

# Small mammal community composition in relation to cattle grazing and associated burning in eucalypt forests of the Northern Tablelands of New South Wales

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## ABSTRACT

The small mammal fauna of cattle-grazed and moderately frequently-burnt forest areas was compared with that of ungrazed forest using systematic replicated live-trapping over 18 months at 12 sites in north-eastern New South Wales. Nine species of small mammals were captured during the study. There was no difference in species richness or diversity of small mammals between grazed/burnt and ungrazed sites, but species composition did differ. The fawn-footed melomys *Melomys cervinipes* was found only at ungrazed sites, all of which were characterised by a dense, complex shrub understorey. In contrast, the New Holland mouse *Pseudomys novaehollandiae*, Hastings River mouse *P. oralis*, and common dunnart *Sminthopsis murina* were found only at the grazed and frequently-burnt sites, which typically had an open, grassy and floristically diverse understorey. The abundance of small mammals was much higher in the ungrazed sites, due to extremely high numbers of the brown antechinus *Antechinus stuartii* and bush rat *Rattus fuscipes* in most ungrazed sites. The abundances of *A. stuartii*, *R. fuscipes* and *M. cervinipes* were highly correlated with overall vegetation complexity. Manipulation of cattle grazing, and more particularly fire regimes, offers a powerful tool for influencing the composition of small mammal communities, but one that must be carefully managed.

**Key words:** small mammals, grazing, livestock grazing, fire, disturbance ecology, native forests, community composition

## Introduction

The physical structure of vegetation has long been known to affect abundance and species composition of mammals and birds, with greater cover usually resulting in higher abundance and diversity (Leopold 1933; Elton 1939; MacArthur and MacArthur 1961; Birney *et al.* 1976). More recent studies in Australia have demonstrated that this is also true, at least in terms of abundance, for small mammals in eucalypt forests (Catling 1991; Bennett 1993; Catling and Burt 1995; Knight and Fox 2000).

Cover may provide increased protection from predators and inclement weather, and reduced agonistic conspecific and heterospecific encounters. It also provides more nest sites, food, seclusion, and increased gradients of nutrients and microclimate (Elton 1939; Huffaker 1958; McCoy and Bell 1991; Petren and Case 1998).

This study was conducted as part of a larger project focusing on the effects of grazing and associated grazer burning on vegetation structure and floristics, invertebrates and small mammals in the Northern Tablelands of New South Wales (Tasker 2002). Grazed forests in the study region are characterised by an open grassy understorey with a few small scattered shrubs, whereas ungrazed forest typically has a dense multi-layered shrub understorey with little ground cover vegetation. Many of the responses of small mammals to fire and livestock grazing can be interpreted as responses to the changes in vegetation structure and floristics resulting from these disturbances, rather than to the disturbances *per*

se (Fox 1982; Sutherland and Dickman 1999; Monamy and Fox 2000). In general, fire itself is not a major cause of mortality of small mammals in forests, particularly when the fire is of low intensity (Cowley *et al.* 1969; Leonard 1972; Catling 1991); there are apparently no described cases of cattle *per se* affecting the species composition or behaviour of small mammals.

In his paper in the original *Conservation of Australia's Forest Fauna*, Catling (1991) recognised the association of structurally simplified habitat with cattle-grazed and frequently-burnt forests throughout eastern New South Wales. He predicted, using general knowledge of species' biology, which were the animals that would be either negatively or positively affected by the structural simplification brought about by grazing and associated burning. In this study we tested these predictions for small ground-dwelling mammals by carrying out a detailed comparison of grazed/burnt and otherwise similar ungrazed forests in north-eastern NSW.

The small ground-dwelling mammal fauna of the forests of the eastern edge of the Northern Tablelands is diverse, with 14 native species and three introduced species known (Table 1). The native mammals consist of six dasyurid marsupials, one burramyid possum, and seven species of rodents. The two most common species of small mammal in the study area – as in the forests of eastern Australia generally – are the brown antechinus *Antechinus*

**Table 1.** Small (< 200 g average mass) ground-dwelling and semi-arboreal mammals known from the study area (Walcha and Kempsey-Wauchope forestry management areas). Data from Walcha/Nundle and Styx River Management Areas EIS and Kempsey-Wauchope Management Areas EIS (Forestry Commission of NSW 1993; Mount King Ecological Surveys 1995). \* denotes introduced species. Protected means protected under the NSW *National Parks and Wildlife Act 1974*. 'TSC Act' is the NSW *Threatened Species Conservation Act 1995*, and 'EPBC Act' is the national *Environment Protection and Biodiversity Conservation Act 1999*.

Scientific Name	Common Name	Conservation Status
<b>Dasyuridae</b>		
<i>Phascogale tapoatafa</i>	Brush-tailed phascogale	Vulnerable (TSC Act)
<i>Antechinus stuartii</i>	Brown antechinus	Protected (NSW NPW Act)
<i>Antechinus swainsonii</i>	Dusky antechinus	Protected (NSW NPW Act)
<i>Antechinus flavipes</i>	Yellow-footed antechinus	Protected (NSW NPW Act)
<i>Sminthopsis murina</i>	Common dunnart	Protected (NSW NPW Act)
<i>Planigale maculata</i>	Common planigale	Vulnerable (TSC Act)
<b>Burramyidae</b>		
<i>Cercartetus nanus</i>	Eastern pygmy possum	Vulnerable (TSC Act)
<b>Muridae</b>		
<i>Melomys cervinipes</i>	Fawn-footed melomys	Protected (NSW NPW Act)
<i>Mastacomys fuscus</i>	Broad-toothed rat	Vulnerable (TSC Act)
<i>Pseudomys oralis</i>	Hastings River mouse	Endangered (TSC and EPBC Acts)
<i>Pseudomys novaehollandiae</i>	New Holland mouse	Protected (NSW NPW Act)
<i>Pseudomys gracilicaudatus</i>	Eastern chestnut mouse	Vulnerable (TSC Act)
<i>Rattus fuscipes</i>	Bush rat	Protected (NSW NPW Act)
<i>Rattus lutreolus</i>	Swamp rat	Protected (NSW NPW Act)
* <i>Rattus rattus</i>	Black rat	-
* <i>Rattus norvegicus</i>	Brown rat	-
* <i>Mus musculus</i>	House mouse	-

*stuartii* and the bush rat *Rattus fuscipes*. The distribution of *A. stuartii* stretches from Kioloa in southern New South Wales north to the ranges on the NSW – Queensland border (Braithwaite 1995; Crowther 2002). This species was previously thought to be far more widespread, and much of the classic ecological research carried out on “*Antechinus stuartii*”, is now known to actually have been on the agile antechinus *A. agilis* (e.g. Hall and Lee 1982; Lazenby-Cohen and Cockburn 1988) or the sub-tropical antechinus *A. subtropicus* (e.g. Wood 1970).

## Management history

Cattle grazing takes place throughout large areas of the eucalypt forests and woodlands of north-eastern New South Wales, and the practice dates back to the arrival of European settlers and their cattle to the region in the mid-1800s. On the eastern edge of the tablelands and neighbouring escarpment cattle are predominant, rather than sheep, because they are less vulnerable to attack by dingoes and dogs, are generally hardier, and the fodder and climate are more suitable for them (Hassall and Associates Pty Ltd 1998).

Cattle grazing in forests occurs on private land, Crown leases, and areas within government-owned timber production forests (State Forests). In north-eastern NSW, 526,000 hectares of State Forest are officially grazed by livestock under permit and lease arrangements (Hassall and Associates Pty

Ltd 1998). Leases are generally unfenced, and livestock are typically put out in the forest for approximately six months over winter to spell adjacent pastures (Forestry Commission of NSW 1993; Hassall and Associates Pty Ltd 1998). In the study area the nominal stocking rate for forest grazing is about 1 animal per 20 ha. In drought years this may be as low as 1 per 70 ha (Forestry Commission of NSW 1995), but in more open forests on fertile soils it may be as much as 1 animal per 4 ha (Forestry Commission of NSW 1993).

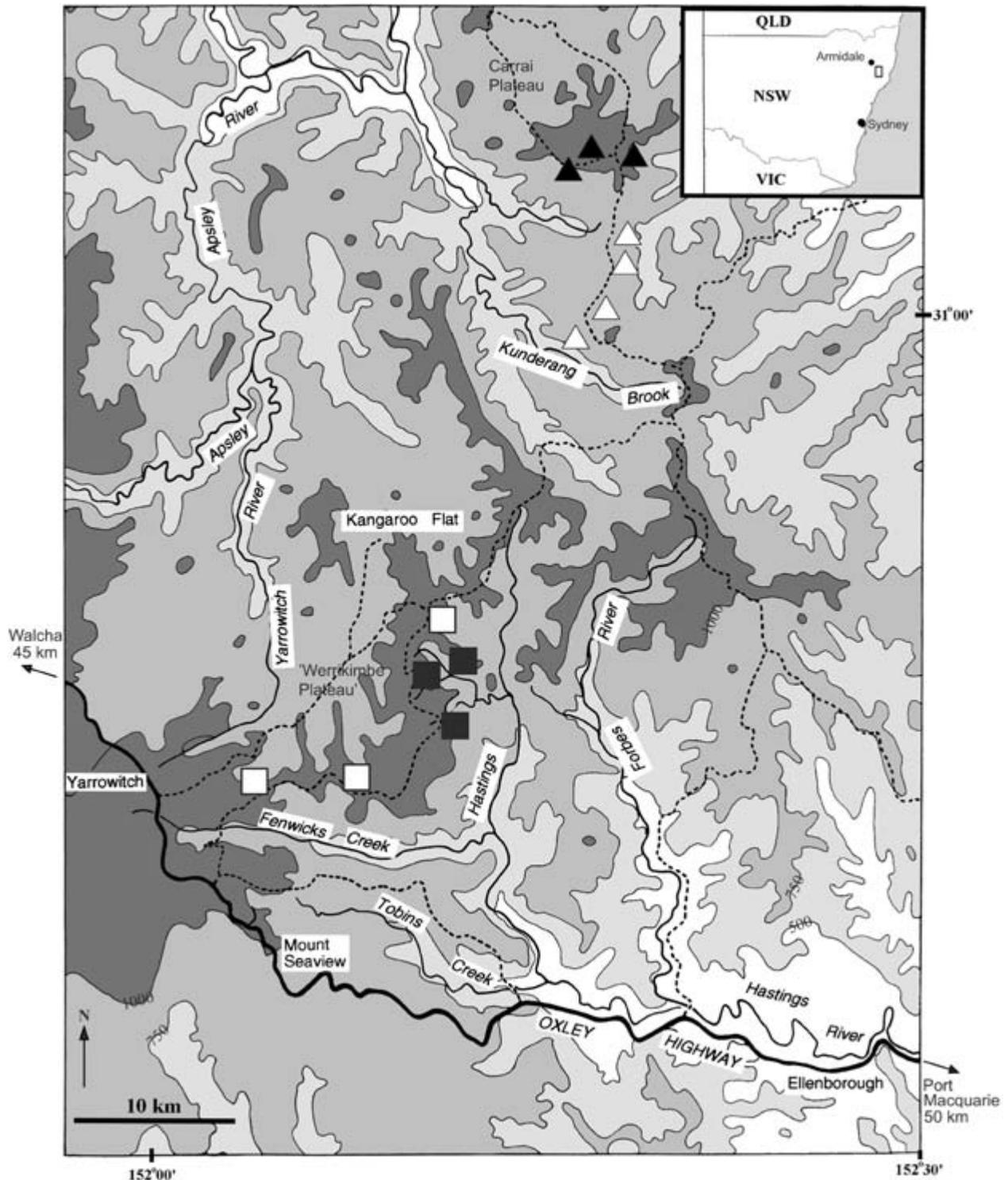
In most grazed forests the graziers periodically set fire to the forest to stimulate fresh growth of what is otherwise poor quality forage, and to maintain an open grassy understorey. This burning is typically carried out in spring, is relatively low intensity, and occurs at roughly 1-5 year intervals (Margules Groome Poyry Pty Ltd 1994; Christensen 1995). The frequency of burning varies considerably between graziers. The historical frequencies and spatial distribution of fires in the study region are poorly documented, as is generally the case, particularly prior to the 1990s.

