencies of three to five years do not produce irreversible change. However, if fires were conducted at this frequency on an ongoing basis, site fertility would decline and permanent changes to the fauna and flora could be expected.

Implications

Lighting pattern and the weather associated with prescribed burning is critical to achieving the desired management outcomes. The importance of the skill and knowledge of the people managing and lighting the fires should not be underestimated. The date of the burn, the time of day a fire is ignited and the lighting pattern are three important factors of prescribed burning that impact significantly on the outcomes achieved.

Implication 19

The skill and knowledge of the people conducting prescribed burning need to be underpinned by a high level of training and competency assessment.

Some important qualifications have been placed on the interpretation of these results and these must not be forgotten when using this information. Some of these qualifications relate to the size of the areas burnt, some to the intensity and patchiness of the fires, some to the amount of fallen timber and other surface discontinuities and some to the age and structure of the forest.

Prescribed burning should not be considered to be the only fire occurring in the landscape. When planning prescribed burns at a landscape scale, the expected impact of wildfires should also be included. Wildfires provide a greater proportion of the larger, higher-intensity, summer fires that make up the landscape level fire regime.

Implication 20

The occurrence of wildfires should be factored into the planning of fire regimes across the landscape, especially in regard to the distribution of prescribed fires of higher intensity and summer wildfires.

This research has shown that the foothill forests are very tolerant and resilient to fire, providing a wide range of management options. However, the research has also shown that there are limits to the resilience of the forest and these limits must not be crossed if potentially irreversible biodiversity changes caused by inappropriate fire regimes are to be avoided.

Implication 21

There is a need for a simpler and clearer monitoring methodology to assess whether or not burn objectives and broader management objectives are being met.

Implication 22

There is a need to have explicitly stated management priorities across the Department of Sustainability and Environment's businesses to better put prescribed burning in context. Similarly, it is important that adequate resources are made available during the limited period of opportunity to conduct prescribed burns.

Implication 23

It is important that the roles prescribed fire can play in the protection of life and property, and in the maintenance of biodiversity, is promulgated more widely in the community.

Introduction

Background

Prescribed fire has been used extensively for bushfire protection, silvicultural purposes and, increasingly, for ecological management across public land in Victoria for about 40 years. Between 1985 and 1995, an annual average of 131 000 ha of the State's total public land estate of 8 million ha was prescription burnt. Suppression of unplanned fires in Victoria has been given high priority since the devastating 1939 fires that burnt about 1.4 million ha of public land and killed 71 people (Stretton 1939).

However, there exists in the wider community a range of views about the effectiveness of prescribed fire in reducing the damage done by wildfires, and about the impact of regular low-intensity fires on biodiversity. Members of the public generally object to prescribed burning due to an aversion to smoke in the atmosphere and a dislike of the sight of recently burnt areas. Conversely, and as recent events have demonstrated, many rural landholders consider the Department of Sustainability and Environment (DSE) to be neglecting its duty of preventing bushfires from burning out of public land and damaging private property, by not doing enough prescribed burning. DSE has legislative requirements to protect human life and property from bushfires, maintain biodiversity on public land and facilitate the production from public land of goods and services, including the yield of water and timber and the provision of tourist and visitor destinations. The responsibilities of DSE for fire management in Victoria are outlined in the *Code of Practice for Fire Management on Public Land* (CNR 1995).

The ecological effects of fire in Victorian forest ecosystems have been researched since the 1920s; the main effort being made in the late 1950s. Despite the longevity of interest in the ecological effects of fire, fire ecology research in Victoria has been limited to short-term projects conducted by tertiary institutions and some longer-term research conducted by the Museum of Victoria and the State department responsible for park and forest management (now the Department of Sustainability and Environment). These projects have often been opportunistic and have generally focused on the effects of a single fire, or the effects of fire on a single part of the ecosystem (e.g. individual species, soil nutrients, etc.). According to Tolhurst et al. (1992a), 'few of these studies included pre-fire measurements, and fewer still had accurate records of fire behaviour and climatic variables.' They noted that the main cause for the lack of progress in furthering and understanding the relationship between fire regimes and biodiversity has been a lack of long-term experimental research.

At a symposium held at Monash University in September 1983, discussions and presentations highlighted that, despite much work, little progress had been made in understanding the effects of repeated fuel reduction burning on Victorian forest ecosystems (Ealey 1984). In response, in January 1984, the then Forests Commission of Victoria initiated a multidisciplinary research program to study the ecological effects of repeated low-intensity prescribed burning in mixed eucalypt foothill forests. Five permanent Fire Effects Study Areas (FESAs) were established in the Wombat State Forest, approximately 80 km north-west of Melbourne.

Study areas and experimental design

A full description of the FESAs and experimental design for the Wombat Fire Effects Study is published in Tolhurst and Flinn (1992). Individual FESAs were around 15 ha in size.

The five Fire Effects Study Areas were established at Barkstead, Blakeville, Burnt Bridge, Kangaroo Creek and Musk Creek (Figure 1.1).

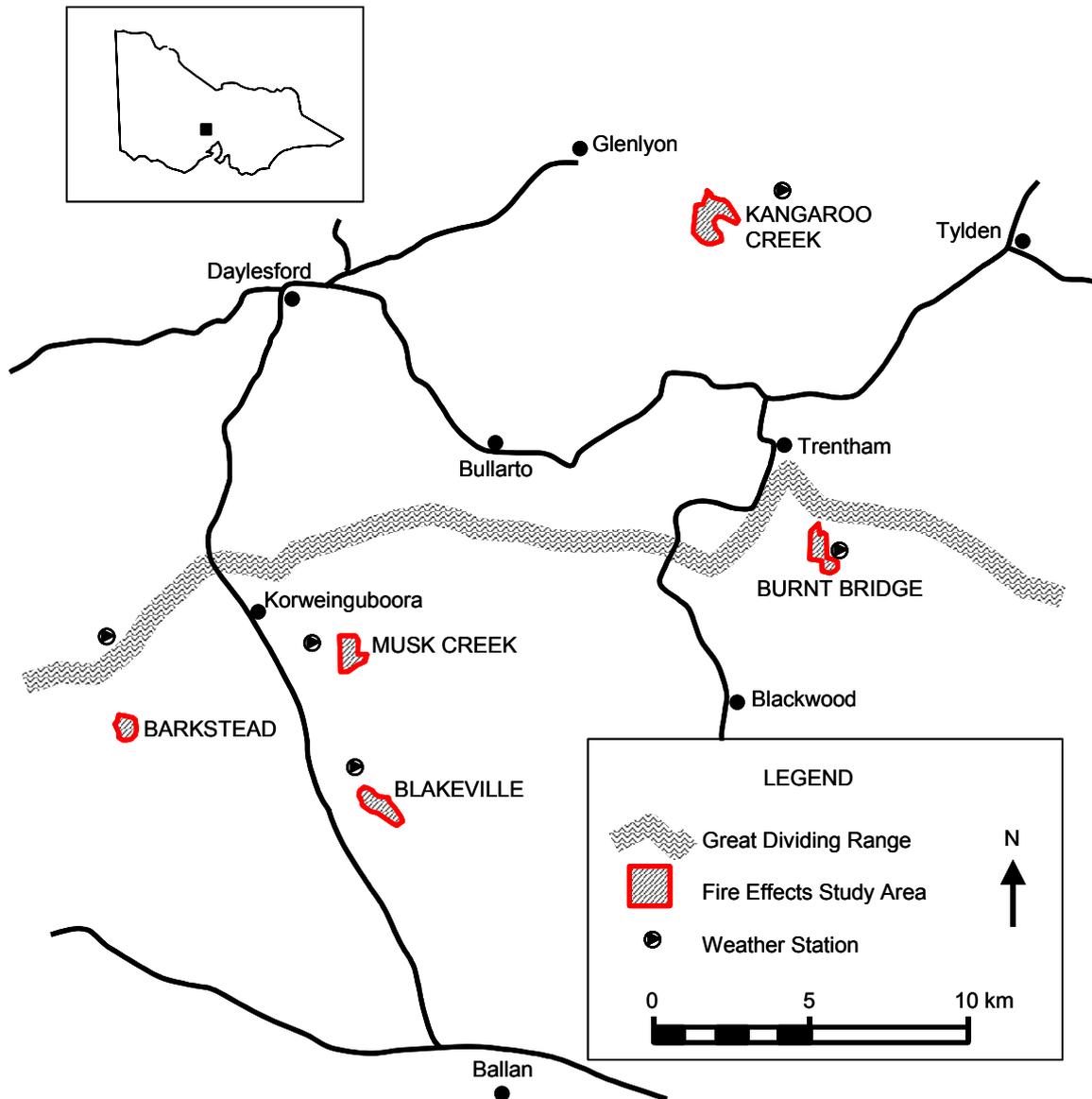


Figure 1.1 Location of the five Fire Effects Study Areas used in the Wombat Fire Effects Study

Within each FESA, five Treatment Areas were established: Control (C), short-rotation spring (Spring 3 = S3), long-rotation spring (Spring 10 = S10), short-rotation autumn (Autumn 3 = A3) and long-rotation autumn (Autumn 10 = A10). The treatment for each Treatment Area was randomly selected. For the spring and autumn treatments, there were two rotations: as often as possible—about every three years—(short-rotation) and once every ten years (long-rotation). Although these were the prescribed rotation lengths, the actual rotation lengths varied slightly due to weather conditions and operational constraints. In this report, the first rotation is annotated as 1R, second rotation is shown as 2R, third rotation is shown as 3R and so on. The year of each prescribed burn is shown in Table 1.1.

Table 1.1 Burning history of the Wombat Fire Effects Study Areas

Fire Effects Study Area and treatment	Burning history				
	Previously burned	1R	2R	3R	4R
Barkstead					
Control (C)	1931				
Spring 3 (S3)	1931	1985	1988	1991	1994
Spring 10 (S10)	1931	1985	1994		
Autumn 3 (A3)	1931	1987	1991	1997	
Autumn 10 (A10)	1931	1987	1997		
Blakeville					
Control (C)	1935				
Spring 3 (S3)	1935	1985	1988	1991	1994
Spring 10 (S10)	1935	1985	1994		
Autumn 3 (A3)	1935	1987	1992	1997	
Autumn 10 (A10)	1935	1987	1997		
Burnt Bridge					
Control (C)	1953				
Spring 3 (S3)	1953	1986	1988	1991	1995
Spring 10 (S10)	1953	1986	1994		
Autumn 3 (A3)	1953	1987	1991		
Autumn 10 (A10)	1953	1987			
Kangaroo Creek					
Control (C)	1944				
Spring 3 (S3)	1944	1985	1988	1991	1994
Spring 10 (S10)	1944	1985	1994		
Autumn 3 (A3)	1944	1987	1992	1997	
Autumn 10 (A10)	1944	1987	1997		
Musk Creek					
Control (C)	1974				
Spring 3 (S3)	1974	1986	1988	1991	1994
Spring 10 (S10)	1974	1986	1994		
Autumn 3 (A3)	1974	1987	1991	1992	2002
Autumn 10 (A10)	1974	1987			

Notes:

Treatment Areas: Spring 3 = short-rotation spring; Spring 10 = long-rotation spring; Autumn 3 = short-rotation autumn; Autumn 10 = long-rotation autumn

Burning history: 1R = first rotation, 2R = second rotation, etc.

The prescribed fires were all 'low-intensity', that is, they were less than 500 kW/m on average (*sensu* Cheney 1981). However, fire intensity at any one point of the area burnt may be a factor of 10 greater or less than the average. Table 1.2 provides a summary of the fire behaviour statistics. There were no great differences in fire behaviour between the spring and autumn burns, or for different rotations of burns. The main differences seemed to be a slightly greater rate of spread with spring fires that lead to a slight increase in the fireline intensity. Other fire behaviour parameters were similar. Another difference was the relatively low surface soil temperature in the 1R spring burns compared with later rotations of spring burns and the autumn burns and the relatively low proportion of area burnt in the 3R autumn burns (73%) compared with other burns (about 90%).

Table 1.2 Summary of average fire behaviour associated with each season and rotation of burning in the Wombat Fire Effects Study

Fire behaviour parameter	Spr 1R (n=10)	Spr 2R (n=10)	Spr 3R (n=5)	Spr 4R (n=5)	Aut 1R (n=10)	Aut 2R (n=8)	Aut 3R (n=4)	Overall (n=52)
Mean flame height (m)	0.5	0.3	0.3	0.3	0.4	0.3	0.2	0.3
Mean rate of spread (m/h)	57	70	51	66	30	31	45	50
Mean scorch height (m)	5.8	8.6	4.9	5.7	8.2	4.5	5.2	6.5
Mean fire intensity (kW/m)	149	324	227	377	162	107	192	214
Max surface soil temp. (°C)	324	591	608	608	582	558	550	532
Mean surface soil temp. (°C)	157	457	437	473	387	328	291	353
Mean area burnt (%)	90	93	95	94	97	91	73	92

Rainfall across the Wombat State Forest has a distinct winter peak: June being the wettest month and March being the driest (Fig. 1.2). During the 14 years of rainfall measurements, the average rainfall for the study areas was 881 mm, ranging from 830 mm at the Kangaroo Creek FESA to 907 mm at the Burnt Bridge FESA.

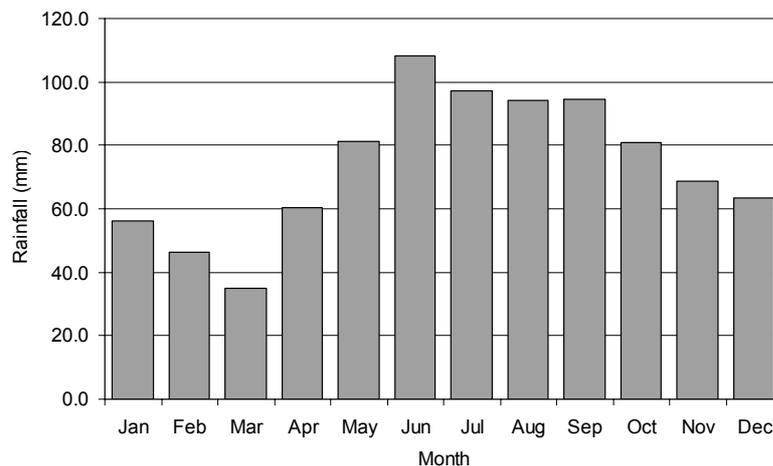


Figure 1.2 Average monthly rainfall for the five Fire Effects Study Areas across the Wombat State Forest. Fourteen year average (1986–99)

The year-to-year variation in rainfall across the Wombat State Forest for the period 1986–99 ranged from 638 mm in 1994 to 1124 mm in 1993 (Fig. 1.3). This represents a variation of about 250 mm, (about 30%) above or below the 14-year average of 881 mm. It is worth noting that there was a generally drier trend in the six-year period 1994–99, and this trend continued until at least 2003.

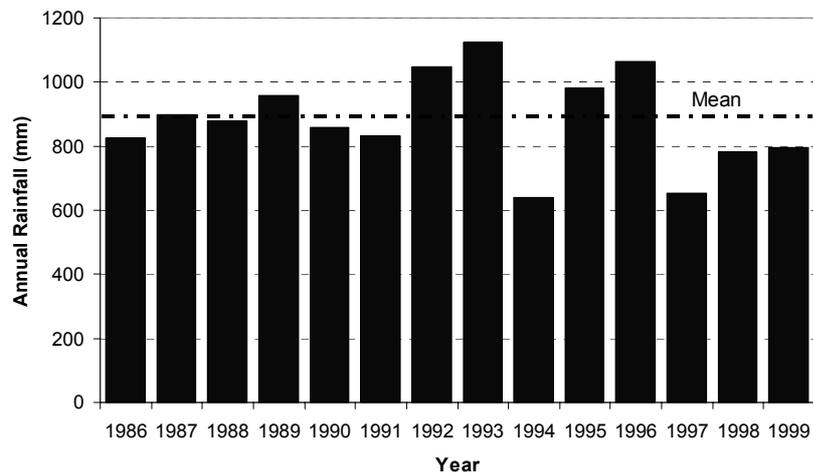


Figure 1.3 Average annual rainfall for the five Fire Effects Study Areas across the Wombat State Forest, 1986–99. Fourteen year mean (881 mm) is shown.

Objectives

The broad objectives of the Wombat Fire Effects Study were to assess and describe the effects of repeated spring and autumn low-intensity fires on:

- flora, fauna and soils
- the functional processes of ecosystems in mixed eucalypt foothill forest
- the short-term and long-term stability of such ecosystems.

The study was divided into ten project areas, and each project area was developed with its own objectives. Although each project was researched individually, it was obvious throughout the study that, generally, the findings from different projects were complementary.

This report is a summary of a more detailed set of papers. Each section of this report has been individually authored and contains the main findings of different components of this study. For greater details about methods and results, refer to the individual reports (available at the website www.dse.vic.gov.au/fires — fire management; fire research).

Fuel dynamics

Kevin Tolhurst and Natalie Kelly

Introduction

All the biomass produced in a forest is potentially fuel that can be consumed in fires. However, it is also an important component of the habitat for many invertebrates, mammals, birds, reptiles and fungi. This report concentrates on aspects of the dead fine fuel, that is, the dead leaves, bark and twigs which are consumed in the flaming zone of a fire and which are therefore most important to the rate of spread of the fire and the flame height.

Of particular interest is the dynamics of fuel accumulation. Fuel accumulation is the build-up of fuel resulting from the balance between litterfall and chemical, physical and biological decomposition. There are times when the rate of litterfall is high due to drought, scorching, disease or insect attack. There are also times when the rate of decomposition can be high or low depending on the seasonal and annual moisture conditions of the fuel and the abundance and activity of decomposing fungi and invertebrates. Given the range of factors affecting fuel dynamics, year-to-year and season-to-season fuel loads are rarely constant.

The accumulation rate of fuel is generally described by the negative exponential model described by Olson (1963). This model takes the form:

$$X_t = X_{ss} (1 - e^{-kt})$$

Where:

X_t is the amount of fuel accumulated at time t

X_{ss} is the amount of fuel accumulated when the fuel bed is in equilibrium (steady state)

k is the decomposition rate

t is the time value in years that the fuel has had to accumulate—usually the time since the last fire.

This model assumes that there is no fuel at time $t = 0$, and that the fuel will reach a steady state where the rate of fuel accession is equal to the rate of decomposition.

Litterfall rates depend on the seasonal conditions, the productivity of the site and the amount of biomass growing on a site. Birk (1979) summarized the typical eucalypt litterfall rates as ranging between 6 and 8 t/ha/yr for tall open forest, 2 and 4 t/ha/yr for open forest and between 1 and 1.5 t/ha/yr for low open forest. However, these averages may be altered by local rainfall or site exposure. Both Fox et al. (1979) and Turnbull and Madden (1983) found that annual litterfall was a function of overstorey basal area while Hatch (1955) and Walker (1979) related annual litterfall to canopy density. Both basal area and canopy density are functions of site conditions as well as disturbance factors such as timber harvesting and disease. Litterfall rates can vary significantly from the long-term average. Pook (1985) reported almost complete canopy litterfall for some eucalypt species during a period of drought. This amounts to about twice the annual average. Simmons and Adams (1986) reported a similar finding during the 1982–83 drought in Victoria. Therefore the value of X_{ss} , the steady state fuel load, is only an average and actual fuel levels in any one particular year could be a factor of two or more greater or less than the average.

All previous studies have shown a sinusoidal pattern of litterfall with a peak in the summer - early autumn period. Seasonal litterfall is correlated to maximum daily temperature (Lee & Correll 1978; Turnbull & Madden 1983) and daily levels of solar radiation (Pressland 1982). The total amount of litterfall for a season is correlated to soil moisture conditions (Pressland 1982). This means that the amount of surface fine fuel in a forest at the end of the fire season can be significantly greater than that at the beginning of the fire season, even in an average year.

The other aspect of surface fine fuel accumulation is the rate of decomposition. This, too, changes with the season of the year and with the time since fire. Water-soluble nutrients, sugars and starch are leached or consumed in the first six months following leaf fall. Decomposition of the more resistant components of the fuel, such as cellulose, depends on having a suitable microenvironment. Woods and Raison (1983) found that the rate of decomposition is greatest when the fuel moisture content exceeds 60% and air temperatures are high. The litterbed structure and the overstorey cover will affect how long these warm and humid conditions last and hence how quickly the litter decomposes. Fox et al. (1979) and Woods et al. (1983) suggest that it may take three to four years after a fire before conditions are suitable for optimum decomposition of the litter. Litter decomposition is therefore disrupted for a period after fire due to a reduction in to the populations of decomposers and the lack of an environment conducive to microbial and arthropod activity in the litterbed.

Tolhurst et al. 1992a presented the first progress report of this study. They reported that humus was significantly affected by spring and autumn burning and that litter and twig material, whilst reduced by approximately 60-70%, had reaccumulated to be not significantly different from the long-unburnt treatment within two to three years of being burnt. The rate of litter reaccumulation was about 2 t/ha/yr. It was noted that there was a significant difference between surface fine fuel levels in spring and autumn, with autumn measurements being about 1.5 t/ha greater than those in spring.

This chapter concentrates on the changes in surface fine fuel due to seasonal and year-to-year variations and the time since fire. All the fires in this study were low intensity (less than 500 kW/m) and were conducted in either spring or autumn, i.e. on the 'shoulders' of the fire season. Unlike most other published work on fine fuels, this study covers a range of fuel ages from 0 to 68 years. This provides a better basis for looking at long-term trends than studies that investigated only 20 or 30 year-old fuels. Fuels thicker than 25 mm, bark on tree boles, canopy fuels and shrub and elevated fuels are not considered in this chapter.

Research results

A total of 6810 fuel samples were taken over the period of the study, 15 from each Treatment Area on each occasion. Table 2.1 provides a summary of the range of values for each of the factors measured at each Treatment Area. These values are relevant in the context of the mixed eucalypt foothill forests of Wombat State Forest. It would be expected that higher surface fine fuel loads would be found in more productive forests, such as the mountain forests.

In the long-unburnt Treatment Areas, the total fine fuel loads ranged between 9.4 t/ha and 26.4 t/ha with an overall mean of 16.0 t/ha. However, some individual samples were three times greater than the maximum average, while others were approximately ten less than the minimum average. This variation in fine fuel across the study areas indicates a mosaic of fuel loads. Typically, the patches of very low fuel load were associated with mounds around tree stumps or ant mounds and the very high fuel loads were associated with depressions caused by stump holes or a build-up behind a log on a steep slope.

The greatest proportion of fine fuel consisted of leaf, bark and twig litter. The partially decomposed and fragmented fuel (humus) made up approximately 25% of the total fine fuel in the forest.

The depth of the fine fuel litterbed in the long-unburnt Treatment Areas ranged from 1.6 cm to 5.8 cm, and averaged 3.3 cm. The greatest litterbed depth measured during the study for a treatment was 11.6 cm in an area of dense Forest Wire-grass.

Table 2.1 Summary of all 454 fuel measurements (6810 samples) taken over a 14-year period from the Wombat Fire Effects Study Areas. Numbers in brackets represent the 119 measurements (1785 samples) taken from the long-unburnt (Control) Treatment Areas.

Fuel parameters	Mean	Standard Deviation	Maximum	Minimum
Humus (t/ha)	3.7 (3.9)	1.2 (1.2)	8.7 (8.7)	0.9 (1.6)
Litter (t/ha)	10.1 (12.1)	3.3 (1.8)	22.9 (17.7)	1.9 (7.8)
Twigs (t/ha)	2.0 (2.3)	0.9 (0.7)	7.7 (5.1)	0.4 (0.6)
Total fine fuel and twigs (t/ha)	15.8 (18.2)	4.3 (2.7)	30.3 (24.9)	5.0 (11.3)
Depth (cm)	2.8 (3.3)	0.9 (0.7)	11.6 (5.8)	0.8 (1.6)

General trends with time since fire

There was a negative exponential trend in litter accumulation with time since fire as shown in Figure 2.1. This trend suggests that litter fuels approach their equilibrium levels within four years after fire, given the fire intensities studied. The equilibrium litter load for this type of forest was about 12 t/ha, although there is a significant level of variation around this mean, most of which falls within ± 6 t/ha of the mean. The regression model that fits this data is:

$$L = 8.28(1.47 - e^{-0.506t})$$

$$(r^2 = 0.64, n = 454)$$

Where:

L is the litter fine fuel (in t/ha)

t is the time since last fire (in years)

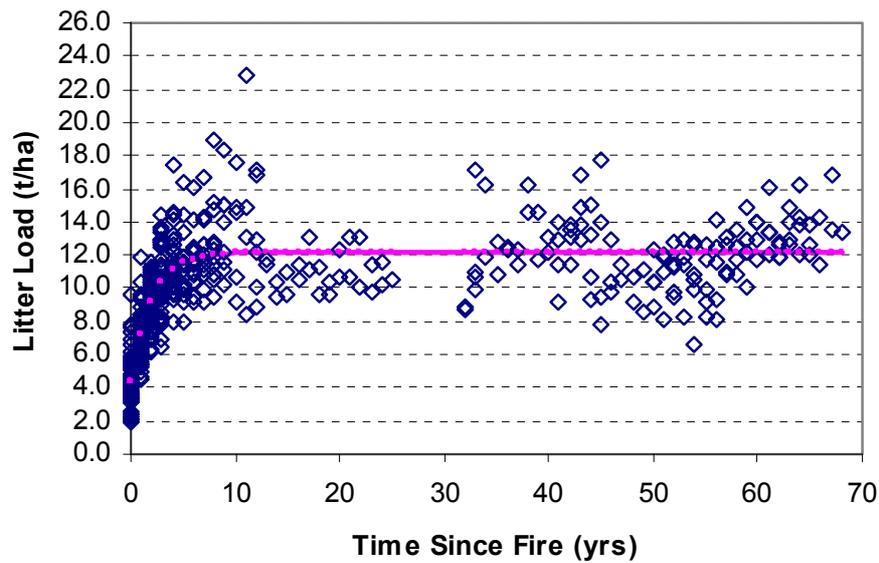


Figure 2.1 Litter fuel load (t/ha) with time since fire (years) for all treatments from all Fire Effects Study Areas

Whilst the equilibrium litter load for this study was about 12 t/ha, the total surface fine (less than 6 mm thick) fuel load was about 16 t/ha, and the total of surface fuel load less than 25 mm thick was about 18.5 t/ha. Litter fuels dominate the general pattern of accumulation and the variation about the mean, and so the graphs of total fine fuel and fuel less than 25 mm show similar patterns to the litter fuel load shown in Figure 2.1.

