



**Presence, breeding activity and
movement of the yellow-footed
antechinus (*Antechinus flavipes*), in a
fragmented landscape of the southern
Mt Lofty Ranges.**

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ABSTRACT

Habitat fragmentation poses a threat to small mammal populations existing in remnant vegetation. Reduced habitat area, habitat isolation and inhibited dispersal are some of the factors that can increase the risk of local and regional extinction. This study was undertaken to examine the persistence of the yellow-footed antechinus (*Antechinus flavipes*) in a fragmented landscape of the southern Mt Lofty Ranges in South Australia. Live trapping was carried out in small, remnant patches and strips of forest and in areas of contiguous forest, to document autecological data for the species and to investigate occurrence, breeding activity and inter-patch movements. Radio-tracking was also conducted to compare home range properties of lactating females, in restricted and unrestricted habitat.

Results of autecological investigations showed mostly comparable population densities and body weights with other *A. flavipes* populations. However, regional differences were detected in juvenile dispersal behaviour and to some extent, breeding times. The study populations showed a male-biased, natal dispersal strategy, which is the first published record of such a strategy for *A. flavipes*. This result differed from Queensland populations, which did not show distinct dispersal phases. Regional differences regarding breeding times have been attributed to varying peaks in insect abundance. The causes of the differences in dispersal behaviour remain unclear. Site differences were also apparent. Forest animals were lighter, were largely of lower population density (particularly in 2000) and were generally later breeders than patch and strip animals. Differences in weight and population density may be due to favourable 'edge effects' in remnants, while variations in breeding times may be due to local climatic or habitat factors. Despite being largely 'isolated' for approximately 50 years, all study sites detected the presence and breeding activity of *A. flavipes* in at least one of the two seasons sampled. Inter-patch movements by males and females were detected during the juvenile dispersal phase and the 2001 breeding season. Most movements were by adult males, while females tended to be recaptured in the same patches. Landscape types potentially traversed included exotic pine plantations, open paddock and narrow sections of native vegetation. Movement between sites indicated the presence of a metapopulation operating amongst local populations. Home range

investigations did not provide conclusive evidence to suggest that *A. flavipes* was able to adapt or change home range behaviour in response to habitat restriction. However, it did show that the species was able to tolerate some degree of home range overlap between resident, lactating females. Investigations also revealed the use of the landscape matrix for resource supplementation. The adaptability, life-history strategies and a tolerance of the landscape matrix shown by *A. flavipes* provide some explanation for the species' success in this fragmented system.

It is considered that the long-term persistence of *A. flavipes* in this landscape will be determined by the ability of females to maintain a presence in the small patches, the ability of unrelated males to move into the patches to breed with resident females and the maintenance or enhancement of the current habitat area and distances between habitat sites. This study illustrates the importance of recognising the occurrence of metapopulations in fragmented landscapes for conservation management purposes.

DECLARATION

This thesis contains no material which has been accepted for the award of any other degree or diploma in any university or other tertiary institution and, to the best of my knowledge and belief, contains no material previously published or written by another person, except where due reference has been made in the text.

I give consent to this copy of my thesis, when deposited in the University of Adelaide Library, being available for loan and photocopying. All references to this thesis or any information therein must be fully acknowledged in any report or publication.

Doreen Marchesan

Dated

12/9/02

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CHAPTER 1: INTRODUCTION AND LITERATURE REVIEW

1.1 Introduction

Habitat fragmentation is one of the major threats to the long-term conservation of flora and fauna worldwide. The resulting reduction of habitat area and the isolation of remaining habitat fragments can increase the risk of local and regional extinction of flora and fauna populations existing in the fragments. The arable regions of South Australia have been extensively cleared for agricultural use, with approximately 96% of native vegetation cleared on the Fleurieu Peninsula, from Cape Jervis in the south to Peterborough in the North (Taylor, 1987). Land clearance has created highly fragmented landscapes throughout the Peninsula. The habitat fragments (or remnants) of these landscapes have become a major part of the natural system in this region. As such, their use by native fauna, and the general effects of habitat fragmentation in this region needs to be understood in order to manage these areas for the long-term persistence of resident populations.

The present study was undertaken to examine the persistence of a small, carnivorous marsupial (*Antechinus flavipes*) in a highly fragmented landscape in the southern Mt Lofty Ranges, on the Fleurieu Peninsula in South Australia. Results from this study begin to address the paucity of information on the effects of habitat fragmentation on small mammal populations in South Australia, and add to the slowly growing, Australia-wide database.

1.2 Thesis Layout

Chapter 1 of this thesis introduces and reviews the issues surrounding habitat fragmentation and associated effects. The review will outline studies done to date in Australia and identify current gaps in knowledge about the ecology of *A. flavipes* in a fragmented landscape, for both Australia and South Australia. The study aims and objectives will then be given. Chapter 2 contains details of the study sites including vegetation descriptions, and the general materials and methods applicable to all phases

of the study. Chapters 3 to 5 present and discuss the results of the main phases of the study. Chapter 3 details the autecological data collected, providing background information on the study species and insight into the effects of habitat fragmentation observed in this study. Chapter 4 presents results of surveys carried out to detect the presence and breeding activity of *A. flavipes* in habitat remnants, and results of investigations into landscape movements between study sites. Chapter 5 presents results of an investigation into the home range behaviour of *A. flavipes*; a behavioural factor that may affect the species' persistence in a fragmented landscape. Chapter 6 contains a general discussion of results, including the implications of findings for the long-term persistence of *A. flavipes* in a fragmented landscape.

1.3 Literature Review

1.3.1 Habitat Fragmentation

Many populations of small mammals world-wide have been subjected to changes in landscape structure through land clearance, and are now faced with surviving and persisting in fragmented habitat. The consequences of habitat fragmentation for such populations have been debated and discussed both in Australia and around the world (eg. Saunders and Ingram, 1987; Saunders and Hobbs, 1991; Simberloff *et al.*, 1992; Fahrig and Merriam, 1994). Despite the interest in the subject there is still a paucity of available, empirical information to assist with the management of populations in fragmented landscapes within Australia.

Habitat fragmentation occurs when native vegetation is cleared for agricultural or urban use. It results in a landscape mosaic of remnant patches of native vegetation set in a matrix of agricultural and/or urban development. The remnants can be of varying sizes and shapes, connected to greater or lesser degrees by strips or other sections of native vegetation. Remnants may also be completely isolated from other areas of native vegetation by the landscape matrix. This isolation from other areas of suitable habitat is the main cause of concern for species existing in remnants, and the underlying factor behind many of the potentially negative effects of habitat fragmentation.

1.3.2 Effects - Habitat loss, isolation and restriction

The most obvious effect of habitat fragmentation is habitat loss and the associated decrease in population size, number and range of species (see Bennett, 1998). In particular, a loss of habitat will affect resource abundance and availability, increasing competition for food, nesting and space resources. Saunders and Ingram (1987) documented a decrease in reproductive success and the resulting reduction in population size and range over time, for Carnaby's cockatoo (*Calyptorhynchus funereus latirostris*) in Western Australia, due to land clearance and habitat fragmentation. A reduction in food resources was one of the main contributors to the decrease in reproductive success of this species (Saunders, 1990). Reduced habitat area can also influence species diversity and abundance. A number of authors have demonstrated a positive correlation between increasing remnant size and species diversity (e.g. Matthiae and Stearns, 1981; Bennett, 1987; Bennett, 1990), while other studies have found a positive relationship between area of remnant and occupancy (e.g. Van Apeldoorn *et al.* (1994) for the red squirrel, *Sciurus vulgaris*). Establishing the minimum area of suitable habitat for viable populations, both genetically and demographically, is a continuing quest for ecologists (Soulé, 1987). The age of the remnant (i.e. time since isolation) can also have an effect on species diversity and species presence (Bennett, 1987). For example, Bolger *et al.* (1997) found that older remnants of vegetation in southern California supported fewer species of native rodents than younger remnants.

