Department of
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Ecological effects of repeated low-intensity fire on terrestrial mammal populations of a mixed eucalypt foothill forest in south-eastern Australia

Research report no. 63

Effects of repeated low-intensity fire on terrestrial mammal populations of a mixed eucalypt foothill forest in south-eastern Australia

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Cover photographs 1. Low-intensity fire in mixed eucalypt forest, DSE/K.Tolhurst 2. Agile Antechinus, DSE/I.McCann

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Foreword

The vegetation, topography and climate of south-eastern Australia combine to make the region one of the most wildfire-prone areas on Earth. 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 here is also considered to have had a more recent influence on these evolutionary processes. Paradoxically, it has been estimated that, in the last one hundred years, two-thirds of all human deaths related to bushfires in Australia and more than half of all significant related property losses have occurred in Victoria.

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 and has been a key component of park and forest management in Victoria since the late 1950s – early 1960s.

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.

In 1984, a multidisciplinary study was established in the Wombat State Forest, 80 km north-west of Melbourne (Victoria), to investigate the effects of repeated low-intensity prescribed burning in mixed eucalypt foothill forest. The study—the Wombat Fire Effects Study—is quantitative and statistically based and includes various aspects of fauna, flora, soils, tree growth, fuel management and fire behaviour.

On the same permanent plots, various methodologies are used to investigate the ecological impacts of fire on understorey flora, invertebrates, birds, bats, reptiles, terrestrial mammals, soil chemistry and the growth, bark thickness and defect development in trees. Local climate and weather, fuel dynamics and fire behaviour are also studied, along with their interactions. Numerous published papers and reports have been produced as a result of the work. Fire Management Research Reports comprising the current (2003) series 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

The foreword to the summary report (Fire Management *Research Report* No. 57) sets out more fully the background to the research, the impact it has had on fire management in the State and the future of the program.

I would like to acknowledge the very considerable efforts of the scientists and technical officers who have contributed to this specific report and more generally to this most significant project.

Gary Morgan AFSM
CHIEF FIRE OFFICER

Department of Sustainability and Environment

2003

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Summary

A total of 17 mammal species, excluding bats, were recorded for the Wombat Fire Effects Study Areas (FESAs). Of these, nine species were terrestrial and eight were arboreal. Two of the terrestrial mammals were the introduced European Rabbit and Red Fox. Only two of the terrestrial species were present in sufficient numbers for quantitative analysis—*Rattus fuscipes* (Bush Rat) and *Antechinus agilis* (Brown Antechinus).

Two main methods of study were used: trap and release, and radio telemetry. Trapping provided a good basis for studying population dynamics and dispersal while radio telemetry provided a good basis for studying habitat usage. The terrestrial mammal study was concentrated in three of the five FESAs, namely Blakeville, Barkstead and Musk Creek.

Five burning treatments were replicated in each FESA: short-rotation spring, short-rotation autumn, long-rotation spring, long-rotation autumn and a long-unburnt control.

The ability of small mammal populations to recover from fire was related to the survival rate of individuals in burnt areas, the presence of unburnt refuges and the rate of vegetation recovery.

Radio telemetry of *A. agilis* indicated that leaf litter and logs were important foraging microhabitat. It was also found that nests for this species were in cavities in trees and fallen logs and in the ground under trees, stumps, logs and tussocks. Radio tracking of *R. fuscipes* indicated that it was mainly restricted to dense thickets of sedges, rushes and ferns in gullies.

A significant decline in *A. agilis* numbers was noted after the 1985 spring and 1987 autumn fires. Spring fires reduced abundance by approximately 50% for 12 months after the fire. However, the population recovered in the second year due to a combination of high breeding rate, a significant proportion of the population surviving in the burnt area and the rapid accumulation of leaf litter in which to forage.

Rattus fuscipes population declined following fire in all treatments. It took three breeding seasons (two years) for the population levels to recover from a single spring fire when more than half their habitat was burned, compared with recovery after one breeding season following a single autumn fire, when less than half of the habitat burned. This relatively slow population recovery rate may be related to the longer recovery time of the dense sedge-dominated vegetation preferred by *R. fuscipes*. *Rattus fuscipes* also has a slower reproductive rate than *A. agilis*.

No single burning treatment favoured either species, but habitat preferences were observed. Although *A. agilis* and *R. fuscipes* have different habitat preferences, the survival and recovery of both species depends largely on retention of unburnt habitat patches.

Background

In 1984, the then Forest Commission in Victoria initiated a research program in response to growing community concerns about the ecological effects of repeated fuel reduction burning. This program looked at the effects of repeated low-intensity prescribed fire on the main elements of a foothill forest ecosystem in the Wombat State Forest, west-central Victoria. Output from this program is continually being incorporated into public land management across the State.

