ACT Government

DISTRIBUTION AND ECOLOGY OF THE BROAD-TOOTHED RAT IN THE ACT

RICHARD MILNER, DANSWELL STARRS,
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Richard Milner*, Danswell Starrs^, Greg Hayes* and Murray Evans*

*Conservation Research, Environment and Planning Directorate,
ACT Government, Canberra, ACT 2601

^Evolution, Ecology and Genetics, Research School of Biology,
Australian National University, Canberra, ACT 0200

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Front cover: Broad-toothed Rat. G A Hoye © Australian Museum.

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Summary

The Broad-toothed Rat *Mastacomys fuscus* is a small to medium sized rodent of the family Muridae. Within the ACT the species is thought to occur in subalpine swamps and the associated riparian heathland, sedgeland and grassland environments above 1400 m. *M. fuscus* is recognised nationally as a declining species endemic to alpine and sub-alpine regions of south-eastern Australia, including Tasmania, Victoria, NSW and the ACT. The species is identified as being at extreme risk to global warming due to shifts in composition and distribution of alpine vegetation communities. Climate change has been identified as a key threatening process to the long-term viability of the species. In NSW *M. fuscus* is classified as vulnerable, with one of the two populations, Barrington Tops, classified as endangered. In Victoria the species is classified as near threatened and in the ACT, where only limited information is available on the species, *M. fuscus* has no special protection status.

The purpose of this study was to determine the presence or absence of the *M. Fuscus* in known habitat for the species (sphagnum moss bogs and fens) in the ACT, and to identify key ecological attributes that determine the presence of the species. A key aim of this study was to establish a long term monitoring program to assess long-term trends in the species' abundance and distribution, which may be affected under projected climate change scenarios.

Fourteen ACT alpine bogs were surveyed across Namadgi National Park, which covers the southern half of the ACT and adjoins Kosciusko National Park. Scat searches were undertaken in 27-132 quadrats per bog to determine the species presence/absence, relative abundance and habitat preference. For each quadrat, information on *M. fuscus* presence/absence, vegetation type, feral animal disturbance and distance to drainage lines was recorded. Bog size and distance to the nearest occupied bog was also recorded.

Results indicate *M. fuscus* presence across 13 of the 14 sites: Cotter Gap, Cheyenne, Rocks Flat, Snowy Flat, Ginini West, Big Creamy, Ginini East, Tom Gregory, Little Bimberi, Rotten Swamp, Murray Flat, Leura and Top Flat. The species showed a positive habitat preference for vegetation communities dominated by heath, sedge and Poa, and was also identified in a further nine dominant vegetation types, including *Eucalyptus pauciflora* (snowgum) woodland. Vegetation composition varied with bog size and *M. fuscus* detection rate increased with increasing bog size. Distance from drainage lines was also important, with detection rates increasing with proximity to drainage lines. Finally the presence of feral animals such as foxes, rabbits and pigs significantly influenced detection rates, with *M. fuscus* detection rate decreasing as signs of feral animal disturbance increased.

This study provides land managers with a baseline assessment of *M. fuscus* condition in the ACT against which future changes can be assessed and key-threatening processes can be monitored.

The results of this study have also been published in:
Milner R, Starrs D, Hayes G and Evans M. 2014. Distribution and habitat preference of the broad-toothed rat (*Mastacomys fuscus*) in the Australian Capital Territory, Australia. Australian Mammalogy. http://dx.doi.org/10.1071/AM14031

1 Introduction

The Broad-toothed Rat, *Mastacomys fuscus*, is a cryptic native rodent that inhabits montane, subalpine and alpine areas, usually in close association with water bodies. In the ACT *M. fuscus* is known to occur in sphagnum moss bogs, although surveys for the species have not been conducted for over two decades and only limited information exists on its distribution and abundance. The species is known to have declined in other jurisdictions, and is listed as threatened in NSW and Victoria. Climate change has been identified as a key threatening process to the long-term viability of the *M. fuscus* due to shifts in composition and distribution of alpine vegetation communities.

The broad goal of this study was to gain a better understanding of the distribution, abundance, ecology and conservation status of *M. fuscus* in the ACT to support appropriate management decisions for the species and its habitat.

Specific aims of study were to:

- Determine the distribution, abundance and status of the Broad-toothed Rat *Mastacomys fuscus* in the ACT.
- Identify key ecological attributes that determine or influence the likelihood of the species occurrence.
- Establish a baseline and method for long term monitoring to assess the effects of climate change on the abundance and distribution of the species.

2 Background

2.1 Description and Distribution

The Broad-toothed Rat, *Mastacomys fuscus*, is a small to medium sized rodent of the family Muridae (Happold 2002). Adult head-body length ranges from 142–175 mm, tail length ranges from 100–130 mm and adult weight ranges from 95–145 g. The species is found across a range of altitudes from sea level to 2200 m where habitat is suitable in the Dandenong and Otway Ranges, Victorian Alps, Snowy Mountains, Barrington Tops and Tasmania;. Across Snowy Mountains and Barrington Tops populations, the species habitat is characterised by mean annual temperatures <10°C and mean annual rainfall >1000 mm (Green and Osborne 2003). While the species is currently restricted to patches of apparently optimal habitat, fossil records indicate a more extensive distribution during the Pleistocene when the climate was cooler (Ride 1956).

Within the ACT there is limited information on the distribution, abundance and ecology of *M. fuscus*. Only three surveys in the ACT have successfully detected the species, all of these were undertaken prior to 1990. In 1973 Eberhard and Schulz (1973) captured a single *M. fuscus* at Murrays Gap when undertaking a vertebrate survey of the Cotter River catchment. In 1986, Lintermans (1986) undertook a survey of *M. fuscus* across six ACT sites (Murrays Gap, Cheyenne Flats, Stockyard Saddle, saddle below Mt Gingera, Mt Gingera and Snowy Flats) from which only one individual was captured (at Murray Gap). From 1988 to 1989, Green and Osborne (2003) surveyed 25 areas within or bordering the ACT with *M. fuscus* recorded within 12 of the sites: Hanging Flat, Murrays Gap, Mt Murray, Ginini Flats, Cheyenne Flats, Stockyard Flats, Brumby Flats, 80 Acres, Blackfellows Gaps, Rolling Ground Gap, Mt Bimberi and Mt Scabby. Runways and scats of *M. fuscus* have been observed at Ginini Flats, Cheyenne Flats, Morass Flats and Snowy Flats (M Evans pers. obs.; D Whitfield pers. obs.).

2.2 Status

M. fuscus is recognised nationally as a declining species endemic to alpine and sub-alpine regions of south-eastern Australia, including Tasmania, Victoria, NSW and the ACT. In Victoria *M. fuscus* is listed as Near Threatened, and in NSW the species is listed as Threatened (Vulnerable) though one of the populations (Barrington Tops) is classified as Endangered. *M. fuscus* is not listed as threatened in the ACT, where there is a paucity of information on the species.

M. fuscus is identified as being at extreme risk to global warming due to loss of habitat (Sphagnum moss bogs and other wet seepage areas) resulting from shifts in the composition and distribution of alpine vegetation communities (DECC NSW 2007), and accordingly climate change has been identified as a key threatening process to the long-term viability of *M. fuscus*. Brereton and colleagues (1995) predicted that with a 1°C rise in temperature the range of *M. fuscus* would decrease by 36% and with a 3 ^{o}C rise in temperature it would decrease by 75%. In 2005, O'Brien showed that extinction of *M. fuscus* at Barrington Tops is expected under almost any climatic warming scenario.

2.3 Habitat

M. fuscus is associated with a variety of habitat types, including rocky outcrops, tall tussock grasslands, reeds, sedges and low spreading heath shrubs, usually within 15 m of watercourses (K Green pers. comm.). The species requires dense ground cover and an abundance of green grass, including snow grasses (Poa spp). In the ACT, *M. fuscus* is apparently associated with subalpine swamps and associated riparian heathland, sedgeland and grassland environments above 1000 m, with a mean annual rainfall greater than 1000 mm and average temperature <10 ^{o}C (Green and Osborne 2003). The species' strong association with refuge habitat close to water limits its distribution to isolated pockets of subalpine swamps and riparian zones. Historical records show that *M. fuscus* may have inhabited Snow Gum (*Eucalyptus pauciflora*) woodlands adjacent to subalpine swamps and grasslands, though recent evidence of their use of such habitat is lacking (Green 2000).

2.4 Diet

M. fuscus feeds primarily on monocotyledons, including snow grass (DECC NSW 2007). Other food eaten includes small quantities of dicotyledons, bark, seeds and fungi. Two heath species commonly found in the diet of *M. fuscus* includes *Phebalium ovatifolium* and *Prostanthera cuneata* (Carron et al. 1990). Bark and dicotyledons appear to be increasingly consumed during winter when grass quality is likely to be lower (Carron et al. 1990). This dietary pattern is consistent with many boreal and subarctic rodents that tend to feed on growing plant material (rich in protein) during summer, and vegetative storage organs, roots, rhizomes and bark (rich in carbohydrates) during winter. There is little evidence to suggest that food is currently likely to be a limiting resource for *M. fuscus*. What effect climate change may have on food availability due to shifts in vegetation community composition is unclear.