1.3.3 Effects - Increased risk of extinction

Habitat fragmentation can increase the risk of local or regional extinction. In a review of various studies, Fahrig and Merriam (1994) concluded that extinction and recolonisation of local populations in fragmented habitat (natural and human induced) is fairly common, with extinction rates often within the range of 10 to 20% per year. Examples of such studies include Schoener and Spiller (1987) for orb weaver spiders (various species), Celada *et al.* (1994) for the red squirrel (*Sciurus vulgaris*) and Hanski *et al.* (1995) for the Glanville butterfly (*Melitaea cinxia*). The main causes of local extinctions are likely to centre around the forces of demographic, genetic and

environmental stochasticity. The effects of each of these are also likely to be felt more profoundly by populations in smaller and more isolated remnants (Krebs, 1994; Lindenmayer and Lacy, 1995).

Demographic stochasticity includes the random fluctuations in natality, mortality, dispersal and migration within a population. They can be powerful factors in influencing the persistence of populations in fragmented landscapes (Lande, 1988; Gaines *et al.*, 1997), but may not always be negative. For example, Bowers and Dooley Jr. (1999) found that mean peak population densities, population growth rate and adult recruitment of meadow voles (*Microtus pennsylvanicus*) were significantly higher in artificially fragmented remnants than in continuous habitat. This was partly attributed to enhanced reproductive activity by females inhabiting remnant edges. The genetic health of a population is another factor to consider, where genetic replenishment is restricted and the risk of inbreeding increased. There has been much discussion in the literature regarding the risks and potential effects of inbreeding (Shields, 1983; Charlesworth and Charlesworth, 1987; Lacy, 1992). Such effects include a loss of genetic diversity, a reduction in fitness and viability within a population and an increase in vulnerability to environmental change (Lacy, 1992; Falconer and Mackay, 1996). However, detecting and documenting the effects of inbreeding in wild populations is difficult. Empirical studies (eg Sarre, 1995) and computer simulation models (e.g. Lacy and Lindenmayer, 1995) have provided some examples of losses (or potential losses) of genetic diversity in populations existing in fragmented landscapes. Environmental stochasticity is the third main factor to pose risks of local extinction. Biotic influences (such as disease and predator invasion) and abiotic events (such as fire, flood and land clearance) could all play a part in causing local extinction in one or more remnants.

The exact causes of local extinction are difficult to explore (particularly those relating to genetics), leading authors to challenge the presumed occurrence and severity of the above factors on the persistence of species in fragmented landscapes (Simberloff *et al.*, 1992). Despite limited empirical evidence, it is acknowledged that the occurrence of local extinctions in fragmented landscapes can be a threat to the local or regional persistence of a species, if functions such as population replenishment, genetic

replenishment and the ability for recolonisation are not maintained. Such functions will be aided by the movement of animals between remnants.

1.3.4 Landscape movements and metapopulations

In the cases of local extinction described by Fahrig and Merriam (1994), regional populations were able to persist if the extant remnants were recolonised by dispersing individuals from source populations within the system. Dispersal and other movements, then, become key functions for recolonisation and population replenishment (Harrison, 1991; Merriam, 1991; Fahrig and Merriam, 1994). Movements within a landscape allow the formation of a metapopulation within a region; that is, a population of local populations connected by dispersing individuals (Hanski and Gilpin, 1991). The formation of metapopulations can allow recolonisation of extant remnants, maintenance of adequate gene flow between populations and wider exploitation of resources by individuals within a population (Merriam, 1991; Fahrig and Merriam, 1994; Sarre, 1995). The maintenance of metapopulations can be critical to the long-term persistence of species in fragmented landscapes.

The types of dispersal and other movements that could occur between populations include juvenile dispersal, mate searching, general migration and wide-area foraging. Such movements can be limited in a fragmented landscape by a number of factors. Habitat remnants exist largely in an agricultural matrix. However, many small Australian animals find agricultural landscapes and other breaks in cover (such as roads) inhospitable (Burnett, 1992; Downes *et al.*, 1997a; Downes *et al.*, 1997b; Goosem and Marsh, 1997) and will not enter or cross them. The degree of inhospitability will depend on the agricultural land use and the particular species. For example, it has been demonstrated that pine plantations can serve as habitat for some populations of native bush rats (*Rattus fuscipes*), whereas other small mammal species of the same region were largely absent from the plantations (Suckling and Heislars, 1978). Other factors influencing landscape movements include the dispersal ability of species. It stands to reason that highly mobile species that can tolerate the landscape matrix would be more inclined to successfully complete landscape movements than

their more sedentary counterparts. Dispersal strategies are also significant to the success of species in fragmented landscapes. Bennett (1987) partly attributed the persistence of the long-nosed potoroo within a fragmented landscape in Victoria to the fact that both sexes dispersed from their natal home range. Such a strategy would allow successful colonisation of extant remnants, assisting in the maintenance of the regional metapopulation. In contrast, a sex-biased dispersal strategy could result in single-sex colonisation of extant remnants, which would be of limited value to the species' regional persistence (Lindenmayer and Lacy, 1995). However, if sex-biased dispersal occurs between inhabited remnants, it may assist in the maintenance of heterozygosity. This was shown by Aars and Ims (1999) in an experiment on the effect of corridors on the transfer of root voles (*Microtus oeconomus*) between habitat patches. Analysis of the expected and observed frequency of heterozygosity showed that that observed frequency among offspring recruited in a habitat remnant was higher than the expected. This was explained by the male sex-biased dispersal behaviour of the species and the use of connecting corridors for mating forays. These issues are particularly pertinent to many Australian small mammal species that exhibit sex-biased dispersal strategies.

1.3.5 Landscape Connectivity

The degree of connectivity between remnants is an important aspect of the landscape structure, as it may influence movement opportunities (Anderson and Danielson, 1997). Bennett (1998) identified two main components of connectivity – structural (length of gaps, distance to be travelled, continuity of fragments) and behavioural (the behavioural response of individuals and species to the landscape structure). He also pointed out that the connectivity that exists in a landscape for one species may be different for another, depending on the species' perception of the landscape.

Structural connectivity can be in the form of vegetation or other types of structures that allow the passage of individuals from one remnant to another. Much of the focus of structural connectivity has been on 'corridors' (eg. Merriam, 1991; Lindenmayer and Nix, 1993; Bennet *et al.*, 1994; Andreassen *et al.*, 1996; Downes *et al.*, 1997a; Tischendorf and Wissel, 1997; Bennett, 1998); relatively narrow strips of native

vegetation (or other suitable habitat) connecting two or more remnants. Corridors have been considered a viable option (and sometimes considered the best option) for providing connectivity in a landscape, attracting much interest and debate (eg. Saunders and Hobbs, 1991; Simberloff *et al.*, 1992). The concept of corridors had been widely adopted as the solution to the potential genetic and demographic problems brought on by habitat fragmentation, despite a lack of empirical evidence about the use and suitability of corridors in fragmented landscapes. This adoption in the absence of supportive data has been criticised, and in some cases shown to be inappropriate to the conservation objectives set (Lande, 1988; Simberloff *et al.*, 1992). Corridor management and construction remains a possible, viable management strategy, pending conclusive results from well-designed studies. However, it should be acknowledged that corridors are not the answer to all habitat fragmentation problems (Lindenmayer, 1994).

Examination of the surrounding, unfavourable matrix as a form of connectivity has been relatively understudied. However, there is evidence to suggest that some species will tolerate the matrix for movements between remnants. For the male hispid cotton rat (*Sigmodon hispidus*), 130 m of forest matrix (typically unfavourable habitat for this species) served as a form of connectivity between suitable patches of grassland habitat (Bowne *et al.*, 1999). Lindenmayer *et al.* (1999) suggested that a pine forest matrix, although not the preferred habitat for residence by small native mammals, may serve as connectivity between native vegetation fragments in an area of pine plantations in south-eastern Australia.

1.3.6 Australian Studies to Date

Studies conducted in Australia to ascertain the effects of habitat fragmentation on resident species have tended to consider the resulting distribution and diversity of species (Suckling, 1984; Laurance, 1994; Lindenmayer *et al.*, 1994; Bentley *et al.*, 2000). However, little attention has been given to how a species is able to persist and move within a fragmented landscape; their ability to persist and breed in habitat remnant, their mobility between remnants, the suitability of the landscape matrix for movement or a specific behavioural response to habitat restriction. Despite the extent

of land clearance in South Australia, little work has been conducted to investigate the effects of such clearance on remaining animal populations. There have been investigations on particular topics, such as the value and management of roadside vegetation (Lewis, 1991; Hyde, 1998). However, there is still much to be investigated, particularly in areas where the majority of the remaining habitat is in the form of habitat remnants.