The fire effects research program comprises five replicated Fire Effects Study Areas (FESAs) located at Barkstead, Blakeville, Burnt Bridge, Kangaroo Creek and Musk Creek in the Wombat State Forest. Five Treatment Areas are located within each FESA:

- frequent fires (approximately every three years) in spring—short-rotation spring, S3
- frequent fires (approximately every three years) in autumn—short-rotation autumn, A3
- infrequent fires (approximately every 10 years) in spring—long-rotation spring, S10
- infrequent fires (approximately every 10 years) in autumn—long-rotation autumn, A10
- fire exclusion (unburnt for more that 20 years)—long-unburnt control, C.

A full description of the program design and methodology is provided in Tolhurst (1992).

This report is a review of studies on terrestrial mammals undertaken by the University of Ballarat as part of the above research program.

Terrestrial mammals have received some attention from a fire management perspective in other research. The relationship between the fate of their nesting sites/food sources and the survival of these animals, however, is not well understood. A clearer indication of the effects of fire on terrestrial mammals is therefore needed.

Program objectives

- To assess and describe the effects of repeated fuel reduction burning in both spring and autumn and applied on short, medium and long rotations (five fire treatments) on the composition and abundance of terrestrial mammal species.
- To determine the home range, habitat requirements and movement patterns of each species using telemetry.
- To have these findings incorporated into the Department of Sustainability and Environment’s fire protection and land management plans, policies and operations.

Introduction

Whilst the majority of the present research concentrated on small terrestrial mammals, incidental observations and some associated projects have provided information on larger mammals that occur in the area.

Johnston et al. (1983) noted 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 areas (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, 3–4 years after fire, species such as *Rattus fuscipes* (Bush Rat) 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).

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.

Small mammal habitat

Antechinus agilis (Brown Antechinus) and *R. fuscipes* are common and widespread small native mammal species inhabiting forest woodlands and heaths of south-eastern Australia. Population dynamics and habitat preferences of these species have been recorded on a local scale (Robinson 1987; Kemper 1990; Lazenby-Cohen 1991; Moro 1991; Catling & Burt 1995; Heislers 1974; Humphries 1994) such that vegetation type can be used to predict the presence of small mammal species within given communities (Braithwaite & Gullan 1978). Structural variation is also important where there is relative homogeneity in floristic composition (Dickman 1991; Lindenmayer et al. 1991; Moro 1991; Bennett 1993; Catling & Burt 1995). 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.

Past research in the Wombat State Forest

Population studies

Prior to the establishment of the Fire Effects study program, Leonard (1970) studied the effect of controlled burning on the reproduction, movements and general ecology of small mammals. Heislers (1974) investigated successional changes in vegetation and mammal populations over the four years following a wildfire in 1969. In a study that was not fire specific but which has implications for fire management, Calder et al. (1979) investigated the relationship between forest management practices and the provision of habitat for hollow-utilising species.

Limited studies were undertaken in the vicinity of the Blakeville FESA in an attempt to determine habitat and microhabitat preferences of arboreal mammals. The most significant studies of small mammals were those of Humphries (1994) who investigated the impact of single spring and autumn prescribed fires from March 1986 to August 1988 and Scuffins (1994), who later investigated the effect of repeated spring and autumn low-intensity burns. Both researchers worked within the Fire Effects Study framework and related variation in abundance data to habitat variables. Both studies focused on the most abundant species, *A. agilis* and *R fuscipes.*

Radio telemetry studies

Recent research in and adjacent to the FESAs utilised radio telemetry to locate and identify the microhabitat usage by *A. agilis* and *R fuscipes.* Studies by Kambouris (1998), Baker (1998) and Solly et al. (1999) obtained microhabitat data from *A. agilis.* Kambouris (1998) also used radio telemetry to study *R. fuscipes* but with limited success. Table 1 shows the research techniques used in these studies.

Table 1 Mammal survey techniques used by researchers in mixed eucalypt foothill forest associated with the Wombat State Forest Fire Effects Study

Research author	Survey techniques used						
	Elliot traps	Opportunistic	Spotlighting	Signs of activity	Nest Boxes	Natural hollows	Radio telemetry
Leonard (1970)	*						
Heislers (1974)	*	*		*			
Calder et al. (1979)					*	*	
Kemp (1989)			*		*		
Pretty (1990)					*		
Humphries (1994)	*						
Scuffins (1994)	*						
Brookman (1997)			*				
Kambouris (1998)	*						*
Baker (1998)	*						*
Solly et al. (1999)	*						*

Each study undertaken in the region involved the collection of information on a range of mammals. Excluding bats, 23 species of terrestrial and arboreal mammals were recorded. Table 2 identifies the animals observed by each researcher in this study, and notes other records from the Atlas of Victorian Wildlife.