2.5 Population dynamics

M. fuscus lives three to four years in the wild (Happold 1989a). Over-winter survival is high in Kosciuszko National Park with between 69–81% of adults surviving the over-winter period. The species falls within the Critical Weight Range (Burbidge and McKenzie 1989) and are therefore at high risk to decline and extinction through predation by introduced predators such as foxes and cats.

The species has a patchy distribution across its range. While large habitat patches are likely to support larger populations, there is no evidence that patch size influences population persistence. There is some evidence, however, that extinction events in *M. fuscus* are more likely in isolated habitat patches (O'Brien 2008). *M. fuscus* are good dispersers and have been shown to recolonise high quality patches following local extinctions, though long distance movement is uncommon, especially if riparian cover or creek lines are absent (Happold 1995; O'Brien 2008). Although evidence of *M. fuscus* has been found in drainage lines linking habitat patches or bogs, *M. fuscus* habitat within these drainage lines is considered to be poor (e.g. forested environments with dense shrub), suggesting that such drainage lines are used sporadically for dispersal rather than as permanent habitat (Keating 2003).

M. fuscus are solitary during summer and communal during winter where they nest with up to three other individuals (Bubela et al. 1993). Activity patterns and home range size varies between seasons. Activity increases during summer and in Kosciuszko National Park male home range falls from 2703 m^2 in summer to 996 m^2 in winter, and in females it falls from 1614 m^2 in summer to 1044 m^2 in

winter (Bubela et al. 1991). Female home range size is determined by metabolic requirements such as food quality and food density (Mace et al. 1983), while male home range size is also subject to sexual selection for increased access to females, with home range decreasing as female reproductive receptiveness decreases (Bubela et al. 1991). While *M. fuscus* do not appear to display territorial defence, there is evidence that both sexes occupy an exclusive/solitary space that averages 802m^2 within their home range during January to autumn (Bubela et al. 1991).

2.6 Breeding biology

M. fuscus display female-only parental care and are believed to be either polygamous or promiscuous (Happold 2011). Mating occurs in mid-late October and is trigged by a gradual increase in ambient temperature, snowmelt, decreased frequency of frost, increasing day length and the growth of new grass (Bubela and Happold 1993). Breeding nests are built under dense shrubs, mattered stems, boulders and fallen logs and are similar to overwinter nests – they have a single entrance and a single internal cavity up to 30 cm in diameter (Happold 2011). Gestation period is 35 days (Calaby and Wimbush 1964) and all females synchronously give birth to their first litter in the last week of November or first week of December (Happold 1989). Births of the second and sometimes third litters are spread out over several weeks from December to early March and litter size ranges from 1–4 (Bubela et al. 1991; 1993). Young *M. fuscus* start to eat grass at three weeks and are weaned at five weeks (Happold 2011) and sexual maturity is reached at 8–10 months.

2.7 Threatening processes

Key threatening process recognised by the NSW Threatened Species Conservation ACT 1995 that are relevant to *M. fuscus* include:

- predation by European Red Fox *Vulpes vulpes* and the Feral Cat *Felis catus*;
- competition and grazing by feral European Rabbits *Oryctalagus cuniculus*;
- competition, disease transmission and habitat degradation by Feral Pigs (*Sus scrofa*);
- invasion of exotic perennial grasses;
- climate change resulting in loss of sub-alpine and alpine habitat, and;
- spread of the plant root fungus *Phytophthora cinnamomi*.

3 Methods

3.1 Study sites

Potential habitat for *M. fuscus* was identified remotely using aerial photographs, vegetation maps (i.e. sphagnum shrub bog, sod tussock Poa grassland, Poa-Danthonia grassland, and Restiad bog ArcGIS layers – CPR GIS system), ACT Wildlife Atlas records, and expert knowledge (D Whitfield pers. comm.). Fourteen study sites were identified, including: Cotter Source, Top Flat, Rotten Swamp, Big Creamy, Rocks Flat, Cotter Gap, Tom Gregory, Murray Gap, Little Bimberi, Leura, Snowy Flat, Cheyenne Flat, Ginini West and Ginini East.

3.2 Survey method

Surveys were undertaken in early autumn (March) 2013. Area-based searches for scats (faecal pellets) were undertaken in 27–130 circular quadrats (2 m radius) per study site. Quadrats were positioned 20 m apart along 5–12 straight-line transects. Line transects were positioned at right angles to the streams and elevational gradient (i.e. transects cut across contours; Map 1) and were placed 50 m apart for study sites/bogs > 35,000 m^2 and 25 m apart for study sites/bogs <35,000 m^2. *M. fuscus* tends to deposit scats (singly but usually in clusters) along tracks (or the more obvious 'runways') that it makes though vegetation. Scat searches were undertaken qualitatively by inspecting sections of *M. fuscus* tracks and runways (Appendix Figure A) and also quantitatively by searching within quadrats (Appendix Figure B). A maximum of three minutes was allowed for searching in each quadrat. Runways and tracks are easily identified; they are constructed beneath vegetation clumps and are highly visible in vegetation openings. *M. fuscus* scats are easily distinguished from other sympatric species as they are green when fresh, fibrous with small fragments of undigested grass and abundant when present. A single *M. fuscus* produces between 200–400 scats per day and the scats can last up to five years (Happold 1989b). Scat searches, including time-limited scat searches, are a reliable and commonly used technique for determining the presence of *M. fuscus* at a site (e.g. Happold 1989; Green and Osborne 2003). Green and Osborne (2003) showed that where scats were detected – they were detected within the first minute of searching in 79% of all sites. The technique, however, requires a sound understanding of *M. fuscus* ecology and habitat preference. By limiting both area and time surveyed (i.e. this study) – an index of relative abundance can be identified and the likelihood of an observer bias can be reduced.

Information recorded for each quadrat consisted of: *M. fuscus* presence/absence, feral animal disturbance (i.e. presence/absence of fox scats, rabbit scats and pig scats and diggings), dominant and co-occurring vegetation/habitat type (presence/absence in the categories: Sphagnum, Poa, Empodisma, Sedge, Juncus, Woodland, Low heath (HeathL: <50 cm high), Medium heath (HeathM: 50-150 cm high), Tall heath (HeathT: >150 cm high; Appendix 1, Figure C-G), Richea, forbs, exotic grass and rock) and observer identification (i.e. to measure observer bias). Quadrat coordinates were recorded and proximity to drainage lines was measured using ArcGIS map software post hoc. Spatial characteristics of each bog/survey site was measured post hoc using ArcGIS. Characteristics measured included bog size and distance to nearest bog occupied by *M. fuscus*.

3.3 Statistical Analyses

The proportion of quadrats with *M. fuscus* scats was plotted against bog size (km^2). Linear regression was used to determine if a significant relationship existed between *M. fuscus* abundance and bog size. Principle component analysis on the covariance matrix was used to explore variation in vegetation types between bogs. Proportional cover of vegetation across 11 bogs was explored (three bogs were excluded due differences in the method of reporting of vegetation types, which did not enable comparison of vegetation cover data). Principle components 1 and 2 were plotted, and distances across Euclidean space interpreted from raw data. Bog size and principle components were plotted to examine if principle components corresponded to bog size. Average proportional cover (± SE) of each vegetation type was plotted to examine the relative importance of each vegetation type.

A habitat selectivity analysis (Manly et al. 2002) was used to examine *M. fuscus* preference for specific habitat types. We assumed that vegetation preferences would not vary between bogs, so data across all bogs was combined to improve sample sizes for each vegetation category. A Type I (Manly et al. 2002) selectivity analysis was conducted in the R package AdehabitatHS (Calenge 2013). The proportion of available habitat was calculated as the proportion of quadrats containing each vegetation type, and *M. fuscus* usage was estimated as the count of quadrats that contained *M. fuscus* scats corresponding to each vegetation type. Habitat selection indices (Wi) were calculated for each vegetation type. Overall significance of habitat selection was determined by the log-likelihood statistic (Manly et al. 2002).