There was no significant long-term trend in the humus or twig fuels as can be seen in Figures 2.2 and 2.3. Humus loads were about 4.1 t/ha with most of the observations within ± 2 t/ha of this mean. Twig loads averaged about 2.4 t/ha with most of the observations within ± 1 t/ha of this mean. Both the humus and twigs showed a negative exponential trend soon after burning in a similar but less pronounced way to the litter.

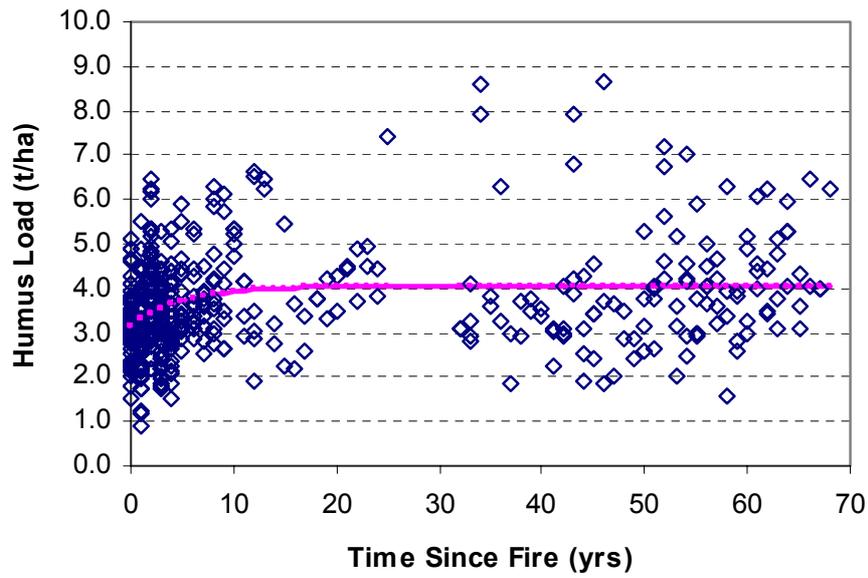


Figure 2.2 Humus fuel load (t/ha) with time since fire (years) for all treatments from all Fire Effects Study Areas

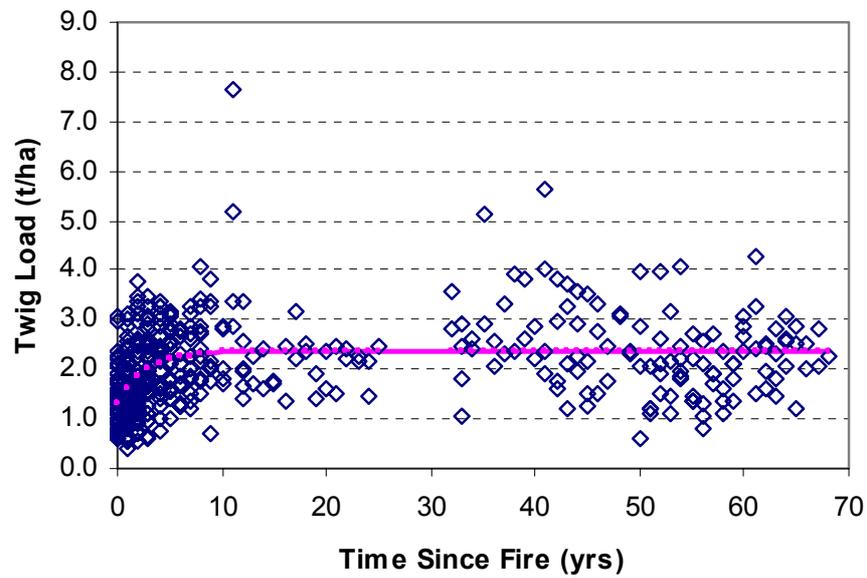


Figure 2.3 Twig fuel load (t/ha) with time since fire (years) for all treatments in all Fire Effects Study Areas

The relationship between the depth of surface fine fuel and time since fire was similar to that of the litter load, showing a negative exponential increase with time since fire (Figure 2.4). The main component of the surface fine fuel is the litter, so this was not surprising. The model of litterbed depth with time since fire is given by the equation:

$$D = 1.98(1.77 - e^{-0.253t})$$

$$(r^2 = 0.44, n = 411)$$

Where:

D is the surface fine fuel depth (cm)
 t is time since fire (years)

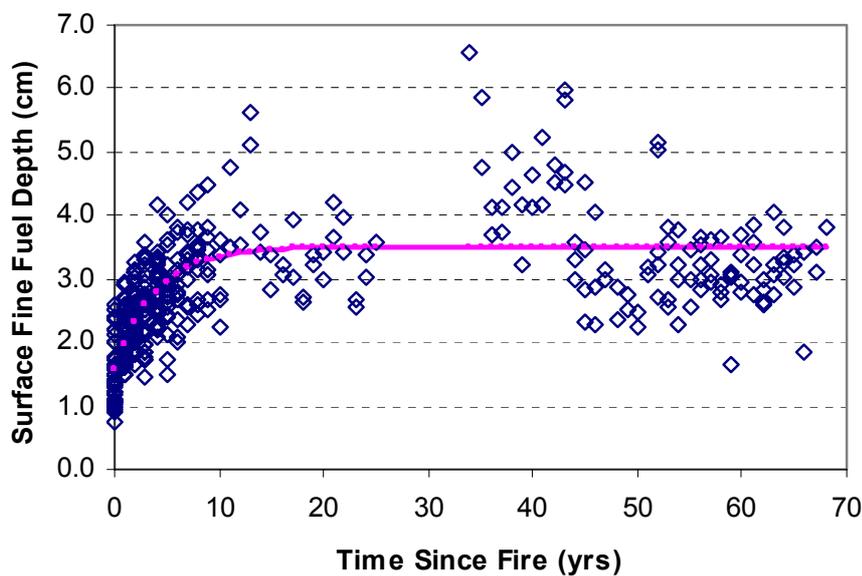


Figure 2.4 Surface fine fuel bed depth (cm) with time since fire (years) for all treatments in all Fire Effects Study Areas

Data point (year 34, depth 11.6) is off the scale and not shown for clarity.

If a one-year offset is used to account for the residual fuel left behind after a burn, the fuel accumulation model, using the Olson (1963) equation is:

$$X_t = 15.9 (1 - e^{-0.318 t})$$

With this one-year offset applied, the general exponential model predicts a fine fuel load immediately after the fire of less than 1 t/ha. The assumed litterfall rate (L) to achieve a steady state total fine fuel load of 16 t/ha is about 5.0 t/ha/yr.

Long-unburnt Treatment Areas

The long-unburnt Treatment Areas in this study were used as Controls. No fire treatment was applied to them, so variations in these areas were due to seasonal conditions, year-to-year variations and any changes taking place due to stand development. All FESAs were effectively cleared by the beginning of the 1900s, therefore the age of the stands when the study began was about 85 years; by the end of the measurements recorded here, the stands were about 100 years old.

There appear to be two distinct trends in the data for the long-unburnt (Control) Treatment Areas. The first is the long-term trend seen in Figure 2.1 where the surface litter load achieves an equilibrium level in a period of about four years after a fire. The second is a shorter-term trend, indicated in Figure 2.5, of a consistent general increase in the amount of surface litter fuel in each of the five FESAs. The average rate of surface litter increase was about 0.2 t/ha/yr (± 0.1) over the 13 years of measurement: 1987-99. This rate of increase is statistically significant ($p = 0.002$, $n = 12$). However, there are indications from some of the latter measurements, and the long-term trend shown in Figure 1.1, that these increases may only be a decadal phase rather than a long-term trend. This decadal phase may be due to some seasonal trend or a stage of development this forest is going through as it ages from about 85 years to 100 years.

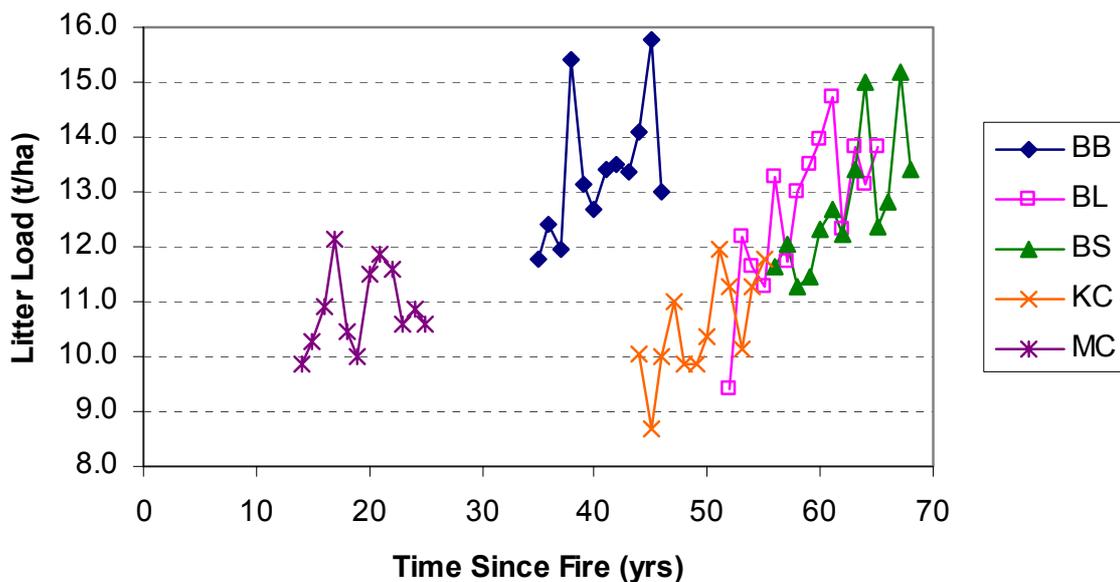


Figure 2.5 Annual litter loads (t/ha) in the Control of each of the Fire Effects Study Areas with time since fire (years)

Key: BB = Burnt Bridge; BL = Blakeville; BS = Barkstead; KC = Kangaroo Ck; MC = Musk Ck

There was also a trend of increasing humus over the period of the measurements (1987-99). Humus at Musk Creek, Kangaroo Creek, Blakeville and Barkstead showed an average increase of about 0.15 t/ha/yr over the 13 years of measurement. No trend was evident at Burnt Bridge, however. Nevertheless, as was seen in Figure 2.2, there is no long-term trend in humus accumulation, so this observed increase was probably due to conditions that have existed over the period of the study.

Twigs showed no clear trend over the period of measurement. This is the same result as was observed from the long-term data in Figure 2.3.

Litterbed depth did not show any clear trends over the period of measurement. This is in contrast to the observed litter load variation (Figure 2.5). As there is a strong correlation between litter load and litterbed depth, this result was surprising. It is, however, consistent with the long-term trend shown in Figure 2.4.

Litterbed depth was 1.2 cm (± 0.24) greater at Burnt Bridge (4.3 cm) compared with the other FESAs (3.1 cm) ($p < 0.01$). There is widespread Wire-grass at Burnt Bridge, which raises the litter above the ground and makes the effective depth of the litterbed greater.

Litter fuels were consistently greater in autumn than they were in spring as shown in Figure 2.6. On average, autumn litter loads were 1.6 t/ha (± 0.3) greater than spring litter loads and this was statistically significant ($p < 0.001$). Litterbed depth was also significantly greater in autumn than in spring by 0.35 cm (± 0.18 , $p = 0.006$). However, there were no significant differences in humus or twig quantities between autumn and spring at the 95% confidence level.



Figure 2.6 Litter loads (t/ha) across all Fire Effects Study Areas measured annually in spring and autumn on the long-unburnt (Control) Treatment Areas

Effects of burning treatments

The experimental fires reduced the litter fuel by 40–80% (Figure 2.7). This equates to a residual litter load of between 2 and 6 t/ha. Litter load increases rapidly for the first two years and then more gradually as it approaches the pre-burn levels. Litter loads reached about 80% of the pre-burn levels within two years of burning in spring, but took about four years to reach similar levels following autumn burning. There is no evidence, from the data collected in this study, that litter loads reaccumulated faster or slower after short-rotation (3 years) or long-rotation (10 years) burning.

Reaccumulation rates in the first three years after burning in both spring and autumn are statistically similar. These rates are about 13–25% of the pre-burn level per annum for the first three years. The only exceptions to this are the two areas burnt in autumn in the twelfth year of the study. Not as much fuel was removed from these areas in the first instance and the rate of reaccumulation had been only about 5% of the pre-burn level per annum for the first three years.

In the first rotation of fire treatments, the autumn fires tended to remove more of the litter than did the spring fires. However, subsequent spring and autumn fires were more similar in their impact on litter load. Clearly, the autumn burns in the twelfth year of the study did not remove as much fuel as did previous burns. Differences in the amount of fuel removed in spring and autumn fires seem to be related to factors other than the number of times an area has previously been burnt and the season in which it is burnt.

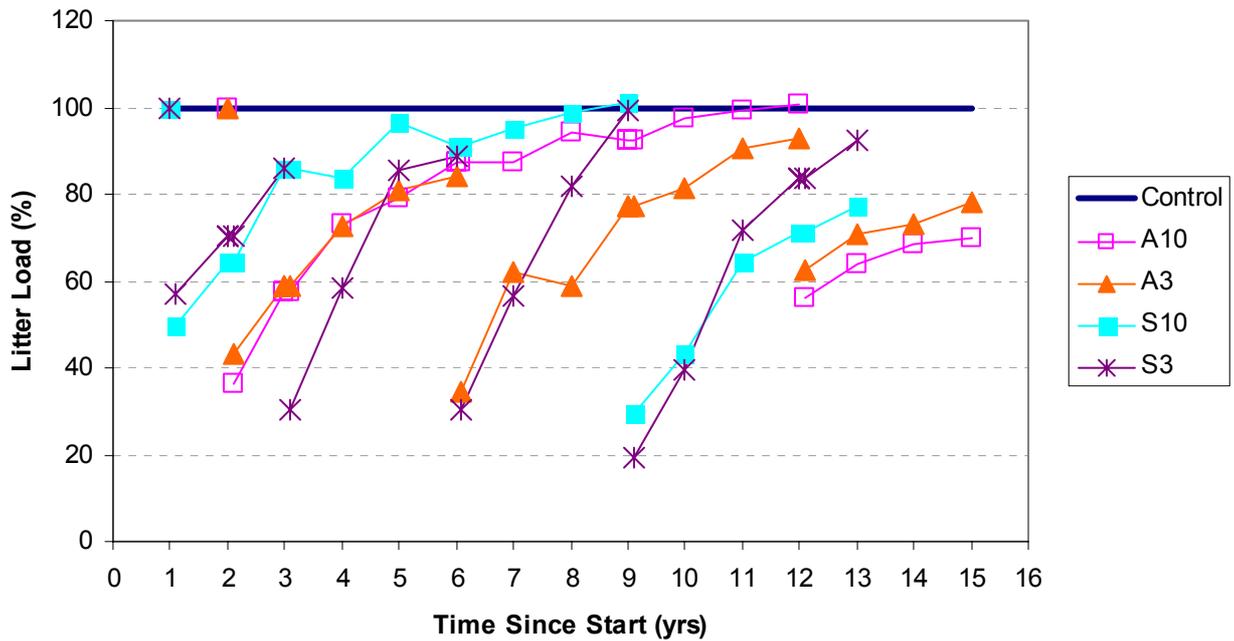


Figure 2.7 Relative litter load (t/ha) in each of the Treatment Areas with time from the beginning of the experiment for all Fire Effects Study Areas combined

Data were standardized to remove the effect of year-to-year changes in Control litter loads as shown in Figure 1.6.

Conclusions

This chapter has addressed only the surface fine fuels in a mixed eucalypt foothill forest; it has not addressed the elevated fuels such as shrubs, nor the bark held on tree boles. However, when considering the impact of prescribed burning on potential fire behaviour, attention should be given to the overall fuel complex, which includes elevated and bark fuels.

Surface fine fuels in this forest have an average steady state level of 16 t/ha, but seasonal variations may result in the fuel loads ranging from 9 t/ha to 26 t/ha. Therefore, fuel accumulation models should only be used as a general, long-term, broad-scale planning tool. To assess the impact of fuels on fire behaviour or hazard levels in any particular year or in any particular place, an on-site assessment must be made.

Surface fine fuels reaccumulate to within 90% of the long-unburnt state within four years of either spring or autumn burning. The impact of prescribed burning on surface fuels is therefore short-lived. However, studies elsewhere (e.g. Tolhurst et al. 1992a; Tolhurst 1995d; McCarthy & Tolhurst 2002) have shown that the effect of prescribed burning on overall fuel hazard levels may last for 15 years or more.

Surface fine fuel decomposition is significantly reduced for two to three years after a low-intensity fire. Litter accumulation is therefore very rapid during this time. Surface fine fuels left behind after a fire become an important part of the reaccumulated litterbed. This study has shown that, unlike the results from several previous studies, the residual litter does not break down rapidly or disappear. Further, the rate of reaccumulation of surface fine fuel is not significantly affected by the season of burn or the burn frequency.

Stand structure and canopy density are important factors in the rate of litterfall and, hence, the amount of fuel that accumulates in this type of forest. Increases in canopy cover result in an increase in the 'steady state' surface fine fuel loads. (Shrub and other elevated fuels were not considered here and these may compensate for loss of overstorey-sourced litter fuels.)

Short-term trends in litter accumulation due to seasonal or stand structural changes are likely to be more significant than the long-term trends in litter accumulation.

This study has demonstrated that some of the conclusions drawn from short-term studies or space-for-time studies can be misleading. Given the longevity of a forest ecosystem, studies undertaken in a period of less than ten years may attribute trends or variations to erroneous causes or overlook underlying trends altogether. Longitudinal studies, such as this one, provide a better insight into the dynamics of the forest ecosystem than do short-term studies.

Vegetation

Kevin Tolhurst

Introduction

Some evidence exists to show that the structure and composition of forest understoreys have been changed in south-eastern Australia due to frequent, moderate- to high-intensity fires. Chesterfield (1984) and Wakefield (1970) both give accounts of increasing domination of fire-resistant plants (Purdie 1977a; —1977b) in areas burnt frequently, and hence totally changing the appearance of the forest understorey from predominantly herbaceous to predominantly shrubby. Shrubby understoreys are typically less diverse, more dense and highly flammable, being dominated by leguminous shrubs, notably *Daviesia* spp., *Acacia* spp. and *Pultenaea* spp., Austral Bracken (*Pteridium esculentum* (G.Forster) Cockayne) and Forest Wire-grass (*Tetrarrhena juncea* R.Br.). Fuel reduction burning, it has been argued, may be a factor in maintaining these highly flammable understoreys.

A series of reports and papers on the vegetation's responses to the spring and autumn burning regimes in this experiment have been prepared previously. The most significant report (Tolhurst & Oswin 1992) was written as part of the 'First Progress Report' (Tolhurst & Flinn 1992). The conclusions in that report were:

- no species had been lost as a result of two rotational burns in spring or a single burn in autumn
- the height of the understorey had been reduced for at least four years after each fire
- the recovery after the second-rotation fires was slower than after the first fires
- the age structure of the plant populations had been simplified to become largely single-aged
- seedling establishment was more pronounced following autumn fires than spring fires
- resprouting was more vigorous after spring burning
- whilst there were differences between spring and autumn burning, none of these differences represented irreversible changes to the plant communities studied.

These conclusions have been reinforced in later papers, such as Tolhurst (1995c, 1998).

A better understanding of how climatic factors affected the response of understorey plants to low-intensity fires was outlined in Tolhurst (1995a). In that report it was shown that the effect of seasonal conditions following a fire could significantly alter the magnitude of the response to the fire. By taking into account the length of growing season after each burn, the responses to fires at different times became more similar. It was found that seasonal weather conditions were a major determinant of the period of active growth in burnt and unburnt vegetation. Soil moisture status, ambient air temperature and solar radiation were major factors determining the growth period of herbs, shrubs and trees in foothill forest. As a result, it was concluded that the effects of the season of burning (spring or autumn) could potentially be masked by unusual weather conditions such as a wet summer or a mild winter, especially during the first year after burning. It was also found that when continuous growth was possible immediately after a fire, vegetative growth was rapid, and the growing period before the first drought influenced the chances of seedlings growing large enough to survive the first year. It was concluded that the effects of fire on vegetation could only be considered in conjunction with the prevailing seasonal weather conditions.

Variability of the fire across the burnt area and in time were also found to be important to the response of the vegetation to fire (Tolhurst 1995b). It was found that the duration of heating of the vegetation and the extent of soil heating varied with season and with successive burns on the same Treatment Area. The duration of heating was related to the size of the trees killed in a fire which, in turn, was related to the thickness of the bark on the

trees. The density of legume germination was related to the soil temperatures induced by the fires; and soil temperatures were higher on average in autumn than in spring. Soil heating in long-unburnt Treatment Areas was generally found to be more variable than those recently burnt, except where there was a relatively small proportion of fuel available, and then soil heating became very variable. These variations in soil heating were reflected in the variability in plant responses.

This report continues the analysis of these experiments. The research reported here tests the hypothesis that repeated low-intensity prescribed fire changes the structure (cover) of the forest understorey, and that there is a quantifiable difference between spring and autumn fires on forest structure in mixed eucalypt foothill forest of west-central Victoria.

Research results

Total understorey plant cover

Over the decade 1988-98, total understorey plant cover averaged about 45% across the five FESAs. Of this, Austral Bracken contributed almost 17%, Forest Wire-grass contributed 16% and *Poa sieberiana* about 4%. Table 3.1 shows the approximate contribution to total plant cover by each major species or life-form group.

Table 3.1 Average cover contributed by major species or life form group across the five Fire Effects Study Areas in the Wombat Fire Effects Study

Species or life-form group	Average cover (%)
Austral Bracken	16.5
Forest Wire-grass	16.0
Poa	4.0
Herbs	3.0
Rushes	2.5
Geophytes	0.8
Shrubs	0.8
Small shrubs	0.6
(Legumes)	(0.5)
Trees (regrowth in understorey)	0.4
Small trees	0.1
Climbers	0.1
Sedges	0.1
Ferns	0.1
Total understorey cover	45.0

In the absence of fire, total understorey plant cover varied from year to year, gradually increasing from 27% at the first measurement in 1985 to about 44% in 1989. Understorey plant cover then fluctuated between about 40 and 45% from 1989 to 1998. This initial steady increase in total understorey plant cover followed a decade of below-average rainfall culminating in the drought and devastating wildfire season of 1982-83. This rise in cover may be due to a gradual recovery after this drought or it may be due to structural changes taking place in the forest as it ages from about 85 years to about 100 years. However, the fact that the cover value seemed to level out suggests that the drought may have been the most likely cause of these changes. These fluctuations represent the background changes to any impact of fire.

The total understorey plant cover in the spring burning treatments showed a rapid regrowth following each of the burns. The first-rotation spring burns (S10-1R and S3-1R) reached the pre-burn levels of cover within four years. However, after the second-rotation burns (2R), the recovery, whilst rapid initially, had not returned to pre-burn levels before it was burnt again with the third-rotation (3R) burning treatment. The impact of the 3R and 4R burns seemed to be progressively greater than earlier burns. A noticeable trend in progressive burns is the increased impact on the level of cover in the first year after the fire. After the 1R burns, the cover of plants dropped to 80%, then 60% after the 2R burns, 30% after the 3R burns and 20% after the 4R burns. This seems to indicate that the plants had not fully recovered from the previous burns before they were burnt again, even though the cover may have apparently returned to near pre-burn condition.

Following autumn burning, the total understorey cover had not returned to the pre-burn condition even after ten years. Understorey plant cover on the first-rotation (1R) long-rotation autumn (A10) treatments dropped to about 70% of the pre-burn condition and progressively increased over the next ten years to be still only 80% of its original value. There appeared to be little change in plant cover on these treatments between five and ten years after the burn. The short-rotation autumn burning treatment (A3-1R) results followed a similar trend until it was reburnt. However, after the second-rotation burn (A3-2R), there seemed to be little recovery from the post-fire level of 60% over the following five-year period. The third-rotation burn (A3-3R) again reduced the total understorey plant cover to just below 60%.

Autumn burning seemed to have a greater impact on the total understorey cover than spring burning. Over the 14-year period of this study, there had been an increase in the total understorey cover of about 15% in absolute terms and about 50% in relative terms. These generalised understorey responses were not uniform across all species or life-form groups.

In terms of absolute cover, most understorey plant groups increased in cover over the period 1985-98 as shown in Table 3.2. The exceptions to this have been the herbs, legumes and trees. Generally, short-rotation fires have reduced understorey plant cover compared to both the long-unburnt (Control) Treatment Area and the pre-treatment levels. There were exceptions, however, such as for herbs burnt in spring, and rushes and shrubs burnt in autumn. On the other hand, legumes have benefited from the longer cycle of spring and, more particularly, autumn burning.