The cattle-grazed/frequently-burnt sites in this study ('grazed/burnt' sites) had all been subjected to a regime of low intensity, seasonal cattle grazing and associated moderately frequent burning for approximately a century. The “ungrazed” sites had not been grazed by cattle, and available fire records indicated that they had only infrequently been burnt by wildfire: none had been burnt within 15 years prior to fieldwork commencing.

The aims of the study were to determine whether: a) composition of the small mammal community differed between grazed/burnt and ungrazed forest; b) abundance of small mammals differed between grazed/burnt and ungrazed forest; and c) abundance of small mammals was related to aspects of habitat structure, in particular vegetation complexity and density.

### Methods

The study area comprised the southern Carrai Plateau and Werrikimbe Plateau east of Walcha, part of the Northern Tablelands of New South Wales. The area is bounded in the north by the Macleay River west of Kempsey (30°45'S, 152°15'E) and in the south by the Oxley Highway (31°20'S, 152°15'E), and varies in elevation from 800–1100 m (Fig. 1).



**Figure 1.** Study area showing the location of the twelve study sites on the south-eastern edge of the Northern Tablelands. Grazed/burnt sites are marked with an open symbol, ungrazed sites with a filled symbol, Carrai Plateau sites with triangles, and 'Werrikimbe Plateau' sites with squares. Illustration by H. Finlay.

Using a Geographical Information Systems database, we generated transparent map overlays of various environmental, vegetation and management attributes at a 1:25,000 scale for use with topographic maps. From these we identified those parts of the region with a particular combination of topographic position, geology, elevation, logging history and forest type. These areas were all ridges or upper slopes, on sedimentary geology, 900-1100 m elevation, unlogged (Werrikimbe) or with as little logging disturbance possible (Carrai). All sites were classified on existing maps as New England blackbutt *Eucalyptus campanulata* forest type, but were found to include varying amounts of other canopy eucalypts: diehard stringybark *E. cameronii*, messmate stringybark *E. obliqua*, Sydney blue gum *E. saligna*, forest ribbon gum *E. nobilis*, and narrow-leaved peppermint *E. radiata* subsp. *sejuncta*.

Within these areas, 12 study sites were then identified. On both Carrai and Werrikimbe plateaux, three sites were located within forest grazing leases, and three were outside mapped grazing leases and as far as possible from cattle access points (i.e. grazing leases or properties), but were otherwise similar. The sites were chosen from maps, and then located in the field. This design allowed the influence of understorey complexity within a single forest type to be examined in a statistically robust manner; this contrasts with previous studies which have generally lacked replication, or have compared the fauna between different vegetation types (e.g. woodland versus rainforest) and made inferences about the importance of associated differences in vegetation structure.

Vegetation surveys of the sites showed that grazed and frequently-burnt sites were characterised by an open grassy understorey and few shrubs (Fig. 2a). The dominant understorey species at the six grazed sites were tussock grass *Poa sieberiana*, spiny mat-rush *Lomandra longifolia*, blade grass *Imperata cylindrica* and bracken *Pteridium esculentum*. In contrast, the six ungrazed and infrequently-burnt sites had a dense multi-layered understorey of shrubs (Fig. 2b), and the dominant understorey species were false bracken

*Calochlaena dubia*, mountain pepperbush *Tasmannia stipitata*, prickly coprosma *Coprosma quadrifida*, blackwood *Acacia melanoxylon* and spiny mat-rush *L. longifolia*, and at some sites *Banksia integrifolia* subsp. *monticola*. A full floristic description of the sites can be found in Tasker (2002).

At each of the sites a small mammal trapping grid of 5 x 5 trapping points was established. Traps were spaced 10 m apart, so that each grid covered an area of 40 x 40 m. At each trapping point an Elliott brand 'type A' folding aluminium live-trap (30 x 10 x 8 cm) was placed at the most suitable microhabitat within 2 m of the trapping point. A strict Elliott trapping protocol was implemented to maximise captures and minimise bias in captures (see Tasker and Dickman 2002 for details). For example, whenever a trap was soiled by a captured animal, the trap was replaced with a clean one. This is because both conspecific and heterospecific odours of animals which occupy similar niches can affect which species and individuals are caught subsequently (Wuensch 1982; Brown 1985; Tasker and Dickman 2002), which in turn can influence apparent species and/or demographic composition.

Trapping was carried out over 18 months in each of the four seasons: sites were trapped every three months from winter 1997 (July) to winter 1998 (July), and then 9 months later in autumn 1999 (April). Two replicates per combination of treatment and sub-region were trapped in winter 1997; subsequently this was increased to three replicates. The third ungrazed replicate site on Carrai Plateau was not established until summer 1998 due to access problems. Repeated temporal sampling was carried out to increase the chances of capturing rare species, to provide data on population processes, and to test whether differences between grazed/burnt and ungrazed treatments were consistent in all seasons. The mammal trapping grid at one of the permanent sites (Fitzroy) had to be abandoned after winter 1998 due to continued disturbance and raiding of the traps by a resident family of the vulnerable spotted-tailed quoll *Dasyurus maculatus* (Dasyuridae). A new site, Hueys Corner, 2.5 km away,



**Figure 2.** Photographs of a) a typical grazed/burnt site, showing the open, grassy understorey dominated by *Imperata cylindrica*, *Poa sieberiana* and *Pteridium esculentum* and the absence of mid-storey layers, and b) a typical ungrazed site, showing the structurally complex mid-storey dominated by – in this case – proteaceous shrubs such as *Banksia integrifolia* subsp. *monticola*, and mesic species such as *Tasmannia stipitata*.

was located as a replacement to maintain three replicates in the grazed/burnt Carrai combination. This new site was located in the same grazing lease, and had the same elevation, aspect, distance from the access road, and general floristic and structural composition.

At each trapping point, one Elliott trap was set each night, apart from autumn 1999 when two traps were set per point, as the trapping success had exceeded 80% on previous occasions (see Southern 1973). Each trap was baited with rolled oats, peanut butter and honey, and provided with a bedding of crushed *Eucalyptus* leaves and cotton wool. Traps were placed in plastic bags in winter, and also in spring and autumn if rain appeared likely. Trapping was carried out for four consecutive nights at each site in each season.

Traps were checked from dawn. Each captured animal was identified, weighed, its reproductive condition noted, and ear-clipped with a unique combination of notches. Bush rats were also given a small, individually-numbered monel tag in one ear, as they may damage their ears in aggressive encounters making identification from ear-clipping unreliable.

An index of abundance of each species was derived using the minimum-number-alive (MNA) method (Krebs 1999). For each species, differences in overall abundance of the cumulative number of individuals caught at each site between grazed/burnt and ungrazed treatments, and between the two sub-regions, were evaluated by analysis of variance using the GMAV software package (Underwood and Chapman 1998). The design was a 2-factor model, with grazing as a fixed factor ("grazed/burnt" or "ungrazed") and sub-region as a random factor (Carrai or Werrikimbe). The unit of replication was the site. Only species with >5 individuals caught were analysed in this way. Data were tested for homogeneity of variances using Cochran's test prior to analysis, and transformed where necessary (Underwood 1997). Post-hoc pooling was carried out if the interaction terms were non-significant at  $P = 0.25$  to increase the power of the tests (Underwood 1997).

Using the cumulative number of individuals per site, species richness and diversity were calculated for each site. Diversity was estimated using the Shannon-Wiener function ( $H'$ ), as expressed by MacArthur, according to Krebs (1999), i.e.:

$$N_1 = e^{H'}$$

where  $N_1$  is the number of equally common species that would produce the same diversity as  $H'$ . This was determined to be the most appropriate diversity index, because it is sensitive to the abundance of rarer species in the community, is widely used and it is easily interpretable (Krebs 1999).