One of the few small mammal species remaining on the Fleurieu Peninsula is the yellow-footed antechinus, *Antechinus flavipes* (Plate 1.1). Much of its habitat now exists as isolated remnants of native vegetation. In 1998 a brief survey (MacKenzie, 2001) was conducted in the southern Mt Lofty Ranges, where *A. flavipes* was detected in small, relatively isolated remnants of native vegetation. It was not determined if these were transient individuals or belonged to functioning populations within the remnants. Results of this survey provided an opportunity to examine this species in a fragmented landscape, and to further the investigation of the effects of habitat fragmentation on South Australian populations of small mammals. The present study also contains autecological data for wild populations of *A. flavipes* in South Australia, which is lacking in the literature.

1.4 Aims and Objectives

The general aim of this study was to investigate if and how *A. flavipes* was able to persist in a fragmented landscape of the southern Mt Lofty Ranges. This was carried out by examining:

- The presence and breeding activity of *A. flavipes* in remnant patches and strips,
- Inter-patch movements and the matrix type traversed by *A. flavipes*,
- Home range behaviour and any changes resulting from habitat restriction and
- The autecology and life history strategies of *A. flavipes*.



Plate 1.1 – *Antechinus flavipes* (by A. Headland, 2001).

CHAPTER 2: STUDY SITE AND GENERAL MATERIALS AND METHODS

2.1 Study Site Description

The study region was located near Parawa, in the southern Mt Lofty Ranges of South Australia (Lat. 35°34'50", Long. 138°15'40"). This region has been subjected to land clearance, the majority of which occurred between 1950 and 1965 (Jones, M. 2001, *pers. comm.*). Land clearance resulted in a landscape mosaic of grazing land, exotic pine plantations and remnants (patches and strips) of native vegetation. The majority of remnants are between 0.5 and 80 ha in size, with the largest area of native vegetation being Deep Creek Conservation Park at 4500 ha. In brief, the vegetation community in this region is characterised by an open forest (Specht, 1972) of messmate (*Eucalyptus obliqua*) or brown stringybark (*E. baxteri*) with an understorey dominated by yaccas (*Xanthorrhoea semiplana* ssp *tateana*), bracken fern (*Pteridium esculentum*) and heathy shrubs (Plate 2.1). The region consists of low, undulating hills with a range in elevation of 250 to 350 m. The average annual rainfall in this region is 884.5 mm, with an average maximum temperature of 17.2°C and average minimum of 10°C. Annual rainfall figures effective for the study period are 866.6 mm (1999), 1010.8 mm (2000) and 1093.2 mm (2001) (Bureau of Meteorology, SA).

Sixteen study sites were selected for this project (Figure 2.1); 13 in remnant patches and strips (sites 1 to 12 and Add) and three in contiguous forest of the Deep Creek Conservation Park (Sites F1, F2 and F3). The number of sites was chosen to allow detailed examination of remnant dynamics, rather than to test for differences between site types. Such testing would require a much higher number of study sites within each site type. Replicates of each site type were also limited by the availability of suitable sites within the landscape and logistical constraints, given the detailed nature of the project.

Study patches and strips ranged from 0.5 to 2.5 ha in size, except for the patch in which site Add was located, which was 6 ha. This 'additional' site was included predominantly for the purposes of detecting inter-patch movements. Each study site



Plate 2.1: Typical vegetation community of study sites. Photograph taken in site F3.

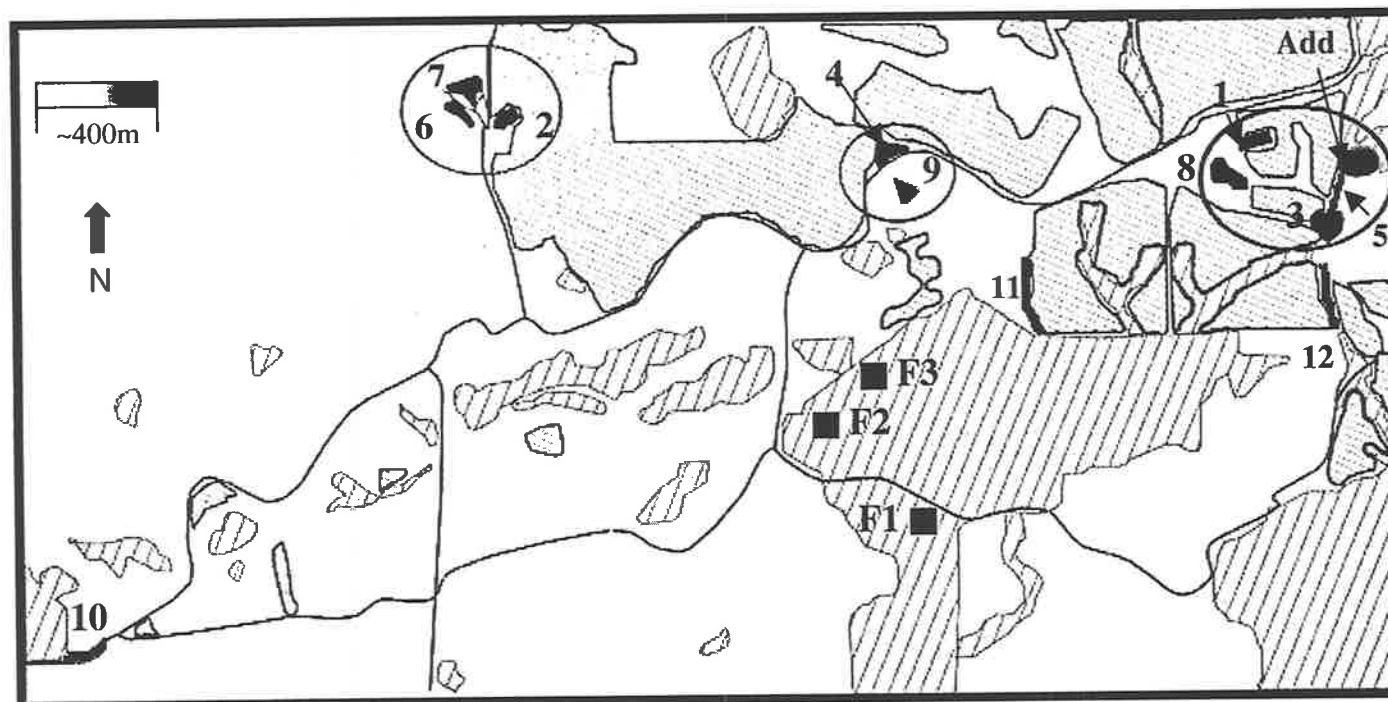


Figure 2.1: Map of study sites (blocked and labeled), other native vegetation (wide hatch) and exotic pine plantation (narrow hatch). Circles show groups of patches used to examine inter-patch movement. 'Add' labels the additional site surveyed primarily for inter-patch movements. Sites 1, 2, 4, 8 and 9 are discrete patches, sites 3, 6 and 7 are semi-detached patches and sites 5, 10, 11 and 12 are linear strips.

was classed as one of four site types: forest sites ($n=3$), discrete patches ($n=5$), semi-detached patches ($n=4$) and linear strips ($n=4$). Forest sites were located in contiguous forest of the Deep Creek Conservation Park, within *E. obliqua* forest; discrete patches were between 1.3 ha and 1.8 ha, totally surrounded by agricultural land (exotic pine plantation and/or grazing paddock); semi-detached patches were between 0.5 ha and 2.5 ha, separated from native vegetation by a narrow barrier such as a road or clearing, yet mostly surrounded by agricultural land; and linear strips were 20 to 30 m wide and 200 to 600 m long, connected at one or both ends to native vegetation. One of the discrete patches did have a road dissecting it. However this road did not appear to block daily movements between segments, and so was treated as one site. An example of landscape structure is shown in Plate 2.2. Attempts were made to decrease variability amongst sites by selecting patches of similar size, vegetation community and quality and degree of isolation (amongst site types). It is likely that these sites had been selectively logged and grazed in the past. One site continued to be grazed during the study period. A summary of disturbance history for each site is contained in Table 2.1.