Table 2 Mammals species recorded (excluding bat species) for each study associated with the Wombat State Forest

Research Author	*Phascolartos cinereus* (Koal	*Antechinus agilis* (Brown Antechinus)	*Antechinus swainsonii* (Dusky Antechinus)	*Phascogale tapoatafa* (Brush-tailed Phascogale)	*Acrobates pygmaeus* (Feathertail Glider)	*Petaurus breviceps* (Sugar Glider)	*Petauroides volans* (Greater Glider)	*Trichosurus caninus* (Mountain Brushtail Possum)	*Trichosurus vulpecular* (Common Brushtail Possum)	*Pseudocheirus peregrinus* (Common Ringtail Possum)	*Rattus fuscipes* (Bush Rat)	*Rattus lutreolus* (Swamp Rat)	*Rattus rattus* (Black Rat)	*Mus musculus* (House Mouse)	Wallabia sp. (Wallaby species)	*Canis vulpes* (Red Fox)	*Oryctolagus cuniculus* (European Rabbit)
	Species recorded																
Leonard (1970)		*	*								*			*			
Heislers (1974)		*		*						*	*		*	*	*	*	*
Calder et al. (1979)		*		*	*	*				*							
Kemp (1989)	*						*	*		*							
Pretty (1990)					*												
Humphries (1994)		*		*							*	*	*	*			
Scuffins (1994)		*									*						
Brookman (1997)	*						*		*	*							
Kambouris (1998)		*									*						
Baker (1998)		*									*						
Solly et al. (1999)		*	*								*						

Taxonomy according to Menkhorst (1996).

The following species have also been recorded in the Wombat State Forest (Source: Atlas of Victorian Wildlife 2000).

Macropus giganteus (Eastern Grey Kangaroo)
Wallabia bicolor (Black Wallaby)
Vombatus ursinus (Common Wombat)
Tachyglossus aculeatus (Short-beaked Echidna)
Dasyurus maculatus (Spot-tailed Quoll) *'accepted' record 1992.*
Rattus norvegicus (Brown Rat)

Methods

Population studies

Trapping of small mammals in the population studies of Humphries (1994) and Scuffins (1994) was carried out using collapsible 30 x 10 x 9 cm Elliott traps (Elliott Scientific, Lower Plenty, Victoria).

Both Humphries (1994) and Scuffins (1994) undertook studies at Blakeville and Musk Creek FESAs, while Humphries also undertook studies at Barkstead FESA. Both researchers used the same trapping method at their respective study areas. At each FESA a trapping grid was established of 120 trap stations with a trap spacing of 40 m on a 12 x 10 configuration. The grids were arranged so that an equal proportion of every Treatment Area was covered by the grid (40 trap stations per Treatment Area). The trapping procedure was a modified version of that used by Leonard (1972) and Heislers (1980). Traps were set in groups of four and systematically moved on successive mornings to untrapped stations. This routine gave wide-ranging individuals the opportunity to move as far as possible in a trapping session and eliminated the chances of an individual being trapped at the same location on successive nights. Populations were studied using the Capture-Mark-Release technique. A combination of ear-notching and toe-clipping was used to mark trapped individuals.

Telemetry

Radio telemetry studies (Baker 1998; Kambouris 1998; Solly et al. 1999) first involved trapping the required animals. Elliot traps were placed in locations likely to harbour the target species (i.e. amongst vegetation or in close proximity to logs).

After the desired number of each species was captured, radio-collars were so fitted around each animal's neck as to avoid discomfort, not be dislodged or be in a position where injury could occur. The transmitter comprised a miniature (1 g) LTM magnetic radio transmitter (Titley Electronics ®, Ballina, NSW). Baker (1998) and Solly et al. (1999) wrapped the antennae of each transmitter around the collar in an attempt to retain them, following recommendations by Kambouris (1998).

A receiver was then used to track the transmitter's signal to the animal's exact location (telemetry location).

Kambouris (1998) monitored each animal's location every morning and every afternoon for ten consecutive days. Baker (1998) monitored individuals more intensively (every two hours) using a discontinuous method and a discrete time period—from 0600 hrs until 1800 hrs (Harris et al. 1990). On two occasions he monitored *A. agilis* for 36 hrs—from 0600 hrs one day to 1800 hrs the next. Solly et al. (1999) noted locations of collared animals at four-hourly intervals for each 24-hour cycle. To avoid disturbance, no attempt was made to approach an animal closer than was necessary to determine its position. This continued for a period of up to eight days per animal or until the transmitter failed to emit a signal.

Baker (1998) monitored two *A. agilis* in an area adjacent to the Blakeville FESA, Solly et al. (1999) monitored thirteen *A. agilis* at four locations around the Musk Creek and Blakeville FESAs and Kambouris (1998) monitored six *A. agilis* and one *R. fuscipes* at both Musk Creek and Blakeville.