Table 3.2 Summary of changes in cover over the 14 years of the study in the different fire treatments in the Wombat Fire Effects Study

Life-form	Control	Spring 3	Spring 10	Autumn 3	Autumn 10
Total understorey cover	+13.4	-4.0	+9.4	-3.7	-0.7
Austral Bracken	+10.2	+1.8	+3.8	-2.3	-1.3
Forest Wire-grass	+6.9	-5.2	+5.2	-1.2	+1.3
Poa	+0.7	+0.6	+1.1	+1.8	+1.3
Herbs	-3.1	+1.0	-0.3	-1.9	-1.8
Rushes	+2.1	+0.5	+0.8	+4.1	+1.6
Geophytes	+0.9	+0.6	+0.2	-0.2	+0.3
Shrubs	+0.8	+0.1	+0.1	+5.6	-0.4
Small shrubs	+0.6	-0.1	-0.1	+0.5	+0.6
Legume	-1.2	-0.6	+0.1	-3.0	+1.2
Small trees	+0.1	-0.4	+0.1	+0.2	0.0
Trees	-0.8	-1.2	-0.3	-1.0	-1.0
Climbers	+0.1	-0.7	0	+0.1	+0.1
Sedges	-	-	-	-	-
Ferns	-	-	-	-	-

Note: There was insufficient data for sedges and ferns

Species composition and abundance

The species composition for each of the 25 Treatment Areas was collected annually. The results from each of the five FESAs were similar so only the results from Blakeville will be reported here.

A total of 85 plant species was recorded on the permanent vegetation plots at the Blakeville FESA. More species were present across the other FESAs, but these results are limited to those species found on the permanent plots in a single mid-slope vegetation type.

Table 3.3a shows that there were between 44 and 58 species in each plot at Blakeville. Over the 14 years of measurements, between half and one-third of the species recorded maintained their cover-abundance in both the long unburnt (Control) and the treated (i.e. burnt) Treatment Areas. Again, about half to one-third of the species in each treatment increased in their cover/abundance rating, and generally less than a third of the species actually reduced their cover/abundance. If anything, the number of species which declined in cover/abundance over the 14 years were more common in the long-unburnt Treatment Area. No species disappeared from a treatment and no new species appeared.

Table 3.3a Number of species in each Treatment Area and the number of species whose cover/abundance increased, decreased or stayed the same over the study period

	Treatment Area				
	Control	Spring 3	Spring 10	Autumn 3	Autumn 10
Number of species	55	58	54	49	44
No change	15	20	21	24	23
Reduced	21	19	11	9	8
Increased	19	19	22	16	23

Summary of the species composition data collected from the Blakeville FESA 1985–99.

Even though the Treatment Areas had been selected on the basis of being in a similar vegetation type, only 36% of the 85 species were common to all five treatments at Blakeville (Table 3.3b). About half (47%) of the understorey plant species were only found in either one or two of the treatment plots. These data demonstrate that being ‘uncommon’ is the norm rather than the exception.

Table 3.3b Analysis of the species distribution between treatments at the Blakeville FESA

Common to all 5 plots	Common to any 4 plots	Common to any 3 plots	Common to any 2 plots	Restricted to a single plot	Total number of individual species
31 (36%)	8 (9%)	6 (7%)	15 (18%)	25 (29%)	85

There were clear differences between the impact of burning in spring and autumn, and clear differences in the impact of successive rotations of burning. This is one of the first studies to look at the effects of repeated burning as distinct from the impact of a single burn or the end result of a sequence of burns. The results here show that there is a progression in the impact of successive burns which can help lead to an understanding of the processes taking place.

Changes in the absence of fire

With a set of annual measurements over a 14-year period it was found that, in the absence of fire, there are significant changes taking place in the relative abundance of individual plant species as well as the cover of these species. These changes differ between species. For the first five years of this study there was a gradual increase in the total understorey plant cover which then appeared to level out for the remaining nine years. This period of increase coincided with the breaking of one of the State’s most severe droughts, which saw the disastrous bushfires of 1982–83. This drought was so severe that whole hillsides of trees were dying—an unusual and extreme event. The increase in understorey cover up until about 1989 may well have been in response to the breaking of this drought and a return to more normal conditions.

Whilst the overall understorey cover in the absence of fire was relatively constant between 1989 and 1998, individual species and groups of plants responded differently. Austral Bracken and Forest Wire-grass generally followed the same trend as the overall understorey cover. Poa, rushes, geophytes, shrubs, small shrubs and climbers, on the other hand, all tended to gradually increase in cover over the whole period while herbs, legumes and trees all gradually decreased in cover. Bracken and Wire-grass make up the bulk of the understorey cover and hence dominate what happens in the understorey.

The groups of species which continued to increase over the period of the study probably did so for different reasons. *Poa* has long-lived tussocks, but also has the ability to spread underground and produce new tillers at a distance from the parent tussock. This ability to spread has probably assisted in its increase. Rushes are also long-lived and are able to expand laterally to occupy suitable sites. Shrubs, small shrubs and climbers are all relatively long-lived and slow-growing, so they continue to increase in cover as they grow. Geophytes are somewhat more opportunistic and are able to proliferate in good seasons and retreat in poor seasons. Geophytes typically grow best in relatively bare ground, which may have been reduced by a heavier litter load on the forest floor resulting from leaf death induced by the 1982-83 drought, and which has subsequently decomposed.

The plant groups that have decreased in cover over the 14-year study period have also done so for different reasons. Herbs are relatively short-lived, but rapidly occupy vacant ground. Many are 'mobile' and rely on relatively short-lived seed to propagate and spread. They include many annuals that are totally reliant on this form of survival. The drought probably provided good opportunities for herbs to spread and germinate following rain, but they would have died again in the absence of further rain. Geophytes, on the other hand, produce an underground organ such as a bulb or tuber and hence are more adaptable to unreliable growing conditions. The legumes and trees are probably the longest-lived plants in the understorey. Their decline has probably resulted from the recovery of the canopy cover following the drought. This, in turn, has increased the competition for light and moisture in the understorey. Legumes and trees may have been able to grow more effectively in the understorey during the drought because of their relatively sheltered position, but declined again as the tree canopy recovered.

Climatic factors

The dynamics in the understorey in the absence of fire show how important to plant cover are climatic factors and the growth stage of the forest. The effect of fire is superimposed on this dynamic background, a factor usually ignored in many other fire ecology studies.

An analysis of the impact of climate on the growth and response of understorey plants was made in this study (Tolhurst 1995a). This analysis showed how many of the apparent effects of fire were in fact due to climatic conditions. The nature of the season in which the fire takes place and the subsequent growing conditions after the fire are particularly important to plant responses.

In general, it has been recognised that resprouting plants are promoted by spring burning (Ashton 1970) as this is the time of active growth and a time when seedlings have a lower chance of survival due to the oncoming summer drought. This study found that Bracken, *Poa*, herbs and geophytes all responded well to spring burning, but Wire-grass, rushes, shrubs, small shrubs, legumes, trees, small trees and climbers did not respond well. There is no clear division between resprouters and seeders in these two response groups. Perhaps the only differentiation between the two response groups is between those that are able to rapidly change their cover/abundance on an annual basis and those that are slower to respond to changed environments. One generalisation that may be made is that Bracken, *Poa*, herbs and geophytes probably respond better to frequent low-level disturbance than those plants in the other group. The Wire-grass, rushes, shrubs, legumes, etc. group are probably favoured more by more intense, but less frequent disturbance. Spring burning represents a lower level of disturbance as a result of less canopy scorch and less soil heating and exposure (Tolhurst 1995b). The effect of repeated spring burning is generally to increase the amount of soil heating and hence make it more like autumn conditions, but the drought-free growing season following spring burning is still only about three months compared with nine months following autumn burning (Tolhurst 1995a).

Repeated spring and autumn burning

An important result from many of the plant groups, as well as for the total understorey, was the progressively greater impact of successive spring burns on the cover of the plants. The cover of the total understorey dropped from 80% to 60%, 30% and 20% in the year following each successive spring burn. The apparent shoot recovery from these levels was quite good, but the results show that the understorey might not have completely recovered. Much of a plant's biomass is underground (e.g. Tolhurst 1996b) and often this is where substantial carbohydrate reserves are stored (Tolsma 2002). These carbohydrate reserves provide the initial energy for resprouting and can be progressively consumed by successive defoliation events such as fire or grazing. If these energy losses exceed the rate of replenishment, the plants will eventually die (Tolsma 2002). It is highly likely that the progressively greater impact of the spring burning is due to the gradual depletion of these energy reserves. However, it was found that the second-rotation burn on the infrequent burn treatment also produced a big reduction in the first-year cover. This indicates that the reduction in cover is probably due not only to reduced energy reserves but also to the direct impact of a fire event and the conditions during the subsequent growing season.

Overall, autumn burning seems to have reduced the understorey cover for at least ten years. Bracken, Wire-grass, small shrubs and geophytes are the main groups of plants exhibiting reduced cover after autumn burning. Wire-grass is showing a strong recovery pattern, but looks like taking 12 to 15 years to return to its pre-burn levels. There were some differences in response between the frequent and infrequent autumn burning. Frequent autumn burning reduced the cover of Wire-grass, Poa, rushes, legumes and trees beyond the effects of a single autumn burn. These groups of plants are generally longer-lived and hence slower to develop and are therefore adversely affected by repeated autumn fires. Herbs, on the other hand, seem to be the only group to be advantaged by frequent autumn burning. Herbs are generally shorter-lived and develop more quickly and include many seed-reproducing species. If a regime of regular autumn burning was applied, the understorey would be expected to become more herbaceous and less shrubby. Conversely, infrequent autumn burns would lead to a more shrub-dominated understorey (Chesterfield 1984).

Species composition

The species composition of all treatments remained the same over the duration of this study. No new species were introduced and no species were lost as a result of any of the applied fire regimes. There was, however, a change in the relative abundance of many species. In broad terms, almost half of the species maintained their relative cover/abundance, about one-third increased and the remaining fifth decreased in their relative cover/abundance over the 14 years of the study. Previous studies have reported species loss as a result of a particular fire regime (Gill & Bradstock 1995) or even a single fire (e.g. Hamilton 1991). Those studies restricted their method of assessing species cover/abundance to limited areas and a limited period of time. The scale of the sampling unit should vary with the time since fire to truly represent what is happening as a result of a particular fire regime. In this study, species presence was determined from an area of one hectare or more; an area large enough for a range of plant microenvironments to exist. In addition, species presence was assessed annually but, from the perspective of determining local extinctions, a period of time up until the next prescribed fire was used rather than the year after a fire. Hamilton's (1991) findings are contingent on a limited amount of sampling in both space and time. Gill and Bradstock (1995) looked at local extinction over an interval of 50 years and in areas of tens of hectares or more. This is a more relevant scale to work on but, even so, Weston (1985) found some plants, apparently unseen for 150 years, reappeared in abundance after a fire in 1972-73.

Conclusions

Over a 14-year period, no plant species was either lost or gained as a result of up to four successive spring fires or three successive autumn fires. The relative cover/abundance increased for about 30% of the species and decreased for about 20% of others.

There was a progressively greater effect of repeated, frequent burns. This was attributed to incomplete recovery of plants between successive burns in conjunction with seasonal and fire intensity conditions.

Short-rotation spring burning favoured Austral Bracken, herbs, geophytes and *Poa sieberiana* and disadvantaged Forest Wire-grass, rushes, shrubs, small shrubs, legumes, trees, small trees and climbers. Infrequent spring burning favoured legumes and, to some extent, *Poa*.

Short-rotation autumn burning favoured only herbs and disadvantaged Bracken, Wire-grass, *Poa* species, rushes, legumes, small shrubs, trees and geophytes. Infrequent autumn burning favoured *Poa*, rushes, legumes and trees by providing a regeneration opportunity followed by a growth period. Bracken, shrubs and small shrubs were the only plant groups that seemed to be disadvantaged by infrequent autumn burning.

This study has shown that the effect of a single fire cannot simply be extrapolated to successive burns. The condition of the plants, the nature of successive fires and the prevailing seasonal conditions all combine to affect the final response to successive burns.

There are subtle changes in the forest understorey in the absence of fire. These changes are only small in magnitude on a year-to-year basis, but amount to significant changes over a period of a decade or more. A long unburnt area should not be assumed to be in a 'steady state' and therefore be expected to be a fixed reference point when assessing the impact of fire on the forest ecosystem.

There is no single fire regime that can be said to be 'ideal'. It is likely that a variable fire regime will provide the greatest biodiversity in space and time.

Long-term fire ecology studies have the ability to discern changes too subtle to be picked up by shorter-term or space-for-time studies. Space-for-time studies should be treated with caution in the same way as long-term studies, and must be viewed with an understanding of the environmental and biological factors operating at the time.

Recommendations

- Long-term studies should continue to be supported to provide information on subtle changes in forest ecosystems that may be hidden to short-term research.
- Spring and autumn prescribed burning should continue to be practised provided recognition is given to the likely effect season will have on the outcome.
- Short-rotation burning is undesirable from a biodiversity point of view and should be avoided where possible.

Invertebrates

Nick Collett and Fred Neumann

Introduction

Detailed studies of the effects of fire on Australian invertebrates are relatively rare. This impacts significantly on the abilities of land managers to develop sustainable strategies that conserve the invertebrate fauna as well as satisfy other management objectives. This report presents the effects of various prescribed fire regimes (in spring or autumn; single or multiple fires; short or long-term rotation) on invertebrates sampled in the same forest over the 14-year period: 1985-99.

Arthropods that frequent the soil layer are especially vulnerable to ground fires that destroy their habitats and desiccate upper soil layers. Earthworms (Annelida) require a continuous film of moisture over their permeable outer skin (cuticle) for respiratory exchange (Lee 1983) and are therefore highly fire-sensitive due to their intolerance to ambient temperatures above 25 °C (Reynolds 1973) and to low moisture levels in the litter/upper soil horizons.

Humus/litter and upper soil horizons provide temporary or permanent habitats for a diverse range of invertebrate macrofauna (greater than 10 mm in length) and mesofauna (0.1-10 mm). These invertebrates, together with microbes (less than 0.1 mm), perform essential ecological functions, such as regulating the decomposition of organic matter, recycling of nutrients, aerating the soil and facilitating biological control among invertebrate populations (Crossley 1977; Greenslade & Greenslade 1983; Spain & Hutson 1983).

Research results

A total of 36 ordinal or lower-level arthropod taxa were identified among the tens of thousands of specimens collected during the course of the study. These represented the classes Arachnida (arachnids), Malacostraca (crustaceans), Chilopoda (centipedes), Diplopoda (millipedes), Symphyla (symphlans), Collembola (springtails), Protura (proturans), Diplura (diplurans) and Insecta (insects). The feeding types of predators, decomposers, herbivores, sapfeeders, omnivores, parasitoids and seed feeders were all well represented in the litter surface of each Treatment Area (Table 4.1). The most commonly trapped taxa were the Coleoptera (beetles), Diptera (flies), Formicidae (ants), Araneae (spiders), Acarina (mites) and Collembola (springtails), and are referred to as the 'major' taxa to distinguish them from the less-commonly trapped 'minor' taxa.

The specimens were collected in pitfall traps recovered from the four fire-treated areas and one unburnt (Control) Treatment Area within the Blakeville FESA. Because of the size of the task, sampling was confined to this single FESA, and thus replication of the study in adjacent forest of a similar type did not occur (Neumann & Tolhurst 1991; Collett et al. 1993; Collett 1998; —2000). Despite this limitation, the results obtained allowed some specific trends to be observed and are detailed below.

Table 4.1 Taxa of litter arthropods trapped in the foothill forest

	Taxon	Feeding type
Arachnida	Scorpionida (scorpions)	Predators
	Araneae (spiders)	Predators
	Pseudoscorpionida (pseudoscorpions)	Predators
	Opiliona (harvestmen)	Predators
	Acarina (mites)	Predators/decomposers
Malacostraca	Amphipoda (landhoppers)	Decomposers
	Isopoda (woodlice)	Decomposers
Chilopoda	Geophilida (earth centipedes)	Predators
	Lithobiida (lithobiid centipedes)	Predators
	Scolopendrida (scolopendrid centipedes)	Predators
	Scutigera (house centipedes)	Predators
Diplopoda	Polydesmida (polydesmid millipedes)	Decomposers
Symphyla	(symphylans)	Decomposers
Collembola	Collembola (springtails)	Decomposers
Protura	(proturans)	Decomposers
Diplura	(diplurans)	Herbivores/predators
Insecta	Thysanura (silverfish, bristletails)	Omnivores
	Blattodea (cockroaches)	Omnivores
	Isoptera (termites)	Decomposers
	Mantodea (mantids)	Predators
	Dermaptera (earwigs)	Predators
	Orthoptera-Tettigoniidae (long-horned grasshoppers)	Herbivores
	Orthoptera – Acrididae (short-horned grasshoppers)	Herbivores
	Pscocoptera (psocids)	Decomposers
	Homoptera (aphids, leafhoppers, scale insects)	Sapfeeders
	Heteroptera (true bugs)	Sapfeeders/predators
	Thysanoptera (thrips)	Herbivores
	Neuroptera (lace wings)	Predators
	Coleoptera (beetles)	Various
	Siphonaptera (fleas)	Parasitoids
	Diptera (flies)	Various
	Trichoptera (caddis flies)	Predators
	Lepidoptera (moth, butterflies)	Herbivores
	Lepidoptera-Coleophoridae (case-bearers)	Herbivores
	Hymenoptera-Formicidae (ants)	Sap/seed feeders
	Hymenoptera-Aprocrita (wasps)	Parasitoids/predators

Effects of one spring burn and one autumn burn

Neumann and Tolhurst (1991) found that, after a single spring burn, Collembola, Diptera and earthworm populations temporarily declined, as did Collembola populations after a single autumn fire, although Collembola and Diptera populations recovered within one year and earthworms within three years. It was observed that fine fuel loads accumulated more rapidly in the first year after the spring fire compared with the autumn fire, although, in the subsequent two years after fire, fine fuel loads increased fairly evenly at both Treatment Areas. These results suggested that the spring fire and, to a lesser extent, the autumn burn induced a short-term decline in the decomposer cycle, especially as the Collembola, larval Diptera and earthworms are important components among decomposers in the forest litter (Greenslade & Greenslade 1983; Spain & Hutson 1983). It was concluded that the effects of a single spring fire coupled with the desiccating conditions of the subsequent summer period impacted more adversely on earthworms than an autumn fire (which is usually much hotter than a spring fire) coupled with the cool and wet conditions following in winter.

A marked decline in activity of Collembola on both the autumn burnt and unburnt Control Treatment Areas during dry conditions persisting from December 1987 to April 1988 suggested this important decomposer taxon is very sensitive to moisture levels within the litter layer. Fuel reduction burning during protracted drought periods is therefore likely to impact adversely on already disrupted populations of Collembola. For similar reasons it is likely that such conditions would also impact on earthworm and larval Diptera populations.

A moderate boost to Formicidae populations was recorded after a single spring burn. This may reflect either: (i) increased activity of existing populations due to drastic changes in microclimate at the litter surface and to greater food availability or, (ii) an increase in trapping efficiency as a result of the simplification of the habitat and a widening of the foraging range (Majer 1980; Andersen & Yen 1985; Andersen 1988). The autumn burn did not enhance ant activity, possibly because the cooler conditions following the fire confined many ants to their nests. This hypothesis requires further investigation. These moderate interactions between low-intensity fire and ants in foothill forest are in marked contrast to the massive increase in populations of the seed-feeding ant *Prolasius pallidus* Clark (Formicinae) observed in response to high-intensity wildfire in mature Mountain Ash (*Eucalyptus regnans* F. Muell.) forest in the Victorian Central Highlands in 1983 (Neumann 1991).

Effects of two short-rotation burns in spring

After each of two short-rotation spring burns, Collett et al. (1993) found activity levels for both Collembola and earthworm populations to be significantly depressed, although population levels did recover within three years. Campbell and Tanton (1981) made similar observations. Declines in activity were accompanied by a marked decline in fine fuel loads immediately after each burn, as observed by Neumann and Tolhurst (1991) after a single spring fire. The prolonged period required by earthworm populations to recover to pre-burn levels after short-rotation spring burns suggests that spring burns, coupled with the dry, warm conditions of the ensuing post-burn summer period, impact adversely on earthworm activity (Neumann & Tolhurst 1991). This situation is likely to be exacerbated during protracted drought periods that impose severe soil moisture stress in these forests. In addition to the loss of litter habitat, the large difference in surface soil temperature between the first and second burn (161 °C compared with 403 °C respectively) may have caused earthworm levels to be severely depressed over an 18-month period after the second fire compared to only 10 months after the first fire.

Coleoptera activity levels were lower after the second spring burn compared to the first burn, which may indicate that repeated fuel reduction burning at close intervals has a debilitating effect on this order. However, as Coleopteran activity on the Control Treatment Area also declined to low levels during the summer after the second burn, it is likely that the observed decline was due to 'other environmental factors', rather than to the effects of the burn. Due to the broad taxonomic approach adopted in this study, it could not be established whether these reductions were spread over a wide range of Coleopteran species or confined to a select few. Dipteran activity was also depressed after the first burn, but not after the second, indicating factors other than burning may have affected their activity levels.

Formicidae activity followed a consistent pattern of peaks during warm periods of the year and troughs during cool periods at both the Control and burnt Treatment Areas. Moderate increases in ant activity occurred after both burns, but populations quickly returned to pre-burn levels. Again, these moderate short-term increases of ant numbers within foothill forest in response to low-intensity prescribed burns in spring are in marked contrast to the very substantial boosts in Formicidae activity following a high-intensity wildfire in 54-year-old Mountain Ash regrowth forest in the Victorian Central Highlands (Neumann 1991; —1992).

Effects of two short-rotation burns in autumn

Collett (1998) found that the results of the two- and three-dimensional χ^2 tests performed on the pre- and post-fire collections of the arthropod taxa at the Control and burnt Treatment Areas provided good indications of the effects of fire on taxon richness. As the null hypothesis of a nil effect was accepted in all of these tests, it appears that neither the burns on their own, nor their cumulative effect, had significantly altered the profiles of the arthropod taxa. This conclusion was supported by the non-significant results of the Mann-Whitney U-tests on pre- and post-fire data sets based on the total arthropod taxa for the Margalef taxon richness and Pielou community evenness indices and also for the general Shannon-Wiener diversity index. The results presented by Neumann and Tolhurst (1991), which also indicated no significant differences for total arthropods before and after a single autumn fire for the aforementioned three population indices, are in line with the above conclusion.

No significant change in activity was observed among total insects, Araneae, Coleoptera and Diptera throughout the 1986–94 study period. This indicated that, despite the application of two low-intensity burns in autumn within the 7.8-year period, the activity of these taxa remained unchanged. Insects therefore appear to form a fire-stable part of the litter fauna of foothill forests. However, when the pre-burn period was compared to the between-burns period, a significant decrease in activity was detected among total arthropod taxa and total non-insects, primarily as a result of a significant concomitant drop in activity of the Collembola (Figure 4.1). A similar result was observed when the pre-burn period was compared with the post-second burn period (i.e. the total taxa and non-insects decreased significantly as a result of a decrease in activity among Collembola and, to a lesser extent, Acarina). These results on Collembola confirm findings by Neumann and Tolhurst (1991) who noted that Collembolan activity significantly decreased following a single autumn low-intensity prescribed burn, most probably due to the fire temporarily eliminating leaf litter, reducing high humidity habitats and exposing the Collembola to desiccation.

Comparing the between-burns period with the post-second burn period showed no significant change in activity among total arthropod taxa, non-insects and Collembola. This suggests that a second prescribed fire within five years may not necessarily impact adversely on these broad arthropod categories, unless other significant events, such as a protracted drought, supervene. However, the second autumn burn was less intense than the first, indicating that sufficient litter cover may have remained, allowing the Collembola to remain relatively unaffected.

A significant increase in ant activity during the post-second burn period was predominantly due to increased activity in the second summer after the second burn. It is uncertain whether this observed boost in ant activity reflected: (i) increased activity of existing populations due to changes in the microclimate at the litter surface; (ii) greater food availability; or (iii) an increase in trapping efficiency resulting from the simplification of the fire-affected habitat and a widening of the foraging range (Majer 1980; Andersen & Yen 1985; Andersen 1988; Neumann & Tolhurst 1991). However, as no such boost in activity occurred among the Formicidae after the first burn, it seems probable that the rise in activity following the second fire was more likely due to some environmental variable other than fire (Collett et al. 1993). The decrease in activity levels of Dermaptera following the second burn coincided with a decrease in activity on the Control Treatment Area indicating that the decrease may have occurred as a result of some variable other than fire.