Detailed vegetation surveys of study sites were not conducted, as there was no intent to rigorously test for site differences using vegetation characteristics. However, it was considered important to record at least general vegetation characteristics to provide base information. Vegetation surveys documented the community type present, the major native and weed species present and obtained simple, relative measures of structural characteristics considered to be potentially significant to the persistence of *A. flavipes*. Vegetation characteristics measured were structure (using Specht, 1972; See Appendix 1a) dominant overstorey species, sub-canopy species, distribution of overstorey, understorey type, understorey condition and disturbances present (using Stokes *et al.*, 1998; see Appendix 1b). The presence of tall *X. semiplana*, the presence of logs, the presence of large, old trees, plant species diversity and species richness were also noted. Relative scores of high, medium and low were given for each (Table 2.2). Vegetation characteristics were then available for preliminary correlation with some of the main findings of the study. Plant species recorded in each site are presented in Appendix 2.

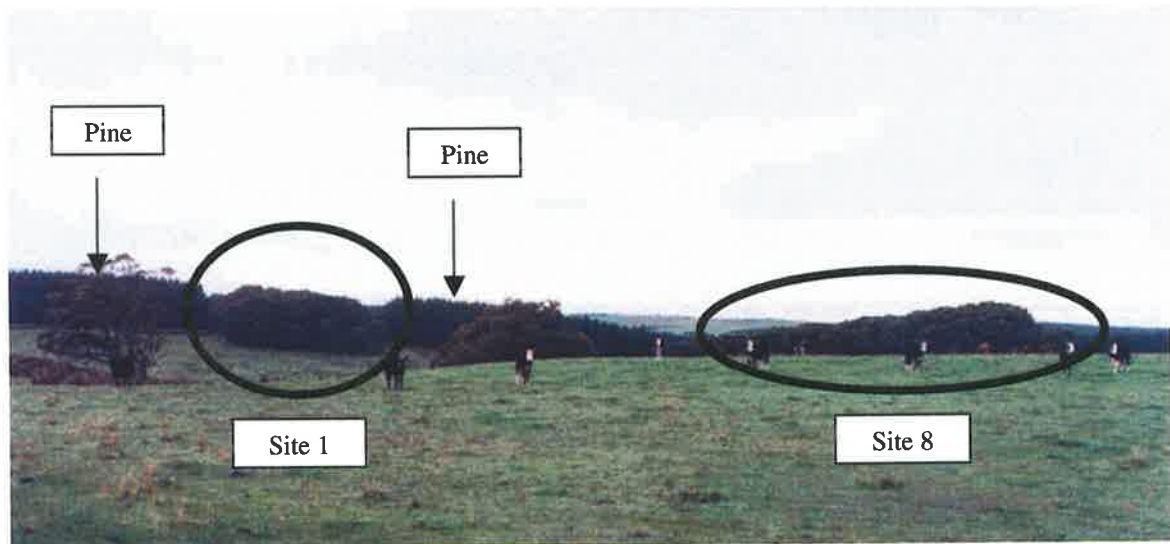


Plate 2.2: Example of landscape structure, showing sites 1 and 8, pine plantations and cattle grazing paddock. View is facing East.

Table 2.1: Size of study patches (habitat area), approximate number of years since land clearance and the disturbance history of study sites.

Site	Size (ha)	~ No. Years Since Land Clearance	Past Grazing	Past Logging
1	1.7	50	Light	Light
2	1.3	30-35	Light	Light
3	2.5	50-55	Light	Light
4	1.8	50	Light	Light
5	0.6	50	Light	Light
6	0.9	50	Light	Light
7	0.5	50	Moderate	Light
8	1.8	50	Moderate	Light
9	1.3	50	Heavy*	Light
10	0.7	50	Light	Light
11	1	50	Light	Light
12	0.8	50	Light	Light
Add	6	50	Moderate*	Light
F1	In contiguous forest	NA	Nil	Light
F2	In contiguous forest	NA	Nil	Light
F3	In contiguous forest	NA	Nil	Light

* Grazed during study period.

Table 2.2 – Vegetation characteristics for each site.

Site	OS	Sub-canopy	Structure	Dist. OS	US Type	US Cond	Xanth	Logs	Old Trees	Diversity	Disturb
1	E.o.		OF	C	SO, B	3	4	L	L	L	CR, AT, CL
2	E.o		OF	C	SO, B	3	3	L	L	L	FA, CR
3	E.o		OF	C	SO, B	3	4-5	M	M	L	CR, FA
4	E.o/P.r*		OF	D	SO, VN	3	1	L	M	M	FA, AT, CR, RD, CL
5	E.o		OF	C	SO, B, JE	4	4	L	H	L	G, CR, FA
6	E.o	P.r*, A.p, E.c, As*	LOF	C	SO, VN	3	1	M	H	H	RD, FA, GE, CR
7	E.o	A.p	LOF	C	SO, SE, VN	3-4	1	L	L	M	G, FA, AT, SP, CR
8	E.o		OF	C	SO, B	4	2	M	L	L	CR, FA
9	E.o		OF	C	JE, B, SO, SE	1	3	H	L	L	G, CR
10	E.o		LOF	C	JE, SO	3-4	3	H	M	M	RD, CR
11	E.o	A.p	OF	C	SH, B, SO	2	1	L	M	H	CR
12	E.o/E.b	A.m	OF	C	SO, VN	2-3	1	L	L	M	CR, AT
Add	E.o		OF	C	SO, B	4	4-3	L	M	L	G, AT, CR, AT
F1	E.o		TOF	C	SO, B	2-3	3	M	L	L	CR, FA
F2	E.o	E.c	TOF	C	SO, B	2-3	2	M	L	L	CR, FA
F3	E.o		TOF	C	SO, B	2-3	3	M	L	L	CR, AT, FA

Legend - see Appendix 1b for descriptions of categories (excluding Xanth, Logs, Old trees and Diversity)

* - Introduced species

OS – E.o (*Eucalyptus obliqua*), P.r (*Pinus radiata*), E.b (*E. baxteri*)

Sub-canopy – P.r (*Pinus radiata*), A.p (*Acacia pycnantha*), E.c (*Exocarpos cupressiformis*), As (ash tree; species unknown), A.m (*A. melanoxylon*)

Structure – OF (open forest), LOF (low, open forest), TOF (tall open forest)

Dist. OS – C (continuous), D (dis-continuous)

US type – SO (other shrubs), B (bare ground/litter), VN (sedges), JE (exotic herbs), SH (shrubs, heath)

US condition – score of 1 (excellent) to 5 (poor).

Xanth – relative score of 1 (low) to 5 (high) for presence of tall *Xanthorrhoea semiplana*.

Logs – relative scoring for presence of logs; L (low), M (medium), H (high).

Old Trees – relative scoring for presence of old, large trees; L (low), M (medium), H (high).

Diversity – relative scoring for species diversity within sites; L (low), M (medium), H (high).

Disturb – CR (coppice regrowth), AT (access track), CL (clearing), FA[#] (feral animals), RD (rubbish dumping), G[#] (grazing), SP (stockpile) ([#] Added to list)

2.2 Sampling Methodology

Data were collected during eight trapping sessions and four radio-tracking sessions (which also included some trapping) between July 2000 and November 2001. Trapping sessions to specifically examine breeding activity occurred in July and September 2000 and July, August and September 2001. Sessions to record inter-patch movements were carried out during the juvenile emergence and dispersal phase in December 2000 and January and March 2001. Inter-patch movements were also determined from data collected in July, August and September 2001. The presence of *A. flavipes* was determined from all trapping sessions. Radio-tracking sessions to assess home range behaviour were carried out in October and November of 2000 and 2001. Different sites were sampled at different times, depending on the phase of the study (Table 2.3). All sites were sampled during at least three trapping sessions.

Live trapping was carried out using aluminium box Elliott traps (33 x 10 x 9 cm). Traps were baited with a mixture of rolled oats, honey and peanut butter and placed in plastic bags to prevent rain entering the traps. Wool fibre was added during the 2001 breeding season to provide bedding. Traps were laid at 20 m intervals in single lines or in a grid, depending on patch or strip shape, with trap lines commencing 15 to 20 m in from the edge of the patch or strip. The number of trap stations ranged from eight to 21, depending on patch or strip size, with two traps set per station (Table 2.4). Only 14 trap stations were used at each site during radio-tracking sessions. Five of the 12 trapping grids within small patches and strips covered the entire patch or strip, while the remainder covered more than 50% (Table 2.4).