Data analysis

Both Scuffins (1994) and Humphries (1994) estimated small mammal populations for each trapping period using the number Known To Be Alive (KTBA) estimate as described by Krebs (1966) and used by Wood (1970). Correlations between each of the microhabitat measures and the frequency of dominant species captures were made by Scuffins (1994) to determine broad habitat preferences.

Scuffins (1994) analysed the percent dissimilarity of small mammal populations between treatments using a simplified polar ordination technique (Ludwig & Reynolds 1988). Scuffins also carried out Two-sample T-tests between the small mammal KTBA estimates in the control (long unburnt area) and the other four treatments. This tested the significance of any differences between burnt and unburnt treatments.

Habitat assessment of population studies

Humphries (1994) and Scuffins (1994) used an assessment of habitat which quantified the percentage cover of microhabitat. These microhabitat variables were assessed within a radius of 5 m of each trapping station (Barnett et al. 1978). Percentage cover was compared to standards in McDonald et al. (1984).

Humphries (1994) used a randomised block experimental design and analysis of variance (ANOVA) to test for differences between treatments and variation due to site differences.

Habitat assessments of telemetry studies

Kambouris (1998), Solly et al. (1999) and Baker (1998) used similar microhabitat assessment techniques. At all telemetry locations, a range of habitat variables was assessed within a 5 x 5 m quadrat. Floristic assessment was made using a modified six-point Braun-Blanquet (1928) scale to estimate the percentage foliage cover of each species in the quadrat (Table 3).

Table 3 Modified Braun-Blanquet scale

Scale	Description
+	Cover 5%, few individuals
1	Cover 5%, any number of individuals
2	Cover 5–20%, any number of individuals
3	Cover 20–50%, any number of individuals
4	Cover 50–75%, any number of individuals
5	Cover 75–100%, any number of individuals

The availability of logs on the ground within the same 5 x 5 m quadrat used for floristic assessment was recorded using a similar scale (Table 4).

Table 4 Log availability scale

Scale	Description
+	Cover <10%, any number of logs
1	Cover 10–20%, any number of logs
2	Cover 20–30%, any number of logs
3	Cover 30–50%, any number of logs
4	Cover 50–75%, any number of logs
5	Cover 75–100%, any number of logs

Where present, leaf-litter was recorded as its percentage of ground cover within the 5 x 5 m quadrat. Leaf-litter depth was also recorded. Leaf-litter was defined as any dead plant material lying on the ground, either attached or unattached to the parent plant and excluding logs and branches greater than 10 cm long (Baker 1998).

Solly et al. (1999) recorded the following variables:

- log diameter and stage of decay
- hollow entrance size
- height above ground and type of tree
- litter depth and the components of the litter
- microhabitat within the study area (i.e. gully, mid-slope and ridge).

The habitat components assessed by Solly et al. (within a five-metre radius of each telemetry location) were:

- log abundance
- litter percentage
- shrub cover percentage
- hollow abundance
- canopy cover percentage
- ground cover percentage.

Results

Species recorded in single and repeated prescribed fire

Humphries (1994) gathered data over the three years 1986–88, trapping 539 individuals, representing six small mammal species, a total of 1175 times. Species trapped by Humphries were *Antechinus agilis*, *Phascogale tapoatafa*, *Rattus fuscipes*, *Rattus lutreolus* and the introduced *Rattus 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).

Rattus fuscipes population estimates

Humphries (1994) found *R. fuscipes* to be common only at the Barkstead FESA, where it was restricted to dense vegetation in minor gullies and drainage lines. *Rattus fuscipes* numbers showed seasonal fluctuations, with the highest densities (6 ha^{-1}) in late winter – early spring when juveniles entered the population. Population numbers were lowest in autumn (0.5 ha^{-1}). Adult males and females that had bred in the previous year tended to die at this time, and this occurred just before juveniles were first trapped (Humphries 1994).

Spring fire effects on *Rattus fuscipes* populations

No pre-burn data of *R. fuscipes* abundance were available for the spring burn in the Barkstead FESA, as burning had taken place prior to the Humphries (1994) study. In the first 16 months post-burn, however, *R. fuscipes* abundance was less than 50% of the control. After a second breeding period and the subsequent recruitment of juveniles, abundances were 28–64% lower than the control. Following a third breeding period and further juvenile recruitment, the abundance of *R. fuscipes* in this area was 64% higher than at the control (Humphries 1994). By the following autumn, numbers had increased slightly. The *R. fuscipes* population was totally restricted to regenerating gully vegetation. Even though *R. fuscipes* was restricted to dense vegetation in drainage lines, within three breeding seasons post fire abundances in the fire treated areas exceeded those in control areas (Humphries 1994).