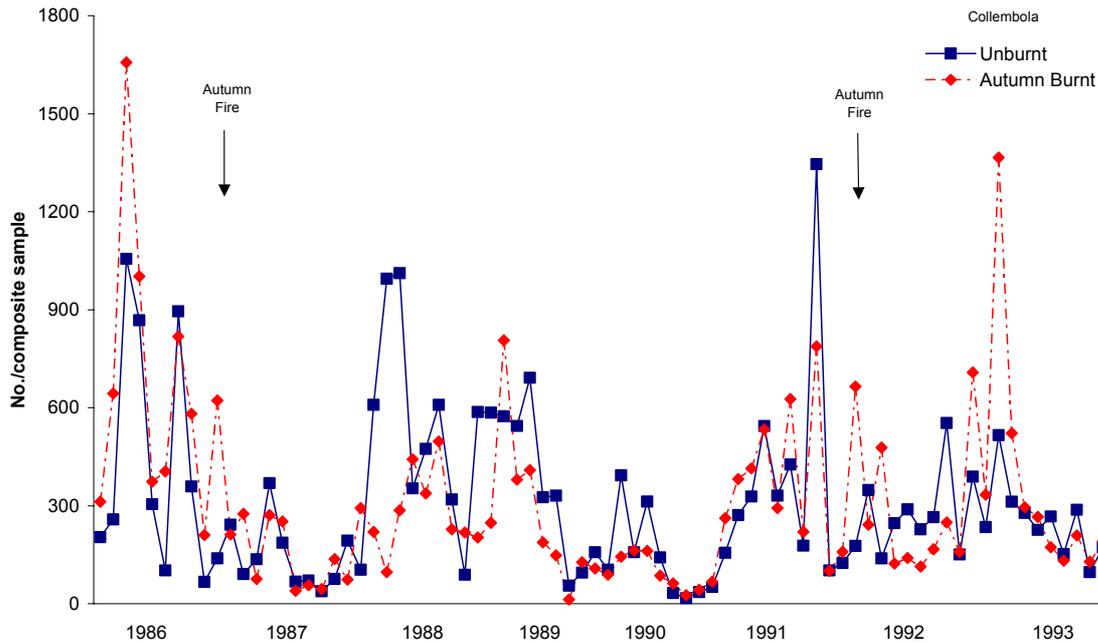


Figure 4.1 Temporal variations in monthly activity of Collembola from April 1986 to February 1994 at the litter surface of the Control Treatment Area and of that burnt in March 1987 and March 1992

Source: Collett 1998

Effects of three short-rotation burns in spring

Strong seasonal trends among the major arthropod groups at the Control Treatment Area were identified (Collett et al. 1993; Collett 1998; —1999), which highlights the need for contemporaneous sampling to achieve valid comparisons in such work. Differences between sampling times associated with pre- and post-burn periods (e.g. post-1985 - pre-1988 and post-1988 - pre-1991 are both three-year periods compared to the one-year pre-burn periods) does not imply that an increase in taxon richness had occurred as a result of the burning. The results of the two- and three-dimensional χ^2 tests performed on the pre- and post-burn collections of the arthropod taxa at the burnt and Control Treatment Areas showed that a null hypothesis (or 'nil effect') was accepted throughout. This indicates that none of the burns in isolation, nor their cumulative effects, had significantly changed the profiles of the arthropod taxa. This conclusion is supported by both the non-significant results of the Mann-Whitney U-test on the Margalef taxon richness community index and the non-significant results of the Mann-Whitney U-tests on pre- and post-burn data sets for activities of total taxa, total non-insects, Araneae, Acarina, Collembola, total insects and Formicidae. All of these tests indicate that three consecutive spring burns have no effect on either taxon richness or activity of the major arthropod groups.

The lack of significant change in activity observed among total taxa, total non-insects, Araneae, Acarina, Collembola, total insects and Formicidae indicated that, despite the application of three prescribed low-intensity spring burns within the 8.6-year period (1985–93), their activity remained unchanged and therefore appear to form a fire-stable part of the litter fauna in foothill forests. Similar stability in arthropod populations was reported after a single and two consecutive spring burns (Neumann & Tolhurst 1991; Collett et al. 1993). However, despite the indications of an apparent 'nil effect' of the three consecutive fires on the litter fauna, the data indicates a significant increase in the Shannon-Wiener general diversity index as a result of significant increases in the Pielou community evenness index. Screening of the data suggested that these trends are primarily due to a marked decline in Coleopteran activity between the pre-burn period and the post-1988 - pre-1991 and post-1991 burn periods, as well as between the post-1985 - pre-1988 burn period and the post-

1988 - pre-1991 and post-1991 burn periods. Collett and Neumann (1995) noted a similar decline in Coleopteran activity following a second spring fire and identified the Staphylinidae, the largest group of all the Coleopteran families found at the FESA, to be the cause. It was also observed that the reduction in Coleoptera—or more specifically, in Staphylinid—activity, in response to spring fires, occurred evenly over the species' range. It is therefore likely that a similar trend may have applied after the three spring burns, although this has yet to be fully investigated.

However, as Coleopteran activity also declined at the Control Treatment Area during the periods following the second and third burns, it is uncertain whether the observed drop in activity was due to the burns rather than other environmental factors. For example, Coleopteran activity prior to the 1985 fire was similar to the post-1985 - pre-1988 fire period, and activity for the post-1988 - pre-1991 fire period was also comparable to the post-1991 fire period. Similarly, the significant increases in Dipteran activity in the periods after the second and third fires were also probably not fire-related due to a concomitant reduction at the Control Treatment Area. Such non-fire related factors may have included changes in the microclimate at the litter layer, greater availability in food and breeding habitat or simplification of the habitat due to the removal of surface fuel by burning, allowing for greater trapping efficiency (Majer 1980; Andersen & Yen 1985; Andersen 1988; Neumann & Tolhurst 1991; Collett 1998).

Effects of long-term rotation burning in spring and autumn

Of the taxa recovered during the study from the long-term rotation Treatment Areas, only the Formicidae (ants) appeared to be sensitive to prescribed burning. This is shown by the increase in activity at the autumn-burnt Treatment Area and, to a lesser extent, at the spring-burnt Treatment Area, in the post-second fire period, without corresponding increases at the Control Treatment Area. As the ants were not identified to species level, it is still uncertain whether the observed increase in activity was due to a single species or a range of species.

The first prescribed burns in both spring and autumn produced different responses among the Formicidae than did second spring and autumn burns, probably due to a much smaller reduction in fine fuel loads in the latter round of fires (Figure 4.2). Lower fine fuel loads at the Treatment Areas following the second post-spring-autumn fires may have created a more favourable microclimate for ant activity on the forest floor and possibly widened the foraging range and consequentially increased food availability. The detected different responses may also be due in part to an increase in trapping efficiency resulting from the simplification of the fire-affected habitat (Majer 1980; Andersen & Yen 1985; Andersen 1988; Neumann & Tolhurst 1991; Collett 1998).

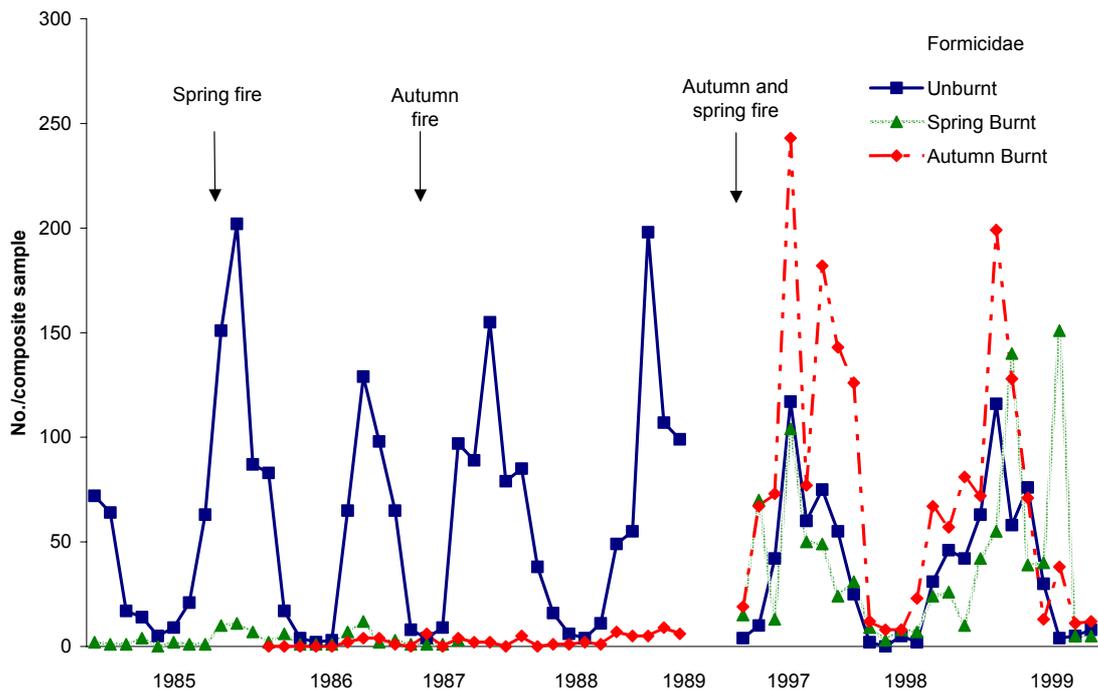


Figure 4.2 Temporal variations in monthly activity of Formicidae from March 1985 to February 1999 at the Control Treatment Area, from March 1985 to November 1987 at the spring burnt Treatment Area, from April 1986 to February 1989 at the autumn burnt Treatment Area, and at all three Treatment Areas from August 1997 to July 1999

Source: Collett 2000

Pitfall trapping

The broad taxonomic approach generally adopted throughout the study assumes that the findings for individual arthropod taxa give a cumulative estimate of the responses of the species within each of the larger taxa groups (Neumann & Tolhurst 1991). Sampling of arthropods was largely restricted to the litter surface and to pitfall trapping. While pitfall trapping is an excellent method for sampling certain taxa, such as Coleoptera and Formicidae, it may not be appropriate for sampling such taxa as Pseudoscorpionida, Isopoda, Chilopoda, Diplura, Thysanura, Neuroptera, Isoptera, Lepidoptera and Mantodea (Neumann & Tolhurst 1991; Collett 1998; Collett 1999; Collett 2000) that may either occupy ecological niches other than forest litter or may not be suited for sampling using the pitfall trap method (Collett et al. 1993; Collett 1999; Collett 2000).

Strong seasonal trends were identified among the major arthropod taxa collected, thereby strengthening the assertion by Stewart-Oaten et al. (1986), Collett and Neumann (1995), Neumann et al. (1995) and Collett (1998) that sampling for fire effects on taxon richness should ideally be done contemporaneously to achieve valid comparisons between burnt and Control Treatment Areas (Neumann et al. 1995). Ideally, similar lengths of time for collection should also be aimed for. In this study, the between-burns period corresponded to approximately five years of sampling compared to a single year of pre-burn sampling and two years of sampling after the second burn period. Thus, the higher taxon totals recorded for the between burns period do not necessarily imply that a substantial increase in taxon richness had occurred as a result of the first fire (Collett 1998).

Conclusions

Neumann and Tolhurst (1991) found that no long-term changes in activity or abundance among invertebrates occurred following a single low-intensity prescribed burn in spring or autumn, with Collembola and Diptera populations recovering to pre-burn levels within one year and earthworms within three years.

Collett et al. (1993) found that two low-intensity prescribed spring burns within three years of each other adversely affected Collembola and earthworm populations, although Collembola activity levels also declined at the Control Treatment Area, indicating there may have been other environmental factors apart from burning responsible for the decline. It was recommended that any scheduled fuel reduction or low-intensity burning for spring be done outside protracted drought periods and at intervals exceeding three years.

It appears that three low-intensity prescribed spring burns within eight years has minimal impact on litter arthropods in foothill forest although the effect of short-term rotational burns on the abundance of the Coleoptera and Diptera is less clear due to the significant change in activity levels also observed on the Control Treatment Areas. Burning in autumn is not considered to be any more beneficial than burning in spring as activity of both Coleoptera and Diptera was found to be similar after both the autumn and spring burns. Notwithstanding the above, it is recommended that any spring fuel reduction burning should be conducted at intervals of no less than three years to enable minimal impact on litter arthropods and, when a period of drought has intervened, the following fuel reduction fire should be delayed until the drought period has ended.

Neumann and Tolhurst (1991) found that the litter/upper soil fauna populations sampled in foothill forest appear to be 'fire-stable' when subjected to occasional low-intensity surface fire. This attribute would appear essential for forest types in which the risk of fire is high between summer and mid-autumn. However, the responses of individual species require further study, as does the effects of high frequency burning. It is therefore recommended that, for broad-scale fuel reduction burning in foothill forests, it is preferable to burn in autumn rather than spring and, for protection of invertebrates generally (in particular earthworms), to prohibit burning during protracted drought periods.

Recommendations

- Prescribed burning should be conducted at intervals of no less than three years to ensure that any fire-affected taxa are given the opportunity to recover or stabilize their populations prior to the impact of the next fire.
- Any broad-scale short-rotation prescribed fires should be scheduled for autumn rather than spring to minimise adverse effects on earthworms and a range of arthropod taxa.
- Burning should be avoided during protracted drought periods as this would exacerbate the negative impacts on decomposers such as Collembola and earthworms and, hence, further depress the decomposer lifecycle.

Bats

Martin Westbrooke, Pat Prevelt, Richard Adler, Marc Irvin and Jeanette Kemp

Introduction

In Australia, less is known about bats than any other order of mammal. They are frequently overlooked in mammal surveys, despite the fact they often occur in large numbers and may often be more diverse than other mammal groups present. The major reason for this is that bats are generally difficult to catch. Techniques for catching bats in flight have limitations, and researchers can put in much effort for few results. These limitations are due to the variety of foraging behaviour exhibited, with some species foraging above the canopy, some below and some even taking insects from the ground (McKenzie 1983). Research on Australian bats has been restricted mainly to colonial or cave-dwelling species with easily accessible roosts (Hall 1981). The study of forest-dwelling species has largely been neglected.

The primary objective of the bat research in the foothill forests of the Wombat State Forest was to compare the data from different burning treatments so that inferences can be made about the effects of fire regimes on bat species. These comparisons are difficult to make for bats as their ranges may cover several Treatment Areas.

Research results

Various researchers used trip lines, nest boxes, vertical trip lines, Skye Ultrasonic Detectors, harp traps, Anabat II bat detectors and radio telemetry, and recorded seven species of Microchiropteran bats in the FESAs (Table 5.1). New techniques were enlisted during the study as technologies developed and became more affordable. Bat activity in the FESAs in relation to abiotic factors (such as temperature, wind speed and lunar cycle) were first measured by van Gaans (1987) using a Skye Ultrasonic Detector. Bat activity was recorded in subsequent studies with recent technologies making measurement of the activity of individual species possible. The use of ultrasonic detectors is a means of identifying the presence of bats without the need for captures (Fenton & Bell 1981; Fenton 1982). Kemp et al. (1992) attempted to distinguish between calls of different species. This proved unsuccessful due to the inability to differentiate between the call of the captured bat and the call of other bats in the vicinity. Adler and Westbrooke (1999) used Anabat II bat detectors (Titley Electronics[®], Ballina, NSW). This allowed measurement of activity levels of individual species.

Table 5.1 Bat species recorded by various researchers in the Fire Effects Study Areas

Scientific and common name	van Gaans (1987)	Fowler (1988)	Kemp (1989)	Pretty (1990)	Burt (1991)	Preston (1992)	Irvin (1998)	Adler and Westbrooke (1999)
<i>Falsistrellus tasmaniensis</i> Eastern False Pipistrelle	*	*	*	*			*	*
<i>Nyctinomus australis</i> White-striped Freetail Bat	*	*					*	*
<i>Nyctophilus geoffroyi</i> Lesser Long-eared Bat	*	*	*	*	*	*	*	*
<i>Nyctophilus gouldii</i> Gould's Long-eared Bat	*	*			*	*	*	*
<i>Chalinolobus gouldii</i> Gould's Wattled Bat	*	*	*	*	*	*	*	*
<i>Chalinolobus morio</i> Chocolate Wattled Bat	*	*		*	*		*	*
<i>Vespadelus darlingtoni</i> Large Forest Bat	*	*	*	*	*		*	*

Bat activity related to fire treatment

The limited data from studies in autumn 1992 indicated differences in bat activity between Barkstead, Blakeville and Musk Creek FESAs (Preston 1992). The more comprehensive study in February 1999 (Adler & Westbrooke 1999), using multiple ultrasonic bat detectors, found no significant difference between the activity levels of bats in any of the fire effects treatments at Blakeville and Musk Creek FESAs. This was despite variations in the treatment histories of these areas. The comparative activity levels at each Treatment Area suggest the value of habitat to bats is equal for all Treatment Areas.

Bats partition their environment into microhabitats by means of their flight morphologies and echolocation designs (Crome & Richards 1988). As data from this survey does not show the activity levels of individual species within different Treatment Areas, it may be premature to suggest that prescribed burning does not have an effect on bat activity (Adler & Westbrooke 1999).

Insectivorous bats have the ability to travel large distances while foraging. *Chalinolobus gouldii* and *Nyctophilus geoffroyi* have been observed travelling 6-12 km to forage (Lumsden & Bennett 1996). Survey of the FESAs only sampled a proportion of the potential foraging area of bats. Results may therefore not give a true representation of foraging effort on a landscape level (Adler & Westbrooke 1999).

The differences in activity between Treatment Areas is likely to be due to lower temperatures during the month of April compared with February; low temperatures being the most probable factor in a decline in activity by insectivorous bats (O'Neill & Taylor 1988). If comparisons of survey results are to be made between different Treatment Areas, it is essential that surveys be done simultaneously or on nights of similar temperature (Kemp 1989).

Temperature

The single most important factor controlling bat activity was temperature (van Gaans 1987; Fowler 1988), with activity limited below 9–10 °C (van Gaans 1987; Kemp et al. 1992). Preston (1992) found an almost linear relationship between bat activity and air temperature. Although Kemp et al. (1992) found a natural logarithmic relationship between bat activity and air temperature, this was based on relatively fewer observed species. Pretty (1990) also found a correlation between temperature and bat activity. Fluctuations in temperature have also been found to affect insect activity (Hill & Smith 1986). O'Neill and Taylor (1988) found insect activity and the number of bats trapped showed similar binomial distributions, suggesting that bat activity is closely related to invertebrate activity. Preston (1992) notes foraging activity of bats will increase with insect activity, which in turn increases with temperature.

Kemp (1989) noted two nights of very high bat activity, which were not at a significantly higher temperature than other survey nights. Kemp suggested that a combination of factors might contribute to the activity, including some not investigated, such as availability of insects and weather patterns over the previous few days. Fowler (1988) and van Gaans (1987) found captures were lowest on nights of a full moon.

Diurnal roosts

The tendency for Australian foothill forests to burn with some regularity requires that forest dwelling species be either more aware of alternative roosts, or be less selective in the nature of the roost (Hall & Woodside 1989). These dynamic and unpredictable features of forest habitats would be reflected in the type of social organisation of the species, colony size and tendency to migrate (Hall & Woodside 1989).

Vespertilionid bats may enter periods of torpor (prolonged inactivity) to overcome inclement conditions, which improves seasonal survival and longevity (Hall & Woodside 1989). The characteristics of roosts used for torpor are unknown and may be different to those used during seasons of higher activity.

Tree cavities offer protection against fluctuations in ambient temperatures and humidity (Kunz 1982). Therefore the type of roosts used by different bat species to some degree reflects the tolerance of particular bat species to different roosting microhabitats. This is supported by the contention that the origin of a hole is of little importance to a bat, provided certain other requirements are met (Tidemann & Flavel 1987).

Tidemann and Flavel (1987) suggest that, in a regional context, one species of tree may provide more roosts than others but, overall, particular trees will be unimportant. Different roost types offered by particular tree species may be more suitable to one species than another. Whilst an individual tree may be unimportant, a natural representation of vegetation communities throughout the forest probably is (Irvin 1998).

Vespadelus darlingtoni prefer fissures in *Eucalyptus ovata* and, with 94% of *E. ovata* occurring in gullies, gully ecosystems are important to the species' ecology (Irvin 1998).

The partitioning of resources by bats to avoid niche overlap has been recognised in flight patterns and foraging behaviour (Kunz 1982; Fullard et al. 1991; McKenzie et al. 1995). The preference by bats for certain roost types is another example of a resource partitioning strategy. Mixed species colonies are not uncommon (Kunz 1982) and this strategy may be important where roosts suitable for several bat species are either unavailable or scarce.

Although many bats appear to be loyal to certain preferred roosts, there is a growing recognition that bats establish and maintain familiarity with one or more alternate roosts (Kunz 1982). Irvin (1998) found this fidelity to a general area was exhibited by *V. darlingtoni* and *N. geoffroyi*. Lunney et al. (1988), in a study on roost selection, found that *N. geoffroyi* and *N. gouldii* both roost and forage within a range of less than 1 km from their cluster of roosts. Likewise *C. morio* and *C. gouldii* confined their activity to an area within 5 km of their roosts (Lunney et al. 1985).

Roost fidelity can be affected by such factors as the relative abundance and permanency of roost sites, the proximity and stability of food resources as well as response to predator pressure and human disturbance. Roost fidelity may change seasonally and can be affected by reproductive condition, sex, age and social organisation (Kunz 1982). If ecological requirements of bats are in short supply or, conversely, are abundant in localised areas, rather than show fidelity to one roost area, it is possible that several preferred areas may be frequented.

Forest-dwelling bats experience changeable environments and are not likely to make regular long-distance forays. Rather, a high degree of home-site fidelity in the short term, with regular surveillance of nearby habitats, is to be expected (Kunz 1982; Lunney et al. 1988; Taylor & Savva 1988; Hall & Woodside 1989). Irvin (1998) found six roosts were used on multiple occasions, with the distance between day-to-day roosts in many cases being only a few hundred metres. Kunz (1982) describes the disadvantages of tree cavities in offering limited roosting space for colonial species as they eventually rot and fall, requiring the periodic relocation of inhabitants. This suggests that forest bats are able to adapt to changes in the environment. These changes may include fuel reduction burning and silvicultural practices, though relocation to a new roost is only possible if new roosts exist. Studies on the choice of roost sites by Microchiroptera in south-eastern Australia by Tidemann and Flavel (1987) and Lunney et al. (1988) concluded that, amongst other things, it is important to retain trees that are of a size to contain bat roosts.

It is thought that fire is a factor that can contribute to the initiation of hollows. Inions et al. (1989) suggested that high-intensity fire would significantly reduce the time before hollows form. Since low-intensity burns are observed to remove most of the loose bark on the lower parts of eucalypt species, the availability of this type of roost site may be reduced (M. Westbrooke², pers. comm., 2000).

Roosts were found in a range of tree sizes and it is significant that different roost types are provided by each tree size (Irvin 1998). Irvin observed bats using roosts in trees above and below 350 mm diameter at breast height (DBH). Lumsden et al. (1994) found *C. gouldii* and lactating female *N. geoffroyi* were located in trees of the size class 100-140 cm DBH, which are probably important to their roosting ecology. Small hollows and splits form as the precursors of larger ones, so bats are probably among the first mammals to recolonise regenerating stands of trees (Tidemann & Flavel 1987).

Bats need to drink in summer. Tidemann and Flavel (1987) suggest it is probably significant that all roosts they found were within a few hundred metres of water. Irvin (1998) recorded all roosts within one kilometre of a reliable water source, however it is probable that free-standing water occurred closer in many cases. It is common also for well-watered sites to act as insect breeding grounds (Tidemann & Flavel 1987) and, as a result, foraging sites. The occurrence of many bat roosts in gullies may be related to the accumulation of permanent and semi-permanent water in these areas. Given the natural availability of water in the foothill forests of central Victoria, it is thought that, while fire dams may affect the distribution of bats, they are unlikely to have an impact on overall populations.

Gullies are protected from timber harvesting in the form of buffer strips along watercourses and drainage lines. Within the study areas these buffer strips of native vegetation meander through pine plantations as well as native forest. Irvin (1998) noted that several roosts were found within these 50-m buffer strips, with bats apparently unaffected by the nature of the surrounding plantation. Whether bats use these buffer strips as a corridor to travel between forage, watering and roosting sites was not determined.

Most research has focused on the numbers of bats or the amount of bat activity. While this research has provided information on species richness in the region, further research focusing on the specific requirements of each bat species will enhance future management. It should not be assumed that all seven bat species studied have similar requirements. Further research should focus on specific requirements of the different species.

² Professor Martin Westbrooke, Head School of Science and Engineering, University of Ballarat.

Hollows

Calder et al. (1979) suggested that natural hollows are likely to be a fundamental habitat requirement of the mammal and bird species observed in them and that the depletion of natural tree hollows may threaten the survival or decrease the abundance and diversity of some species. Bats may choose natural hollows in preference to otherwise suitable artificial nest boxes. Artificial nest boxes can complement existing habitats as well as provide a means for long-term monitoring. In regard to monitoring, if bats use hollows at a certain time of year, then their presence or absence could be established by inspecting hollows during that time (Calder et al. 1979).

The significance of isolated trees close to forests needs to be investigated for bats. Recent studies overseas have indicated that some species of bats avoid thicker vegetation or 'clutter' (Grindal 1996). The extent to which the reduction in understorey by fire has the potential to change the species mix within bat guilds is not known in Australia.

Conclusions

Habitat protection is vital to the conservation of bats as different bat species have different requirements. The data set out here from the research so far provides a good knowledge-base from which to question current management practices where appropriate, and refine questions for further study. Allocation of resources to expand our currently insufficient knowledge of bats will pave the way for current and future information to be applied to forest management.

Bats travel at least 1-5 km while foraging, therefore the study of bat ecology on a landscape scale, much larger than the FESA Treatment Areas is necessary. Since bats move widely in the landscape and are not confined to forest habitat, extending studies beyond forest boundaries to adjacent treed areas may also be required.

Very little bat activity was recorded when air temperatures were below 10 °C, with activity greatest at around 20 °C. Temperature at the time of study is critical to the level of activity observed. There is a need also to assess populations of flying nocturnal insect in each fire treatment area and relate these to bat activity. Impacts of fire on insect populations in these areas will probably also be reflected in the observed activity of bats.