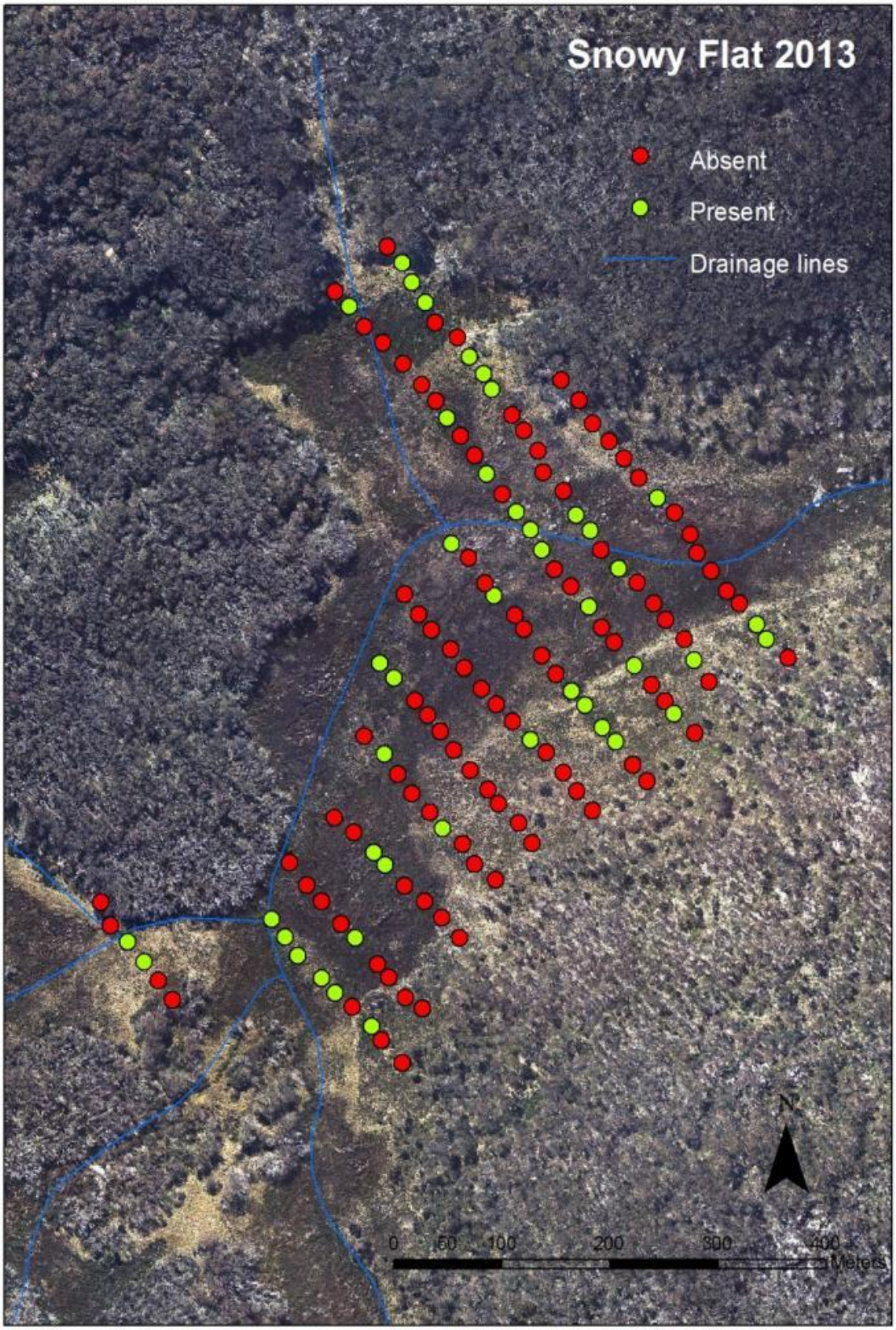
A total of three additional observers assisted the lead author in identifying *M. fuscus* scats at five bogs (Cheyenne, Ginini East, Leura, Snowy Flat and Tom Gregory). Only one additional observer was present at any one time. Observer bias was explored by examining the relative proportion of *M. fuscus* scats discovered by an observer, after controlling for bog-level differences in *M. fuscus* abundance. If observer bias was present, it would be expected to manifest itself through differences in *M. fuscus* scat detection between observers. *M. fuscus* scat abundance was modelled against observer, with bogs included as a random factor to account for bog-level differences in *M. fuscus* abundance.

The relationship between proportion of quadrats with *M. fuscus* scats and distance (transformed by the square-root to meet assumptions of statistical normality) from nearest drainage line was explored in a GLMM framework. Bog ID was included as a random factor, to account for possible differences between bogs. In a similar manner, the relationship between *M. fuscus* abundance and incidence of feral animal disturbance was explored in a mixed statistical model. All feral animal disturbance indicators (fox, rabbit and pig) were combined into a single category. While it is plausible that the relationship between *M. fuscus* and these species differs due to ecological relationships (competition, predation), we chose to combine these categories due to limited data for each feral species. Mixed effects models were implemented in R with the package lme4 (Bates et al. 2011). Plots were produced in R, with the fixed factors plotted for each level of the random factor (bog ID).

A final model, fitting *M. fuscus* abundance against the three positively selected vegetation types, feral animal disturbance and square-root of linear distance to drainage (m). Bog ID was included as a random factor. The presence of spatial autocorrelation in the model residuals was explored by way of Moran's I, in the R package 'ape' (version 3.2, Paradis et al. 2003). No significant spatial autocorrelation was detected (P = 0.84). Model fitting was performed in R (version 2.13; R Development Core Team, 2011). Model selection was conducted in 'glmulti' (Calcagno and de

Mazancourt 2010), using an exhaustive screening of all possible models with 2-way interactions. The best model was identified using AIC. Figures were produced in Sigmaplot (version 10; Systat Software Inc., London).

Map 1. Survey design and site establishment for *Mastacomys fuscus* in the ACT (Snowy Flat).



4 Results

4.1 Bogs

Fourteen alpine bogs in Namadgi National Park were surveyed for *M. fuscus* scats and vegetation characteristics in March, 2013. A total of 13 vegetation/habitat types were recorded from quadrats across all bogs. A Principle Components Analysis revealed that the proportion of vegetation type could provide separation between bogs in Euclidean space, with PC1 and PC2 explaining 46.39% and 20.05% of the total variation in vegetation between bogs respectively (Figure 1). Proportion of vegetation in the categories of Exotic plants, Empodisma, HeathL, Sphagnum, Richea, and HeathT were particularly informative in describing differences between bogs, whereas the proportion for categories of Poa, rocks, Forbs and woodland were not (Figure 1a). Tom Gregory and Cotter Gap bogs differed from the other bogs due to their relatively high presence of exotic species (33% and 16.7%, respectively), while Little Bimberi, Cheyenne Flat and Ginini differed from the other bogs in their relatively higher proportions of *Sphagnum* and *Richea* (Figure 1b). *Poa* was the most abundant vegetation type in 11 of the 14 bogs, being present in 75% ± 3.63 SE of all quadrats (Figure 2). HeathL, HeathM, Empodisma and Sedge were present in more than 20% of quadrats on average, across all bogs (Figure 2). Conversely, the categories of Forbs, Richea and Juncus, and rocks were relatively rare in bogs, each being present at less than 5% of all quadrats. The abundance of exotic plant species was relatively low in bogs; this category was recorded at just three bogs (Tom Gregory, Cotter Gap and Murray Gap), with bog Tom Gregory having the highest proportion of quadrats containing exotic plants (33%).

4.2 Distribution of *M. fuscus*

Across all bogs, *M. fuscus* scats were found in 26.5% of quadrats examined. At the landscape-scale, *M. fuscus* density (proportion of quadrats within a bog that had *M. fuscus* scats) varied greatly between bogs (Figure 3a). *M. fuscus* density was greatest at Cotter Gap bog (69%), and none were present at Cotter Source bog (0%; Table 1; Figure 3a). The probability of detecting *M. fuscus* scats was not related to bog size, however when excluding an outlier (bog Cotter Gap), which supported an unusually high density of *M. fuscus*, there was a significant positive relationship between bog size and *M. fuscus* density (Figure 3b). PC1 was negatively correlated to bog size, suggesting that vegetation types vary by bog size (Figure 4). However, *M. fuscus* abundance was not related to PC1 or PC2.

4.3 Habitat preference of *M. fuscus*

Pooling data across all bogs, habitat selectivity indices (Wi) suggested that *M. fuscus* show preferences for specific vegetation types (log-likelihood statistic: $X^2 = 31.75$, df = 12, $P < 0.001$) (Figure 5). Specifically, after bonferroni correction for multiple comparisons, *M. fuscus* scat presence was found to be significantly positively associated with medium height heath (HeathM) (Figure 5; Appendix Figure C). *M. fuscus* also showed weak but non-significant preferences for Poa and Sedge (Appendix Figure D-E). Conversely, there was a weak, non-significant negative association with Empodisma, woodland and low heath (HeathL) (Appendix Figure F-G). Apparent preferences for rarely occurring vegetation types (such as Forbs, *Richea*, *Juncus*) and rock, were given a low confidence due to the low abundance of these vegetation types and hence small sample sizes. The

probability of *M. fuscus* scats being observed was negatively related to straight-line distance from drainage lines, with the likelihood of detecting *M. fuscus* scats declining by 1.1% per m distance from a drainage line (Figure 6; Table 2). This relationship was found to be reasonably consistent between bogs, after accounting for the high bog-level variance (59%) (Table 2; Figure 6).