Traps were set at dusk, checked at first light then left closed during daylight hours. Any *A. flavipes* captured were identified, weighed, sexed, marked by ear-notching, checked for sexual condition (Smith, 1984) then released at the site of capture. Other species captured were noted but not marked or measured. Radio tracking methodology is detailed in Chapter 5.

Table 2.3: Sites sampled during each session (♦). 'F' denotes a site not trapped due to fox disturbance, which included incidents of animals being taken from inside the traps and/or trap disturbance at the majority of trap stations in a study site.

	Site												Add	F1	F2	F3	
	1	2	3	4	5	6	7	8	9	10	11	12					
Jul-00	♦	♦	♦	♦	♦				♦	♦	♦	♦					
Sep-00	♦	♦	♦	♦	♦					♦	♦	♦		♦	♦		
Oct-00		♦								♦				♦			
Nov-00	♦				♦												♦
Dec-00	♦	♦	♦	♦	♦	♦	♦	♦	♦				♦				
Jan-01	♦	♦	♦	♦	♦	♦	♦	♦	♦				♦				
Mar-01	♦	♦	♦	♦	♦	♦	♦	♦	♦				♦				
Jul-01	♦	♦	♦	♦	♦	♦	♦	♦	♦	♦	♦	♦	♦	♦	♦	♦	♦
Aug-01	♦	♦	♦	F	♦	♦	F	♦	F	♦	♦	♦	♦	♦	♦	♦	♦
Sep-01	♦	♦	♦	F	♦	♦	F	♦	F	♦	♦	♦	♦	♦	♦		
Oct-01	♦	♦								♦							
Nov-01					♦									♦			♦

Table 2.4: The number of trap stations, total number of traps, patch size and approximate % of patch covered by trapping grid for each site.

Site	No. trap stations	Total no. traps	Size (ha)	~% patch covered
1	20	40	1.7	100
2	14	28	1.3	90
3	20	40	2.5	60
4	20	40	1.8	100
5	10	20	0.6	100
6	20	40	0.9	100
7	8	16	0.5	50
8	20	40	1.8	60
9	15	30	1.3	90
10	15	30	0.7	100
11	20	40	1	70
12	15	30	0.8	70

CHAPTER 3: AUTECOLOGY OF *Antechinus flavipes* IN A FRAGMENTED LANDSCAPE

3.1 Introduction

When investigating the effects of habitat fragmentation on a species, autecological data can complement and provide important insight into the biogeographic data collected (Bennett, 1987). Such biogeographic data will follow in Chapters 4 and 5. To date there is a paucity of autecological data on *A. flavipes* in the southern part of its range, with very little published work found on populations of *A. flavipes* in South Australia (Inns, 1976; Cockburn *et al.*, 1983). Most autecological information available for *A. flavipes* concerns populations in Queensland, New South Wales, the ACT and Western Australia (Marlow, 1961; Dickman, 1980; Smith, 1984; Goldingay and Denny, 1986; Wardell-Johnson, 1986; Dickman, 1991; Watt, 1997). Furthermore, few studies Australia-wide have recorded autecological or biogeographic data on *A. flavipes* in a fragmented landscape (Burnett, 1992; Laurance, 1994; Goosem and Marsh, 1997; Bentley *et al.*, 2000). This Chapter presents the autecological data collected for the study populations, which will provide base data for the effects of habitat fragmentation on *A. flavipes* investigated in the following Chapters.

3.1.1 Introducing *Antechinus flavipes*

General information about the genus *Antechinus* has been widely documented (see (Wakefield and Warneke, 1967; Lee and Cockburn, 1985a; Lee and Cockburn, 1985b). *Antechinus* spp. fall within the family Dasyuridae (subfamily Phascogalinae), within the order of Dasyuromorphia, the carnivorous marsupials (Robinson *et al.*, 2000). The genus is characterized by particular life history traits, described by Lee *et al.* (1982) and labeled as 'Strategy I'. This strategy entails a brief, "highly synchronized and predictably timed" (Lee and Cockburn, 1985b, p87) mating period, the complete mortality of males shortly after mating, and females that produce only one litter a year. Although the timing of the breeding season is mostly synchronous and predictable amongst local populations, it can vary between regions (Dickman, 1982). Nipple number also varies between regions (Cockburn *et al.*, 1983).

Antechinus flavipes is the most widespread of the *Antechinus* species (Strahan, 1995), occurring along the eastern seaboard from north-eastern Queensland (sub species *A. f. rubeculus*) to south-western Victoria (*A. f. flavipes*), and also occurring in the south-west of Western Australia (*A. f. leucogaster*). Its distribution in South Australia is restricted to the Mt Lofty Ranges and the south-eastern corner of the State.

Antechinus flavipes is known to occur in a wide range of habitats, from tropical vine forests, to swamps, stringybark forests and dry mulga country (Strahan, 1988). The species has previously been recorded in fragmented habitat (Burnett, 1992; Laurance, 1994; Laurance and Laurance, 1995; Goosem and Marsh, 1997; Bentley *et al.*, 2000), and has even been described as an edge-favouring species (Laurance, 1994).

Antechinus flavipes is predominantly a nocturnal species, but is also known to be crepuscular and active during the day (Wakefield and Warneke, 1967; Carthew, S. 2000, *pers. comm.*; Petersen, B. 2000, *pers. comm.*; Marchesan, D. 2001, *pers. obs.*). They are scansorial in their foraging behaviour, feeding largely on insects. They will also feed on nectar, small birds and house mice (*Mus musculus*) (Strahan, 1995).

From the little evidence available, the mating season for *A. flavipes* in south-eastern Australia appears to occur between late June and late July (Inns, 1976). The mating period is considered to be brief in all *Antechinus* species but the actual duration is not well known (Lee and Cockburn, 1985b). Marlow (1961) noted that the duration of oestrus in captive females ranged from two to seven days, while (Lee and Cockburn, 1985b) postulated that mating could occur over a two week period. Gestation lasts for approximately 31 days (Marlow, 1961). After parturition young are carried in the pouch for approximately five weeks, before being deposited into a nest for a further two to three months (Smith, 1984). The dispersal patterns for weaned juveniles have not previously been documented for southern populations of *A. flavipes*. Queensland populations of *A. f. rubeculus* do not display a defined juvenile dispersal strategy (Smith, 1984; Watt, 1997), which is evident in other *Antechinus* populations (Cockburn *et al.*, 1985). Life expectancy is 11 months for males and one to three years for females (Smith, 1984).

The aim of this Chapter is to present and discuss the autecological data collected during sampling sessions carried out in the patch, strip and forest sites within the main

study. Results will be used in the final discussion to complement the main outcomes of the study and provide insight into the main findings.

3.2 Materials and Methods

3.2.1 Sampling

All study sites, as described in Chapter 2, were included in the collection of autecological data. Data were recorded for animals trapped during 12 sampling sessions between July 2000 and November 2001. Details of the general trapping methodology are presented in Chapter 2. The sampling sessions included two breeding periods (July to September), two lactation periods (October to November) and one period of juvenile emergence and dispersal (December to March). Data collected included animal weights, sexual condition, nipple number and life history events. Radio-telemetry data collected during October and November sessions provided information on nesting sites.

3.2.2 Data Analysis

Parametric tests were used where data were normally distributed and of equal variance. Where this was not the case, non-parametric tests were used. A two-tailed *t*-test was used to examine weight differences between emerging male and female *A. flavipes*. Mann-Whitney U-tests were used to compare weights between adult males and females, first and second year females and between males in forest and non-forest sites (males in patches and strips were pooled into 'non-forest sites' as the site of residency of adult males amongst patches and strips could not be reliably ascertained). A Mann-Whitney U-test was also used to compare within-site distances moved by males and females. A Kruskal-Wallis test was used to examine differences in female weights between those in patch, strip and forest sites. Where a significant difference was found, the source of the difference was examined using a non-parametric multiple comparisons test suited to unequal sample sizes (Zar, 1999).