The use of Anabat to determine the distribution of bats in the forest landscape is in the early stages of development. The recent development of linking Anabat detectors directly to computers in the field will greatly facilitate our understanding of bat foraging activity. Rather than gaining activity data for insectivorous bats in general, individual bat species will be recognised and assessed via their echolocation calls.

Tree hollows do not form within the span of any currently acceptable fire rotation. All insectivorous bats of this study require some form of hollow or cavity in trees for shelter, rearing of young and protection from predators. Whilst all eucalypt species of the FESAs form hollows, the type of hollow produced depends partly on the tree species and partly on the stage of maturity of the tree. No wintering roosts (used during periods of torpor) or maternity roost trees were identified in the present study.

In this study *Vesperdelus darlingtoni* were found to prefer roosts in *Eucalyptus ovata*, which grows primarily in wet gullies and drainage lines. Low-intensity prescribed burning strategies should be examined to determine the extent to which fire imposes change in the pattern of tree species within forest communities. The preferred habitat of various bat species will depend on this information.

Radio telemetry studies of insectivorous bats are possible within the sometimes dense and rugged terrain of foothill forests. Radio telemetry is a viable means of collecting diurnal roost information for insectivorous bats. Further telemetry studies will provide important information of the effects of fire and silvicultural management.

Recommendations

The information gained from this research program is insufficient to predict the effects of repeated low-intensity burning or other management practices on bats. However, to date, the data suggests no effect in the differing Treatment Areas. Ongoing research to better understand bat ecology and management effects on bat ecology should be undertaken. Attempts to answer practical research questions can provide a sound knowledge base, which can be applied as a management tool for sustainable use of forest ecosystems.

There is therefore a need to:

- obtain a more detailed understanding of the use of Treatment Areas by individual bat species by using Anabat detectors linked to laptop computers for the collection of field data. Wherever possible, this research should aim to collect data prior to burning, and at a range of strategic times in the months following burning.
- conduct further assessments of roost site usage by application of radiotelemetry tracking of the full range of seven species present in the Wombat State Forest. The use of roosts by bats for different purposes throughout the night and throughout the year is not clear. Long-term monitoring of the use of roosts by bats is needed to ensure fire does not detrimentally impact on the roosts.
- initiate studies on the relationship between forage areas used by bats and aerial insect activity within study areas
- investigate the movement of bats within cluttered (dense understorey) and burnt areas, since reduction of understorey by fire may influence the bat species guilds present
- further investigate the relationships between where bats are active, why they are active and how far they will travel both in the short and long term.

Reptiles

Rob Humpries, Martin Scuffins, Donna Moody, Marc Irvin and Martin Westbrooke

Introduction

Detailed ecological studies of Australian reptiles remain relatively rare, which impacts significantly on the community's ability to develop appropriate strategies to conserve them (Cogger 1993). Most information related to the effects of fire on reptiles has been from studies in mallee woodlands, heathlands and northern Australian savanna forests where reptilian diversity is high. Few studies have been undertaken in southern temperate areas (Wilson 1996).

Australian skinks make use of the full range of habitats occupied by terrestrial vertebrates, with the possible exception of the outer tree canopy (Hutchinson 1993). Much of the physical forest habitat is at least partially consumed as fuel for fires. The degree to which different microhabitats are modified by a fire event and the rate of recovery in the post-fire period will determine the impact of the fire and fire regime on different species (Wilson 1996).

Research Results

A number of researchers recorded more than 1000 individual reptiles with a total of eight scinid lizard species and one elapid snake, across the five FESAs. The species recorded and survey techniques used are shown in Table 6.1.

The short-term fire responses of the five common scinid lizards to single spring and autumn low-intensity prescribed fires, the effects of repeated spring and autumn fuel reduction burning on populations of reptiles as well as their activity patterns and microhabitat utilisation were investigated through a number of studies.

Effects of fire on microhabitat and reptile abundance

Scuffins (1994) found significant differences in microhabitats between treatments. Pooled data from all five Treatment Areas showed an overall significant difference in the grass/herb cover and bare ground between treatments. As expected, there was less bare ground in the Control Treatment Areas and the spring and autumn ten-year treatments respectively, with greater amounts found in the spring and autumn three-year treatments.

Scuffins (1994) found litter levels were still significantly different from the Control Treatment Area 26 months after burning in the spring and autumn three-year Treatment Areas. The combined differences in litter cover, and the lower percentage cover of grasses and herbs in these Treatment Areas may be responsible for the lower numbers of *Pseudemoia coventryi* and *P. entrecasteauxii* in these blocks.

Sixty-five per cent of *P. entrecasteauxii* were observed in litter, although grasses and herbs (28%) and logs (28%) were utilised in the Control Treatment Areas (Scuffins 1994). The high dependence of *P. entrecasteauxii* and *P. coventryi* on litter and ground vegetation microhabitats may account for the lower numbers of these species in burnt Treatment Areas. In those Treatment Areas where time since fire is greater (spring and autumn ten-year rotations), numbers recorded were intermediate between those of spring and autumn three-year Treatment Areas and the Control Treatment Areas (Scuffins 1994).

Table 6.1 Reptile species recorded by each researcher from surveys and survey techniques

Researcher	Fire Effects Study Area	Survey technique	Species recorded	
			Scientific name	Common name
Moody (1991)	Blakeville * Barkstead *	Pitfall-driftline Transect searching Stationary census	<i>Pseudemoia coventryi</i> <i>Pseudemoia entrecasteauxii</i> <i>Nannoscincus maccoyi</i> <i>Pseudechis porphyriacus</i> <i>Eulamprus tympanum</i>	Coventry's Skink Grass Skink Maccoy's Skink Red-bellied Black Snake Southern Water Skink
Humphries (1994)	Blakeville Barkstead Musk Creek *	Pitfall-driftline Walked transects Opportunistic Stationary census	<i>Egernia whitii</i> <i>Lampropholis guichenoti</i> <i>Pseudemoia coventryi</i> <i>Pseudemoia entrecasteauxii</i> <i>Nannoscincus maccoyi</i> <i>Pseudechis porphyriacus</i> <i>Eulamprus tympanum</i> <i>Tiliqua nigrolutea</i> <i>Tiliqua scincoides</i>	White's Skink Garden Skink Coventry's Skink Grass Skink Maccoy's Skink Red-bellied Black Snake Southern Water Skink Blotched Blue-tongued Lizard Eastern Blue-tongued Lizard
Scuffins (1994)	Blakeville Barkstead Burnt Bridge Musk Creek Kangaroo Creek	Opportunistic Stationary census	<i>Pseudemoia coventryi</i> <i>Pseudemoia entrecasteauxii</i> <i>Lampropholis guichenoti</i> <i>Eulamprus tympanum</i> <i>Tiliqua nigrolutea</i>	Coventry's Skink Grass Skink Garden Skink Southern Water Skink Blotched Blue-tongued Lizard

Taxonomy is consistent with Cogger (1993).

* Pitfall-driftline traps used at these Fire Effects Study Areas only.

Numbers of *Pseudemoia coventryi* and *P. entrecasteauxii* were significantly different between Treatment Areas (Scuffins 1994). Scuffins also found numbers of both species recorded were significantly lower in burnt Treatment Areas (notably S3 Treatment Areas, 32 months after burning and A3 Treatment Areas, 18-32 months after burning) than in the Control. Humphries (1994) found *P. coventryi* and *Nannoscincus maccoyi* forage on and in the ground/litter layer, the microhabitat most affected by burning. The relatively high recorded abundances of these species immediately after burning were probably due to their increased detectability as a result of reduced cover (Humphries 1994). Webb (1985) found 92% of *P. coventryi* were observed in litter. Within the fire effects study Scuffins (1994) found 49% of *P. coventryi* were observed in litter and 17% in association with grass/herb cover. The use of grasses and herbs was lowest in burnt Treatment Areas (0-14%) and highest in Control Treatment Areas (64%), suggesting that this species will utilise any available cover at ground level, probably switching from one cover type to another when ground vegetation or litter is consumed by fire. This habitat switching may be responsible for the low correlation coefficients that were obtained (Scuffins 1994).

Although no pre-burn observations of microhabitat use by *N. maccoyi* were available to Humphries (1994), Robertson (1981) identifies its preferred microhabitat as deep, moist litter. All observations of this species were after fires, with almost all specimens found sheltering under fallen logs. The lack of *N. maccoyi* sightings in the Control Treatment Area, even under logs, suggests a temporary shift in microhabitat selection after fire due to the loss of preferred foraging sites with the loss of litter. The decrease in observed abundance of this species after a further 12 months of post-burn recovery probably indicates the return of *N. maccoyi* to the recovering litter layer where this cryptic species is difficult to detect (Humphries 1994).

Humphries (1994) found the effects of single spring and autumn prescribed fires to be similar. Four of the five species Humphries studied either depended on or largely relied on the litter layer and showed fluctuations in abundance. The rapid recovery rate of fine fuels following fire (Tolhurst et al. 1992b) resulted in litter levels reaching pre-burn levels two to four years after burning. The importance of this rapid recovery of a major habitat component is shown by the increase in the relative abundances of *P. coventryi* 28 months after spring burning (Humphries 1992). *Nannoscincus maccoyi* (a species dependent on deep litter) is likely to be locally eliminated by high frequency and moderate- to high-intensity burning regimes (Humphries 1992).

The abundance of *Eulamprus tympanum* remained relatively stable following both spring and autumn fires (Humphries 1994; Scuffins 1994). This is likely to be due to the abundance and condition of its primary habitat (fallen logs and branches) not being significantly affected by fire (Tolhurst et al. 1992b). Scuffins (1994) noted that highest numbers of *E. tympanum* were in the autumn ten-year treatments at Barkstead and Musk Creek seven years after burning. It is not known why this is the case, but it may be due to the degree of decomposition of logs in those treatments or inherent differences in the Treatment Areas before burning. Webb (1985) found that *E. tympanum* was rarely located more than 1 m away from a log. Scuffins (1994) also found this species depended on logs with 70% of individuals found either on logs or immediately adjacent to them.

Humphries (1994) noted that lizard abundance appeared to be high after both spring and autumn fires, however this increase was often followed by a marked decrease some 7 to 14 months after the fire. This response is probably a result of increased detection of lizards due to a lack of cover initially after the fire, and decreased detection of lizards once litter cover has increased. The amount of cover of litter, shrubs and logs can all effect the observability of reptiles (Scuffins 1994), especially when undertaking transect counts (Humphries 1992). Despite this, Scuffins (1994) recorded greater numbers of *P. coventryi* and *P. entrecasteauxii* in locations where it would be expected that detection rates would be low due to dense cover of litter, ground vegetation and logs.

Variation in lizard abundance was considerable between Treatment Areas prior to burning, and between years within Treatment Areas. This demonstrates a number of limitations in reptile census techniques such as local small-scale differences in microhabitat between apparently similar Treatment Areas and variability and observer bias at these Treatment Areas (Humphries 1994). Humphries (1994) also found local differences in microhabitat suitability between apparently similar Treatment Areas (i.e. vegetation cover, number and suitability of available basking and sheltering sites) may explain variation in reptile numbers from location to location.

Burning regimes

The impact of specific burning regimes on particular species will depend on the ecological requirements of the species in question (Heatwole & Taylor 1987). The degree to which the ecological requirements are affected, and their relative rates of recovery, will largely determine the impact of a particular burning regime on the reptile community and species present. Fyfe (1980) and Caughley (1985) found that burrowing species in the arid interior were more common at recently burnt sites than long-unburnt sites. Burrowing species avoided the immediate impact of the fires, and the resulting openness of communities following burning favoured their foraging requirements. Species that depend on ground litter and vegetation were most affected by the fires; only one out of six species classified as requiring litter shelters was recorded after the fire in Fyfe's study.

Fires generally affect 60–80% of the area being burned (Hodgson & Heislars 1972). The unburnt areas left in the small-scale mosaic of burnt and unburnt areas resulting from a fuel reduction burn are crucial to local survival and can act as the source for recolonisation of burnt areas by both invertebrates (Leonard 1972) and small mammals (Leonard 1972; Heislars 1980). Humphries (1992) notes that these unburnt patches are also likely to act as refuges for small skinks.

Most lizard species studied have definite home-ranges (Heatwole & Taylor 1987). This behaviour permits lizards to familiarise themselves with their immediate environment and thus the nearest point of shelter in case of attacks from predators or danger. The sizes of lizards' home-ranges vary considerably. Mather (1978) determined the mean home-range of *E. tympanum* to be 7.6 m² in open-forest at Toolangi in central Victoria. It is therefore likely that, in a low-intensity fire, entire or portions of existing home-ranges of *E. tympanum* and other small skinks would be left relatively intact.

Clearly no single fire regime is favourable to all species. There is no data on the optimum frequency of fire for conserving reptile species. However, a fire regime that allows the build up of litter and other microhabitats as well as adequate food supplies should enable reptiles to thrive (Suckling & Macfarlane 1984).

Direct effects of fire on reptiles

Lunney et al. (1991) noted that the survival of *Lampropholis guichenoti* during fire is attributed to its ability to take refuge underground. Scuffins (1994) recorded only five individual *L. guichenoti*; three associated with litter and two with logs. These observations are too few for conclusions to be drawn.

The effect of temperature on the number of reptiles recorded during the census counts follows a normal distribution (Scuffins 1994). Preferred temperatures in burnt Treatment Areas appear to be around 25 °C and approximately 27–28 °C in the Control Treatment Areas. The activity of lizards at higher temperatures in the Control Treatment Areas is probably due to the presence of the vegetation cover in those areas (Scuffins 1994). Lunney et al. (1991) suggested that, after fire in the Mumbulla State Forest (southern NSW), the critical thermal maximum temperature of Australian scincid lizards would be quickly reached and sustained for longer in the exposed and blackened ridges of recently logged forest in the immediate post-fire weeks than in other habitats. It was noted in the Wombat State Forest that high survival rates of *L. guichenoti* in gullies was consistent with these being moister, less exposed habitats (Scuffins 1994).

Humphries (1994) made observations of species surviving or being killed by low-intensity fire. During the autumn burn at Barkstead in April 1987, the comparatively strong-limbed species, *E. tympanum*, was observed moving ahead of the fire front—which had a maximum forward rate of spread of 1.2 m/min (Tolhurst et al. 1992b)—and taking refuge in hollow logs and stumps. The security of these refuges depends on a number of factors, including their size, degree of decomposition, seasonal dryness and fire intensity. At the Musk Creek FESA, Humphries (1994) collected numerous individuals of *N. maccoyi* from under large logs the day after both spring and autumn burns. *Nannoscincus maccoyi*, a short-limbed and slow-moving species, often showed signs of being burnt in the fire, with scales and limbs melted and fused together. Many individuals probably died as a direct result of the fire.

Predation

Increased predation due to the lack of suitable cover is often assumed to reduce lizard abundance following burning (Fox 1978). Skinks have the ability to drop their tails when alarmed, a trait known as tail autotomy (Jenkins & Bartell 1980). The obvious benefits of tail loss relate to escapes from predation attempts. The incidence of tail loss in a population of lizards can give an indication of predator pressure (Robertson 1981). Tail loss in skinks can also be caused by intraspecific competition. These interactions may have increased after burning due to competition for limited resources or increased levels of stress. Although small sample sizes were used, Humphries (1994) found a statistically significant difference between burnt treatments and Control treatments of lizards with and without tails.

Raptors, such as *Accipite fasciatus* (Brown Goshawk), were observed running over bare ground looking for reptiles after the spring burn at Burnt Bridge (Loyn et al. 1992). *Dacelo novaeguineae* (Kookaburra) and *Halcyon sancta* (Sacred Kingfisher) have also been seen feeding on reptiles in the Wombat State Forest, with the numbers of Kookaburras feeding from bare ground generally increasing after fire (R. Loyn³, pers. comm., 2000).

Limitations of survey techniques

Scuffins (1994) and Humphries (1994) found significant differences in skink abundance between Treatment Areas and, in the latter study, between years within a single Treatment Area. Webb (1985) and Heatwole and Taylor (1987) recorded similar results. Moody (1991) found that air and soil temperatures and cloud cover could potentially impact on reptilian activity. Reptiles were most active when soil temperature was between 15 and 25 °C, air temperature was 20–30 °C and there was little or no cloud cover. The variation in reptile activity in relation to temperature, cloud cover and suitable microhabitat availability (Webb 1985) indicates that a standard, repeatable census method with equal sampling intensity is crucial for a study, particularly when comparisons between relative abundance in sites with different treatment histories are required. Any variation in sampling techniques (i.e. sampling frequency, observer variation) or abiotic factors of FESAs and/or Treatment Areas can lead to inconsistencies when analysing data. Stationary census counts combined with active searching under logs, rock and other debris was found to give the best estimates of species abundance (Scuffins 1994). Scuffins found no significant difference in temperature or amount of cloud cover, suggesting that variation in skink numbers was due to other factors.

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Conclusions

None of the reptile species studied favoured a single given burning treatment. If diversity and rapid recovery of reptile populations are management objectives, low-intensity prescribed fires need to be managed to ensure that refuges of suitable microhabitat remain.

All species observed before burning were still present following a single spring or autumn low-intensity fire, however the numbers of each species were influenced. The conclusions discussed below have been drawn from knowledge produced from research in mixed eucalypt foothill forests. The success of reptiles in these forests depends on the management of their habitat and, as such, issues relating to microhabitats are repeatedly raised.

The impact of specific burning regimes on particular species will depend on the ecological requirements of the species in question. Those species that depend on ground litter and vegetation are most affected. Some skinks (*E. tympanum* in this study) are able to outrun low-intensity fires, although some are likely to be injured or die as a direct result of burning. The availability of a range of microhabitats will benefit a number of reptile species.

Unburnt microhabitats (particularly logs, deep beds of leaf litter and areas frequently missed by low-intensity fire, such as gullies) provide important refuges as well as food, shelter and oviposition sites in the post-fire period. While some reptiles that survive fire can utilise alternative microhabitats, the rapid recovery of preferred microhabitat components is of major importance in the post-fire survival and recovery of reptiles.

The small-scale mosaic of burnt and unburnt areas that generally results from low-intensity prescribed fires are crucial to local reptile survival. Due to their small home-ranges, some species of skinks are likely to be eliminated from local areas by high frequency and moderate- to high-intensity burning regimes. Widespread patchiness is beneficial to reptilian survival and recovery. Fires with large perimeter-to-burn ratios are preferred because they contain more refuge areas inside the fire perimeter and allow for recolonisation by individuals from adjacent unburnt areas as habitat recovers.

The short-term removal of shrub and ground cover due to infrequent prescribed fires may benefit some species (*E. tympanum*, *P. coventryi* and *P. entrecasteauxii*) through an increase in basking sites, but may disadvantage other species by removal of their food source and means of protection.

Choosing appropriate survey techniques is paramount in obtaining useful results. The rarity or absence of some species in the survey results discussed here is a reflection, at least in part, of their cryptic nature rather than their relative abundance. The dense cover of leaf litter, logs and ground vegetation prior to burning conceals these cryptic species compared to the bare, dark earth immediately after fire and makes definitive statements regarding reptile abundance somewhat artificial.

The longevity of small skinks can vary from one to seven years, and is probably influenced by environmental factors such as climate. Further research of reptile life histories is required to avoid generalisations in the application of longevity knowledge to the management of prescribed fire.

Recommendations

From this work it is recommended that:

- burning regimes create a mosaic of burnt/unburnt patches across the burnt area
- fires with large perimeter:area ratios are planned to enhance the rate of recolonisation
- logs are retained where possible.

Terrestrial Mammals

Rob Humphries, Peter Kambouris, Martin Scuffins, Kevin Tolhurst and Martin Westbrooke

Introduction

Whilst the majority of the research concentrated on small terrestrial mammals, incidental observations and several associated projects have provided some information on larger mammals that occur in the general area. Small carnivorous marsupial (Dasyuridae) genera such as *Antechinus* and *Sminthopsis* and native members of the Rodentia form the major component of the small mammal fauna of mixed eucalypt foothill forests in south-eastern Australia.

Johnston et al. (1983) note that, of all the animal groups, mammals have received by far the most study in terms of fire effects in Australian forest ecosystems. Many mammals, even highly mobile species such as kangaroos and wallabies, are killed in high-intensity fires. Conversely, few deaths are recorded in low-intensity fires, as individuals are able to outrun the fire, take refuge underground or under rocks, or escape to unburnt patches (Johnston et al. 1983).

Mammals colonise burnt areas following the recovery of plants that make up their habitats. In south-eastern Australia, Dunnarts generally prefer the early stages of regeneration following fire. Later, three to four years after fire, species such as *Rattus fuscipes* thrive. There is evidence that some mammal populations are favoured by a 20–40 year period without fire (Johnston et al. 1983). Pyric response patterns of small mammal species are closely tied to their shelter, food and breeding requirements (Friend 1993).

Wildfire removes the ground cover upon which *R. fuscipes* depends. Individuals usually survive a fire by remaining in their burrows, but fail to breed in the following season. A burnt area thus becomes available for colonisation by juveniles dispersing from unburnt areas (Lunney 1983).

To understand how fire affects small mammals, it is necessary to consider the habitat requirements of individual species and the way in which fire modifies vegetation structure and floristic composition.

Research results

Five hundred and thirty-nine individuals representing six small mammal species were trapped a total of 1175 times during Humphries' (1994) study, which gathered data over the three years 1986–88. Species trapped by Humphries were *Antechinus agilis*, *Phascogale tapoatafa*, *R. fuscipes*, *R. lutreolus* and the introduced *R. rattus* and *Mus musculus*. Only *A. agilis* and *R. fuscipes* were trapped in sufficient numbers to justify detailed analysis. Scuffins (1994) captured only *A. agilis* and *R. fuscipes*. He trapped 72 individuals a total of 92 times (data gathered in 1994). Other species of terrestrial mammals recorded for the various FESAs are presented in Table 7.1.

Humphries (1994) found fuel reduction burning had little or no short-term (two to three years) effect on populations of *A. agilis* and *R. fuscipes*, although populations of both species were affected during the first 18 months post-fire.

Table 7.1 Mammal species recorded (excluding bat species) for each study associated with the Wombat Fire Effects Study Areas

	<i>Phascogale tapoatafa</i> (Koala)	<i>Antechinus agilis</i> (Brown Antechinus)	<i>Antechinus swainsonii</i> (Dusky Antechinus)	<i>Phascogale tapoatafa</i> (Brush-tailed Phascogale)	<i>Acrobates pygmaeus</i> (Feathertail Glider)	<i>Petaurus breviceps</i> (Sugar Glider)	<i>Petauroides volans</i> (Greater Glider)	<i>Trichosurus caninus</i> (Mountain Brushtail Possum)	<i>Trichosurus vulpecular</i> (Common Brushtail Possum)	<i>Pseudocheirus peregrinus</i> (Common Ringtail Possum)	<i>Rattus fuscipes</i> (Bush Rat)	<i>Rattus lutreolus</i> (Swamp Rat)	<i>Rattus rattus</i> (Black Rat)	<i>Mus musculus</i> (House Mouse)	<i>Wallabia</i> sp. (Wallaby species)	<i>Canis vulpes</i> (Red Fox)	<i>Oryctolagus cuniculus</i> (European Rabbit)
Kemp (1989)	*						*	*		*							
Pretty (1990)					*												
Humphries (1994)		*									*	*	*	*			
Scuffins (1994)		*									*						
Brookman (1997)	*						*		*	*							
Kambouris (1998)		*									*						
Baker (1998)		*									*						
Solly et al. (1999)		*	*								*						

Note: Taxonomy according to Menkhorst (1999)

Survival and post-fire recovery

The ability of small mammal populations to recover from fire is related to the survival rate of individuals in burnt areas, the presence of unburnt refuges and the rate of vegetation recovery. Refuges are an important source of recolonising individuals (Leonard 1972; Heislars 1980), particularly if the survival rate in burnt areas was low.

The amount of refuge habitat remaining after the fire depends on fire intensity (Suckling & Macfarlane 1984) and the amount of the total fuel actually burnt (Tolhurst et al. 1992a). A fuel reduction burn generally results in 60-80% of the area being burnt to some degree, thus creating a small-scale mosaic effect (Hodgson & Heislars 1972). The amount of fallen logs on the ground also significantly increases the patchiness of a burn. Fallen logs are less likely to burn in low-intensity fires (Humphries 1994). Humphries (1994) found the proportion of area burnt during his study was relatively high (69-99%). Fire intensity and the proportion of total fuel burnt are largely dependent on prevailing weather conditions.

Humphries and Tolhurst (1992) reported significantly more *A. agilis* in areas with patches of unburnt habitat. Unburnt islands can act as refuges for litter invertebrates, thus increasing the food supply for predatory small mammals.

Antechinus agilis

Humphries and Tolhurst (1992) noted significant declines in *A. agilis* numbers after the 1985 spring and 1987 autumn fires. Spring fires reduced abundance of this species by approximately 50% for 12 months after the fire. Humphries' (1994) results suggested that recovery of the population in the second year was due to a combination of the high breeding rate of *A. agilis*, a significant proportion of the population surviving within the burnt area and the rapid accumulation of leaf litter in which to forage. This rapid recovery after low-intensity fire concurs with the findings of Cowley et al. (1969), Linsdell (1969) and Leonard (1970 and 1972). However, a significant population decrease was observed after a single autumn fire and numbers had not recovered after two mating periods, 16 months post-burning (Humphries 1994). Fox and McKay (1981) also observed that *A. agilis* did not appear in population numbers until well into the second year after fire. Newsome et al. (1975) found a fire at the Myall Lakes National Park (NSW) had little affect on numbers of *A. agilis*.