4.4 Feral animal disturbance

The probability of detecting *M. fuscus* scats decreased significantly if evidence of feral animal disturbance was present (Table 3). Feral animal disturbance was observed in 11.2% of quadrats across all bogs, with Cotter Source having the highest incidence of feral animal disturbance (31.1%; Table 1). Of the observed feral animal disturbance in bogs, evidence of pigs was most common (80.2%), with fox and rabbit representing 18.1% and 1.7% respectively. With one exception, modelling revealed a consistent decrease in *M. fuscus* density at the bog level, suggesting that feral animals have a consistent, negative effect on *M. fuscus* abundance. The exception occurred at Cotter Gap, where *M. fuscus* density increased with feral animal disturbance (Figure 7).

A model explaining *M. fuscus* scat presence with respect to the three most preferred vegetation types, distance from drainage and presence of feral animal disturbance was found to provide the best fit, compared to models with a subset of these variables (Table 4). Likelihood of detecting *M. fuscus* scats declined with distance from drainage line, and presence of feral animal disturbance. Conversely, likelihood of detection increased significantly with the presence of Poa, Sedge and medium height heath (Table 4). All interaction terms were found to be non-significant when the main effect was present within the model. The random factor (bog ID) explained 34% of variation.

4.5 Observer bias

Observer bias was found to be non-significant. After accounting for bog-level variance, there were no significant differences in scat detection rates among observers (Table 5). This is evidenced by the minor differences between observer coefficients, relative to standard errors (Table 5).

Figure 1. Principle component analysis of 13 vegetation types recorded across 11 alpine bogs. a) Ordination plot of principle component 1 and principle component 2, explaining 46.39% and 20.05% of variation in vegetation types, respectively, and b) vector plot, showing the relationships between bogs based upon similarities and differences in vegetation types.

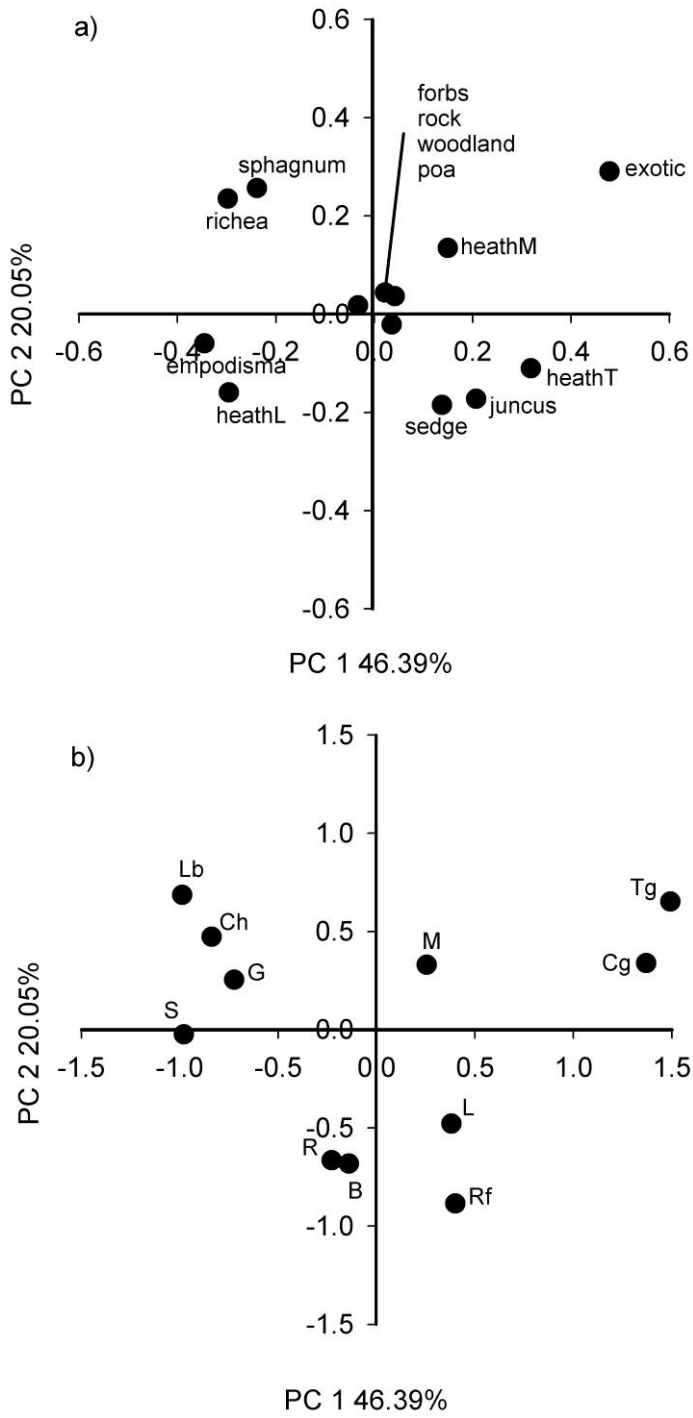


Figure 2. Average percentage of each vegetation type present at each site along transects for 11 alpine bogs. *Poa* was the most common vegetation type, followed by *Empodisma* and medium heath.

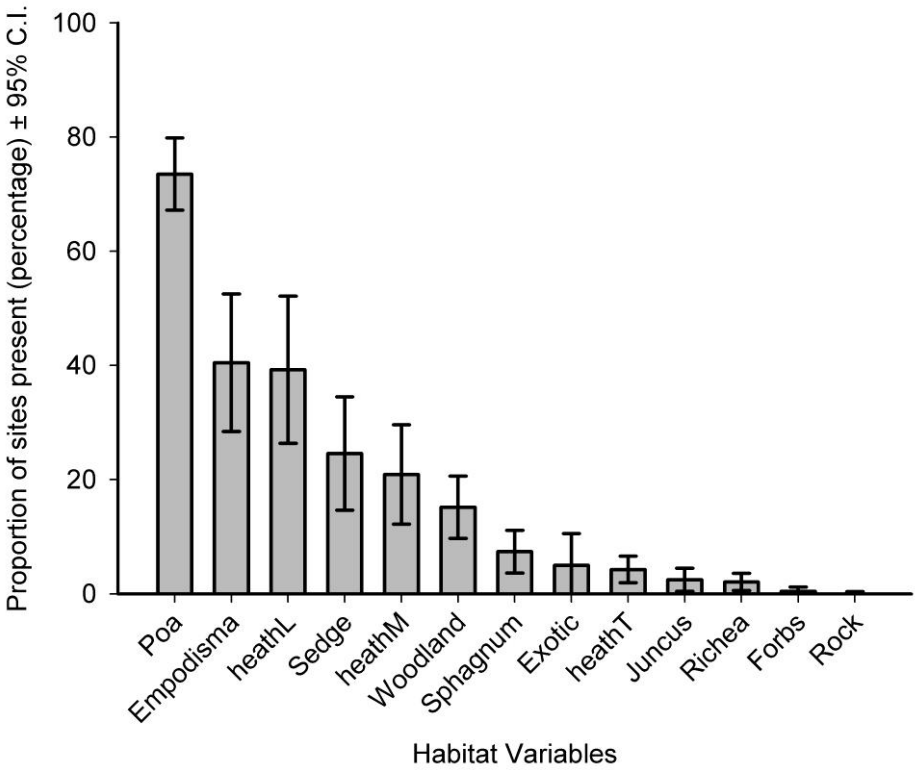


Table 1. Summary of survey effort, *M.fuscus* detection rate and feral animal disturbance per bog.

Bog	Transects	Plots	Plots occupied	Detection rate %	Feral animal disturbance %
Cotter Gap	10	42	29	69	17
Cheyenne	11	77	33	43	4
Rock Flat	8	70	24	34	13
Snowy Flat	11	133	44	33	6
Ginni West	12	100	30	30	13
Big Creamy	10	84	21	25	10
Ginni East	8	131	31	24	18
Tom Gregory	11	78	18	23	5
Rotten Swamp	6	72	13	18	8
Little Bimberi	11	61	11	18	11
Murray Gap	10	63	10	16	2
Leura	5	28	4	14	7
Top Flat	10	53	7	13	19
Cotter Source	10	45	0	0	31

Figure 3. a) Proportion of spots that contained *M. fuscus* scats, by bog ID, and
b) relationship between index of *M. fuscus* abundance (from panel a) and bog size (km^2).
Linear model is fitted to 13 bogs with an outlier (Cg; Cotter Gap) excluded (open circle).