In order to determine the influence of various habitat factors on small mammal abundances, we correlated the cumulative abundance of those mammal species which occurred in high enough numbers for statistical analysis, with a number of habitat variables:

1. percent cover vegetation >2 m in height (tall shrubs, small trees, tree ferns)

2. percent cover of shrubs 0.75 – 2 m in height
3. percent cover of shrubs <0.75 m in height
4. percent cover of ground cover vegetation (herbs and monocots)
5. total (percent) cover of all vegetation
6. vegetation complexity score
7. number of logs
8. leaf litter abundance

The first five were measured by the point-intercept method using a densitometer at 20 points on each of five regularly spaced 100 m transects per site; the resulting 100 +/- scores were then summed to give a single value for each layer for each site. The vegetation complexity score was a single value for each site (Newsome and Catling 1979), and was the sum of a visually estimated cover-abundance score for each of five vegetation layers (using the same classes as points 1 – 4 above, plus a category for tree canopy). The number of logs per site was the number of logs (defined as trunks or large branches > 10 cm diameter) intersected by a 20 x 50 m belt transect, running across the predominant slope. Leaf litter was assessed visually along this same transect (on the return walk) and the entire site allocated to one of five cover-abundance classes:

1. < 33 % leaf litter cover
2. 33 – 95 % leaf litter cover
3. > 95% leaf litter cover, but generally less than 1 cm deep
4. 100 % leaf litter cover, and 1 – 5 cm deep
5. 100% leaf litter cover, and >5cm deep

Abundance of each mammal species (expressed as the cumulative total number of individuals per site) was correlated with leaf litter cover across the 12 study sites using Spearman's rank correlation. All other correlations were performed using Pearson's correlation coefficient, as variables were continuous. Correlations were done in Systat v. 9 (SPSS 1998), and significance levels calculated manually as follows. Because there were many correlation coefficients, significance levels of all Pearson correlations were corrected to maintain an overall global Type I error rate of 0.05 for each species, using a sequential Bonferroni correction according to the Dunn-Sidak method (Sokal and Rohlf 1995). However, as Bonferroni corrections are by no means universally applied (Cabin and Mitchell 2000), the significant results before correction are also indicated in the results. To check for normal distribution of the data, plots of all correlations were inspected (Sokal and Rohlf 1995).

We tested for differences in the composition of the community between grazed/burnt and ungrazed sites and between regions using non-metric multi-dimensional scaling (nMDS) and analysis of similarity (ANOSIM, using all possible permutations: 100) in the program PRIMER (Clarke and Warwick 1994). A square-root transformation was applied to the data prior to analysis, because an examination of the raw data indicated that the numerically dominant species were more variable, as well as to increase the sensitivity of the analysis to rare and uncommon species (Clarke and Warwick 1994).

We also wished to know whether the differences between grazed/burnt and ungrazed sites were the same in all seasons. This was tested using separate ANOVAs for each season, rather than using an analysis with more factors or a repeated measures ANOVA, for the following reasons. Because the same grids were sampled in each season, many of the same animals were caught: this meant that seasons were not statistically independent and therefore could not be included as a factor in a more complex ANOVA (Underwood 1997). Because the number of sites trapped in each season was not constant until summer 1998 and during the study one site had to be substituted with another (due to quoll disturbance), a repeated measures ANOVA could not be used. Only *A. stuartii* and *R. fuscipes* were abundant enough to carry out separate analyses in each season.

Because of the low number of replicates in winter 1997, no formal statistical analysis of this season was conducted. In spring 1997, the Carrai ungrazed combination was one site short (due to access difficulties), so the group average was substituted as a dummy variable and the degrees of freedom in the residual reduced by one (Underwood 1997). As there were relatively few replicates the reduced degrees of freedom had the potential to substantially affect the results, so the analysis was repeated without the dummy substitution but with all other combinations of grazing treatment and region having one randomly chosen site eliminated (i.e. there were two replicates in each combination). As this did not change the results substantially, the analyses with the dummy variable are presented.

## Results

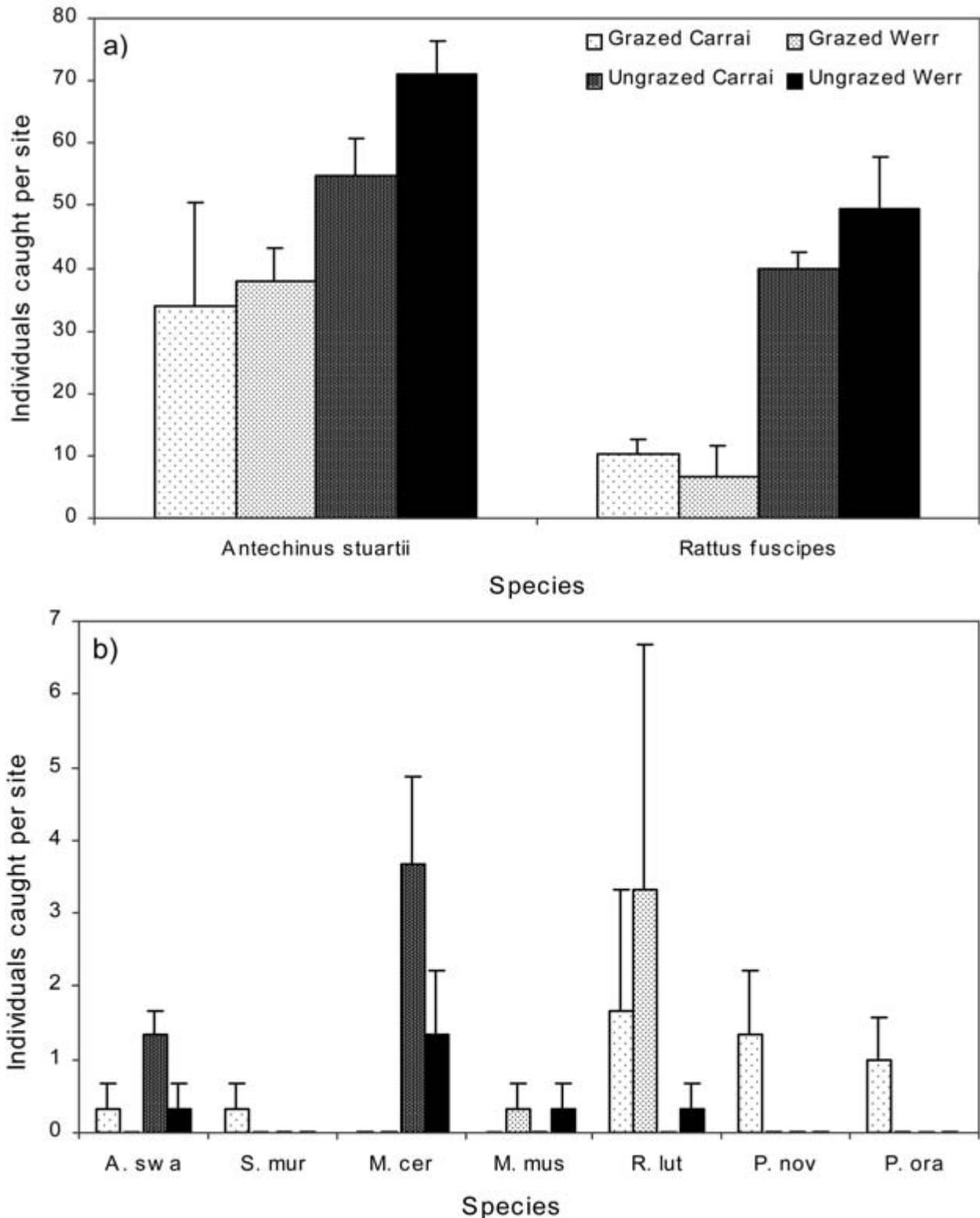
### Abundance and community composition of small mammals

Nine species of small mammals were captured at the 12 study sites during 6,705 trap-nights. The most common species by far were *A. stuartii* (639 individuals caught 1,697 times), and *R. fuscipes* (388 individuals captured 1,145 times). The swamp rat *Rattus lutreolus* and the fawn-footed melomys *Melomys cervinipes* were the next most commonly trapped species, with 16 individuals captured 30 times, and 15 individuals captured 28 times, respectively. Only 16 individuals were caught of the remaining species; the dusky antechinus *A. swainsonii* (6), common dunnart *Sminthopsis murina* (1), house mouse *Mus musculus* (2), New Holland mouse *Pseudomys novaehollandiae* (4), and Hastings River mouse *P. oralis* (3).

There was no significant difference in the species richness or diversity of small mammals between the grazed/burnt and ungrazed sites, with average species richness being 3.50 (3.29 including the Hueys Corner site) and 3.83, respectively, and average diversity 2.20 (2.07) and 2.28, respectively (Table 2). There was, however, a difference in small mammal diversity between the sub-regions, with Carrai (2.51) having a significantly higher diversity of small mammals than Werrikimbe (1.94; Table 2).



**Figure 3.** nMDS plot comparing small mammal species composition in grazed/burnt and ungrazed sites. Open symbols = grazed/burnt sites, closed symbols = ungrazed sites, triangles = Carrai sub-region, and squares = Werrikimbe sub-region. Grazing treatment Global  $R = 0.741$ ,  $P = 0.01$ ; region Global  $R = 0.278$ ,  $P = 0.07$ .



**Figure 4.** Total abundance of a) *Antechinus stuartii* and *Rattus fuscipes*, and b) *A. swainsonii*, *Sminthopsis murina*, *Melomys cervinipes*, *Mus musculus*, *Rattus lutreolus*, *Pseudomys novaehollandiae* and *P. oralis*, in grazed/burnt and ungrazed sites in Carrai and Werrikimbe sub-regions. Means represent the mean of the cumulative number of individuals at each site from all trapping sessions (winter 1997 – winter 1998, autumn 1999) ± S.E.