Population density (for each site) was estimated using resident females during the period of mating and lactation. A resident is defined here as an individual captured in

more than one trapping session at the same site. Females were used for density estimates as they proved to be the most spatially stable members of the population at this time, ensuring independence of sites sampled. Density was calculated as the number of residents per true (or effective) habitat area. To obtain a measure of the true habitat area, the mean home range area (0.27 ha, at the 95% kernel home range estimate; see Chapter 5) was converted to a square, with half the width added as a boundary to the edges of sites adjacent to paddocks or forest, but not pine (home range data showed that lactating females ranged, to some extent, outside of the trapping grid, into adjacent paddocks and forest, but not into pine). A measure of relative density, such as captures per 100 trap nights, was also considered. However, an analysis of the capture frequency (or 'trap happiness') of individuals revealed inconsistent trends, which would have led to biased results. The number of captures per 100 trap nights is used only as an indication of trap success. A two-tailed, paired *t*-test was used to examine differences in population densities between the 2000 and 2001 seasons.

Recapture rates were calculated to determine any extended occupancy of sites and assist in determining dispersal/migration movements. Recapture rate is defined here as the percentage of individuals (from the cohort born in 2000) that were captured during more than one trapping session.

3.3 Results

3.3.1 Captures

A total of 12,171 trap nights resulted in 864 captures of 113 female and 109 male *A. flavipes*. The overall trap success was 7%, with variation between sites and trapping sessions ranging from 0 to 29% (Table 3.1). Other species captured were the bush rat (*Rattus fuscipes*), the swamp rat (*R. lutreolus*), the introduced black rat (*R. rattus*) and the introduced house mouse (*Mus musculus*) (Table 3.2). *R. fuscipes* constituted the majority of the non-target captures, recording up to 38% trap success.

Population densities of *A. flavipes* ranged from 0 to 6 resident females per ha (Figure 3.1), with an overall mean density of 1.95 (\pm 0.3 s.e.) per ha. The variability in population density amongst sites and site types led to difficulty in comparing density

Table 3.1: Number of trap nights and trap success (percentage), for *Antechinus flavipes*, per site and trapping session. Dashes (-) indicate sites not trapped. Data for radio-tracking sessions are not given, as trapping patterns were not consistent with other sessions.

Site	No. trap nights	Trapping Session							
		Jul-00	Sep-00	Dec-00	Jan-01	Mar-01	Jul-01	Aug-01	Sep-01
1	1120	18	3	16	14	9	19	18	13
2	860	11	12	26	19	2	18	15	1
3	1120	7	6	5	3	1	15	21	8
4	820	5	5	3	4	3	3	-	-
5	720	8	7	6	7	6	15	28	15
6	920	-	-	3	4	1	1	0	0
7	240	-	-	2	0	0	0	-	-
8	880	-	-	5	0	2	8	14	5
9	540	0	-	5	3	2	8	-	-
10	660	7	3	-	-	-	2	11	7
11	640	3	3	-	-	-	3	3	1
12	420	6	4	-	-	-	4	3	0
Add	904	-	-	8	4	2	11	10	4
F1	400	-	8	13	-	18	14	29	13
F2	320	-	2	-	-	-	4	13	-
F3	312	-	-	-	-	-	5	19	-

Table 3.2: Number of trap nights and trap success (percentage), for non-target species, for each site. Data for radio-tracking sessions are not included, as trapping patterns were not consistent with other sessions.

Site	No. Trap Nights	<i>R. fuscipes</i>	<i>R. lutreolus</i>	<i>R. rattus</i>	<i>M. musculus</i>
1	1120	22	0	0	<1
2	860	30	0	0	2
3	1120	38	0	0	<1
4	820	11	0	0	1
5	720	11	1	0	<1
6	920	32	0	<1	2
7	240	22	0	2	9
8	880	33	0	0	<1
9	540	0	0	0	<1
10	660	30	0	0	<1
11	640	20	0	0	0
12	420	17	0	0	0
Add	904	10	0	0	<1
F1	400	32	0	0	0
F2	320	31	0	0	0
F3	312	10	0	0	0

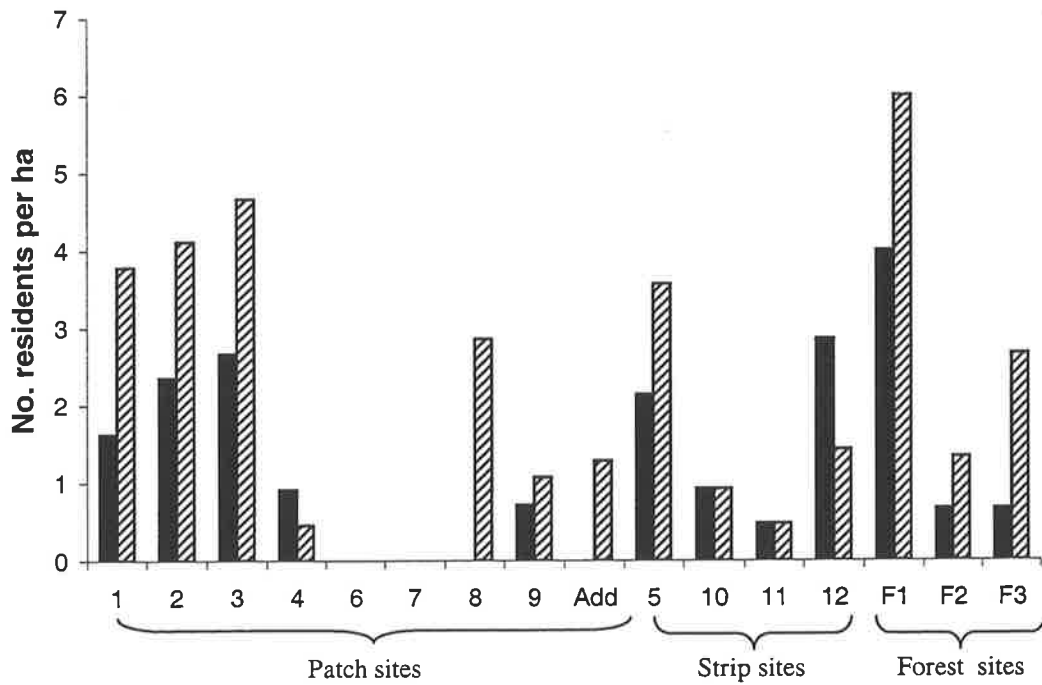


Figure 3.1: The number of resident females per ha for all sites during the 2000 season (filled) and 2001 season (hatched). Sites 6, 7, 8 and Add were not sampled during the 2000 season, while sites 6 and 7 had no females resident for the 2001 season.

between site types. Forest site F1 had consistently higher densities than all other sites. However, results show that 89% and 38% of patch and strip sites contained higher population densities than the remaining two forest sites, for the 2000 and 2001 seasons respectively. Further to this, correlations between population density and vegetation characteristics (outlined in Chapter 2) were investigated to allow a preliminary examination of the variability detected between sites. Results gave a moderate, positive association between density and the presence of tall *X. semiplana* (Pearson's $r = 0.69$), but no association with the presence of logs (Pearson's $r = -0.01$), presence of old trees (Pearson's $r = 0.03$) or understorey condition (Pearson's $r = 0.08$). There were also negative, moderate to weak associations found between population density and vegetation diversity and species richness (Pearson's $r = -0.67$ and -0.32 respectively). Population density results also show a significant increase in mean density from the 2000 (1.67 ± 0.3 s.e.) to the 2001 season (2.54 ± 0.5 s.e.) ($t = -2.56$, 11 d.f., $P = 0.027$). An increase was seen in eight of the 12 sites for which density was measured in both seasons.

3.3.2 Breeding

Mating times were estimated from likely parturition dates (Figure 3.2) using an estimated gestation period of 31 days (Marlow, 1961). From this, the mating period most likely commenced from mid July in most patch and strip sites, and from mid-late August in forest sites. This period also coincided with peak weights measured for males (see below), as expected (Inns, 1976; Smith, 1984). Post-mating deterioration was detected in some males by early August, with a loss of body weight and the occasional pendulous scrotum. Other symptoms described by Inns (1976) and Smith (1984) such as fur loss and ulcers were not observed. Most males had died by early September in both seasons, although three males were caught in early September 2000. These males were still in good condition showing no visible signs of post-mating deterioration, although weight loss could not be assessed as each male was captured on only one occasion.