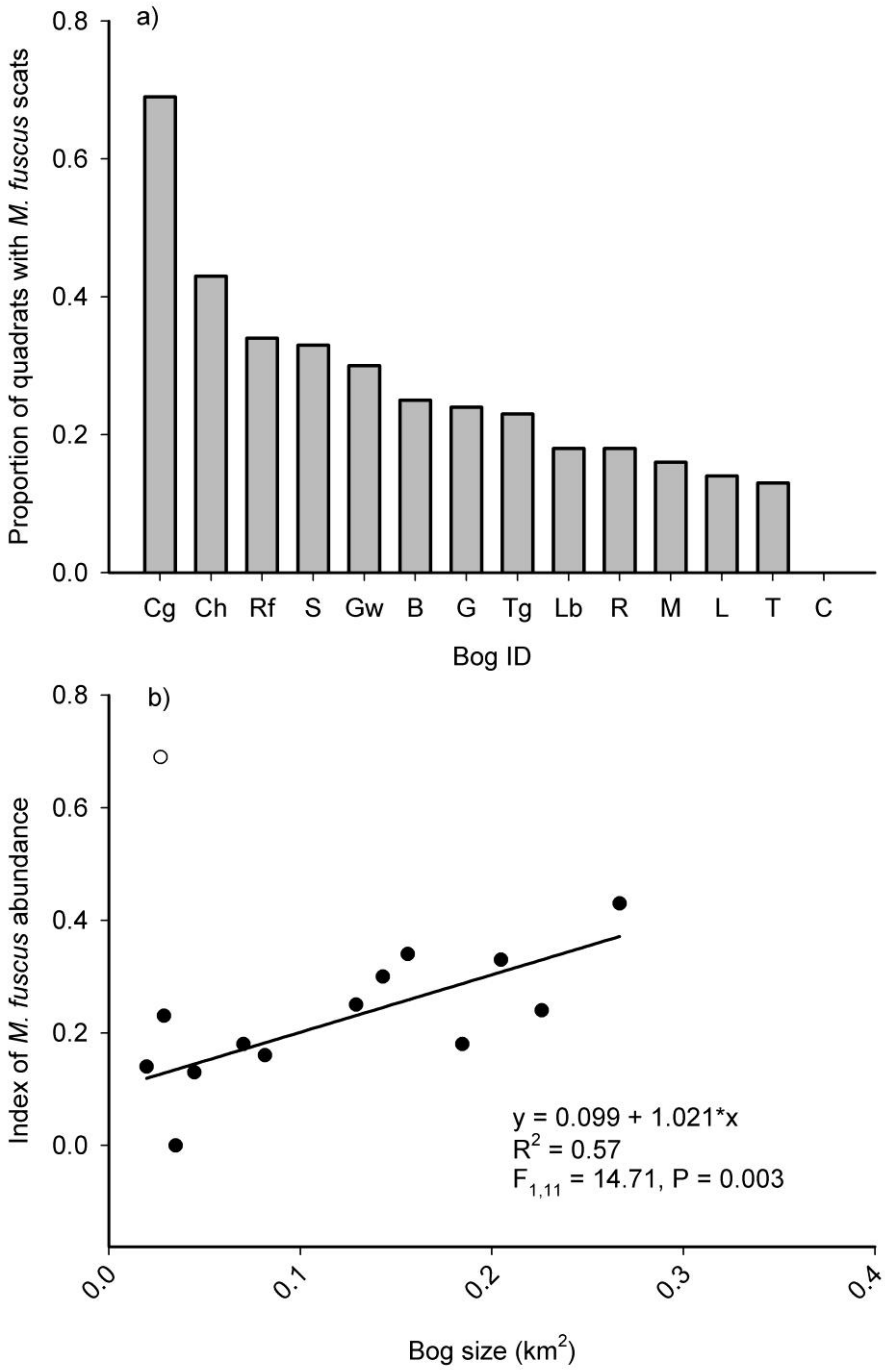


Figure 4. Bog size (km^2) plotted against principle component 1 (PC1) from the PCA analysis. An outlier (bottom left; bog Lb; Little Bimberi) was excluded from the linear model. When included, model fit is still significant ($F_{1,9} = 9.37$, $P = 0.01$), although quality of fit is reduced ($R^2 = 0.51$).

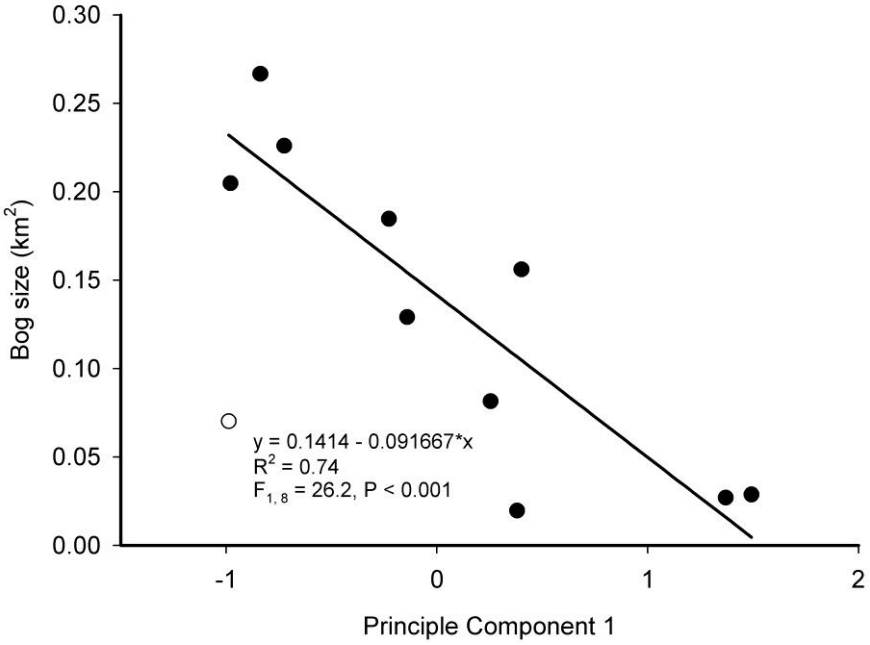


Figure 5. Habitat selection index (W*i*) for *M. fuscus*, averaged across 11 alpine bogs. Vegetation types greater than the dashed line show positive selection by *M. fuscus*, while those below the dashed line are negatively selected by *M. fuscus*.

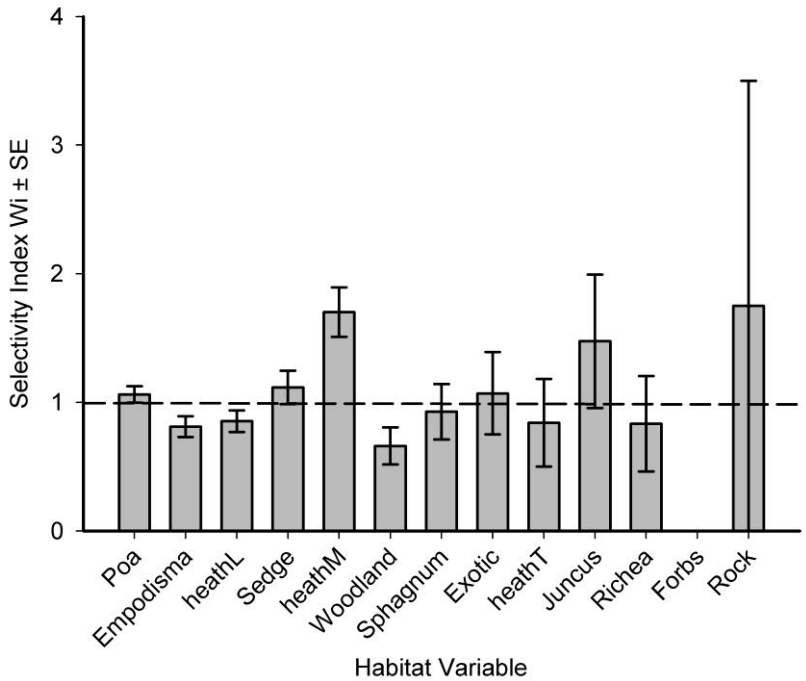


Table 2. Summary of general linear mixed effects model on likelihood of detecting *M. fuscus* scats, and square-root linear distance from drainage, across 14 alpine bogs in Namadgi NP. Bog ID was included as a random factor (random intercept and slope) to account for bog-level variance.

Fixed Factors	Coefficient	SE	Z	P
Intercept	-0.66	0.245	-2.698	0.007
$DistDrainL^{1/2}$	-0.088	0.026	-3.38	<0.001
Random Factors		**Variance**		
Bog1 (intercept)		0.59		
$DistDrainL^{1/2}$ (slope)		<0.001		
Residual		1		

Figure 6. Observed and fitted relationship between *M. fuscus* presence-absence and linear distance from the nearest drainage line (metres) across 14 alpine bogs. Mixed effects logistic regression model is fitted in pink, while observed data is plotted in blue. Figure shows a consistent decrease in *M. fuscus* presence with increasing distance from nearest drainage line, however relationship varies between bogs.