Solly et al. (1999) suggested the higher number of *A. agilis* individuals captured compared with other species may indicate that this species recovers more quickly from the effects of fire. Humphries and Tolhurst (1992) and Whelan et al. (1996) observed that *A. agilis* populations recover well after fire, soon moving back into burnt areas to take advantage of the bare ground to forage for soil invertebrates.

Rattus fuscipes

Humphries (1994) found the population of *R. fuscipes* declined following fire in all Treatment Areas. This result was supported in Lunney et al. (1987), who found that fire dramatically reduced its numbers.

Rattus fuscipes numbers showed seasonal fluctuations, with the highest densities (6/ha) in late winter - early spring when juveniles entered the population. Population numbers were lowest in autumn (0.5/ha). Adult males and females, which had bred in the previous year, tend to die at this time, and this is before juveniles are first trapped (Humphries 1994).

The species composition of the dense sedgy vegetation preferred by *R. fuscipes* varied between FESAs. At Musk Creek and Blakeville the gully vegetation was predominantly *Lomandra longifolia*, which was more open and burnt more easily. This rush community supported a smaller *R. fuscipes* population than the more developed sedge-rush communities of the wetter gullies. In areas dominated by *L. longifolia*, *R. fuscipes* was not trapped in the first two years after the fires (Humphries 1994).

In the Humphries and Tolhurst (1992) study, the *R. fuscipes* population took three breeding seasons (two years) to recover from a single spring fire when more than half of the habitat was burned, compared with recovery after one breeding season following a single autumn fire, when less than half of the habitat burnt. Humphries (1994) found that adults were absent from the first six months following spring burning until the following breeding season, when all age classes were represented. In contrast, all age classes were represented in the first year following the autumn fire. This comparatively slow population recovery rate may be related to the longer time required for the dense sedgy vegetation preferred by *R. fuscipes* to recover. *Rattus fuscipes* also has a slower reproductive rate than *A. agilis*, producing fewer young in each breeding season.

Changes in movement patterns and size of small mammal home ranges following prescribed fires

Prescribed fire potentially affects the movement of small mammals in two ways. First, the construction of firebreaks may form barriers to movement and, second, movements and sizes of home-ranges may be influenced by changes in the availability of food and shelter resources.

Humphries (1994) found that earth firebreaks influenced the movements of *R. fuscipes* but did not inhibit those of *A. agilis*. In contrast, Barnett et al. (1978) found tracks and roads significantly impeded small mammal movement. Humphries (1994) notes these conflicting results may be influenced by track width and the lack of overstorey disturbance in the FESAs when tracks were formed.

Humphries (1994) was able to identify home-ranges for male and female *A. agilis*. Home-range sizes appeared to be larger in the open understorey of the Blakeville FESA than the denser understorey of the Barkstead FESA. Spring or autumn fires did not significantly change the home-range size, although ranging distances appeared to be up to 50% greater during the first year after each fire. Humphries (1994) suggested that this increase was probably a result of a need to search a larger area in order to find sufficient food (invertebrates) in unburnt or partially burnt patches of litter and that the increase ranging-distance was likely to be a temporary response.

Humphries (1994) identified a large proportion of animals to be transient (i.e. only captured once). Throughout the study, the proportion of transients averaged 48% in the Control Treatment Areas; this rose significantly from 44% eighteen months prior to the autumn fire, to 75% six months after the autumn fire. In the 7-16-month period after autumn treatment the proportion of transients decreased to 30%. There was no significant change following the spring fires. In the first year following spring burning, numbers of transient animals rose to 67% compared to 44% in the Control Treatment Areas.

Movements into and out of Treatment Areas were significantly affected as a result of the Treatment Areas being burnt (Humphries 1994). Although sample sizes were small, there appeared to be a trend of movement from areas burnt in both spring and autumn, to unburnt areas in the first 12 months following fire. These movements partially explain the observed reduction in abundance of both *A. agilis* and *R. fuscipes* following prescribed fire at this scale (Humphries 1994). It may not be possible for individuals to move to large patches of unburnt habitat if control burning is practiced at a larger scale.

Influence of burning season on small mammal populations

The single spring fire did not significantly affect the survival of individuals of *A. agilis* within Treatment Areas, but Humphries (1994) did record a decrease in numbers after autumn fires. Spring fires occurred after the mating period and coincided with the time of lowest population numbers caused by the cyclic death of all males. Because of this, there was probably less competition among remaining animals for the reduced resources for at least four to six months after fire. Autumn fires occur during a period of relatively high population levels and, although most individuals survived the fire, the reduction in suitable habitat may have led to a reduced population in the following 12 to 18 months

(Humphries 1994). The increase in competition for resources was not immediately apparent, but took four to six months to become so at all three FESAs and was still apparent at the Barkstead FESA 16 months after autumn burning (Humphries 1994). Recovery of litter and associated invertebrate inhabitants was slower during the winter period following autumn fires compared with the recovery during the summer following the spring fires (Neumann & Tolhurst 1991). Predation following fire may also have been significant in reducing population numbers (Cowley et al. 1969; Christensen & Kimber 1975).

The capture rate of *R. fuscipes* was not significantly different between treatments (Scuffins 1994). As this species appears to recover from fire slowly, *R. fuscipes* populations may still have been recovering at the end of the studies detailed in Humphries (1994) and Scuffins (1994).

Habitat preferences

Antechinus agilis

The geographic range of *A. agilis* covers diverse forest types, from structurally complex vegetation to sparsely vegetated areas, and the species is known to utilise a variety of microhabitats for foraging, shelter and nesting. The microhabitat preference of this species varies between forest types and seasonally at particular locations (Wood 1970; Braithwaite et al. 1978; Wilson et al. 1986; Moro 1991).

Antechinus agilis is generally associated with fallen logs, well-developed leaf litter and dense understorey vegetation (Barnett et al. 1978; Stratham & Harden 1982; Laidlaw & Wilson 1989; Lindenmayer et al. 1991; Bennett 1993). Bennett (1993) suggests that dense understorey vegetation supports abundant invertebrate populations as well as provides protection from predators. Moro (1991) suggests that the importance of logs is reduced at sites with dense ground cover. Litter is an important foraging substrate for this species (Gullan & Robinson 1980; Dickman 1988). Other foraging substrates include loose fallen bark at the base of eucalypts, the trunks and lower limbs of trees and bare ground under logs and between shrubs or tussocks (Dickman 1988; Moro 1991).

Humphries (1994) captured more animals in areas with a greater abundance of fallen logs. Radio telemetry studies also indicated that the abundance of logs was an important factor in habitat preference of this species (Baker 1998; Solly et al. 1999). Hollow fallen logs are used by *A. agilis* for nesting (Baker 1998) and refuge from fires (Humphries 1994). Hollow fallen logs are more likely to be destroyed by fire than solid logs (Humphries 1994). However, burning had no net effect on the abundance of fallen logs (Tolhurst et al. 1992a). Unburnt litter was often associated with fallen logs, adding to the foraging habitat available to the proportion of the population that survived the burns (Humphries 1994). Radio telemetry observations also indicate that leaf litter and logs are important to *A. agilis*, accounting for 73% of identified foraging locations (Baker 1998).

Nests of *A. agilis* were found in cavities in trees and fallen logs and in the ground under trees, stumps, logs and tussocks. Baker (1998) found female *A. agilis* nesting in small tree hollows 1–3 m above ground. In order to exclude predators and to have a nest with a stable microclimate, *A. agilis* appear to select hollows with entrances a little larger than themselves—averaging 23.5 mm in width. This behaviour is common to other small mammals, such as bats (Tidemann & Flavel 1987). Kambouris (1998) found *A. agilis* displayed no preference for either decaying or non-decaying logs. Dickman (1991), Lazenby-Cohen (1991) and Lindenmayer et al. (1991) suggest *A. agilis* favours cavities in non-decaying logs and tree trunks with small entrance holes.

Rattus fuscipes

Rattus fuscipes was mainly restricted to dense thickets of sedges, rushes and ferns in gullies (Humphries 1994; Scuffins 1994). Humphries' (1994) observations suggest that burning caused no change to the habitat preference of *R. fuscipes*. Where sufficient dense ground vegetation remained after burning, so did *R. fuscipes*. This habitat is less likely to be burned but, when it is, recovery time is relatively slow.

At the Musk Creek and Blakeville FESAs, the populations of *R. fuscipes* were found in *Lomandra longifolia* thickets (Humphries 1994). This habitat carried lower numbers of *R. fuscipes* than the denser sedge-rush communities in wetter gullies (Humphries & Tolhurst 1992). Scuffins (1994) recorded no *R. fuscipes* captures in A3 or S3 Treatment Areas at Musk Creek, suggesting that colonisation of these Treatment Areas had not taken place as of 32 (S3 treatment) to 39 (A3 treatment) months after burning. Beds of *L. longifolia* were showing signs of recovery but may not to that time provide suitable habitat for *R. fuscipes* (Scuffins 1994). Likewise, no *R. fuscipes* were captured in the S3 Treatment Area 32 months after burning at the Blakeville FESA and only one individual was captured in the A3 Treatment Area (28 months after fire).

Influence of burns on health, age structure and breeding success

Humphries (1994) found no significant difference in mean body weight of animals between Treatment Areas. Heislars (1980) also recorded no significant differences in mean body weight for *A. agilis* following wildfire. However, Newsome et al. (1975) reported lower male body weights after intense wildfire but offered no explanation for this.

The age and sex structure of the *A. agilis* population after spring and autumn fire was unchanged (Humphries 1994). Female *A. agilis* born before the spring and autumn fires were captured in burnt Treatment Areas in that study, indicating that these animals either survived the fire events or migrated from unburnt areas. The absence of females in the first two post-spring fire assessments suggests that migration of animals from unburnt areas is probably important for maintaining population numbers (Humphries 1994).

Neither spring nor autumn fires appeared to affect the timing or success of *A. agilis* breeding thereby promoting rapid recovery of populations after fire (Humphries 1994). In contrast, low abundance of juvenile *R. fuscipes* following the spring fire treatment probably reflected relatively low reproductive success or unsuitability of the burnt habitat for new recruits (Humphries 1994).

Limitations of telemetry studies

The application of radio telemetry technology to the study of the movements and habitat requirements of small mammals is relatively new and techniques still need further refinement. For example, all of the radio-collars fitted by Kambouris (1998) lost their antennae before the conclusion of the study, resulting in a dramatic reduction in the effective detection distance. It was recommended that the transmitter antenna be wrapped around the collar to limit protrusion and therefore improve durability. Despite this, the radio telemetry studies were successful in collecting habitat data for *A. agilis*, but not for *R. fuscipes* (Kambouris 1998). Both transmitters fitted to *R. fuscipes* individuals were destroyed within 24 hours, apparently chewed by other rats. Future studies of this species should investigate alternative methods of transmitter attachment.

Conclusions

No burning treatment was favoured by either species, but habitat preferences were observed. Although *Antechinus agilis* and *Rattus fuscipes* have different habitat preferences, the survival and recovery of both largely depend upon retention of unburnt habitat patches.

Whilst a range of microhabitats was used by *A. agilis* (leaf litter, vegetation below 1 m and bare ground), logs were found to be an important component of their habitat pre- and post-fire. Although log abundance is not significantly affected by low-intensity prescribed fire, hollow logs, which are used by *A. agilis*, are consumed in fires more readily than solid logs.

Within the Wombat State Forest Fire Effects Study, *R. fuscipes* populations are largely restricted to areas of dense sedgy vegetation in or near gullies.

The seasonal fluctuation of *R. fuscipes* numbers is an important factor in assessment of both pre- and post-fire populations. Populations are highest in winter and early spring when juveniles enter the trappable population and are lowest in autumn after some adult mortality. Surveys should be consistent and timed to recognise and minimise seasonal influences.

Rattus fuscipes populations are reduced as a direct result of burning and the amount of suitable habitat remaining unburnt is of paramount importance to the recovery of *R. fuscipes* populations in burnt areas.

Small-scale, single prescribed fires had little short-term effect on *A. agilis* populations. Populations were initially reduced as a result of direct mortality, probable predation and some emigration out of burnt areas in the first few months after treatment.

Following spring fires, *A. agilis* populations recovered to pre-fire levels within two breeding periods (24 months).

Due to the timing of breeding and juvenile recruitment of *A. agilis*, autumn burns were found to have a greater impact on the population than spring burns. *Antechinus agilis* populations require at least 16 to 24 months to recover to pre-fire levels after autumn burns.

Two to three years following spring and autumn burns, populations of *A. agilis* were significantly higher than in long-unburnt areas.

Up to three-quarters of the *A. agilis* population after autumn burning are transients. Unburnt areas are an important source for immigration of *A. agilis* back into burnt areas, and recolonisation will occur as suitable microhabitats recover.

Populations of *R. fuscipes* took three breeding seasons (36 months) to recover when more than half of its preferred habitat was burnt during spring, but there was no recovery during the same period when the entire habitat was burned.

A single spring fire has a greater impact on *R. fuscipes* populations in the first 12 months post-fire than a single autumn fire.

When autumn fires only marginally reduced *R. fuscipes* habitat, they had negligible impact upon the population dynamics. However, when the entire habitat was burned, there was no sign of recovery during the first six months.

Recommendations

Although six small mammal species were captured, not all were found in numbers suitable for meaningful analysis. New techniques need to be developed to properly assess the habitat ecology of all species. Further radio telemetry studies are recommended in this regard.

From this work it is recommended that:

- fire management prescriptions aim to minimise the burning of gully vegetation and burn no more than 70–80% of other areas. This will ensure that litter and ground vegetation remain as sources of refuge and colonisation by litter invertebrates and allow recovery (within three years) of *R. fuscipes* and *A. agilis* populations. Size of burns should be limited to allow movement of small mammals to unburnt areas.
- as the response of small mammal populations to fire is largely related to the impact of fire on their required microhabitats, it is important that these requirements are understood. Further telemetry studies are therefore recommended for *R. fuscipes*, *R. lutreolus* and *A. swainsonii* to achieve this.
- monitoring of *Rattus* species via telemetry may be impracticable until the damage these species cause to the transmitters is overcome. Future telemetry studies should be aware of these problems and aim to develop techniques that eliminate them.
- the influence of low-intensity prescribed fire on populations of *A. swainsonii* is unknown, as this species was not recorded in sufficient numbers for analysis. Further research on this species and others, and the development of suitable survey techniques are required to properly assess populations.

In any research, the application of an increased survey effort will be beneficial to the study at hand. The telemetry studies in this report show how the applications of various survey efforts influence the amount of data collected. By incorporating this concept with proper project design, future telemetry research will provide vital, cost-effective ecological information.

Birds

Richard Loyn

Introduction

Fire is recognised as one of the main natural agents that has helped shape the Australian flora and fauna (Gill et al. 1981). Prescribed fire is one of several tools used in flora and fauna conservation, and has been used in various ways by indigenous people through to modern managers. Any attempt to use fire as a management tool will have ecological consequences and land managers need to understand those consequences in order to be effective. Various studies have been made to assess the effects of fire management on vegetation and fauna (e.g. Caughley 1985; Recher et al. 1985; Lunney et al. 1987; Tolhurst et al. 1992a; Friend 1993; Adams et al. 1994; Wilson 1996). Remarkably few studies have been designed in a rigorous way to address basic questions about the effects of fuel reduction burning on vertebrate or invertebrate fauna and long-term studies are particularly rare (Friend 1999). The study in Wombat State Forest is an example of a study conducted over many years that uses a rigorous design to address questions about effects of fuel reduction burning on various components of the forest ecosystem, including birds. Methods used for the bird studies were described by Loyn et al. (1992).

Research results

About 100 bird species were recorded on or near the Wombat Fire Effects Study Areas, and 55 species made regular use of the Treatment Areas themselves. The most commonly observed species include Striated Thornbill (*Acanthiza lineata*), White-naped Honeyeater (*Melithreptus lunatus*), Brown Thornbill (*Acanthiza pusilla*), Spotted Pardalote (*Pardalotus punctatus*), White-throated Treecreeper (*Cormobates leucophaeus*), Grey Fantail (*Rhipidura fuliginosa*), White-browed Scrubwren (*Sericornis frontalis*) and Superb Fairy-wren (*Malurus cyaneus*). All species were grouped into guilds for subsequent analysis, with Table 8.1 showing percentage abundance of each guild based on number of birds/count.

Early work summarised by Loyn et al. (1992) showed that changes after prescribed fires were generally subtle, with few species lost or gained from particular Treatment Areas and many species recovering in abundance after initial decreases. Trends found in the initial analysis on data from the 1992-94 assessments have been confirmed with subsequent data, with a high degree of stability in terms of total bird abundance, species per count and species composition.

Table 8.1 Percentage of birds (grouped in bird guilds) found in the Wombat State Forest Fire Effects Study Areas, 1992–94

Guild	Description	% of all birds
Brood parasites	Cuckoos, laying eggs in active nests of other species	0.54
Large hollow nesters	Mainly parrots and Laughing Kookaburra	6.81
Small hollow nesters	Mainly treecreepers and Striated Pardalote	14.11
All hollow nesters	Sum of both small and large hollow nesters	20.98
Bark foragers	Treecreepers, sittellas, shriketit and two honeyeaters	11.45
Low seed-eaters	Taking seeds from ground or low vegetation	0.04
Large seed-eaters	Mainly parrots, taking seeds, galls, etc. from all levels	5.35
Frugivores	Taking soft berries or mistletoe drupes	0.17
Aerial insectivores	Taking insects, often above the canopy	0.49
Open-ground foragers	Taking invertebrates mainly from dry open ground	9.82
Damp-ground foragers	Taking invertebrates mainly from damp leaf litter below trees or shrubs	2.31
Low-understorey foragers	Taking invertebrates from dense understorey or damp ground below	6.64
Shrub foragers	Taking invertebrates from shrubs and wattle trees	10.41
Canopy foragers	Taking invertebrates from eucalypt canopy	39.94
Nectarivores	Taking nectar mainly from eucalypts, also invertebrates	12.43
Carnivores	Taking vertebrates, sometimes also large invertebrates	3.02
Summer migrants	Absent from forests or rare for period each winter	6.30
Uncommon species	Fewer than 850 records in Emison et al. 1987	1.35
Introduced species	Introduced to Australia	0.04

Note that some of these groups were represented on these assessments by a few or a single species (e.g. introduced birds, low seed-eaters and low understorey foragers): compare percentages of total records for each group and its constituent species. Species assigned to two feeding groups were scored as half in each. Birds classed as foragers have diets consisting mainly of insects and other invertebrates.

Insectivorous birds foraging from open ground increased in abundance on burnt Treatment Areas compared with Controls, especially on Treatment Areas burned frequently in autumn (Table 8.2). Nectarivores showed complex patterns, apparently responding to fire positively or negatively at different times. Other changes were subtle, but birds feeding in shrub layers tended to be less common on burnt Treatment Areas than Controls, especially on Treatment Areas burned frequently in spring. Overall, burning season had no significant effect on any group except as an interaction with frequency. Frequent fires served to maintain early successional stages of vegetation, and this pattern benefited some species and may have disadvantaged others. All of the birds that made use of open-ground habitats soon after fire also made use of trees and shrubs in burnt and unburnt parts of the landscape. Patterns of colonisation after fire showed that some of these species were highly mobile within the forest landscape, and were able to exploit temporary flushes of food at particular stages after the fire.

Table 8.2 Mean bird abundances (birds per count) of bird guilds found in each Treatment Area of the Fire Effects Study Areas, 1992–94

Guild	Mean bird abundance (birds per count)				
	Control	Spring 3	Spring 10	Autumn 3	Autumn 10
Brood parasites	0.22	0.18	0.12	0.16	0.20
Large hollow nesters	2.25	2.26	1.73	2.75	1.95
Small hollow nesters	4.95	4.90	4.52	4.22	4.09
All hollow nesters	7.22	7.17	6.29	6.97	6.07
Bark foragers	3.72	3.77	3.67	3.47	3.77
Low seed-eaters	0.00	0.00	0.00	0.07	0.00
Large seed-eaters	1.89	1.79	1.32	2.19	1.42
Frugivores	0.15	0.02	0.08	0.03	0.00
Open-ground foragers	2.37	3.34	3.02	4.81	2.26
Damp-ground foragers	0.80	0.59	0.82	0.99	0.51
Low-understorey foragers	2.08	1.63	2.38	2.41	2.17
Shrub foragers	3.67	2.69	3.71	3.10	3.58
Canopy foragers	13.59	12.34	13.1	12.99	12.18
Nectarivores	4.70	4.92	3.47	3.65	3.25
Carnivores	0.99	0.97	0.87	1.21	0.83
Summer migrants	1.96	1.94	2.02	2.46	1.74
Uncommon species	0.64	0.58	0.41	0.36	0.18
Total birds	33.73	31.52	31.75	34.22	29.54

Measured habitat variables added little to this picture, except that hollow-nesting birds responded strongly and positively to hollow density indices for each treatment. Inclusion of pre-treatment data as covariates did not improve models produced, although the data were helpful in describing qualitative changes and stability in bird communities.

There were only slight variations in mean abundance of birds between the five FESAs (Table 8.3). Canopy foragers were clearly the most abundant guild across all five FESAs.

Table 8.3 Mean bird abundances (birds per count) of bird guilds for each Fire Effects Study Area, 1992–94

Guild	Mean bird abundance (birds per count)				
	Barkstead	Blakeville	Burnt Bridge	Musk Creek	Kangaroo Creek
Brood parasites	0.15	0.10	0.25	0.10	0.26
Large hollow nesters	1.39	3.12	2.01	1.98	2.45
Small hollow nesters	1.88	6.09	3.28	5.26	6.18
All hollow nesters	3.28	9.26	5.30	7.24	8.64
Bark foragers	1.95	4.31	3.64	3.79	4.71
Low seed-eaters	0.00	0.02	0.00	0.00	0.05
Large seed-eaters	1.28	2.02	1.64	1.54	2.13
Frugivores	0.00	0.08	0.10	0.00	0.09
Open-ground foragers	1.75	4.66	4.52	2.37	2.48
Damp-ground foragers	0.62	0.69	0.95	0.34	1.11
Low-understorey foragers	3.38	0.74	3.28	0.83	2.43
Shrub foragers	3.98	2.15	5.19	2.01	3.42
Canopy foragers	10.41	13.42	12.42	14.04	13.92
Nectarivores	2.36	2.80	3.88	3.09	7.85
Carnivores	0.35	1.76	1.03	0.87	0.85
Summer migrants	1.40	1.37	1.74	1.63	3.98
Uncommon species	0.05	0.43	0.38	0.31	1.01

Rare or threatened birds

Square-tailed Kites (*Lophoictinia isura*) were observed occasionally on the FESAs, hunting for birds and lizards through the forest canopy. This species is listed as rare nationally and endangered in Victoria. Two species are noteworthy in this context because they were found to have small and isolated populations on the FESAs, although they are not listed as threatened: Spotted Quail-thrush (*Cinclosoma punctatum*), a pair of which was resident on slopes with sparse understorey at Musk Creek, and Yellow-tufted Honeyeater (*Lichenostomus melanops*), a small population of which was resident in a disturbed gully at Kangaroo Creek. Populations of all these species were too sparse or localised to be expected to enable sensible conclusions be made about the effects of fuel reduction burning. Another noteworthy species, Red-browed Treecreeper (*Climacteris erythrope*), was found to be widespread on the FESAs, and is the subject of a broader study monitoring effects of forest management in the Wombat State Forest.

Introduced birds

Only two introduced bird species were observed on the FESAs. European Goldfinches (*Carduelis carduelis*) occurred as occasional visitors in extremely low numbers. Pairs of Common Blackbirds (*Turdus merula*) were found to be resident and breeding at one or two Treatment Areas only (e.g. in disturbed gully vegetation at Kangaroo Creek). Both species are common in nearby farmland and gardens, along with a range of introduced birds that rarely enter the forest.

Conclusions

Fuel reduction burning in patches of less than 40 ha appears to have minor effects on bird communities in this forest (compared with no burning), and may benefit some species which forage from open ground, at least in the short term. Autumn burns are marginally preferable to spring burns. Greater effects would be expected in forests with different shrub structures. Effects of broad-scale burning would depend on the mosaic of burnt and unburnt areas produced.

The data for birds show that effects of fuel reduction burning are far milder than those of severe wildfires (Recher et al. 1985; Woinarski 1990; Reilly 1991a; Reilly 1991b; Loyn 1997; Woinarski & Recher 1997), especially on nectarivores and canopy-foraging insectivores. Hence, if fuel reduction burning is effective in reducing the extent or severity of wildfire, these groups may benefit from such protection. However, any benefits incurred by this management strategy could be lost if there were long-term changes in the vegetation. The risks of such changes are greatest under regimes of frequent burning (Woinarski & Recher 1997). The present study suggests that the greatest short-term benefits of fuel reduction to birds arise with frequent burns, but it is important to note that this is because the greatest response occurs in early successional stages, not because of any inherent advantage in frequent burns. A policy of burning some areas frequently and others rarely or never appears to have merit, if areas zoned for frequent burning can be selected to maximise their strategic value for reducing wildfire. However, long-term studies are needed to determine the ecological resilience of particular forest types to numerous cycles of frequent burning.

The study suggests that the effects of burning in spring or autumn are quite similar (on subsequent bird abundance compared with unburnt areas) and there is no clear case for burning in one season and not the other. However, the beneficial effects of fire appear to be somewhat greater with autumn burns (probably because they are more intense) and the detrimental effects somewhat greater with spring burns (probably because birds are nesting at that time). It may be prudent to conduct fewer spring than autumn burns, but to maintain diversity by continuing to conduct some burns in both seasons.