Pouch development in females was first detected as early as August 2nd, and as late as September 2nd. Pouch development was not seen in three, presumably un-mated females in 2000. Most females sampled (112 out of 113) had 10 teats. The exception

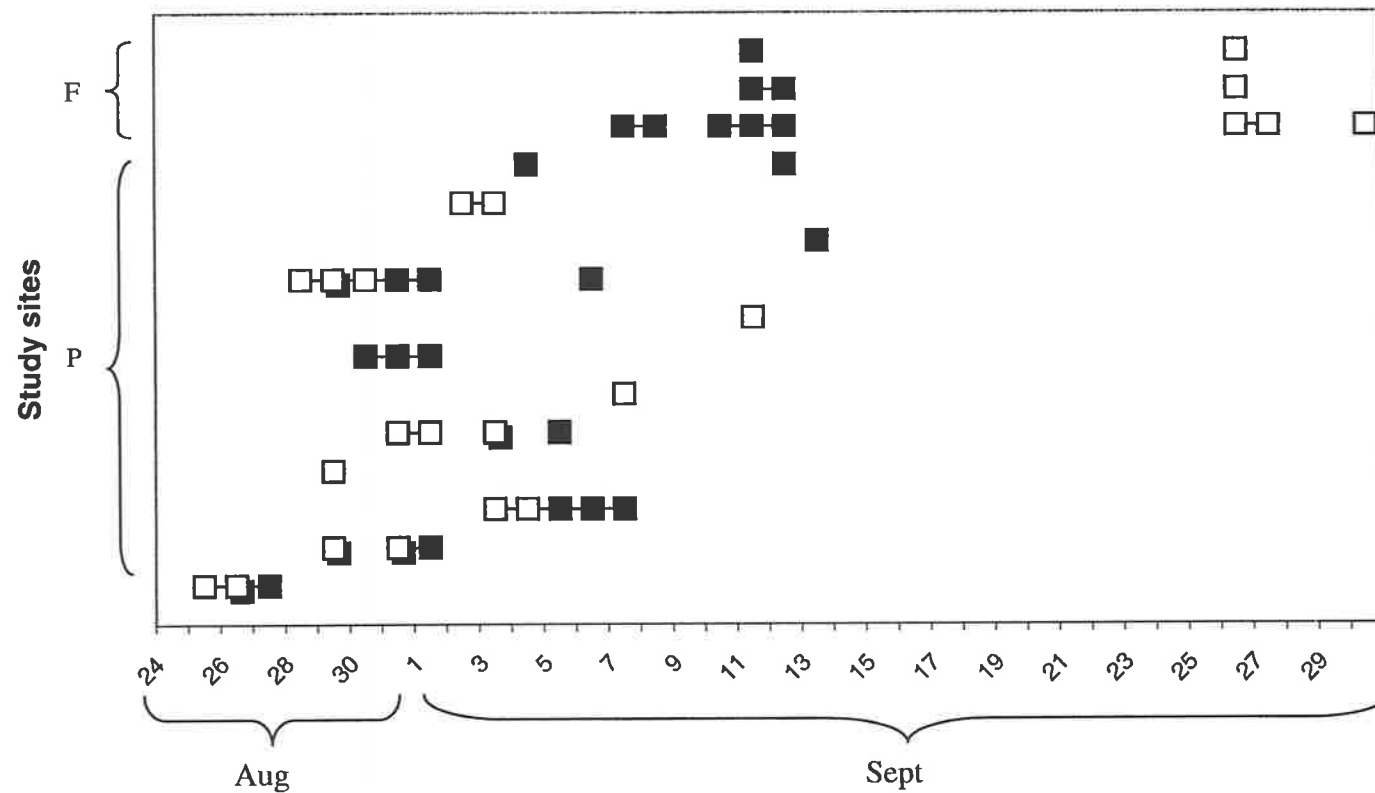


Figure 3.2: Estimated parturition dates for females within sites 1-12 (P) and forest sites F1-F3 (F). Boxes may represent one or more individuals. Open boxes represent parturition dates for the 2000 season, shadowed boxes for the 2000 and 2001 seasons and closed boxes for the 2001 season, where data were available.

was a female in a forest site that had eight teats. Females gave birth to between eight and 10 young (Plate 3.1) from approximately late August to late September. Parturition dates were estimated from the size of pouch young, with reference to Marlow (1961) and personal observations of young known to be one day old. Within a given site, in a given year, most females appeared to give birth within five days of each other, with the maximum range being approximately nine days (Figure 3.2). Parturition dates varied between sites, with the overall parturition period extending for about five weeks in 2000 and about three weeks in 2001. In 2000, females in the three forest sites gave birth in late September, whereas in 2001 they gave birth in early-mid September. Data from all other sites showed relatively consistent parturition times between seasons. By early to mid October, most females had nest young and were lactating.

Nest sites were identified during the radio-tracking sessions of October and November 2000 and 2001. A nest site was considered to be a site at which an individual had repeated observations during day-time tracking. Faecal material could also be seen at the base of some nest sites. Of the 20 females tracked, 15 were tracked to nest sites. The five other females were located at different positions during each day-time observation, and so a nest site could not be identified. The majority of nest sites (10 of 15) were located in the crowns of large *X. semiplana*, which had crown heights ranging from 0.95 to 3 m. The remaining nest sites were in tree hollows or stags of *E. obliqua* (Table 3.3). Exact locations of the nests within the *X. semiplana* or *E. obliqua* were not found. However, all appeared to be located at least 1 m above ground level, with the absence of any obvious nesting material below this height. Tracking also revealed day-time activity for 12 of the 20 females.

3.3.3 Juveniles

The timing of juvenile emergence differed slightly between sites due to the variation in mating and parturition times. The first juvenile was captured on the trapping grids in early November 2000 in Site 1, with the majority of juveniles emerging and captured in December (see Plate 3.2). After the initial emergence event in November/December, juvenile males were generally not detected on the trapping grids. This produced a low recapture rate in January and March 2001 (Figure 3.3). The low recapture rate for



Plate 3.1: Pouch young (photograph by K Hillyard, 2001)

Table 3.3: Number of *A. flavipes* nests located in *X. semiplana* (Xanth) and *E. obliqua* (Euc), mean crown height or tree height, mean girth and the number of nests located at the edge of a patch (E), between edge and middle (ME) and the middle of a patch (M). Girth for Xanth encompassed fronds hanging vertically about the trunk. E = 0 to $x/4$ m from edge, ME = between $x/4$ and $3x/4$ m from edge and M = between $3x/4$ and x m from edge, where x is the shortest distance (m) from an edge to the middle of the patch, in line with the nest. The last three columns were not relevant to five nests occurring in forest sites.

Nest Type	No. Nests	Mean Crown/Tree Height (\pm s.e.)	Mean Girth (\pm s.e.)	E	ME	M
Xanth	10	1.92m (\pm 0.18)	4.38m (\pm 0.31)	5	1	1
Euc	5	7.4m (\pm 1.09)	2.15m (\pm 0.13)	1	0	2



Plate 3.2: A juvenile *A. flavipes* captured in December 2000.

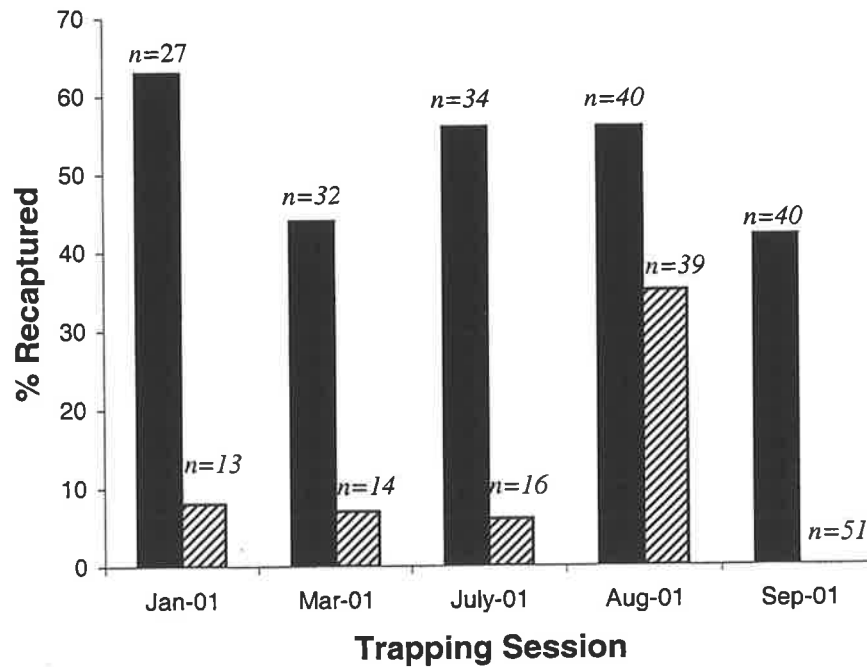


Figure 3.3: Percentage of female (filled) and male (hatched) *A. flavipes*, of the cohort born in 2000, recaptured after initial capture. Only sites sampled during all periods are given. The number of individuals (n) differs between sessions due to the increase in individuals captured over time.