The composition of the small mammal communities did differ markedly between the grazed/burnt and ungrazed treatments (Fig. 3; grazing global  $R = 0.741$ ,  $P = 0.01$ , region global  $R = 0.278$ ,  $P = 0.07$ , all possible permutations). In addition there were

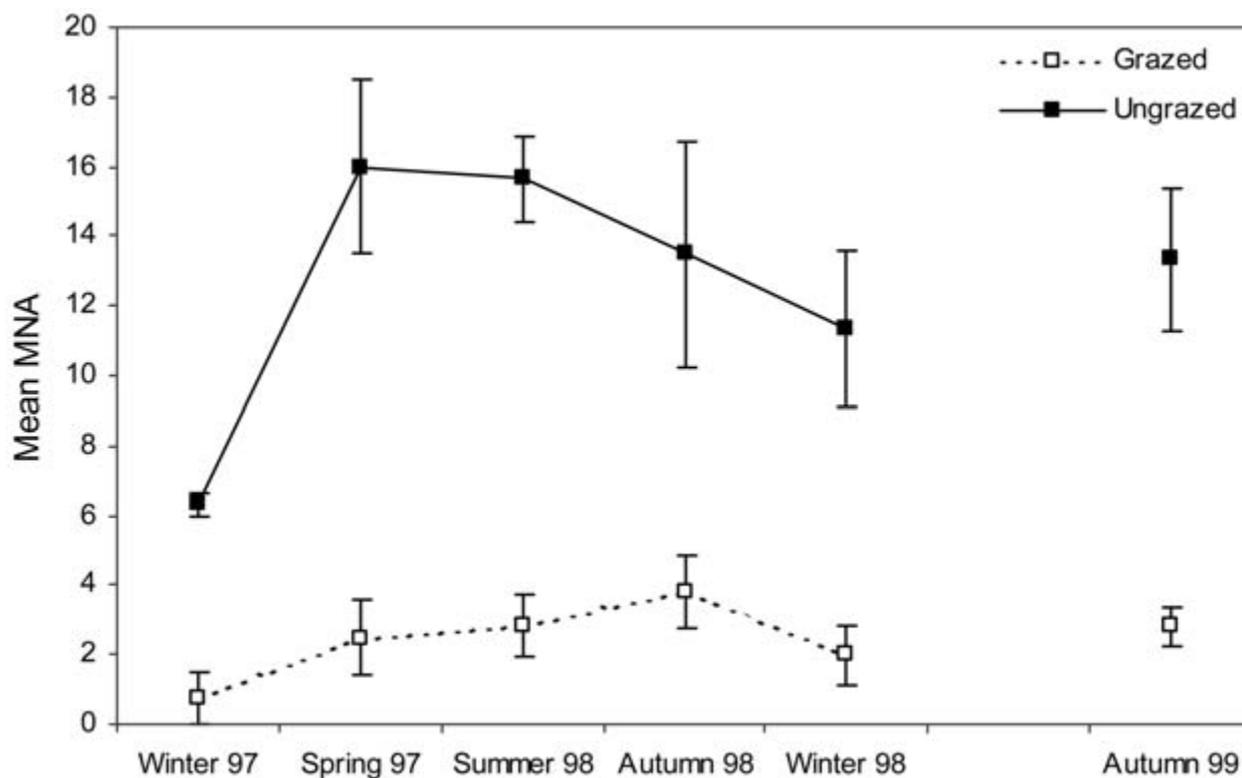
significantly and consistently more small mammals at the ungrazed sites, in particular *A. stuartii*, *R. fuscipes*, *A. swainsonii* and *M. cervinipes* (Fig. 4, Table 2). In fact, *M. cervinipes* was caught only at the ungrazed sites. In contrast, three small mammal species were caught only

**Table 2.** Analyses of diversity and abundance of small mammals at grazed/burnt and ungrazed sites. "Treatment" is the comparison between grazed/burnt and ungrazed sites (fixed factor), "Region" is the comparison between Carrai and Werrikimbe sub-regions (random factor); T = transformation applied; Cochran's C is that after transformation, if relevant; \*\*\*  $P < 0.001$ ; \*  $P < 0.05$ ; "ns" no significant difference. For SNK tests "gr" = grazed/burnt treatment, "un" = ungrazed treatment, C = Carrai, W = Werrikimbe; "<" or ">" indicate a significant difference.

Sources of Variation	df	Species richness Shannon-Wiener			Abundance of <i>A. stuartii</i>		Abundance of <i>R. fuscipes</i>		Abundance of <i>A. swainsonii</i>		
		MS	F	Cochran's C = 0.76 <sup>ns</sup>	MS	F	MS	F	MS	F	
Treatment = tr	1	0.03	0.36 <sup>ns</sup>	0.01	0.03 <sup>ns</sup>	1564.08	5.93*	3816.33	45.13***	1.33	5.14*
Region = re	1	0.32	4.20 <sup>ns</sup>	0.97	9.41*	114.08	0.43 <sup>ns</sup>	16.33	0.19 <sup>ns</sup>	1.33	5.14*
Tr x re	1	0.09		0.29	2.79 <sup>ns</sup>	310.08		147.00		0.33	
Residual = R	8	0.08		0.10		257.83		76.75		0.25	
Pooled (tr x re + R)	9	0.08				263.64		84.56		0.26	
SNK results:				C > W		Gr < Un		Gr < Un		Gr < Un C > W	

in the grazed/burnt sites; *S. murina*, *P. novaehollandiae* and *P. oralis*, and another, *R. lutreolus*, was much more common at the grazed/burnt sites, with only one individual of the 16 caught at an ungrazed site. All the species caught predominantly in grazed/burnt sites were uncommon and present only at very few sites. Because there were no *M. cervinipes* in the grazed/burnt sites, and *R. lutreolus* was present at only 3 of the 12 sites it was not possible to statistically compare treatments for either species.

*Rattus fuscipes* was significantly more abundant in the ungrazed sites at all times of year in both regions (Fig. 5, Table 3). The separation between the grazed/burnt and ungrazed sites tended to be greater in Werrikimbe sub-region due to both lower numbers in the grazed/burnt sites and higher numbers in the ungrazed sites, but neither the region, nor treatment x region effects, were significant (Table 3). The population sizes showed considerable seasonal fluctuation, particularly in the ungrazed treatment, with the fewest animals caught in winter.



**Figure 5.** Seasonal abundance of *R. fuscipes* in grazed/burnt and ungrazed sites in both sub-regions combined. Figures are the mean number of individuals known to be alive (MNA) at each site ( $\pm$  S.E.), and sub-regions have been combined as they showed the same pattern. Greater trapping effort was expended in autumn 1999, when additional traps were set on the final night if fewer than 20% of traps the previous night remained unoccupied.

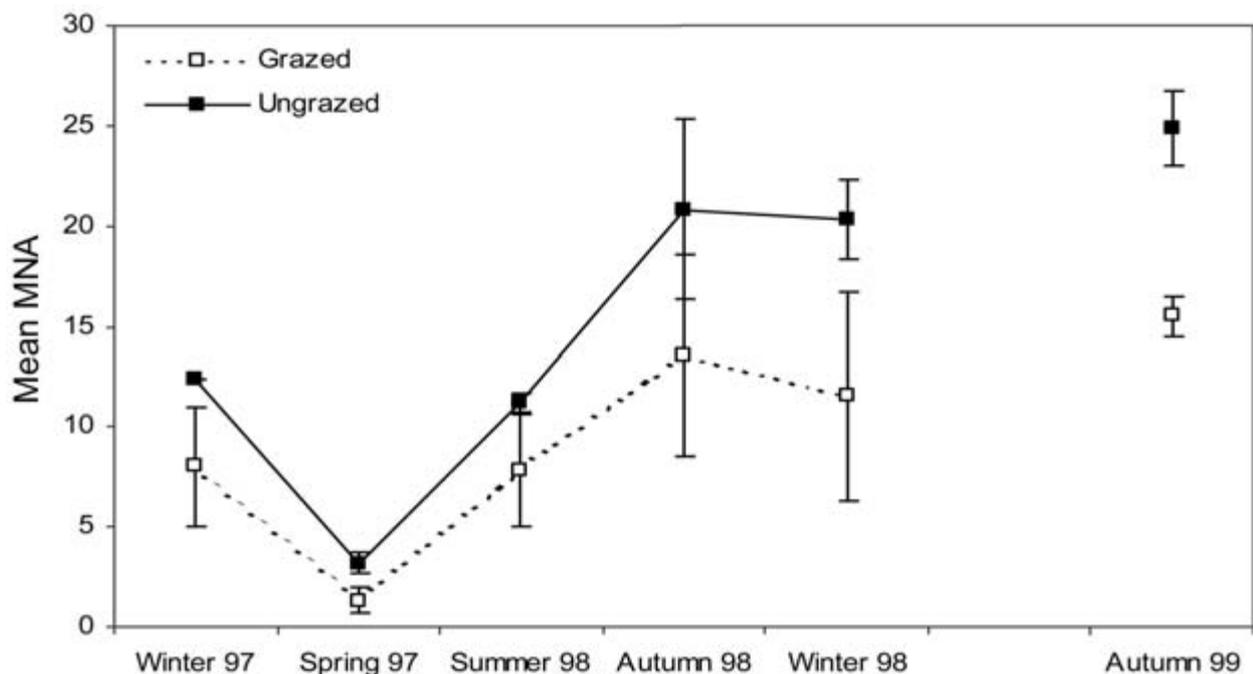
**Table 3.** Analyses of abundance of *Rattus fuscipes* (minimum-number-alive – MNA) in grazed/burnt and ungrazed sites in different seasons. “Treatment” is the comparison between grazed/burnt and ungrazed sites (fixed factor), “Region” is the comparison between Carrai and Werrikimbe sub-regions (random factor); \*\*\*  $P < 0.001$ ; \*\*  $P < 0.01$ ; \*  $P < 0.05$ ; “ns” no significant difference. For SNK tests “gr” = grazed/burnt treatment, “un” = ungrazed treatment, C = Carrai, W = Werrikimbe; “<” or “>” indicate a significant difference.