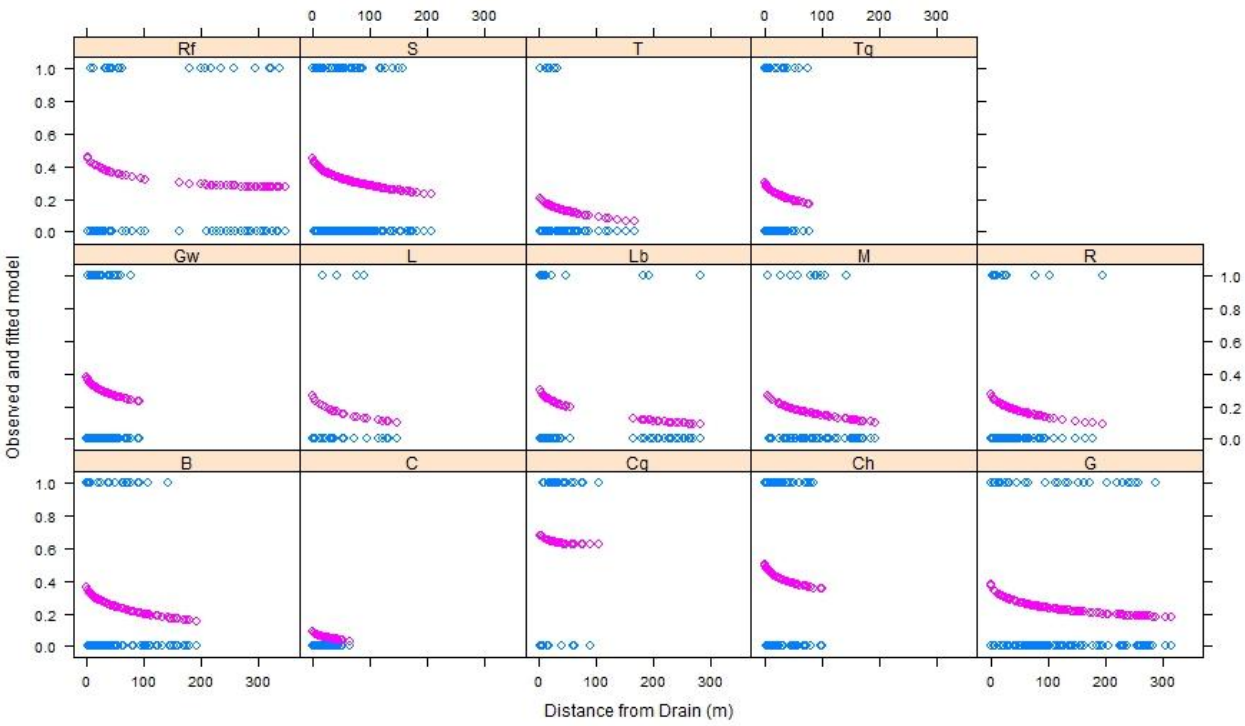


Table 3. Summary of general linear mixed effects model of likelihood of detecting *M. fusucs* scats and incidence of feral animal disturbance, across 14 alpine bogs in Namadgi NP. Bog ID was included as a random factor (random intercept and slope) to account for bog-level variance.

Fixed Factors	Coefficient	SE	Z	P
Intercept	-1.07	0.22	-4.9	<0.001
FeralAnimDis	-1.68	0.44	-3.8	<0.001
Random Factors		**Variance**		
Bog1 (intercept)		0.56		
FeralAnimDis (slope)		0.8		
Residual		1		

Figure 7. Fitted relationship between *M. fuscus* presence and incidence of feral animal disturbance (fox faeces, rabbit faeces and pig diggings) across 14 alpine bogs. A mixed effect logistic regression model is fitted, demonstrating that *M. fuscus* presence declines with feral animal disturbance, at all except 1 bog (Cg; Cotter Gap).

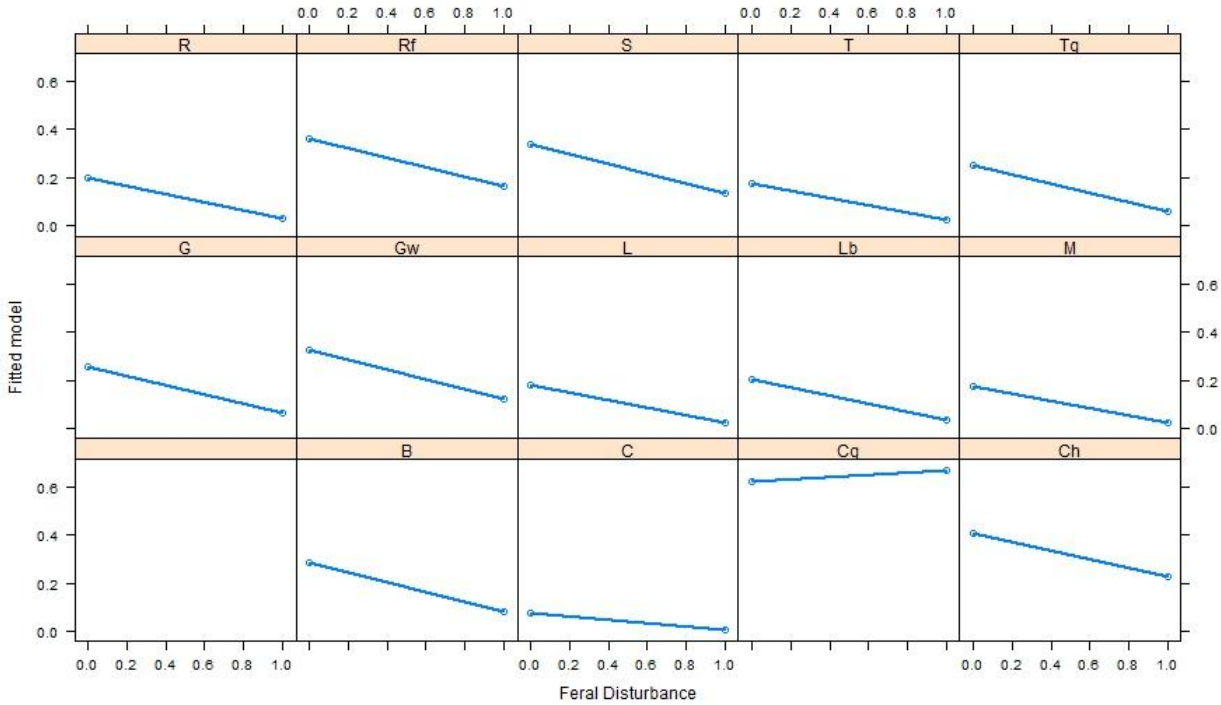


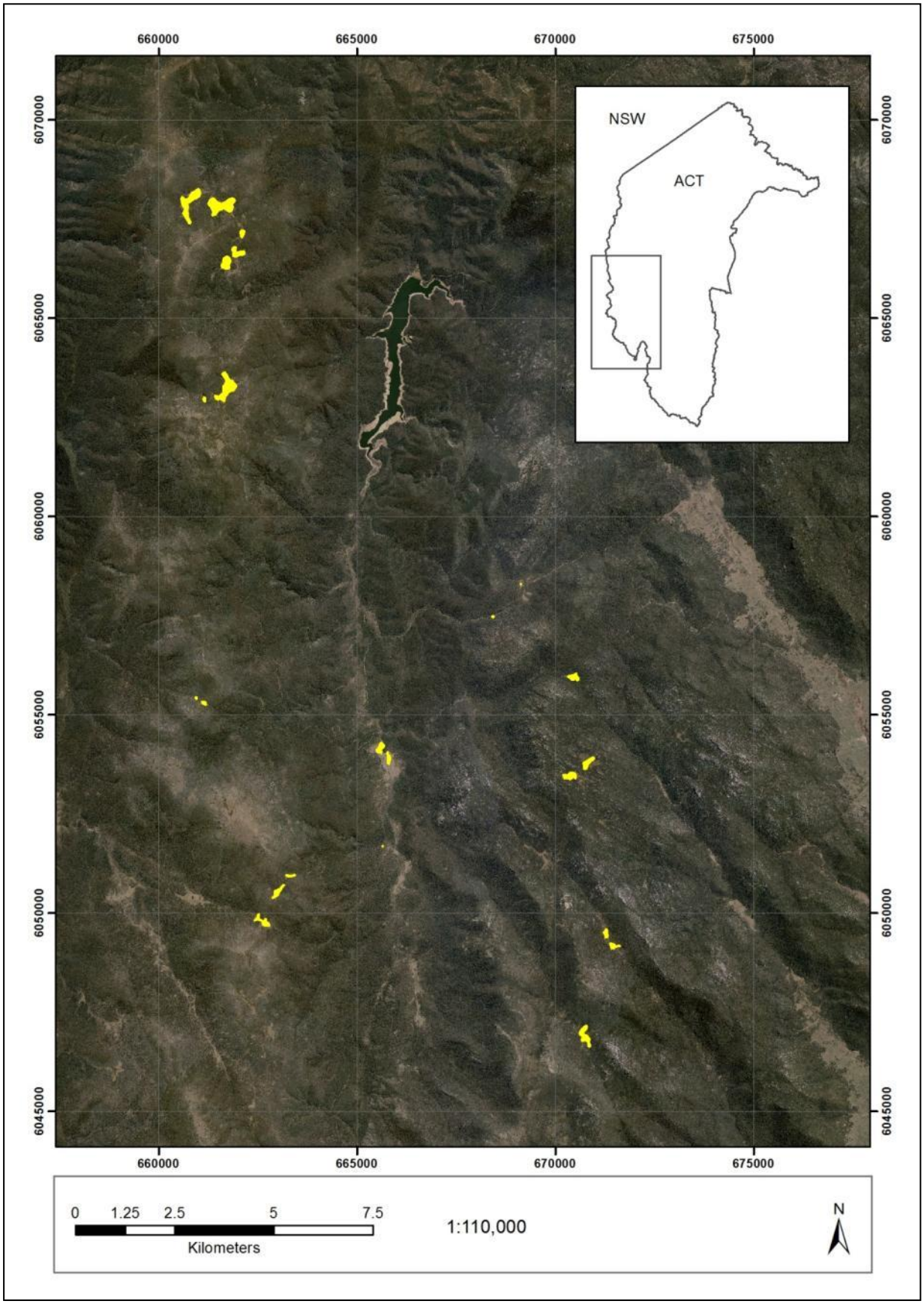
Table 4. Summary of general linear mixed effect model on likelihood of detecting *M. fuscus* scats and key vegetation types (*Poa*, heathM and sedge), distance from drainage, and incidence of feral animal disturbance, across 11 alpine bogs in Namadgi NP. Bog ID was included as a random factor (random intercept) to account for bog-level differences in mean *M. fuscus* abundance.