A policy of extensive fire exclusion would be undesirable as some bird species respond positively to fire, and some may depend on it in this forest type, for example White-winged Cough (*Corcorax melanorhamphos*), Spotted Quail-thrush (*Cinclosoma punctatum*), Blue-winged Parrot (*Neophema chrysostoma*) and Red-browed Finch (*Neochmia temporalis*). Fuel reduction burning serves to provide ephemeral patches of bare-ground habitat at the landscape scale, although it does not mimic the patterns expected under a regime of occasional extensive wildfires.

A policy of extensive broad-scale burning would be highly undesirable, as birds may need access to unburnt vegetation within their home-ranges, especially in the immediate aftermath of the fire as noted for mammals by Newsome et al. (1975), Catling and Newsome (1981), Humphries (1994), Tolhurst (1996a) and Friend (1993). The present study provides no information on the ability of birds to persist in areas subject to such treatment. The precautionary approach is to ensure that any broad-scale burning is done patchily, to produce a mosaic of burnt and unburnt vegetation with many large unburnt areas. At a national level, many threatened species depend on habitats that remain unburned for long periods (Woinarski & Recher 1997), and identification and protection of such habitats should be an important management goal.

Recommendations

Research recommendations

The study has provided valuable insights into the effects of fuel reduction burning in mixed eucalypt foothill forests. General inferences can be made about such effects with a known degree of confidence, because advantage was taken of the full replicated design (five FESAs by five Treatment Areas). Studies of some other fauna groups (e.g. invertebrates and

mammals) have focused on subsets of these FESAs and Treatment Areas, greatly restricting the power to make general inferences. It is strongly recommended that these groups of animals should be studied across all five FESAs. Lack of base-line data should not be used as an argument against such work, as base-line data on birds proved to be of only marginal and qualitative value in interpreting recent data. The suggested studies could be done as snapshot retrospective studies over a single year, now that the fire regimes have been established over many years. Sampling may need to be undertaken over several seasons. Long-term monitoring may add to the picture, but is not necessary in interpreting effects of the established fire regimes. The FESAs provide a unique opportunity for cost-effective snapshot retrospective studies of all fauna groups.

The main current need for research on birds is to extend the work to different forest types and geographical regions. This need could be met through targeted retrospective research in comparable sets of areas that have been subjected to disparate fire regimes (e.g. different Fuel Management Zones). Priority should be given to studying forest types where fuel reduction burning is used extensively and where practical alternative management decisions are likely to have beneficial outcomes. It is strongly suggested that such research should be broadly based and focused especially on habitats where vegetation structure is likely to be more sensitive to fuel reduction burning than it is in the Wombat State Forest (e.g. in forests with more uniformly open or dense understorey, as in heathy forests). The work in the Wombat State Forest can provide a detailed backdrop for such studies, in most cases obviating the need for time-consuming before-after research. New work of this sort should focus on operational burning regimes, not on expensive experimental burns. The main aims of this new research would be to:

- quantify effects of fuel reduction burning in a range of forest types and geographical areas
- identify special habitats that may be eliminated or greatly reduced in areas subject to fuel reduction burning or wildfire, so that such habitats can be given proper protection in fire management planning.

Management recommendations

The study shows that effects of fuel reduction burning on birds in this sort of forest (at this experimental scale) are much less than the expected effects of wildfire. Hence fuel reduction burning can be an appropriate tool for managing parts of these forests, if used strategically to reduce the extent or severity of wildfire. Some more specific recommendations follow.

- The current data suggest that it will be prudent to conduct fewer spring burns than autumn burns, but to maintain diversity by continuing to conduct some burns in both seasons. Spring burns are typically more patchy than autumn burns, and this can be an advantage although the potential benefits were not apparent in the data examined here.
- A policy of burning some areas frequently appears to have merit, if such areas can be selected to maximise their strategic value for reducing wildfire. However, such a policy would increase the risk of long-term vegetation changes, and more needs to be known about the ecological resilience of different forest types to such management over multiple burning cycles.
- The observations on bird mobility and colonisation patterns suggest that it will be important to maintain a patchwork of recently burnt and long-unburnt areas within the forest, at suitable spatial scales. Patchy burns may provide such a mosaic at the fine scale and priority-zoning systems are currently used to provide such diversity at the broad scale.
- Habitats known to be sensitive to fuel reduction burning should be identified (on a local or regional basis) and excluded from fuel reduction burning programs or high-priority burning zones. Some special habitats may deserve targeted protection from wildfire or managed burns.

Soils: carbon, nitrogen and phosphorus

Peter Hopmans

Introduction

The impacts of fire on soils, nutrient cycling and forest hydrology were reviewed at the commencement of the program (Flinn et al. 1984) and specific studies on the effects of fire intensity on soil chemistry and nitrogen mineralisation were completed early in the program (Tolhurst & Flinn 1992). A considerable body of knowledge exists about the effects of single fires on forest soils and, in general, the degree of change in soil properties is linked with fire intensity and resultant soil heating. In contrast, few studies had examined the cumulative effects of repeated prescribed fires on soils, nutrient cycling and site productivity (Flinn et al. 1984; Raison et al. 1993; Adams et al. 1994).

A long-term study on the impacts of repeated prescribed fires on soil chemistry was initiated in 1985 at five experimental FESAs in the Wombat State Forest (Tolhurst & Flinn 1992). Soil profile samples were collected prior to the commencement in 1985 of three- and ten-year cycles of prescribed fires and again in 1998. Soils of two of the experimental FESAs were examined in detail to determine the variability in carbon (C), nitrogen (N) and phosphorus (P) within and between Treatment Areas and the changes in soils in response to fire treatments (Hopmans 2000). Analysis of soils collected from the remaining three FESAs was completed in 2001 and results for all experimental FESAs are presented in this report.

Research results

Comparison of the initial levels of total C and N, and extractable P in soil profiles at the five FESAs showed small but statistically significant differences (Table 9.1) between the various locations in the Wombat State Forest. Soil profiles at Blakeville, Kangaroo Creek and Barkstead FESAs were quite similar and more uniform compared with Burnt Bridge and Musk Creek FESAs. Results obtained demonstrated the importance of taking into account the initial variation between experimental areas when evaluating changes induced by forest management practices, in this case the effects of repeated prescribed fires.

Table 9.1 Average concentrations of carbon and nitrogen (g/kg) and extractable phosphorus (mg/kg) in soil profiles in 1985 prior to prescribed fire treatments at five experimental Fire Effects Study Areas in the Wombat State Forest

Element	Depth (cm)	Blakeville		Kangaroo Creek		Burnt Bridge		Musk Creek		Barkstead		F value*	PLSD ⁺
		Mean	SE [#]	Mean	SE	Mean	SE	Mean	SE	Mean	SE		
Carbon	0-2	118	6.4	108	5.2	114	5.4	98	7.4	100	4.6	NS	-
	2-5	51	2.0	46	2.0	64	4.2	45	4.7	46	2.1	***	10
	5-10	33	1.6	28	1.5	42	2.6	31	3.2	31	2.2	***	7
	10-20	24	1.2	18	1.1	29	1.9	23	2.4	22	1.6	***	5
Nitrogen	0-2	3.9	0.20	4.1	0.18	4.4	0.24	3.3	0.25	3.5	0.22	**	0.7
	2-5	2.0	0.09	2.0	0.12	2.7	0.20	1.8	0.18	1.8	0.13	***	0.5
	5-10	1.4	0.07	1.3	0.10	1.9	0.13	1.4	0.14	1.3	0.12	**	0.4
	10-20	1.1	0.05	0.9	0.07	1.4	0.10	1.1	0.11	1.0	0.07	**	0.2
Phosphorus	0-2	27	1.8	25	1.6	22	2.1	17	2.4	18	1.7	*	6
	2-5	12	0.9	11	0.5	11	0.7	9	1.3	9	0.5	NS	-
	5-10	8	0.7	7	0.3	7	0.6	6	0.7	6	0.6	NS	-
	10-20	5	0.5	5	0.2	5	0.5	4	0.5	4	0.4	NS	-

* Differences between means are significant at $P < 0.05$ (*), $P < 0.01$ (**), $P < 0.001$ (***), or not significant (NS).

+ PLSD, Fisher's Protected Least Significant Difference at $P = 0.05$.

SE, standard error of the mean.

The comparison of the initial levels of C, N and P in soil profiles between assigned treatments—i.e. Control, Spring 3 (S3), Autumn 3 (A3), Spring 10 (S10) and Autumn 10 (A10)—showed that the mean values across the five experimental FESAs were generally similar (Figure 9.1). However, variability was high with coefficients of variation ranging between 20% and 30% for the upper layers in the soil profile. Consequently, comparatively large changes are required to test the hypothesis that repeated prescribed fires have affected these soil properties.

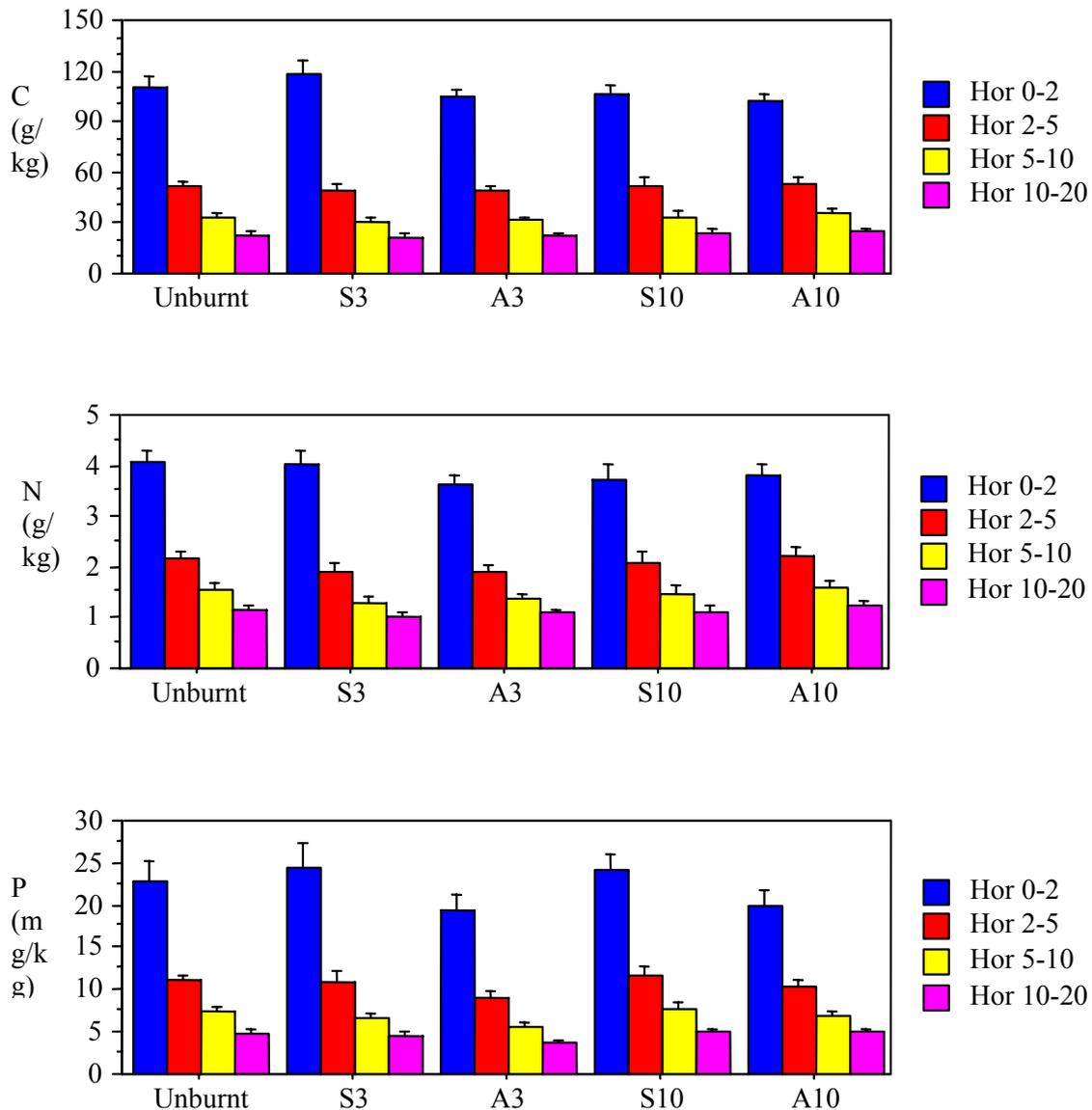


Figure 9.1 Total C and N and extractable P in soil profiles of designated fire Treatment Areas at Blakeville, Kangaroo Creek, Burnt Bridge, Musk Creek and Barkstead Fire Effects Study Areas in 1985

Note: Bars indicate standard errors of the means.

For example, a change in C of 18 g/kg in the surface soil (15% of Control) represents a statistically significant effect at the 5% level of probability. Likewise, the least significant difference (LSD at $P < 5\%$) for extractable P in the surface soil in 1998 was 3.2 mg/kg, or a change of 33% with respect to the Control. These comparatively large LSD values are a reflection of the inherent variability between the experimental FESAs and a consequence of the low level of replication within Treatment Areas (two or three replicates). With the benefit of hindsight, greater replication within treatments and increasing the number of sampling plots from three to six would have improved the sensitivity of the analysis, but this would also have increased the already substantial cost of this soil study about two-fold. Because of the low level of replication within treatments, it is important to evaluate the effects of fire across all five experimental FESAs (Hopmans 2000).

Based on a comparison of soil profiles collected in 1998 at the completion of the prescribed fire treatments, extractable P was not affected by any of the fire regimes (Table 9.2). However, a comparatively large change in P was required for a statistically significant difference between treatments. The change in P from the initial level showed a substantial decline across all fire treatments, including the Control, due to the high initial levels of extractable P in 1985 compared with the post-treatment samples (Figure 9.2). The results obtained are not robust and cannot be used to discern a more subtle impact of fire on extractable soil P. However, it cannot be assumed that fuel reduction burning will necessarily increase extractable P, as is often the case with high-intensity regeneration or slash fires. Repeated low-intensity fires increased extractable P in the surface soils in some studies (McKee 1982; Adams et al. 1994) while other work has shown little if any change in soil P (Binkley et al. 1992; Boyer & Miller 1994).

Table 9.2 Average concentrations of C and N (g/kg) and extractable P (mg/kg) in soil profiles in 1998 following repeated fires at three- and ten-year cycles in spring (S3 and S10) and autumn (A3 and A10) at the five Fire Effects Study Areas in the Wombat State Forest. Standard errors of the means are shown in parenthesis.

Soil	Control	S3	A3	S10	A10	F value [#]	PLSD [‡]
Carbon (g/kg)							
0–2 cm	122 (7.5)	103 (8.2)	91 (5.8)	105 (6.7)	107 (7.5)	**	18
2–5 cm	52 (4.2)	46 (3.4)	44 (3.3)	52 (4.4)	50 (4.0)	*	6
5–10 cm	33 (3.3)	31 (2.4)	31 (2.8)	35 (3.3)	34 (2.7)	NS	-
10–20 cm	21 (2.3)	21 (1.8)	21 (2.0)	23 (2.4)	23 (2.0)	NS	-
Nitrogen (g/kg)							
0–2 cm	4.3 (0.25)	3.7 (0.28)	3.3 (0.22)	3.8 (0.26)	3.8 (0.21)	*	0.5
2–5 cm	2.1 (0.16)	1.8 (0.14)	1.8 (0.14)	2.1 (0.19)	2.0 (0.15)	NS	-
5–10 cm	1.4 (0.13)	1.3 (0.11)	1.4 (0.12)	1.5 (0.14)	1.5 (0.11)	NS	-
10–20 cm	1.0 (0.11)	1.0 (0.08)	1.0 (0.09)	1.1 (0.10)	1.1 (0.08)	NS	-
Phosphorus (mg/kg)							
0–2 cm	9.6 (0.5)	10.0 (1.1)	11.1 (1.5)	9.7 (0.7)	11.7 (1.2)	NS	-
2–5 cm	4.5 (0.3)	4.3 (0.4)	4.0 (0.3)	4.7 (0.4)	4.4 (0.4)	NS	-
5–10 cm	3.3 (0.2)	3.1 (0.3)	2.8 (0.2)	3.5 (0.3)	3.4 (0.4)	NS	-
10–20 cm	2.3 (0.1)	2.1 (0.2)	1.9 (0.1)	2.5 (0.2)	2.7 (0.3)	NS	-

[#] Differences between means are significant at $P < 0.05$ (*), $P < 0.01$ (**), $P < 0.001$ (***), or not significant (NS).

[‡] PLSD, Fisher's Protected Least Significant Difference at $P = 0.05$.

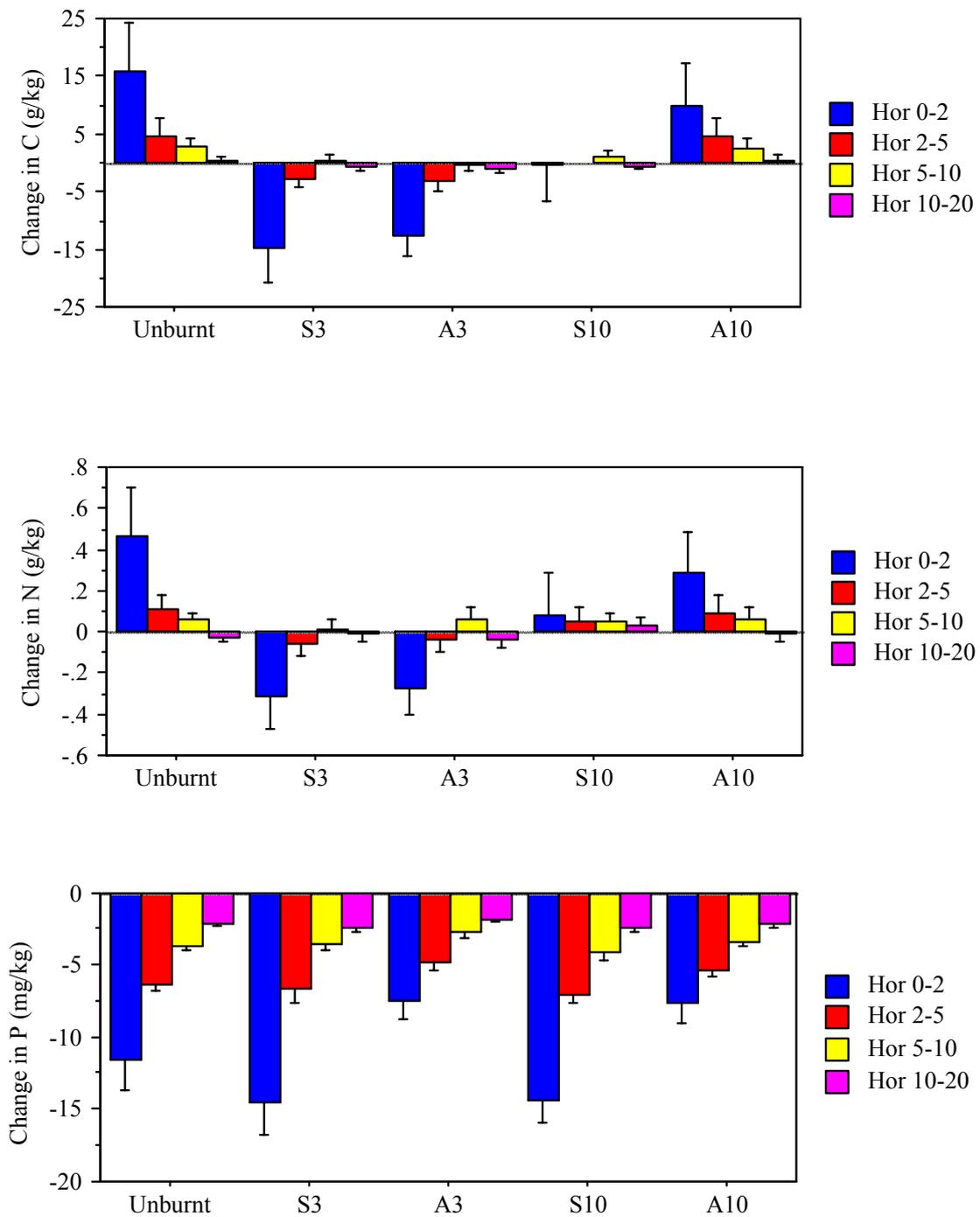


Figure 9.2 Changes in total C and N and extractable P in soil profiles following repeated fires at 3- and 10-year cycles in spring (S3 and S10) and autumn (A3 and A10) at the five Fire Effects Study Areas in the Wombat State Forest

Note: Bars indicate standard errors.

The long-term study of the effects of repeated prescribed fires across all FESAs showed an accumulation of C and N in the surface soil (0–2 cm) and to a lesser extent in the next layer (2–5 cm) of the soil profile in the Control treatments (Figure 9.2). In the absence of fire, C and N in the surface soil increased by 16 g/kg and 0.47 g/kg respectively, equivalent to a change of 14% and 11% relative to their initial levels. In contrast, a decline in C and N was evident in surface soils of Treatment Areas with spring and autumn burning at three-year intervals. Decreases were estimated at 15 g/kg (13%) and 13 g/kg (12%) for C, and 0.31 g/kg (7.8%) and 0.27 g/kg (7.6%) for N in Spring 3 and Autumn 3 burning treatments respectively. Differences between spring and autumn burning were not significant, indicating little, if any, impact from seasonal conditions.

At the less-frequent burning intervals (ten years nominally) changes were negligible for the Spring 10 burning treatment. However, small increases in C and N of 10 g/kg (9.5%) and 0.28 g/kg (7.4%) were noted for Autumn 10 burning. It should be noted that these changes in the surface soils of the Spring 10 and Autumn 10 treatments were not significantly different from the Control.

The effects of fire on C and N in soils were mainly limited to the surface layer (0–2 cm) which includes a relatively large proportion of partly decomposed litter and coarse organic matter. This material is readily volatilised or combusted during low-intensity fires (e.g. 50–350 kW/m) with surface temperatures often in excess of 200 °C (Humphreys & Craig 1981; Neary et al. 1999). The impacts on C and N in the surface soil are dependent on a rather complex range of conditions, including fuel characteristics (e.g. load, size, spatial distribution and moisture content), weather conditions (e.g. temperature, wind and humidity), fire behaviour (e.g. intensity, duration and rate of spread) and soil conditions (e.g. moisture content, bulk density, heat capacity and conductance). Therefore, the effects of a single prescribed fire on soil C and N can be expected to vary considerably between fire events, depending on conditions at the time of the fire (Humphreys & Craig 1981; Raison et al. 1993; Neary et al. 1999). In an attempt to address this inherent variability associated with single fire events, fuel characteristics and fire behaviour, the present study examined the longer-term effects of repeated fires at the five FESAs in the Wombat State Forest.

Results showed a general decline in surface soil C and N associated with low-intensity fires at three-year intervals. However, there was significant variation between Treatment Areas. For example, the changes in C and N were negligible at Blakeville where fires were mostly classed as 'cool' burns (K.Tolhurst⁴, pers. comm., 2002). In contrast, significant changes in C and N were observed at Kangaroo Creek where fires were generally more intense, often resulting in substantial crown scorch. It is suggested that the results of the present study of soil C and N need to be examined in the context of the observed differences in fire behaviour between the FESAs. This would provide a better understanding of the relationship between fire conditions and their effect on soil C and N. Furthermore, this could also be developed into an important 'tool' for the management of prescribed fires and as an indicator of the sustainable management of native forests.

The changes in soil C and N due to repeated fires were comparatively small but statistically significant and limited to the surface soil. This result is consistent with the small changes in C and N due to repeated fuel reduction burning reported from other similar studies (e.g. McKee 1982; Raison et al. 1993; Boyer & Miller 1994). The present study also showed a slight change in the C/N ratios indicating an increase in C relative to N where there was an accumulation of organic matter (unburnt Control). In contrast, a decrease in C relative to N, or conversely an increase in N relative to C, was observed where there was a loss in organic matter due to frequent low-intensity fires.

It is suggested that the input of N from symbiotic nitrogen fixation by native legumes regenerating in response to fire may have contributed to the change in C/N ratios. A study by Hamilton et al. (1993) using ¹⁵N natural abundance and isotope dilution procedures at the

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Musk Creek FESA estimated the amounts of atmospheric N fixed by two *Acacia* species over two years at 26 to 57 mg/plant. The accretion of N to the forest ecosystem depends largely on the density of legumes regenerating after fire. In this case plant density was low (2000 to 5000 plants/ha) and therefore the estimated input of N over the two-year period at Musk Creek was small (0.1 kg/ha). The input of N was also low compared with the range of 1 to 15 kg/ha/yr reported for other ecosystems (Hamilton et al. 1993; Raison et al. 1993). Because of the sparse regeneration of legumes at Musk Creek, the input from symbiotic N fixation following the fire is unlikely to compensate for the expected volatile losses of N (range 30 to 100 kg/ha) due to low-intensity fires in foothill forests. It is suggested that a long period between fires (e.g. 15 years) and hotter fires may be needed to stimulate the regeneration of legumes and to replenish N losses from these forests.