Sources of Variation	df	Spring 1997 Cochran's C = 0.44 <sup>ns</sup>		Summer 1998 Cochran's C = 0.62 <sup>ns</sup>		Autumn 1998 Cochran's C = 0.74 <sup>ns</sup>		Winter 1998 Cochran's C = 0.61 <sup>ns</sup>		Autumn 1999 Cochran's C = 0.50 <sup>ns</sup>	
		MS	F								
Treatment = tr	1	520.08	25.67***	494.08	64.06***	280.33	7.72*	261.33	13.84**	363.00	24.14***
Region = re	1	0.75	0.04 <sup>ns</sup>	0.75	0.10 <sup>ns</sup>	21.33	0.59 <sup>ns</sup>	5.33	0.28 <sup>ns</sup>	5.33	0.35 <sup>ns</sup>
Tr × re	1	24.08		0.75		56.33		5.33		12.00	
Residual = R	8 <sup>1</sup>	19.71		8.58		33.83		20.58		15.42	
Pooled (tr × re + R)	9 <sup>1</sup>	20.26		7.71		36.33		18.89		15.04	
SNK results:		Gr < Un		Gr < Un		Gr < Un		Gr < Un		Gr < Un	

<sup>1</sup>The degrees of freedom for the residual were reduced to 7 in spring 1997, due to a missing replicate substituted with a dummy variable. The pooled degrees of freedom were therefore also reduced to 8 for this analysis. See text for details.

*Antechinus stuartii*, in contrast, showed less distinct differences between grazed/burnt and ungrazed sites (Fig. 6), particularly in summer and autumn 1998, with Carrai Plateau showing no difference in abundance between the grazed/burnt and ungrazed sites for the first year of the study. This was a consequence of very low numbers of *A. stuartii* at the ungrazed sites in Carrai in the first year of trapping, possibly as a result of a long-term population fluctuation (Tasker 2002), and due to one grazed/burnt site in Carrai that had consistently high numbers of *A. stuartii*. The latter site was intermediate in structural complexity between grazed/burnt and ungrazed sites and this apparently resulted in the higher abundance of *A. stuartii*. The differences in

abundance of *A. stuartii* between grazed/burnt and ungrazed sites were not significant other than in winter 1998 (Table 4), but there was also a strong tendency in autumn 1999. There were large seasonal fluctuations in the numbers of animals caught, with very few animals present in spring, following the virtually complete male die-off after mating in winter. Surprisingly, four male *A. stuartii* were found to live to a second breeding season, and this represented approximately 1% of males trapped. This is apparently the first time that male *A. stuartii* have been recorded to breed a second year in the wild, although it has previously been found that males kept apart from other males in captivity may survive for up to two years (Woolley 1966).

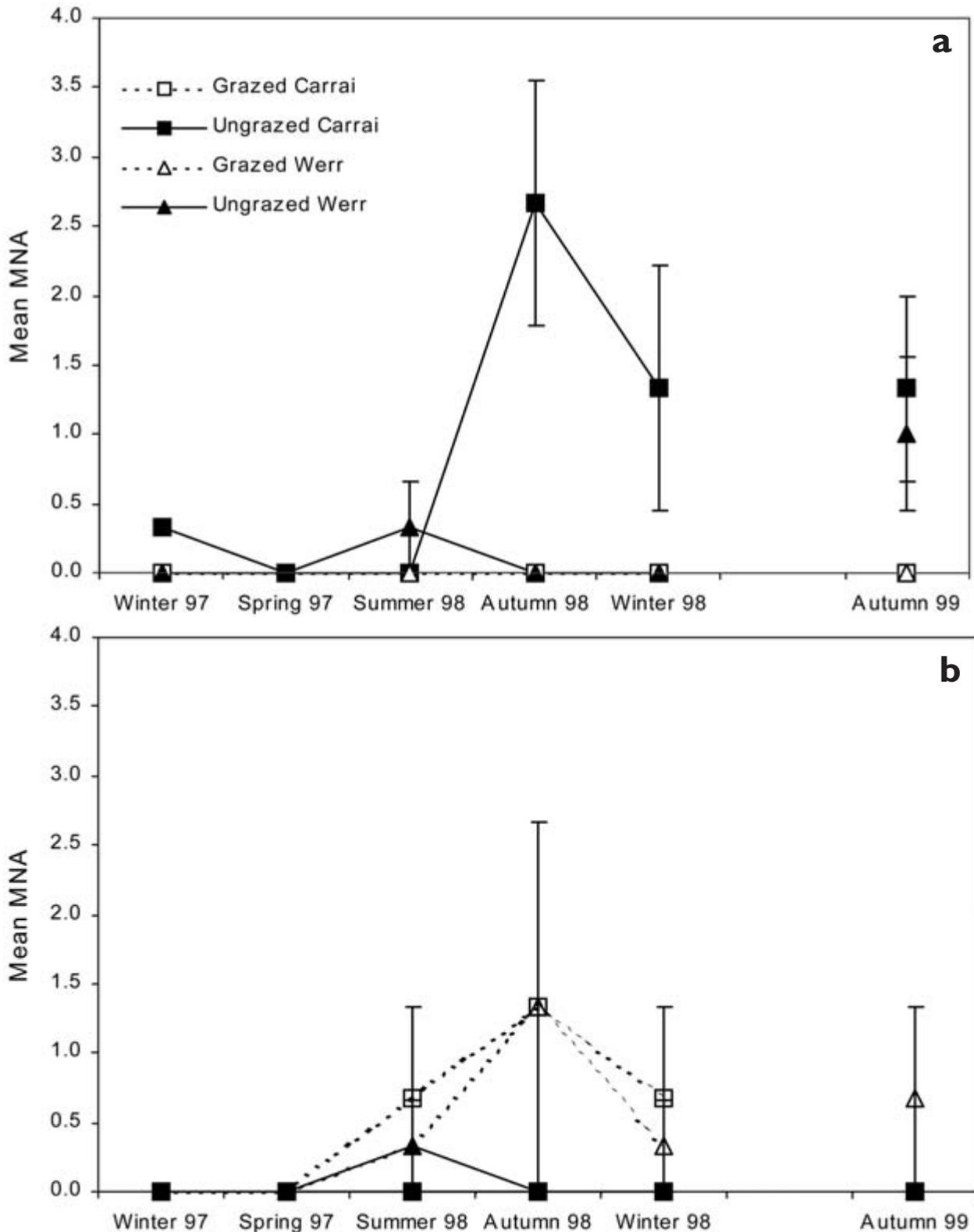


**Figure 6.** Seasonal abundance of *A. stuartii* in grazed/burnt and ungrazed sites in both sub-regions combined. Figures are the minimum-number-alive (MNA) at each site ( $\pm$  S.E.), and sub-regions have been combined as they showed the same pattern. Greater trapping effort was expended in autumn 1999, when additional traps were set on the final night if fewer than 20% of traps the previous night remained unoccupied.

The trappable populations of the other species also showed seasonal fluctuations (Fig. 7), with both *M. cervinipes* and *R. lutreolus* peaking in autumn 1998, although there was very large variance as a result of the low numbers of individuals caught.

### Relationships between habitat variables and small mammals

The abundances of *R. fuscipes*, *A. stuartii* and *M. cervinipes* showed a very close positive relationship with



**Figure 7.** Seasonal abundance of a) *Melomys cervinipes* and b) *Rattus lutreolus* in grazed/burnt and ungrazed sites in both Carrai and Werrikimbe sub-regions. Figures are the mean number of individuals known to be alive (MNA) at each site ( $\pm$  S.E.). Greater trapping effort was expended in autumn 1999, when additional traps were set on the final night if fewer than 20% of traps the previous night remained unoccupied.

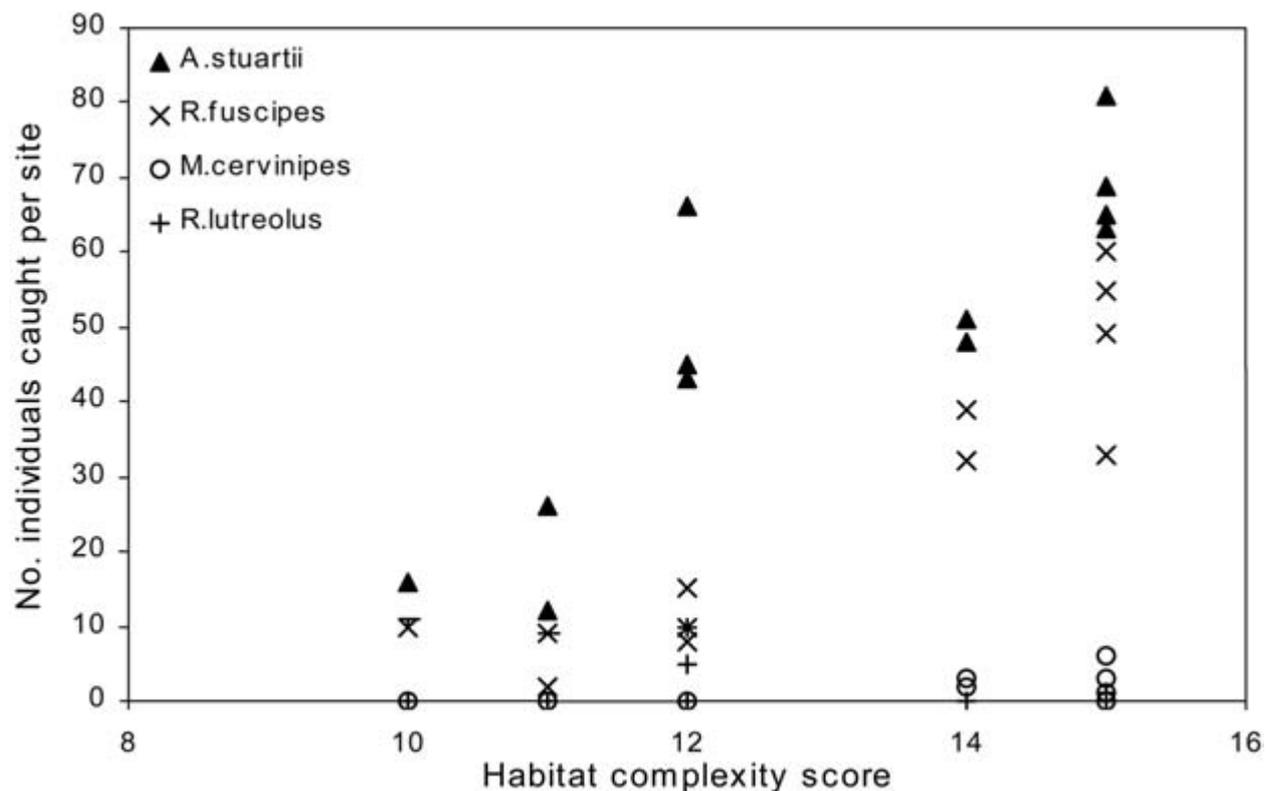
**Table 4:** Analyses of abundance of *Antechinus stuartii* (minimum-number-alive – MNA) in grazed/burnt and ungrazed sites in different seasons. "Treatment" is the comparison between grazed/burnt and ungrazed sites (fixed factor), "Region" is the comparison between Carrai and Werrikimbe sub-regions (random factor); T = transformation applied; Cochran's C is that after transformation, if relevant; \*\* P<0.01; \* P<0.05; "ns" no significant difference. For SNK tests "gr" = grazed/burnt treatment, "un" = ungrazed treatment, C = Carrai, W = Werrikimbe; "<" or ">" indicate a significant difference.