Fixed Factors	Coefficient	SE	Z	P
Intercept	-1.24	0.3	-4.08	<0.001
Poa	0.64	0.2	3.2	0.001
heathM	1.26	0.22	5.65	<0.001
FeralAnimDis	-1.07	0.37	-2.94	0.003
Sedge	0.46	0.21	2.2	0.028
$DistDrainL^{1/2}$	-0.07	0.022	-3.2	0.001
Random Factors		Variance		
BogID (intercept)		0.34		
Residual		1		

Table 5. Summary of model examining observer bias in detection rates of *M. fuscus* scats across 5 alpine bogs in Namadgi NP. Most notably, no significant differences exist between observers, indicating non-significant differences in *M. fuscus* scat detection rates, between observers. Bog ID was included as a random factor in each model to account for bog-level differences in *M. fuscus* abundance.

Factor	Coefficient	SE	Z	P
Observer #1	-0.81	0.205	-3.981	<0.001
Observer #2	-0.98	0.238	-4.108	<0.001
Observer #3	-0.77	0.312	-2.458	0.014
Random Factor		Variance		
Bog ID		0.04		

Map 2. ACT *Mastacomys fuscus* distribution 2013.



5 Discussion

5.1 Distribution, status and abundance

Mastacomys fuscus was detected across 13 of 14 bogs surveyed in the ACT. Detection rate increased as bog size increased and decreased as distance to drainage line increased. The species showed a positive habitat preference for heath, sedge and *Poa* dominated vegetation types. Feral animal disturbance negatively influenced detection rate in almost all bogs.

The 14 ACT subalpine bogs surveyed in this study ranged in altitude from 1028 m to 1629 m, and incuded Cotter Gap, Cheyenne, Rocks Flat, Snowy Flat, Ginini West, Big Creamy, Ginini East, Tom Gregory, Little Bimberi, Rotten Swamp, Murray Flat, Leura and Top Flat. *M. fuscus* was detected in 13 of these bogs; the only bog surveyed that did not support detectable levels of *M. fuscus* was the Cotter Source bog. Feral animal disturbance in this bog was high with over 31% of quadrats showing some sign of feral animal disturbance. The results of this study increase the number of known sites to have supported *M. fuscus* in the ACT from 14 to 23 (Map 2), thereby extending the species' known distribution in the region. Previous evidence of the species in the ACT comes from only three surveys from over 23 years ago and opportunistic observations at some sites of runways and scats. Due to the nature of the previous surveys (i.e. presence/absence) it is not possible to identify trends in population size over the past 40 years. The information collected in the current study can be used as a baseline for longer-term monitoring of *M. fuscus* distribution, abundance and habitat use and also for detecting shifts in the composition and distribution of bog vegetation communities, which might be expected in response to climate change. Shifts in composition and distribution of alpine vegetation communities due to climate change have been identified as a key threat to the persistence of *M. fuscus* across its range (DECC NSW 2007).

5.2 ACT Habitat

M. fuscus was found to occur across a variety of habitat types in the ACT including: rocky outcrops, tussock grasslands, sedgelands and heathlands, frequently within close proximity to watercourses. Evidence of woodland habitat use in areas adjacent to subalpine bogs and grasslands was also found, although results of a habitat selectivity analysis indicated a weak negative association with the vegetation type, most likely due to a lack of appropriate ground cover. Overall, *M. fuscus* was strongly associated with sites dominated by heath (50–150 cm high) and/or areas dominated by *Poa* and sedges. Surprisingly *M. fuscus* showed a weak, negative association with low spreading heath, which would be expected to provide excellent cover and protection from predators (DECC NSW 2007). However, the accuracy of this result is uncertain because scat detectability may have been reduced due to poor visibility.

Proximity to drainage lines is apparently important to *M. fuscus*, with the likelihood of detecting the species decreasing 1.1% every metre away from a drainage line. These results are consistent with observations from Kosciuzsko National Park that indicate the species is most frequently encountered within 15 m of watercourses (K Green pers. comm.). Drainage lines are likely to support a higher density and complexity of vegetation, as well as a higher abundance of grasses, both of which are key habitat attributes for the species. *M. fuscus* feed primarily on monocotyledons and is generally associated with dense vegetation cover (Carron et al. 1990). Nil spatial autocorrelation in model

residuals indicate that in the present study, linear distance from nearest drainage is an adequate predictor of *M. fuscus* distribution in relation to drainages.

5.3 Bog size

The detection rate of *M. fuscus* increased as bog size increased. In a number of animal species, habitat patch size has been shown to influence population persistence (Banks et al. 2007). Larger habitat patches are likely to support larger populations as well as provide a greater 'target' for individuals dispersing into the area (Hanski and Simberloff 1997). Interestingly O'Brien et al (2008) showed that there is no evidence of patch size influencing population persistence, although habitat patch sizes for *M. fuscus* in the Barrington Plateau tended to be larger than those found in the ACT. There is evidence of populations of *M. fuscus* in the ACT region declining in isolated swamps, most likely the result of reduced dispersal success through predation by introduced predators (O'Brien et al. 2008). *M. fuscus* are good dispersers and have been shown to recolonise high quality patches following local extinctions, though long distance movements are uncommon, especially if riparian cover or creek lines are absent (Bubela et al. 1991; Happold 1995; O'Brien 2008). Drainage systems between bogs should therefore be managed as a key habitat resource for the species, aiding dispersal and thereby the persistence of the species in the region.

5.4 Vegetation composition

Vegetation composition varied with bog size. Across all bogs surveyed, *Poa* grass was the most abundant species recorded, followed by heath (<50 cm and 50–150 cm high), Emposdisma and sedges. Exotic grass species were only recorded at Cotter Gap, Tom Gregory and Murray Gap. Unlike the other sites, Cotter Gap, Murray Gap and Tom Gregory are all located on the Great Alpine walking trail or are easily accessible by vehicle and therefore are at increased risk to the introduction of weeds. While bog size influenced both *M. fuscus* detection rate and vegetation composition, it is not clear whether the increase in likelihood of detecting *M. fuscus* in larger bogs is due to changes in vegetation composition and/or due to an increase in habitat size or other unmeasured habitat attributes.