Autumn fire effects on *Rattus fuscipes* populations

The autumn burn at Barkstead was of low intensity and resulted in 69–85% of the area being burned and 14–27% of fine fuels reduced. Less than 50% of the gully vegetation was burned, with only the margins affected. This contributed greatly to the survival of *R. fuscipes* (Humphries 1994). In the first six months following the burn, Humphries recorded an 18–60% reduction in *R. fuscipes* abundance. After breeding recruitment in the following autumn, abundance was 20–40% higher than in the control.

The gully vegetation at Musk Creek and Blakeville FESAs, the understorey of which consists mainly of *Lomandra longifolia*, was almost completely burnt. No *R. fuscipes* were caught following either the spring or autumn fires at these FESAs.

Spring and autumn fire effects on age and sex structure of *Rattus fuscipes*

Humphries (1994) found that adult *R. fuscipes* were absent for 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.

Home range movements of *Rattus fuscipes* populations

Female home ranges of *R. fuscipes* overlapped considerably. The mean size of individual home ranges was 0.5 ha and the adjusted range length was 90 m. Eight movements were recorded across Treatment Area boundaries. Of these, five were post-burn movements and three were from a burnt to an unburnt area during the first 12 months following burning. The remaining two were in the reverse direction, 24 to 36 months following burning. Although the sample size is small, there appears to be a trend of movements out of burnt areas soon after the treatment. More intensive and frequent trapping before and after treatment would be required before this trend can be validated.

Habitat preferences

The capture of *R. fuscipes* was positively linearly correlated with the abundance of vegetation below 1 m height in the unburnt control and autumn burned areas. This vegetation structure was associated with minor drainage lines and gullies. The spring fire burnt 97% of the Treatment Area, consuming almost all vegetation below 1 m (Tolhurst et al. 1992). A strong relationship remained between captures of *R. fuscipes* and abundance of vegetation under 1 m following autumn burning at Barkstead FESA.

Age and sex structure of *Antechinus agilis* populations

Humphries (1994) captured female *A. agilis* born before the spring and autumn fires, in burnt treatments, indicating that these animals either survived the fire events or migrated from unburnt areas

Home range and movements of *Antechinus agilis*

In control areas, *A. agilis* movement ranged 54–66 m. No significant difference in ranging distance was found between sexes, between FESAs or with the time of the year.

Humphries (1994) identified a large proportion of animals as transients (i.e. only captured once). Throughout that study, the proportion of transients averaged 48% in controls areas. This proportion rose significantly from 44% 18 months prior to the autumn burn to 75% 6 months after the autumn burn. In the period 7–16 months after autumn treatment the proportion of transients decreased to 30%. There was no significant change following spring burning. In the first year following spring burning transient numbers rose to 67% compared to 44% in the control.

Effect of fire on habitat

Habitat preferences

The incomplete burning of litter and understorey layers aided the survival of individual *A. agilis*. Unburnt areas such as these accounted for 36% of the autumn Treatment Area. During the first seven months following autumn burning, 55% of *A. agilis* captures were from these unburnt areas (Humphries 1994).

Logs significantly affect the distribution of unburnt litter. Logs covered 4.3% of the area but were associated with 33.4% of the area of unburnt litter. Logs interrupted the progress of fire and maintained moisture in the litter underneath them. Apart from the unburnt autumn treatments at Blakeville and Barkstead, the capture rate of *A. agilis* was significantly correlated with abundance of fallen logs in all unburnt areas (Humphries & Tolhurst 1992). Following spring and autumn burning, there was no correlation between *A. agilis* captures and log abundance. Burning had no net effect on the abundance of fallen logs (Tolhurst et al. 1992). Humphries (1994) found that logs were added to the ground layer as a direct result of the burns and an approximately equal proportion of logs was consumed by fire. Logs that have a number of connected hollow entrances tend to be totally consumed in fire, whilst solid logs or logs with a single entrance were only charred. Solid logs were not used for nesting and shelter.

Provided that enough suitable cover remained (primarily hollow logs for nesting and a litter substratum in which to forage), areas occupied by *A. agilis* prior to burning were also occupied after burning.

Effect of fire on small mammal populations – repeated fire treatment

Antechinus agilis and *Rattus fuscipes*

Scuffins (1994) recorded no significant difference in *R. fuscipes* or *A. agilis* captures at Blakeville between treatments. At Musk Creek, differences in *A. agilis* captures between treatments were significant, but not for *R. fuscipes*. *A. agilis* captures showed a significant increase from the control to the A10 treatment, as well as an increase from the A3 to the S10 treatments (Scuffins 1994).

Movements and home range

Twenty small mammals were recaptured during the Scuffins (1994) study, five were *R. fuscipes* and 15 were *A. agilis.* Average maximum range distance was 44 m for *R. fuscipes* and 37 m for *A. agilis*. However, range distances of up to 130 m were recorded.