Sources of Variation		Spring 1997 Cochran's C = 0.75 <sup>ns</sup> T = Sqrt (X + 1)		Summer 1998 Cochran's C = 0.74 <sup>ns</sup>		Autumn 1998 Cochran's C = 0.42 <sup>ns</sup>		Winter 1998 Cochran's C = 0.50 <sup>ns</sup>		Autumn 1999 Cochran's C = 0.70 <sup>ns</sup>	
	df	MS	F	MS	F	MS	F	MS	F	MS	F
Treatment = tr	1	0.62	2.63 <sup>ns</sup>	58.52	0.71	161.33	1.67 <sup>ns</sup>	234.08	5.87*	261.33	3.88 <sup>ns</sup>
Region = re	1	0.03	0.14 <sup>ns</sup>	117.19	14.24**	133.33	2.97 <sup>ns</sup>	0.08	<0.001 <sup>ns</sup>	108.00	1.60 <sup>ns</sup>
Tr x re	1	0.02		82.69	10.05*	96.33	2.15 <sup>ns</sup>	36.75		8.33	
Residual = R	8 <sup>1</sup>	0.27		8.23		44.83		40.25		74.75	
Pooled (tr x re + R)	9 <sup>1</sup>	0.24						39.86			
SNK results:					W: Gr < Un				Gr < Un		

<sup>1</sup>The degrees of freedom for the residual were reduced to 7 in spring 1997, due to a missing replicate substituted with a dummy variable. The pooled degrees of freedom were therefore also reduced to 8 for this analysis. See text for details.

vegetation complexity across all 12 sites (Fig. 8; Table 5); this was strongest for *R. fuscipes* and weakest for *R. lutreolus*. Density of vegetation > 2 m was important for *R. fuscipes* and *M. cervinipes*, and total vegetation cover was important for *A. stuartii*. Abundance of *M. cervinipes* was also correlated with the number of logs. Interestingly, when the correlations between habitat variables were examined separately for the grazed/burnt and ungrazed sites (Table 5), although the results are no longer significant because of the smaller sample size, abundance of *A. stuartii* in the grazed/burnt sites remained closely correlated with

vegetation complexity and total vegetation cover, but was also correlated with shrubs of medium height. As only vegetation complexity appeared important in ungrazed sites (Table 5), this indicates that medium and low vegetation cover assumes a greater importance in more open grazed forest where there is significantly less cover of tall understorey. There were no strong correlations of *R. fuscipes* with any site-wide features at the grazed/burnt or ungrazed sites, indicating that the abundance-vegetation complexity relationship was weak across the limited range of abundances within each treatment type.



**Figure 8.** Association between small mammal abundance for the four most common species and the vegetation complexity score at each of the 12 permanent sites. See text for details.

**Table 5.** Correlations between small mammal abundance (cumulative total numbers) and aspects of habitat at all sites ( $n = 12$  sites). Pearson correlations were used for all data, other than leaf litter, for which Spearman's correlation was used as this employed ranked cover class data. Significance results are shown for correlations after sequential Bonferroni correction of the global significance level for each species (see text); \*\*  $P = 0.01$ , \*  $P = 0.05$ . Figures in bold but with no asterisk indicate correlations that were significant before Bonferroni correction.

	<i>A. stuartii</i>	<i>R. fuscipes</i>	<i>M. cervinipes</i>	<i>R. lutreolus</i>
<b>All Sites (<math>n = 12</math>)</b>				
Vegetation > 2 m	0.456	<b>0.807*</b>	<b>0.748*</b>	-0.438
Shrubs 0.75 – 2 m	0.551	<b>0.701</b>	0.201	0.004
Shrubs < 0.75 m	-0.116	-0.485	-0.221	0.228
Ground cover vegetation	-0.257	<b>-0.643</b>	-0.376	0.319
Total vegetation cover	<b>0.701</b>	0.487	0.462	0.271
Vegetation complexity score	<b>0.847**</b>	<b>0.910**</b>	<b>0.639</b>	-0.206
Number of logs	0.222	0.359	<b>0.646**</b>	-0.431
Leaf litter abundance	0.466	0.535	<b>0.557</b>	0.199
<b>Grazed/burnt Sites (<math>n = 6</math>)</b>				
Vegetation > 2 m	-0.545	0.189		-0.336
Shrubs 0.75 – 2 m	<b>0.828</b>	0.648		<b>0.824</b>
Shrubs < 0.75 m	0.578	0.030		0.019
Ground cover vegetation	0.537	0.331		0.033
Total vegetation cover	<b>0.914</b>	0.474		0.698
Vegetation complexity score	<b>0.822</b>	0.292		0.586
Number of logs	0.128	0.405		-0.508
Leaf litter abundance	0.293	0.099		0.693
<b>Ungrazed Sites (<math>n = 6</math>)</b>				
Vegetation > 2 m	-0.673	-0.217	0.484	
Shrubs 0.75 – 2 m	0.157	0.680	-0.246	
Shrubs < 0.75 m	-0.146	0.329	<b>0.880</b>	
Ground cover vegetation	0.435	-0.029	0.274	
Total vegetation cover	-0.633	-0.211	0.396	
Vegetation complexity score	<b>0.853</b>	0.605	0.000	
Number of logs	-0.549	-0.417	0.579	
Leaf litter abundance	-0.655	-0.131	0.266	

## Discussion

The composition of the small mammal communities differed markedly between the grazed/burnt and ungrazed sites. This was a particularly striking result considering that the sites were all in the same habitat type – tableland eucalypt forest dominated by *E. obliqua*, *E. cameronii* and *E. campanulata*, on ridges on the same soil type and geology, at the same elevation, and with a similar logging history (within each of the sub-regions). The grazed/burnt and ungrazed sites differed, however, in the structure and composition of the understorey vegetation as a consequence of the differences in their history of cattle grazing and burning (Tasker 2002). There was no difference between the grazed/burnt and ungrazed sites in their proximity to riparian areas, which often have higher densities of small mammals than surrounding areas (Newsome and Catling 1979), or their proximity to rainforests, which may act as refuges during wildfire or drought (Press 1986).

These results demonstrate that the general findings of Catling and colleagues (Newsome and Catling 1979; Catling 1991; Catling and Burt 1995; Catling and Coops 1999) comparing the abundance of small mammals among different vegetation types and forest types also apply within a restricted sub-set of forests that differ essentially only in their management. These in turn result in differences in understorey structure. This finding confirms that aerial mapping of the structural complexity of understorey vegetation as devised by Catling and Coops (1999) offers considerable value in predicting areas of high abundance of these species.

There were also floristic differences between the grazed/burnt and ungrazed sites, however these are unlikely to markedly influence the common species given the very broad range of habitats they utilise, their generalist diet, and their lack of strict floristic association (Braithwaite 1995; Lunney 1995). Subtle seasonal changes in the use of different floristic 'groups' at the trap scale have been demonstrated for *R. fuscipes* (i.e. rats were caught more

often in traps placed within a certain floristic 'group'; Braithwaite *et al.* 1978; Hall and Lee 1982), but given the scale and manner in which this was assessed – within a 5 or 10 m radius around each trap – it may also reflect the fact that certain species of plants tend to have particular growth and cover characteristics, which the animals prefer, rather than any particular plant species *per se*.

Both grazed and ungrazed forest had similar numbers of small mammal species overall, but some species were present only in grazed or ungrazed forest. This contradicts the prediction of Catling and Burt (1995) that eucalypt forest with fewer understorey shrubs would have fewer species of small mammals. In this study, moderately frequent disturbance appears to result in habitat suitable for early-mid successional species. This discrepancy may result from differences in the disturbance regimes between the grazed sites of Catling and Burt, and those of this study, and perhaps also from differences in trapping intensity between this study and that of Catling and Burt (1995). More intensive trapping, as in this study, results in more rare and uncommon species being recorded (Tasker and Dickman 2002).

We now briefly review the factors influencing each of the small mammal species trapped.

### *Antechinus stuartii* – Brown Antechinus

There have been contradictory reports of the habitat requirements of *A. stuartii*. Some studies have concluded that *A. stuartii* is more often caught in dense vegetation (Dickman and Woodside 1983; Knight and Fox 2000), like its smaller and better-studied congener *A. agilis* (Dickman 1980; Moro 1991; Bennett 1993; Lindenmayer *et al.* 1994; Sutherland and Predavec 1999). Others have identified strong associations between *A. stuartii* and indices of tree hollow abundance (Dickman and Steeves 2004), while yet further studies have found no relationship, or a trivial one with few of the many variables measured (Statham and Harden 1982), again similar to findings for *A. agilis* (Stewart 1979; Hall and Lee 1982). The lack of association of *A. stuartii* with habitat variables in these latter studies can be attributed to the way in which the variables were measured, a lack of statistical power, or the inadequacy of the sampling design. Most studies have assessed vegetation cover in a small area around each trap (e.g. Stewart 1979). This is actually evaluating microhabitat use, which can reflect activity levels and interspecific behavioural interactions as well as habitat preferences. For example, a dominant species may exclude a sub-dominant one from preferred habitat (Dickman and Woodside 1983; Thompson and Fox 1993; Luo and Fox 1995; Monamy and Fox 1997; Maitz and Dickman 2001; Tasker and Dickman 2002). Sutherland and Predavec (1999) found that *A. agilis* did not respond to increased cover at the trap-scale, but did respond to cover when measured in a 10 m radius around each trap.