While the changes in C/N ratios were small, they are indicative of subtle changes in the pools of labile C and N. For example, the studies reported by Raison et al. (1993) measured various 'available' forms of N and showed significant reductions in N mineralisation due to repeated fires, indicating important changes in microbial activity of soil in sub-alpine snowgum forest. Likewise, repeated low-intensity fires decreased not only total C and N, but also mineralisable N and phosphatase activity in the surface soil of heathland at Wilsons Promontory in Victoria (Adams et al. 1994). Both of these studies showed that even comparatively small changes in total C and N pools in surface soils are often associated with significant changes in the biological processes and availability of labile forms of C and N at the litter-soil interface.

Conclusions

The present study showed a significant decline in C and N in surface soils due to low-intensity fires at three-yearly intervals. Furthermore, there was some evidence of a change in the quality of organic matter. In contrast, observed changes in extractable P were not attributable to fire. In the long term, low-intensity fires managed at three-yearly intervals can be expected to lead to a decline in soil organic matter and soil fertility in these foothill forests. Based on Montreal Process Criteria and Indicators for sustainable forest management (DPIE 1998), with particular reference to the maintenance of soil resources (fertility and organic matter), prescribed fires at this frequency would be unsustainable as a long-term fire management strategy for these forests. In contrast, the study showed little, if any, change in C and N from less-frequent fires (ten-year interval), indicating that this strategy can be expected to maintain soil organic matter in the long term.

It is recommended that serious consideration be given to the continuation of the program to get a better understanding of the long-term changes in soil processes affected by low-intensity fires. This would also provide the opportunity to examine the differences in the impacts of fire management strategies based on short (three-year) and intermediate (10- to 15-year) intervals on fire intensity, the regeneration of legumes and the replenishment of volatile losses of N through symbiotic fixation of atmospheric N. This provides important information on the period required for recovery of the ecosystem between fires and the maintenance of soil fertility. This information is essential for the long-term sustainable management of fire in these foothill forests.

Recommendations

Based on the present study of long-term changes in selected soil properties due to repeated prescribed fires it is recommended that:

- prescribed fires in mixed eucalypt foothill forests be conducted at intervals of at least ten years in order to minimize the impacts on soil C and N and to maintain long-term soil fertility
- consideration be given to continuation of the program of experimental prescribed burning in the Wombat State Forest to get a better understanding of the impacts of inter-fire period and fire intensity on long-term soil fertility (organic matter, N and P) and the N cycle in foothill forests.

Tree growth and bark thickness

Tina Bell and Karen Chatto

Introduction

Adequate bark thickness on trees is considered to be the first level of protection for buds, timber and underlying phloem, cambium and sapwood against fire, insect attack and disease (Luke & McArthur 1978; Gill 1981). Physical characteristics of bark, such as thickness, fissures, thermal diffusivity, flammability, moisture content and longevity determine the level of protection afforded the cambium and epicormic buds from excessive heat from fire (Stocker & Mott 1981). Several studies (Gill 1981; Shea et al. 1981) have suggested that bark thickness and fire intensity are the key factors determining the extent of wood defects and tree mortality. Despite this knowledge, few quantitative field studies have been conducted.

Characteristics such as tree age, vigour and rates of bark growth influence the level of protection by bark afforded to underlying plant tissues (Stocker & Mott 1981). In general, bark thickness increases with stem diameter, which in turn depends on tree age and species. Younger or smaller trees and thinner-barked species are more likely to be affected by fire. Given that the duration of a heat pulse required to kill cambial tissue (60–65 °C) is exponentially related to bark thickness, small differences in bark thickness can produce large differences in fire resistance (Hare 1965). This equation is complicated by uneven burning of bark around the circumference of the trunk and a general decrease in the amount of bark burnt with flame height (Fahnestock & Hare 1964; Gill 1981). Unevenness in burning around and up a tree stem may lead to superficial damage and charring, death of part of the underlying cambium (forming a fire scar or 'dryside') or death of the stem (Gill 1981). Greaves et al. (1965) and Vines (1968) suggest that tree death after fire is most likely due to secondary effects caused by fungal or bacterial damage to bark and underlying tissue and weakening of the tree by termite infestation of dead wood, rather than as a direct effect of fire.

The study presented here was established to determine the effect of repeated low-intensity prescribed fires on bark thickness of both Messmate (*Eucalyptus obliqua* L'Herit.) and Candlebark (*E. rubida* Dean & Maid.) and on tree growth in terms of changes in mean diameter at breast height under bark. The study commenced in June 1985 and final measurements were taken in 1999. Tree parameters—height, diameter at breast height over bark and bark thickness at 50 cm and 130 cm above ground level on the uphill side (BT50 and BT130 respectively)—were recorded before and after burning events for both species within all Treatment Areas at the Blakeville FESA. Similar data were collected for only *E. obliqua* from the Control Treatment Areas, S10 and A3 Treatment Areas at Barkstead and Kangaroo Creek FESAs, and S3 and A10 Treatment Areas at Burnt Bridge and Musk Creek FESAs. Preliminary results have previously been presented in the first progress report of the Wombat Fire Effects Study (Kellas 1992).

Research Results

Relationship between bark thickness and diameter at breast height over bark

Bark thickness at breast height (BT130) measured prior to any burning in June 1985 was strongly correlated with diameter at breast height over bark (DBHOB) for both species (Figure 10.1). Linear regressions between bark thickness and DBHOB showed strong correlations for regrowth trees less than 300 mm in diameter (*E. obliqua*: $r^2 = 0.68$; *E. rubida*: $r^2 = 0.71$) but not for overwood trees greater than 300 mm diameter (*E. obliqua*: $r^2 = 0.32$; *E. rubida*: $r^2 = 0.42$) at the Blakeville FESA. Similar patterns were reported for overwood and regrowth trees of *E. obliqua* at all other FESAs with r^2 values for regrowth trees ranging from 0.45 to 0.94 and from 0 to 0.64 for overwood trees (Kellas 1992).

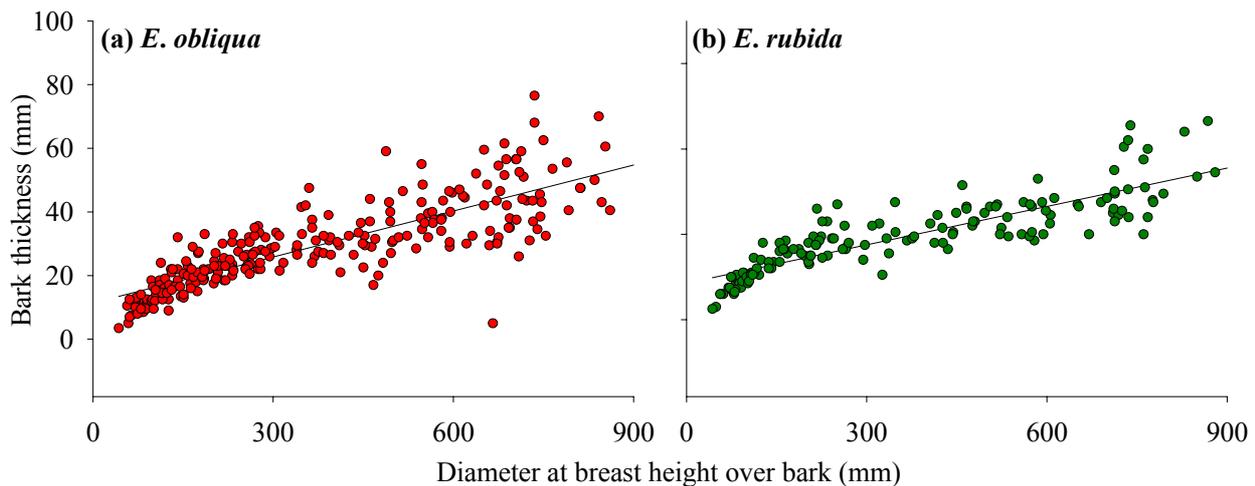


Figure 10.1 Relationship between bark thickness measured at breast height and diameter for (a) *Eucalyptus obliqua* and (b) *E. rubida*. Data points represent all trees (overwood and regrowth) in Control and Treatment Areas in all five Fire Effects Study Areas measured in June 1985 prior to any treatment burning.

There was a gradual increase in bark thickness of unburnt trees in the Control Treatment Areas when measured at two heights—50 cm and 130 cm—and at varying intervals over 15 years. The bark of *E. obliqua* was generally 10 mm thicker than that of *E. rubida* when measured at the 130 cm height and half as thick when measured lower down (50 cm).

Data shows that prescribed burning provided a significant stimulus to tree growth. All burning treatments produced a greater increase in stem diameter for the 1985–99 period compared to the unburnt trees. Earlier in the study it was suggested that one or two low-intensity fires had no major impact on the growth of overwood or regrowth trees (Kellas 1992). With additional burning and further measurements, the preliminary conclusions were revised to incorporate the prolonged effect of prescribed burning regimes on tree growth. It has been suggested that fires of low intensity do not significantly alter characteristics of forest soil that are likely to influence tree growth—e.g. pH, nitrogen, exchangeable cations, total soluble salts and organic carbon (Hopmans 2000). The soil component of the Wombat Fire Effects Study showed that extractable phosphorus was not affected by the prescribed burning regime imposed, however, carbon and nitrogen in the soil surface declined after repeated burning (Hopmans 2000). Changes in soil nutrient availability are therefore unlikely to have contributed to the increases in growth of *E. obliqua* and *E. rubida* and must be attributed to other factors such as increased water availability or reduction in competition.

Reduction in bark thickness due to fire

Mean pre- and post-fire bark thicknesses presented in Figure 10.2 are for *E. obliqua* and *E. rubida* trees at Blakeville and *E. obliqua* trees at Kangaroo Creek FESAs. Burning reduced bark thickness of both species of eucalypts. However, the effect was more marked in *E. obliqua* than *E. rubida*. For *E. obliqua*, this reduction was evident at both measured heights and a significant interaction was found between the year of burning and treatment area ($p < 0.001$). Greatest differences in bark thickness before and after a fire were generally recorded in 1985 or 1987 after initial treatment burning or after ten-year rotational burning (e.g. 1994). Similar patterns were observed for *E. obliqua* trees measured at Barkstead, Burnt Bridge and Musk Creek FESAs (data not shown).

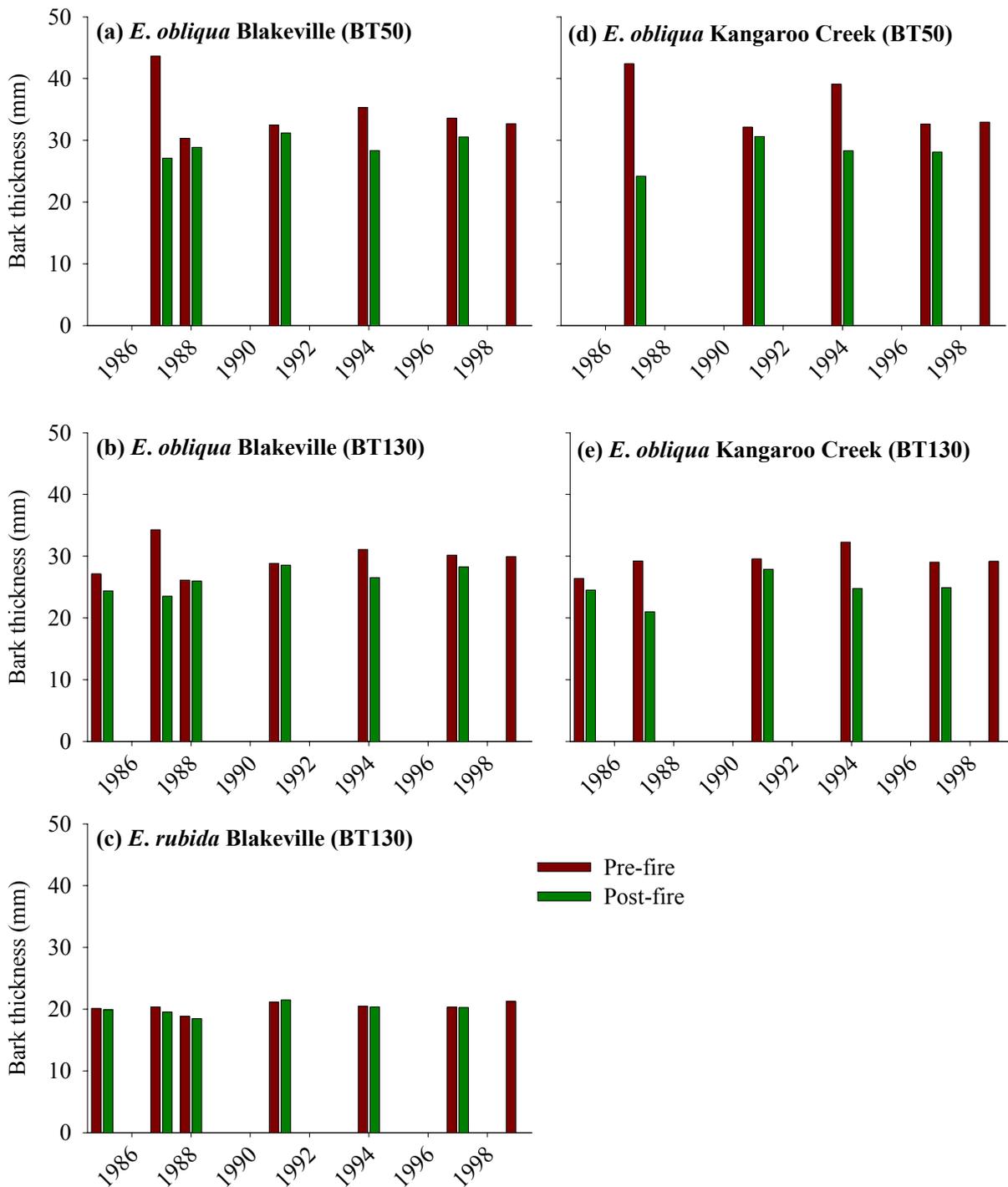


Figure 10.2 Comparison of bark thickness measured a) and d) at 50 cm (BT50) and b), c) and e) 130 cm (BT130) for a), b), d) and e) *Eucalyptus obliqua* and c) *E. rubida* at Blakeville and Kangaroo Creek. Data represent means of pooled data for overwood and regrowth trees and burning treatments.

Significant interactions were found between Treatment Areas and pre- and post-burn bark thickness when measured at both 50 cm ($p < 0.001$) and 130 cm above ground level ($p = 0.007$). By comparing pre- and post-fire bark thickness within similar Treatment Areas in different FESAs, it can be suggested that the amount of bark loss due to fire is dependent on the season and frequency of fire (Table 10.1). For all burning treatments, substantially more bark was lost at 50 cm than at 130 cm. It can be further suggested that the loss of bark due to autumn burning is greater than spring burning for both short (every three years) and long (every ten years) rotations. This seasonal effect was more strongly related to Soil Dryness Index (SDI) than fire intensity (Kellas 1992). Bark loss on regrowth trees could be explained to some extent by fire intensity ($r^2 = 0.46$, Kellas 1992; autumn fires averaged 259 kW/m, spring fires averaged 152.3 kW/m, Tolhurst et al. 1992b) but was not a valid explanation for overwood trees. Longer rotations (i.e. 10-year) caused greater bark loss than shorter rotations (3-year) for both spring and autumn burning. This pattern is again strongly related to SDI and is poorly explained by fire intensity—214.7 kW/m for three-year rotations and 213.7 kW/m for 10-year rotations (Tolhurst et al. 1992b).

Table 10.1 Mean pre- and post-fire bark thicknesses measured at 50 and 130 cm for each burning treatment. Mean values represent pooled data for all Fire Effects Study Areas and all trees (*Eucalyptus obliqua* and *E. rubida*, overwood and regrowth).

Treatment	Mean bark thickness at 50 cm (mm)			Mean bark thickness at 130 cm (mm)		
	Pre-burn	Post-burn	Difference in means (%)	Pre-burn	Post-burn	Difference in means (%)
Spring 3	65	59.6	5 (8.4)	51	48	3 (6.1)
Spring 10	70	57.1	13 (18.6)	53	48	6 (11.0)
Autumn 3	73	61.3	12 (15.8)	57	52	5 (9.1)
Autumn 10	70	56.0	14 (20.2)	55	46	10 (17.4)

Bark loss due to prescribed burning was greater at 50 cm above ground level than at 130 cm. Given that flame height is related to fire intensity and therefore SDI, the above arguments for loss of bark at different heights also apply. The average flame height across all FESAs was 37 cm—range 5–150 cm (Tolhurst et al. 1992b) supporting the greater loss of bark at lower levels. A number of other studies have also found that the amount of damage to a tree stem varies with the height above the ground and have related findings to flame height and fire intensity (Tunstall et al. 1976; Gill 1980; Gill 1981). These studies did not causally link bark loss to SDI or other measures of dryness of weather or fuel.

A significant interaction was found between tree diameter class (overwood and regrowth) and pre- and post-burn bark thickness measured at 130 cm ($p = 0.03$) but not at 50 cm ($p > 0.05$). This suggests that any comparison of bark thickness of overwood- and regrowth-sized trees should take into account the species involved, the area of study and the year of burning. The interaction between fire and tree size class suggests that more bark was lost from overwood trees than from regrowth trees (7.7 and 4.1 mm respectively) during fire. However, both size classes experienced a similar proportional loss (i.e. approximately 10%) of bark (Table 10.2). This suggests that it is not necessarily the amount of bark lost that is important for tree survival but rather the percentage of bark lost. In particular, the greater the loss of bark on regrowth stems, the more likely the underlying tissues are to be affected (Gill et al. 1986). Overwood trees generally have thicker bark than regrowth trees and therefore have more bark available for combustion (Figure 10.1). The average amount of bark lost at 130 cm was up to 60% greater for overwood trees than for regrowth trees.

Table 10.2 Mean pre- and post-fire bark thickness at breast height (BT130) for the two broad tree size classes (regrowth less than 300 mm diameter, overwood greater than 300 mm). Mean values represent pooled data for all five FESAs, all Treatment Areas (S3, S10, A3 and A10) and all trees (*Eucalyptus obliqua* and *E. rubida*).

Tree class size	Mean bark thickness at 130 cm (mm)		
	Pre-fire	Post-fire	Difference in means (%)
Regrowth	45	41	4 (9.1)
Overwood	74	66	8 (10.4)

The restoration of bark after fire is important for the protection of trees from future fire or other disturbances (Gill 1980). In the absence of fire there was a gradual increase in bark thickness measured at 130 cm for both species (*E. obliqua*: 0.66–0.81 mm/yr; *E. rubida*: 0.28 mm/yr). At 50 cm above ground, bark of *E. obliqua* increased at rates of 0.46–1.07 mm/yr. In the presence of fire, bark thickness at 130 cm was generally reduced immediately after burning and remained relatively constant over the course of the study period. Bark thickness measured at 50 cm above ground level never attained the same thickness as was measured before initial burning despite having potentially high growth rates. Similarly, Gill (1980) found slow restoration of bark in *E. dalrympleana* when measured seven years after fire and suggested that regrowth of bark depended on plant vigour and size.

Tree growth

Significant differences in diameter at breast height under bark (DBHUB) were found between area, species, burning treatments and tree class size (Table 10.3). Overwood trees showed a much greater change in mean diameter (39 mm) when compared to regrowth trees (17 mm).

Table 10.3 Mean change in diameter at breast height under bark (in mm) for: (a) each species (*Eucalyptus rubida*, *E. obliqua*) and Fire Effects Study Area and: (b) treatment. Mean values represent pooled data for all trees.

(a)

Area-by-species	Mean change in diameter (mm)
Blakeville, <i>E. rubida</i>	31
Blakeville, <i>E. obliqua</i>	22
Barkstead, <i>E. obliqua</i>	18
Burnt Bridge, <i>E. obliqua</i>	21
Musk Creek, <i>E. obliqua</i>	18
Kangaroo Creek, <i>E. obliqua</i>	23

(b)

Treatment	Mean change in diameter (mm)
Control	17
Spring3	25
Spring10	38
Autumn3	23
Autumn10	26

Tree mortality

A total of 43 trees (5.5% of the total number of trees measured in the study) died during the period of the study. No thorough examinations were carried out to determine cause of death; thus mortality may or may not have been due to burning treatments. The Kangaroo Creek FESA had the greatest death rate (11.9% of all trees measured in the FESA) followed by Barkstead (8.8%), Blakeville (5.3%) and Burnt Bridge (2.2%) FESAs. No deaths were recorded at Musk Creek FESA. Across all FESAs, five dead trees were found in Control Treatment Areas (2.4% of all trees measured in treatment), three in S3 (2.1%), 14 in S10 (10.0%), 19 in A3

(13.3%) and two in A10 Treatment Areas (1.4%). Chi-square analysis showed that the observed number of deaths were significantly different from the expected number of deaths across both FESA-by-species ($p < 0.01$) and treatments ($p < 0.001$). Despite this variation, mortality rates were similar to other studies (e.g. 20%—Burrows 1987). Mortality of trees from fire may be due to interplay of a number of factors, including tree age and size, season of burn, previous silvicultural treatments, climatic and topographical conditions, fuel hazard levels, disease and parasites, soil conditions, rates of recovery and growth rates (Gill 1980; Gill et al. 1986).

Methodological and management considerations

Bark itself is extremely difficult to measure. It varies in thickness not only vertically, but also horizontally around the stem (Fahnestock & Hare 1964; Gill 1981). Although this natural variation can and should be accounted for with good data collection, it is possible for human error to have a greater influence than recommended. For example, as with many long-term studies, numerous people are often involved in the collection of data and the inherent variation must be taken into account. In this study, measurement of bark thickness began in 1985 with use of the Swedish bark gauge and was changed, from 1996 onwards, to the Gill-type needle gauge. Again, this source of variation was kept to a minimum with extensive comparison of means and variance of data produced by the two methods (K.Tolhurst⁵, pers. comm., 2002).

The data presented in this study is defined as the amount of bark lost during treatment fires. When reporting on 'bark loss', only the amount of bark consumed during a fire was considered. No quantification was made of bark that may have been killed by fire but which desiccated and decorticated at a later stage (Gill 1980). This is particularly important for *E. rubida*, a 'smooth-' or 'gum-' barked species, which can lose substantial amounts of bark (6–8 mm) for up to 12 months following fire (Gill 1980). *Eucalyptus rubida* generally holds little potentially flammable bark material at breast height, so minimal bark loss would be expected during burning (Kellas 1992). Gum-barked species also undergo annual decortification of bark (approximately 1–2 mm/yr). As bark thickness was not measured at any time other than immediately before and after a fire (within a month of burning), it was assumed that all bark loss happened during the fire and no further bark loss occurred until the next fire event.

The current study has shown that bark loss is greater at 50 cm than at 130 cm above ground level. This has implication in fire management in terms of fuel reduction. McCarthy et al. (1999) indicates that the amount of bark retained on a tree stem significantly influences fire behaviour, in particular spotting potential. With this in mind, fuel reduction operations should target the potential hazard presented by bark accumulation, along with surface fine fuel and elevated fuel hazards. This study shows that fire of low intensity (12–748 kW/m) removes less than 20% and 17% of bark at 50 and 130 cm respectively. Although low-intensity fires may not reduce the total amount of bark fuel available for consumption by fire, they can reduce the amount of loose fibrous bark, particularly that held on *E. obliqua*.

No consideration was made as to whether the bark loss observed was due to a head fire, flank fire or backing fire. It is assumed that, in a fire where the fire perimeter is not enlarging in a circle (i.e. unequal rates of spread in all directions) and all other conditions are equal, then fire intensities will increase from the backing fire, to the flank fires, to the head fire (Luke & McArthur 1978). The amount of bark loss would depend on where individual trees are located in relation to these fire sections. Fahnestock and Hare (1964) found that head fires caused more damage to tree trunks than backing fires due to increased crown scorching and burning. In addition, the leeward side of a tree tends to burn more than the windward side (Gill 1981).

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This study did not investigate the utilisation of bark by insects, birds and mammals. The removal of loose bark and deep fissures on lower portions of eucalypt trees may cause a reduction in roost sites for bats (Westbrooke 2000). However, low-intensity prescribed burns are unlikely to reduce available roost sites at most FESAs given that an average of less than 10 mm of bark was lost after each fire at 130 cm above ground level and overall bark thickness was not reduced for any treatment over the course of the study.

Conclusions

In this study, bark thickness of *E. obliqua* and *E. rubida* varied widely between species and Treatment Areas and within an individual tree. Such variation is not surprising given that it has been recorded previously for other eucalypt species. Gill (1981) recognised that growth rate, diameter of stem, height above ground and stand history are all important variables that must be taken into account when interpreting bark thickness. Despite the known variability, strong relationships between diameter and bark thickness have been established and proved to be a reliable indicator of the effect of different prescribed burning regimes.

Bark thickness of unburnt *E. obliqua* and *E. rubida* increased linearly with increasing stem diameter. This study did not strictly measure the range of natural variation in tree size and bark thickness, as representative trees were selected according to specific size classes. This selection process may have had the effect of producing a linear increase in bark thickness rather than curvilinear. Such an example highlights the need to take into account the full range of site and population characteristics prior to descriptive or manipulative studies.

Bark loss due to burning was shown to be significantly dependent on burning treatment. Both season and frequency of fires could be considered to be important in the loss and recovery of bark after fire. Bark loss was found to be greater as a result of autumn burning compared to burning in spring. This pattern was found to be strongly related to SDI and poorly related to fire intensity.

Mortality rates during the study were relatively low and not attributable to one particular treatment. There was considerable variation between FESAs with fewer deaths recorded for areas with relatively 'cooler' burns (e.g. Blakeville) and more deaths at Treatment Areas where higher-intensity burns often resulted in crown scorch (e.g. Kangaroo Creek).

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Appendix A

Complete list of related publications/reports for the Wombat Fire Effects Study

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