Microhabitat

Scuffins (1994) found four out of seven habitat variables at Blakeville were significantly different between burning treatments. These were:

- grass and herb cover
- shrubs <l m
- bare ground
- litter.

Grass and herb cover was lowest (mean 10.8%) in the control treatment. Mean litter coverage in the control was 69%. The S3 treatment had the highest mean cover of shrubs < l m (13.5%); this consisted largely of *Acacia* regrowth. Mean bare-ground proportion was highest in the S3 (6.1%) and A3 (4.6%) Treatment Areas and comparatively low in the S10 (1.4%) and A10 (0.75%) Treatment Areas. Proportion of bare ground was lowest in the control (0.5%).

Scuffins (1994) found bare ground, log and rock cover were significantly different at Musk Creek. The highest proportions of bare ground were recorded in the A3 and S3 treatments (6.6% and 6.7% respectively). The S10 treatment had the highest mean cover of logs (12.8%), followed by A10 (11%), C (8.6%), S3 (8.5%) and A3 (6.8%) Treatment Areas. Rock cover was low in all Treatment Areas.

Antechinus agilis captures at Blakeville were too few for statistical analysis. At Blakeville, *R. fuscipes* were weakly correlated with the cover of shrubs less than l m in the A3 treatment, as well as logs, grass and herb cover in the S10 Treatment Area. *Rattus fuscipes* was also positively correlated with the cover of shrubs less than l m and negatively correlated with litter cover in the S10 area (Scuffins 1994).

At Musk Creek, Scuffins (1994) recorded weak correlations between *A. agilis* and shrubs less than l m in the control and between *A. agilis* and logs coverage in the A10 Treatment Area.

Telemetry results

Species recorded

Eighty–eight different microhabitat locations were identified for *A. agilis*. However, radio telemetry of *R. fuscipes* proved unsuccessful.

Rattus fuscipes telemetry

Kambouris (1998) monitored two *R. fuscipes*. In both cases the signal was emitted for only one day. One radio collar was later retrieved and found to be extensively gnawed, presumably by another rat. It was assumed that the collar on the second *R. fuscipes* suffered the same fate (Kambouris 1998).

Antechinus agilis telemetry

Antechinus agilis monitored by Baker (1998) carried pouch young for the length of the radio tracking period. Both individuals monitored appeared to have a single arboreal diurnal nest 1–3 m above the ground and made regular nocturnal foraging forays into terrestrial microhabitats. Only Solly et al. (1999) recorded diurnal movement, and for only two animals. Some communal nesting was recorded (Kambouris 1998; Solly et al. 1999). Solly et al. found that the same microhabitat locations were visited on a number of occasions, while Baker (1998) recorded one microhabitat visited more than once. Overall movement by *A. agilis* was regular, however Solly et al. (1999) found an individual confined to its nest for five consecutive days before movement was again recorded. Movement of radio-tracked individuals ranged from 8 m to approximately 350 m from point of capture.

The average numbers of telemetry locations observed per transmitter by each researcher were:

- 1.6 locations using 12 transmitters (Kambouris 1998)
- 3.9 locations using 13 transmitters (Solly et al. 1999)
- 8.5 locations using 2 transmitters (Baker 1998).

Variation in the average numbers of locations was observed between studies and is probably a reflection of the different monitoring intensities used.

Of the 12 transmitters placed on *A. agilis* by Kambouris (1999), one failed to emit a signal after two nights, despite extensive searching. Other transmitters lasted between six and nine nights. Of the six radio collars retrieved at the conclusion of the study, none retained its antenna. The two transmitters used by Baker (1998) lasted eight and nine days respectively. Solly et al. (1999) reported five transmitters lasting two days or less, with the remaining eight lasting between three to eight days.

Microhabitat use

Of the 17 *A. agilis* observations by Baker (1998), two (12%) were in arboreal nests. Nests were consistently visited for long periods. The remaining 15 observations were considered foraging sites as they were only visited once. Of these, 29% were in logs 5–20 m long and 15–33 cm in diameter with hollow entrances 13–48 mm wide. A further 35% were amongst logs. Twelve percent of observations were in large tree hollows ranging in size from 13 x 6 cm to 23 x 30 cm and were 2–3 m above the ground. The remaining 12% were under bark on living trees with a diameter at breast height (DBH) range of 28–97 cm.

Nesting sites of female *A. agilis* were characterised as tree hollows 1–3 m above ground (Baker 1998). The single entrance of each hollow had widths of 19–28 mm. Each *A. agilis* stayed within its respective nest between 0600 and 1800 hrs. After 1800 hrs, *A. agilis* left the nest to forage. The DBH of the two nest trees were 81 cm and 73 cm respectively.