5.5 Threats and Management Implications

Climate change

Climate change is recognised as a key threatening process to the long-term persistence of *M. fuscus.* Sphagnum bogs, fens and other associated vegetation communities across the alpine and subalpine region are likely to be significantly impacted by climate change. Drought, increased temperatures, increased frequency of wildfires and increased occurrence of invasive species are predicted to significantly impact the functioning of the endangered ecological community and result in the loss of many ecosystem services (Macdonald 2009). Lower rainfall or less evenly distributed rainfall predicted under climate change may result in drying of drainage lines leading to changes in density and composition of vegetation. While there is evidence to suggest that these changes will be detrimental for *M. fuscus*, there is also evidence to suggest that climate change may increase available habitat for the species. For example, an increase in occurrence of wildfires, as is predicted with climate change, may lead to an increase in the number of sedgelands and/or grasslands and a reduction in sphagnum (McDougall and Broome 2007; Macdonald 2009). *M. fuscus* show a positive

association with both *Poa* grasslands and sedgelands and negative association with sphagnum. However, with elevated summer temperatures, woody vegetation is likely to increase in alpine areas (Williams and Costin 1994). *M. fuscus* were shown in this study to utilise woodlands adjacent to high quality habitat, though results of the habitat selectivity analysis showed that *M. fuscus* tend to avoid woodland habitat and therefore the expansion of woody vegetation may reduce available optimal habitat and lead to an overall reduction in habitat complexity and structure. Furthermore, while restricted altitudinal movement is recognised as a significant impact of climate change for *M. fuscus* populations in the Barrington Plateau (NSW), the significance of such processes in ACT populations are currently unknown. Populations of *M. fuscus* have been identified across a range of altitudinal gradients (1028 m to 1629 m) and, unlike the Barrington Plateau population, altitudinal movement for the species in the ACT is less restricted. Indeed, populations of *M. fuscus* have been identified at relatively low altitudes in the ACT (e.g. 1028 m).

Fire

An increase in fire frequency within alpine bog communities is predicted to increase the number of peaty sedgelands, grasslands and heath, while decrease the number of *Sphagnum* peatlands (MacDonald 2009). While an increase in sedgeland, grassland and heath vegetation is likely to increase available habitat for *M. fuscus* in the long-term, high intensity fire is likely to significantly impact populations of *M. fuscus* in periods immediately following fires through direct mortality and loss of resources (e.g. food and shelter). Even in unburnt areas, predation by foxes on *M. fuscus* and other small mammals has been shown to increase following fires, with predation concentrated in areas still carrying populations of small mammals (Green and Sanecki 2006). The wildfire in 2003 burnt most of the habitat for *M. fuscus*, though individuals managed to persist in unburnt habitat patches as evidenced by recent scats and runways in these patches (Carey et al. 2003). While *M. fuscus* are good dispersers and have been shown to recolonise high quality patches following extinction (i.e. following high intensity fires), re-colonisation following such events will depend on the level of connectivity between populations as well as proximity from source populations. While the Top Flat population is relatively isolated from other known populations of *M. fuscus,* the population is well connected to other populations via suitable dispersal habitat along the Cotter River. Nevertheless, actions to reduce high intensity fire risk should be incorporated into the management of such relatively isolated sites.

It should be noted that a number of heath species associated with *M. fuscus* require fire for reproduction. A low intensity fire frequency of 10–40 years is likely to be appropriate for maintaining or improving heath habitat (NPWS 2005) for *M. fuscus*. *Sphagnum* peatlands (an Endangered Ecological Community under the EPBC Act 1999) and surrounding woodland habitat should not be subject to this fire regime, and instead fire should be excluded from these habitats, particularly the fire-sensitive *Sphagnum* moss bog communities. While peatlands in good condition are relatively resistant to low intensity fire, increased evaporation and reduced rainfall with a changing climate (Lucas et al. 2007) is predicted to dry out *Sphagnum* peatlands and significantly increase the risk and impact of fire within the community (Macdonald 2009).

Habitat degradation through feral animal disturbance

Habitat degradation by feral pigs and rabbits poses a significant threat to populations of *M. fuscus* in the ACT. Damage caused by rabbits and pigs was recorded in all 14 bogs surveyed and was significantly correlated with lower *M. fuscus* detection rate across all bogs except Cotter Gap. Feral

pig and rabbit activity is likely to result in erosion, a reduction in food resources and a change in habitat structure and composition (Green and Osborne 2003).

Feral animal activity is likely to contribute significantly to the establishment and spread of weeds and it is possible that feral pigs could play a key role in the spread of *Phytophthora. Phytophthora* poses a significant threat to the species habitat – leading to a loss of heath and other vegetation essential to the species survival. Bogs with *M. fuscus* and other threatened species should be priority target areas for landscape scale feral animal control programs in Namadgi National Park.

Weeds

The invasion of exotic plant species into key *M. fuscus* habitat poses a significant threat to the species survival. The invasion of exotic grasses is likely to result in the loss of important ground layer vegetation such as Snow grass with the potential to reduce the availability of that food source for *M. fuscus*. Yorkshire fog *Holcus lanatus* was recorded at relatively high density at the Cotter Gap bog, though there is currently no evidence to indicate that this introduced species is having an adverse affect on the population as *M. fuscus* detection rate in this bog was high. *H. lanatus* has, however, been identified as a potential threat to populations of *M. fuscus* in the Barrington Tops where there is concern that this grass may displace significant native ground layer species such as Snow Grass, which *M. fuscus* is known to rely heavily on as a food resource (DECC NSW 2007). Careful monitoring of the Cotter Gap bog, and other bogs that can be accessed by the public on foot or by vehicle, is recommended to ensure that infestations of *H. lanatus* and other weeds do not increase and that the presence of the species is not having an adverse affect on the population of *M. fuscus*. It should be noted that *H. lanatus* is unlikely to be a food resource for *M. fuscus*.

Competition

A number of both exotic and native species compete with *M. fuscus* for food, shelter and space. Evidence of rabbits was recorded in nine of the 14 sites surveyed, with disturbance highest at Cotter Gap, Cotter Source and Tom Gregory. Rabbits compete directly with *M. fuscus* for food; and grazing by rabbits can result in the removal or modification of tussock structure that provides shelter and nesting habitat for *M. fuscus*. Control of rabbits and foxes is likely to be most effective when undertaken concurrently at a landscape scale, and control of these species is particularly important at the following three bogs: Cotter Source, Cotter Gap and Tom Gregory.

Competition for resources from native species, such as the Bush Rat *Rattus fuscipes*, may pose an additional competitive threat to *M. fuscus*. For example, a significant increase in the number of *R. fuscipes* trapped in Nolans Swamp (NSW) coincided with a dramatic decrease in the number of *M. fuscus* captured, indicating a potential dominance of *R. fuscipes* over *M. fuscus* (Keating 2003). *R. fuscipes* were detected in four of the 14 bogs surveyed (incl. Cheyenne, Murray Gap and Ginini East and West) in the ACT. However, *R. fuscipes* is naturally sympatric with *M. fuscus* and therefore it is unknown if and under what circumstances *R. fuscipes* poses a potential competitive threat to *M. fuscus* in ACT populations.

Predation

M. fuscus fall within the Critical Weight Range (CWR; Burbidge and McKenzie 1989) and the Vulnerable Terrestrial Vertebrate category (Smith and Quin 1996) indicating a high risk of population decline and extinction through predation from foxes and cats. There is also evidence of selective

predation by foxes on *M. fuscus* over other sympatric CWR rodent species. For example, Green (2002) showed that *M. fuscus* in the alpine regions of the Snowy Mountains were five times more frequently found in fox scats than were *R. fuscipes* despite occurring at relatively similar densities. The disparity in predation rate has been attributed to behavioural difference. For example, *M. fuscus* are less aggressive than *R. fuscipes*, they are slower, they use well-established runways and they nest communally above the ground during winter.

Evidence of foxes was found in two of the 14 bogs surveyed, including Ginini West and Cotter Source.

5.6 Conclusion

M. fuscus is present at a range of sites in the higher elevation areas of the ACT where there is suitable habitat. Whilst the wildfire of 2003 burnt most of the habitat for the species, individuals survived in unburnt patches, and 13 years post-fire the species appears to still maintain its pre-fire distribution in the ACT. To what extent the 2003 fires reduced populations of *M. fuscus* in the ACT and resulted in the species going through a genetic 'bottleneck' is not known. Although it is not possible to determine a trend in abundance from a single 'snapshot' study, the high occupancy rate across the majority of bogs surveyed in the ACT subalpine region suggests that the populations of the species are currently relatively stable, albeit within a small and patchy distribution. On-going monitoring is required to determine whether ACT populations of *M. fuscus* are experiencing longer-term declines that appear to have occurred in other areas. This study provides baseline information on vegetation parameters and an index of density of *M. fuscus* and of introduced predators and competitors in bogs against which future changes can be assessed and key-threatening processes can be monitored. Population monitoring of *M. fuscus* and vegetation assessments undertaken every 5-10 years would enable long-term trends to be identified. The methods used in this study are an efficient and repeatable method for identifying trends in *M. fuscus* abundance and distribution.

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Personal Communications
Murray Evans, Conservation Research section, ACT Government.
David Whitfield, Parks and Conservation Service, ACT Government.
Ken Green, Office of the Environment and Heritage, NSW Government.

Appendix

Figure A. *Mastacomys fuscus* runway.

Figure B. Mastacomys fuscus scats.

Figure C. Vegetation community dominated by medium heath.

Figure D. Vegetation community dominated by *Poa*.

Figure E. Vegetation community dominated by sedge.

Figure F. Woodland vegetation community.

Figure G. Vegetation community dominated by low heath.