Many of the studies cited above also assessed vegetation in many small and specific categories. For example, Hall and Lee (1982) measured vegetation cover in 12 x 30.5 cm height intervals, and then grouped these into seven groups with varying proportions in each of the layers. It

makes intuitive sense that any one of these categories has little importance to a small antechinus foraging on the forest floor, although when taken collectively, i.e. cover in general, probably are important. Principal Components Analysis is one way to avoid the problem of measuring many elements that on their own are not important, as it assesses the effect of all habitat variables together (e.g. Knight and Fox 2000). The vegetation complexity score, or "habitat complexity score" of Newsome and Catling (1979; see also Barnett *et al.* 1978; Dickman and Woodside 1983; Catling and Burt 1995) is effectively acting in the same way, by summing the effects of each vegetation component, and this is probably why it has proved to be so effective in predicting the abundance of *A. stuartii* in widely divergent habitats and localities (Catling and Coops 1999). That it is total cover that is relevant to *A. stuartii* is supported by the similar correlation values for the vegetation complexity score and total vegetation cover, and by the high abundance of *A. stuartii* in rainforest environments (e.g. Press 1986), which have very high total cover even though they often have very little understorey (Specht 1970; Specht and Morgan 1981).

Some studies have found a correlation of *A. stuartii* abundance and logs (e.g. Barnett *et al.* 1978; Cox *et al.* 2003), but many have not (e.g. Statham and Harden 1982), and it is likely that logs may not be important where there is abundant vegetation cover, but crucial where there is not. *Antechinus stuartii* use logs as runways, foraging sites, and sometimes nesting sites (Dickman 1991), although in this area they prefer to nest well above the ground (unpubl. radio-tracking data). The lack of tight associations of *A. stuartii* with understorey elements probably reflects its semi-arboreal habits; provided that trees are present (which are used extensively for foraging and nesting) it can probably tolerate sub-optimal understorey more than a strictly ground-dwelling species can.

Thus much of the apparent inconsistency in the habitat associations of *A. stuartii* is probably a result of the ways in which the association has been measured, as well as the variable importance of each habitat component, depending on the context. Many studies were also carried out at only a single site, thus they lack statistical power.

The lack of significant differences in abundance of *A. stuartii* between the grazed/burnt and ungrazed treatments at Carrai sub-region in any one trapping period is likely due to a number of factors. Firstly, one grazed/burnt site at Carrai was, in many ways, intermediate in structure between a grazed and an ungrazed site. It had a higher shrub cover – a patchy and dense mid-storey of brush pepper bush *Tasmannia insipida* – and a much higher number of logs than any other of the grazed/burnt sites. It also had a much higher cumulative number of *A. stuartii* than any other grazed site, as high as many ungrazed sites. In addition, one ungrazed site at Carrai had a very low number of *A. stuartii* until winter 1998, although there was no apparent reason for this.

Numbers of *A. stuartii* at the ungrazed sites were often extremely high. For example, in autumn 1998, 33 individuals were captured at one Werrikimbe site, and > 20 individuals were commonly present in any given season

on trapping grids in ungrazed sites. The trapping grids covered an area of only 40 x 40 m, and although the area actually sampled is considerably larger, other studies using a similar grid array have not caught such high numbers of this species. In fact, this is the apparently the highest density of *A. stuartii* recorded.

### ***Antechinus swainsonii* – Dusky Antechinus**

*Antechinus swainsonii* is most often associated with dense vegetation cover, particularly of *Blechnum* ferns, and thick leaf litter, which are most often found along creek lines (Stewart 1979; Hall and Lee 1982; Dickman and Woodside 1983; Hocking and Guiler 1983; Dickman 1986) and this close correlation was exactly the pattern found in this study. All of the traps where this species was caught had an extremely dense cover of *Blechnum nudum* and *B. cartilaginum* and leaf litter, or in the case of the one grazed/burnt site where a juvenile male was found, of *Pteridium esculentum*. In fact, the sole grazed/burnt site where this species was recorded was the only grazed site with a dense cover of ferns. *Antechinus swainsonii* is uncommon on the Northern Tablelands and escarpment of the study region (NPWS Wildlife Atlas, Truyard Pty. Ltd. 1992; Forestry Commission of NSW 1993; Mount King Ecological Surveys 1995). A regime of cattle grazing and associated burning that reduces understorey cover, particularly of ferns, is likely to have a negative effect on this already scarce species.

### ***Rattus fuscipes* – Bush Rat**

*Rattus fuscipes* is well-known to prefer dense vegetation cover, with all studies to date finding a significant relationship between local distribution and abundance and vegetation density or the structural complexity of the vegetation (e.g. Leonard 1972; Fletcher 1977; Stewart 1979; Dickman and Woodside 1983; Maitz and Dickman 2001) and also leaf litter abundance (Fox and McKay 1981; Maitz and Dickman 2001). This was also the case in this study. *Rattus fuscipes* showed an almost linear relationship with the vegetation complexity score. The majority of previous studies have quantified the response of *R. fuscipes* to habitat at a trap-level, and often in terms of numbers of captures not numbers of individuals. This records activity levels in particular parts of a site, rather than abundance at a site, although these two measures may be related. It is apparent, nevertheless, that *R. fuscipes* has a strong preference for dense vegetation. Some authors have concluded that rainforest is optimal habitat for *R. fuscipes* (Hockings 1981), however extremely high densities of *R. fuscipes* in our (sclerophyll forest) sites suggest that the apparent selection for closed forest is a by-product of selection for dense overall vegetation cover. For example, 26 individuals were captured on a Werrikimbe grid in autumn 1998, and 23 on another in spring 1997, apparently the highest recorded densities of this species.

Given the significant effect that cattle grazing and associated burning has in reducing understorey cover (Henderson and Keith 2002, Tasker 2002), such a management regime will have a negative impact on this common species.

### ***Rattus lutreolus* – Swamp Rat**

*Rattus lutreolus* is another rodent for which a dependence on dense vegetation has been demonstrated in many previous studies, and in particular very dense low vegetation appears to be an essential habitat requirement (e.g. Fox and McKay 1981; Fox 1984; Norton 1987; Bennett 1993; Haering and Fox 1995; Monamy 1995; Maitz and Dickman 2001). This species also appears to prefer areas with impeded drainage, and this is most likely related to its dietary preference for the lower stems of monocots, and probably accounts for the association that Braithwaite and colleagues (1978) found with vegetation type. At both of the sites where there were resident populations of *R. lutreolus* (both grazed/burnt) there was an extremely dense ground cover of *Imperata cylindrica*, *Lomandra longifolia*, and abundant sedges, which are characteristic of impeded drainage. A transient male of this species was caught on a single occasion at one ungrazed site.

### ***Melomys cervinipes* – Fawn-footed Melomys**

*Melomys cervinipes* most commonly occurs in rainforest and wet sclerophyll forest, and is semi-arboreal, particularly utilising vines and creepers and small trees (Wood 1971). It constructs a leaf nest in rainforest trees and feeds on leaves, shoots and fruit (Wood 1971; Redhead 1995). *Melomys cervinipes* lives in pairs, in a home range approximately 50 m in diameter, and seems to exclude others from this territory (Redhead 1995). This species was found only in ungrazed sites in this study, all of which were characterised by a complex mid-storey of tall shrubs and small trees. The ungrazed sites on the Carrai Plateau, which had a higher number of *M. cervinipes*, had a diverse mid-storey of rainforest trees, vines and creepers. Because of its requirement for a mid-storey, particularly of rainforest plants, this species is likely to be the most adversely affected by frequent burning.

### ***Pseudomys oralis* – Hastings River Mouse**

*Pseudomys oralis* is considered one of Australia's most endangered rodents (Jerry *et al.* 1998) and is listed as endangered on the NSW *Threatened Species Conservation Act 1995* and on the national *Environment Protection and Biodiversity Conservation Act 1999* (NSW NPWS 1999).

This species is known from about 50 localities in north-eastern NSW and south-eastern Qld, and most populations are known from only one or two individuals (Read and Tweedie 1996; Pyke and Read 2002). Consequently, its occurrence at the study sites is significant. The two animals caught at one site were breeding adults (one heavily pregnant and the other a scrotal male, both caught in spring), indicating that they were residents although caught only a few times. The other animal was a sub-scrotal male.

The floristic and structural vegetation research carried out as part of the broader project (see Tasker 2002 for details) taken in conjunction the published literature on *P. oralis* reveals some interesting patterns in the habitat associations of this species, particularly with respect to fire regimes.

The distribution of this species in NSW correlates almost entirely with areas of higher rainfall eastern tablelands forest that are grazed by cattle at low intensities and subjected to associated burning (NSW NPWS Atlas records). Almost all of the locations in National Parks and State Forests where it is known to occur, were recently, or still are, subject to grazing and associated burning, e.g. Barrington Tops (part), Chaelundi, Mount Royal, Nymboida, Oxley Wild Rivers and Werrikimbe National Parks (NSW NPWS 1999), Blicks River Flora Reserve and Hyland, Marengo, Chaelundi, Dalmorton, Billilimbra, Mount Royal, Carrai, Styx River and Gambubal (Qld) State Forests (Lee 1995), and vacant Crown land on the Carrai Plateau. Nevertheless, a number of authors have concluded that it is threatened by disturbance from livestock grazing and burning (Lee 1995; NSW NPWS 1999).

The diet of *P. oralis* consists predominantly of leaves in winter, and a mixture of leaves and seeds in summer. These are from herbs (e.g. *Glycine* spp., *Geranium* spp., *Viola hederacea*), grasses and sedges (e.g. *Carex* spp., *Poa* spp., *Oplismenus* spp., *Microlaena stipoides*, *Lomandra* spp.); the composition is similar throughout the range of the species, thus indicating quite specific dietary requirements (Fox *et al.* 1994; Read and Tweedie 1996; Smith *et al.* 1996). This mix of plant species is typical of the grazed/burnt sites, and this diverse ground flora is actually promoted by moderately frequent burning associated with some grazing leases. The ungrazed sites lack this well-developed and floristically diverse layer of herbs and monocots.