Of the 20 locations assessed by Kambouris (1998) for *A. agilis* habitat use, six were arboreal nests found in living and dead standing trees, five were in logs and nine were under logs; others were in stumps and amongst ground vegetation tussocks. Kambouris (1998) did not differentiate between nests and foraging sites, calling all locations 'nests'. Tracking intensity by Kambouris (1998) was insufficient to draw conclusions regarding differences in arboreal and terrestrial habitat use.

Of the 51 *A. agilis* observations by Solly et al. (1999), three were arboreal nest hollows in living and dead standing trees. Nests were also located within hollow logs. Individuals were generally in nests between 0900 hrs and 1800 hrs. Nests were located for all 13 individuals and communal nesting was observed.

Other microhabitat locations identified by Solly et al. were terrestrial and subterranean, and were found in fallen trees, under logs, within logs and in stumps, although most were amongst ground vegetation. The most common microhabitat used in all Treatment Areas was ground vegetation. Within and under logs was the next most common microhabitats used in the burnt areas. Microhabitats were located in wet gullies along drainage lines or creeks and on ridges or mid-slopes.

Microhabitat assessment

Baker (1998) assessed ground leaf-litter cover within the areas to range from 50 to 95%. Leaf-litter depth varied from 0.3 to 31 cm. Cover of logs greater than 100 mm diameter ranged from less than 10% to 50–75%.

Kambouris (1998) found leaf-litter cover within the assessed areas (Musk Creek and Blakeville) ranged from 10% to 90%. The depth of leaf-litter varied from a low of 1.5 cm at both FESAs, to highs of 8.5 cm at Musk Creek and 12.5 cm at Blakeville. Cover of logs suitable as small mammal habitat ranged from less than 10% to 50–75%.

Effect of fire on microhabitat

Solly et al. (1999) conducted habitat assessments at each of the 51 positional fixes found across all areas and included burnt and unburnt areas. They found more logs at the fire-affected areas after two burns (Appendix 1) than at the control areas. However, due to the effects of fire, cover of litter, hollows and vegetation in the burnt areas were all considerably less than in the control. Logs in the two unburnt areas had a larger diameter (means of 65.8 and 45.9 cm) and showed obvious signs of decay. Overall, logs at the two fire-affected areas were much smaller in diameter (means of 20.2 and 15.8 cm), charred and showed little sign of decay.

Solly et al monitored *A. agilis* in the A10 treatment at Barkstead and the S10 treatment at Musk Creek. A control area was also monitored near each of these two FESAs. The Appendix provides the fire history of all FESAs.

Discussion

Humphries (1994) found fuel reduction burning had little or no short-term (2–3 years) effect on populations of *Antechinus agilis* and *Rattus fuscipes*, although populations of both species were affected during the first 18 months post-fire.

Fire 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 un-burnt refuges and the rate of vegetation recovery. Refuges are an important source of recolonising individuals (Leonard 1972; Heislers 1980), particularly if the survival rate in burnt areas was low.

The amount of refuge habitat remaining after a fire depends on fire intensity (Suckling & Macfarlane 1984) and the amount of the total fuel load actually burnt (Tolhurst et al. 1992). A fuel reduction burn generally results in 60–80% of the area being burned to some degree, thus creating a small-scale mosaic effect (Hodgson & Heislers 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 to be relatively high (69–99%). Fire intensity and the proportion of total fuel burnt are largely dependent on prevailing weather conditions.

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

Antechinus agilis

Humphries & Tolhurst (1992) noted significant declines in *A. agilis* numbers after the 1985 spring and 1987 autumn fires. Spring fires reduced abundance by approximately 50% for 12 months after the fire. The results from Humphries (1994) 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, 1972). 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). Newsome et al. (1975) found a fire at the Myall Lakes National Park (New South Wales) had little affect on numbers. However, Fox & McKay (1981) observed *A. agilis* did not appear in population numbers until well into the second year after fire.

Solly et al (1999) suggest 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 treatments. This result is supported by Lunney et al. (1987) who found that numbers of *R. fuscipes* were dramatically reduced following fire.

The species composition of the dense sedgy vegetation preferred by *R. fuscipes* varied between study areas. At Musk Creek and Blakeville the gully vegetation was predominantly *Lomandra longifolia*, which was more open and burned more readily. This rush community supported a smaller *R. fuscipes* population than the more developed sedge–rush communities in 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 study by Humphries and Tolhurst (1992), 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 burned.

This comparatively slow rate of population recovery 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 small mammal movement patterns and home-range size following burns

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 the width of the tracks and the lack of overstorey disturbance in his study area when the tracks were formed.