Abundant food, however, is not the only essential habitat component, as *P. oralis* is known only from restricted areas of grazed and burnt forest. The other required habitat component, which was present in the grazed/burnt study sites where it was found but not at those where it was not found, was abundant cover that may be used as shelter. This requirement for cover has been noted previously (King 1985; NSW NPWS 1999). The only two of our grazed/burnt sites at which this species was found had by far the highest number of logs and mid-storey shrubs ("Rolf" site), and the densest cover of ferns ("Fitzroy" site) of any of the grazed/burnt sites.

Thus, although the moderately frequent burning associated with many cattle-grazed areas produces an ideal food supply, too-frequent burning or more intense grazing (as in other grazed forests), may remove the essential shelter component for this species. The fire ecology of *P. oralis* is a topic that warrants further study and manipulative experimentation.

### *Pseudomys novaehollandiae* – New Holland Mouse

*Pseudomys novaehollandiae* is found in low heath, woodland and dry sclerophyll forests with a dense ground cover or low shrub layer, in Queensland, New South Wales, Victoria and Tasmania (Lock and Wilson 1999) and is listed as endangered and threatened in the latter two States, respectively. The successional age of the vegetation appears to be an essential characteristic of suitable habitat for this species (Posamentier and Recher 1974; Fox 1982),

and the species peaks in abundance in vegetation aged 2-9 years post disturbance (Fox and McKay 1981; Lock and Wilson 1999). The distribution of *P. novaehollandiae* is strongly correlated with the density of vegetation below 50 cm (Posamentier and Recher 1974; Kemper 1977; Fox and Fox 1978; Fox and Fox 1984; Lock and Wilson 1999) and in some cases with plant diversity (Braithwaite and Gullan 1978; Cockburn 1978; Fox and Fox 1978), both of which are at their maximum shortly after fire. In NSW the species is most common in coastal heath and heathy woodland, but on the Northern Tablelands its distribution correlates remarkably closely with areas of frequently-burnt forest, which in general are those in grazing leases. Again, this species is not common in grazed/burnt forest, and this is probably because of its strong preference for early to mid-seral stage vegetation, the location of which is constantly shifting over time, and because it also requires soft substrates to allow construction of burrows (Kemper 1977; Fox and Fox 1978), which limits the areas of suitable habitat. There was a small resident population at one site (one reproductively active female caught repeatedly over a year, and two males), and this was the most disturbed site and that with the highest floristic diversity of any of the study sites.

### Other species

Two other species of small mammals were caught in extremely low numbers, although for different reasons. The common dunnart *Sminthopsis murina* rarely enters Elliott traps (Fox 1995), and thus is probably more common in the study area than the trapping indicates. It is likely to be advantaged by the open grassy understorey at grazed and burnt sites (Catling 1991). Only one introduced small mammal was caught during the study, the house mouse *Mus musculus* and this is likely to be testimony to the low to moderate levels of disturbance in the study area. Only two individuals were caught in over 6,700 trap nights, and both were within 1 km of a field that was aerially sown with pasture seed several months previously. Both species were caught in such low numbers they are not discussed further.

### Factors affecting small mammal community composition

Although the patterns of small mammal response can be explained in terms of what is known of the ecology of the component species, it is still possible that other factors caused the observed patterns. For example, species that are less abundant in the grazed/burnt sites may be less abundant because they suffer greater mortality from fires, are more directly affected by the presence of the cattle, or because there is less food at the grazed/burnt sites.

Both *A. stuartii* and *R. fuscipes* survive even high-intensity fire (Newsome *et al.* 1975; Recher *et al.* 1975; Catling *et al.* 1982). *A. stuartii* nest in tree hollows, and *R. fuscipes* in burrows under the soil or inside large logs, all of which are well insulated against fire. Neither species is likely to be severely affected by the fire itself. Post-fire mortality may be high, however, probably due to a lack of shelter leading to increased predation, or insufficient food, though the mechanisms that lead to post-fire decline are not well-

known (Sutherland and Dickman 1999). Changes to the vegetation caused by an intense wildfire initiate a more chronic impact, particularly compared to low intensity, patchy fires, such as typically result from grazer burning.

There may be a difference in food availability for *A. stuartii* and *R. fuscipes* between the grazed/burnt and ungrazed sites, in particular of leaf litter invertebrates, which are a major dietary component of both species (Watts and Braithwaite 1978; Statham 1982; Cheal 1987). As part of the broader project (Tasker 2002), leaf litter and tree trunk invertebrates were sampled at all sites. There was no overall difference in the abundance of tree trunk invertebrates between grazed/burnt and ungrazed sites, though the composition did vary significantly (see also Bickel and Tasker 2004). The data on leaf litter invertebrates are still pending sorting and analysis.

Another factor which could give the appearance of differences in the abundance of small mammals between the grazed/burnt and ungrazed sites is a differential response to traps depending on the habitat. In other words, if *A. stuartii*, *R. fuscipes* and *M. cervinipes* entered traps more readily in the ungrazed sites this could result in higher numbers of animals trapped. However, trappability (i.e. likelihood of being trapped, Pollock *et al.* 1990) was calculated as part of a mark-recapture analysis (Tasker 2002) and did not differ significantly between the grazed/burnt and ungrazed sites.

The most likely explanation for the high abundance of *A. stuartii* and *R. fuscipes* in the ungrazed sites is that there are important benefits conferred by dense vegetation cover. Those aspects that are probably most relevant include protection from predators, both mammalian (Green and Osborne 1981; Newsome *et al.* 1983; Banks 1999) and avian (Loyn *et al.* 1986; Lazenby-Cohen and Cockburn 1993), amelioration of adverse climatic elements (Geiger 1965; Birney *et al.* 1976), and the reduction of agonistic encounters between conspecifics and heterospecifics. Small mammals may alter their foraging patterns to use denser cover in order to reduce perceived predation risk (e.g. Dickman 1992; Banks 1998), and Bennett (1993) and Sutherland and Predavec (1999) attributed the higher densities of animals in dense vegetation to a lower predation risk in those patches. Dense vegetation often reduces the efficiency of predators (Elton 1939; Lima and Dill 1990; Finke and Denno 2002), and as a consequence may also enable foraging during daylight by subordinate individuals or species (Birney *et al.* 1976). The effect of habitat complexity in reducing agonistic encounters has apparently not been studied in either *A. stuartii* or *R. fuscipes*, but it has been demonstrated in some rodents such as *Mus musculus*, *Microtus pennsylvanicus* and *Sigmodon* spp. (Crowcroft 1966; Terman 1974) and non-mammalian species (Petren and Case 1998; Finke and Denno 2002).

A number of authors have suggested that a structural vegetation classification is likely to be more successful in predicting the composition of many animal communities than one based on floristics (Elton and Miller 1954; Catling and Coops 1999; Catling *et al.* 2001). However, this study shows that although vegetation structure is important to

small mammals, and allows an accurate prediction of the occurrence and abundance of some species e.g. *A. stuartii* and *R. fuscipes*, other species can be predicted only by a combination of floristic composition and cover. Even so, it is apparent that habitats defined by the understorey will still be better predictors of small mammal abundance than the canopy, as has been concluded by others (e.g. Fox and McKay 1981; Hockings 1981).

## Conclusions and their applicability to the conservation of forest fauna

The community composition of small mammals differed between grazed/burnt and ungrazed forests. Moderately frequent burning and sporadic cattle grazing benefited several species of native small mammals, including the endangered *P. oralis*, through the provision of a floristically diverse ground flora of herbs and monocots. However, the loss of mid-storey shrubs that results from this burning regime disadvantaged other small mammal species, including the uncommon *M. cervinipes* and *A. swainsonii*, and the common *A. stuartii* and *R. fuscipes*. Other studies have also found that livestock grazing results in an increase in some species of animals and a decrease in others (e.g. Landsberg *et al.* 1996).

The stratified habitat sampling within a constrained set of environmental variables in this study demonstrated the relationship of small mammals with understorey more clearly than many previous correlative studies. In the more structurally complex ungrazed sites, the abundances of *R. fuscipes* and *A. stuartii* were consistently higher than at grazed/burnt sites, and at some ungrazed sites the densities were the highest yet found for these species. The abundances of *A. stuartii* and *R. fuscipes*, and to a lesser degree *M. cervinipes*, were correlated strongly with the “vegetation complexity score”, a simple cumulative measure of vegetation cover.

In this study the overall diversity and species richness of small mammals did not differ between the grazed/burnt sites and the ungrazed sites, yet in contrast, they are substantially lower in many grazing leases that are burned every 1-2 years. Surveys by Tasker (2002) have shown that these areas have a greatly reduced understorey vegetation cover and floristic diversity, and depauperate small mammal communities. High frequency fire is listed as a key threatening process under the *NSW Threatened Species Conservation Act 1995*, but in spite of this, fire frequency in many areas is largely determined by the actions of individual graziers. Given the significance and scale of impacts of frequent, low-intensity fires on forest fauna and flora, this is unsatisfactory: these disturbances need to be managed in an active and informed manner.

For example, in grazing leases that are currently burned every year or two, reducing the frequency of burning, as well as making the length of the intervals between fires more *variable* (see Keith *et al.* 2002 for discussion of these concepts), will undoubtedly improve habitat quality for most species of small mammals and allow more diverse plant communities to persist. Guidance on what may be appropriate fire intervals for particular plant communities

can be drawn from recent research in plant fire ecology, in particular the use of plant “functional response groups” (Gill *et al.* 2002).

In order to manage faunal communities by manipulating disturbance regimes, it is necessary to decide for which species, or suites of species, we wish to manage the habitat in any given area. This study has shown that maximising habitat suitability for some species will reduce its suitability for others. For example, management of former grazing leases incorporated into National Park for the conservation of *P. oralis* in our view will require

sufficiently frequent fire disturbance to maintain an open and floristically diverse ground cover while still maintaining adequate shelter cover for the species.

Finally, it is clear that well-designed experimental research – particularly manipulative studies – on the impacts of different fire and grazing regimes is desperately needed. It will be essential in allowing us to disentangle the key elements that are driving these complex processes. In turn, this understanding will enable us to manage and manipulate these disturbances to more effectively conserve our forest fauna.

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