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

Movements into and out of Treatment Areas were significantly affected as a result of the areas being burnt (Humphries 1994). Although sample sizes were small, there appeared to be a trend of movement from burnt areas, both spring and autumn, to unburnt areas in the first 12 months following fire. These movements partially explain the observed reduction in abundances 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 individual *A. agilis* within Treatment Areas, but Humphries (1994) did record a decrease after autumn fires. Spring fires occurred during the time of lowest population numbers—after the mating period and death of all males—and therefore competition among animals for the reduced resources was probably lower for at least 4 to 6 months. Autumn fires occur during a period when population levels are relatively high and, although most individuals survived the fire, the reduced habitat led to a reduced population in the following 12 to 18 months (Humphries 1994). This competition was not immediately apparent, but took 4 to 6 months to become so at all three FESAs and was still apparent at Barkstead 16 months after autumn burning (Humphries 1994). Recovery of litter and the associated invertebrates was slower during the winter period following autumn fires compared with the recovery during the summer following 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 slowly from fire, *R. fuscipes* populations may still have been recovering at the end of the studies by 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 species' microhabitat preference varies between forest types and seasonally at particular locations (Braithwaite et al. 1978; Moro 1991; Wilson et al. 1986; Wood 1970).

Antechinus agilis is generally associated with fallen logs, well developed leaf litter and dense understorey vegetation (Lindenmayer et al. 1994; Bennett 1993; Laidlaw & Wilson 1989; Statham & Harden 1982; Barnett et al. 1978). 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 (Solly et al. 1999; Baker 1998). 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, the burns had no net effect on the abundance of fallen logs (Tolhurst et al. 1992). 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). Observations by Humphries (1994) 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 *R. fuscipes* population was 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 the A3 or S3 treatments at Musk Creek, suggesting that colonisation of these areas had not taken place as of 32 (S3 treatment) to 39 (A3 treatment) months after burning. *Lomandra longifolia* beds 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 32 months after burning at Blakeville and only one individual was captured in the A3 treatment 28 months after fire.

Influence of burns on health, age structure and breeding success

Humphries (1994) found no significant difference in mean body weight between treatments. Heislers (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 fires was unchanged (Humphries 1994). Female *A agilis* born before the spring and autumn fires were captured in burnt treatments 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. This assisted the chances of rapid population recovery (Humphries 1994).

Humphries (1994) proposed that the low abundance of juvenile *R. fuscipes* following the spring fire treatment reflected relatively low reproductive success or unsuitability of burnt habitat for recruits.

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 need further refinement.

All of the radio-collars fitted by Kambouris (1998) lost their antennae before the conclusion of his study, resulting in a dramatic reduction in the effective detection distance. Kambouris recommended that the transmitter antenna be wrapped around the collar to limit its protrusion and improve its durability.

The radio-telemetry studies here 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.

Conclusion

No burning treatment was favoured by either *Antechinus agilis* or *Rattus fuscipes*, but habitat preferences were observed. Although these species 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, *R. fuscipes* populations are largely restricted to areas of dense sedgy vegetation in or near gullies. Seasonal fluctuations in *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 burns, *A. agilis* populations recovered to pre-fire levels within two breeding periods. 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. The source for immigration of *A. agilis* into burnt areas is therefore present and recolonisation will occur as suitable microhabitat recovers.

Populations of *R. fuscipes* took three breeding seasons to recover when more than half of its preferred habitat was burned during spring, but there was no recovery in 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-burn than single autumn fires. 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.

Fire management prescriptions should 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.

Recommendations for further study

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.

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 *Rattus fuscipes*, *R. lutreolus* and *Antechinus swainsonii* to achieve this.

Monitoring of *Rattus* spp. 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 development of suitable survey techniques is required to properly assess its 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 improved project design, future telemetry research will provide vital, cost-effective ecological information.

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Appendix

Fire treatment history of the fire effects research program, Wombat State Forest

Appendix Fire treatment history of the fire effects research program, Wombat State Forest

FESA	Treatment	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998
Blakeville	S 3	15/10			3/11			23/10			16/11				
	S 10	15/10									16/11				
	A 3			8/4					24/3					16/4	
	A 10			8/4										16/4	
Barkstead	S 3	13/11			28/10			15/10			18/11				
	S 10	13/11									18/11				
	A 3			27/4				11/4						17/4	
	A 10			27/4										17/4	
Burnt Bridge	S 3		30/9		11/11			18/11				16/11			
	S 10		30/9								23/11				
	A 3			27/3				10/4							
	A 10			27/3											
Musk Creek	S 3		10/11		3/11			23/10			17/11				
	S 10		10/11								17/11				
	A 3			26/3				7/5	24/3						
	A 10			26/3											
Kangaroo Creek	S 3	1/10			10/11			14/11			14/11				
	S 10	1/10									11/11				
	A 3			24/3					20/3					12/4	
	A 10			24/3										11/4	

(Source: K.G. Tolhurst, Forest Science Centre, pers. comm. 2000)