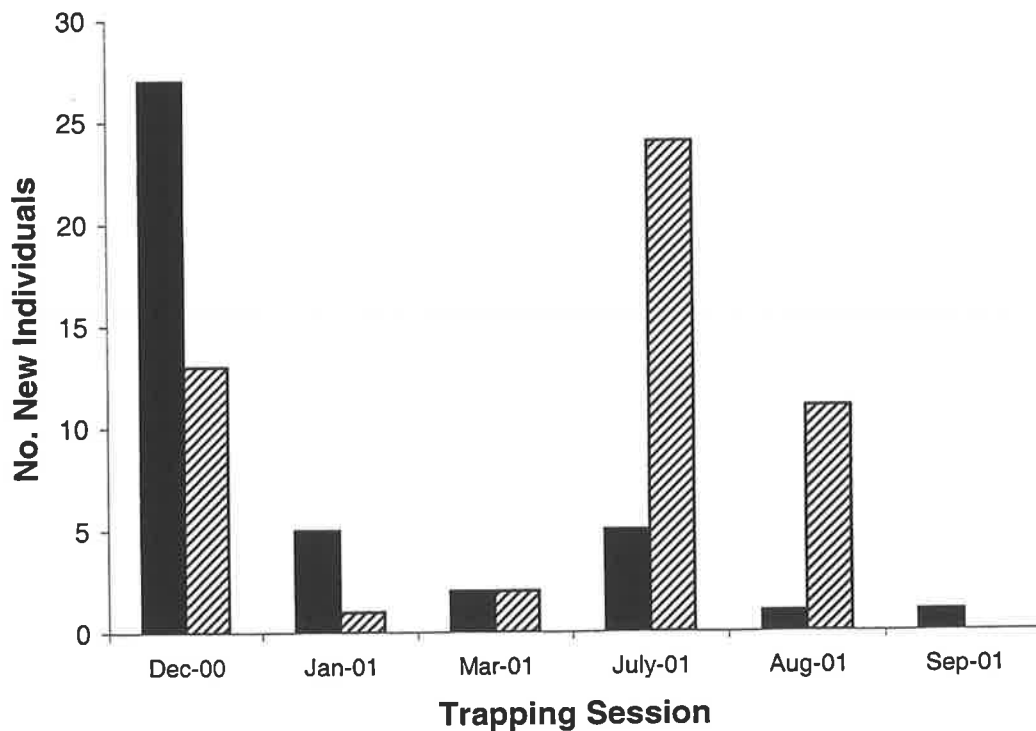


Figure 3.4: The number of new female (filled) and male (hatched) *A. flavipes* captured during each trapping session between the emergence of juveniles in December 2000 and the end of the mating season in September 2001.

males was not the result of a large number of new males occupying traps in January and March, as the capture of new males was also low during this time (Figure 3.4). Females behaved differently to males, with much higher recapture rates during January and March, and throughout other trapping sessions. Furthermore, all but two females were consistently captured in the same site as their initial capture. The capture of new females was also low during early 2001, and continued to be relatively low throughout the 2001 breeding season. The pattern of recapture rates for males and females is indicative of a male-biased, natal dispersal strategy. With such a strategy, males disperse from natal home ranges shortly after weaning, while females have a greater tendency to be philopatric, remaining closer to their natal areas.

3.3.4 Weights

Mean weights for male and first year female *A. flavipes* from the time of emergence to parturition are shown in Figure 3.5. Mean emergence weight was 13.34 g (± 0.28 s.e., $n=28$) for females and 12.82 g (± 0.29 s.e., $n=14$) for males, which increased to a mean adult weight (during mating) of 28.19 g (± 0.59 s.e., $n=58$) for females (first years only) and 54.7 g (± 1.15 s.e., $n=67$) for males. Although weight differences between the sexes were not significant at emergence (t -stat = -1.17, 40 d.f., $P = 0.186$), strong sexual dimorphism of weights was evident during mating ($U = 0$, $P < 0.001$). Post-mating weights tended to decrease for males with the onset of die-off, while female weights tended to increase with the production of young. The mean weight for second year females at the time of mating was 33.84 g (± 2.17 s.e., $n=9$), which was significantly higher than the mean for first years of 28.19 g ($U = 129.5$, $P = 0.015$).

Weight differences were also noted between males and females in different site types. Weights were significantly heavier for males in non-forest ($n=49$) than forest sites ($n=17$) ($U = 123.5$, $P < 0.001$) and significantly different for females between forest ($n=13$), patch ($n=36$) and strip ($n=9$) sites ($H = 15.66$, 2 d.f., $P < 0.001$; Table 3.4). An examination of the source of difference for females showed that mean weights of animals in patches and strips were greater than those in forest sites ($CV_{0.05} = 2.394$, $Q = 3.4$ and 2.4 respectively) but did not differ from each other ($CV_{0.05} = 2.394$, $Q = 0.76$) (Table 3.4).

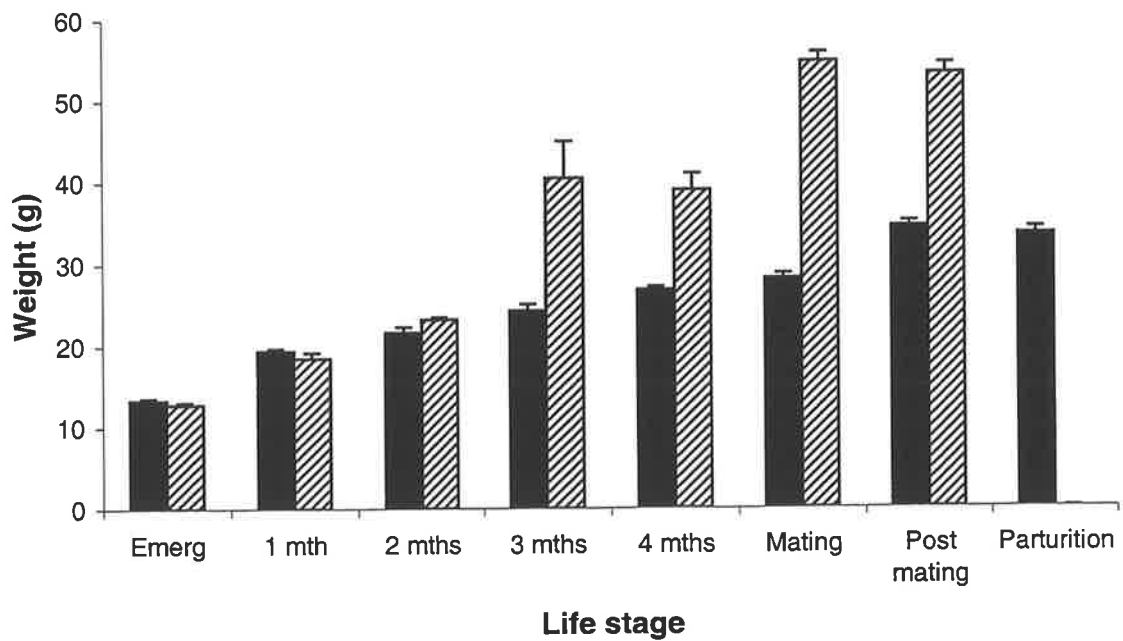


Figure 3.5: Mean weights for female (filled) and male (hatched) *A. flavipes*, from emergence (Emerg) to parturition. 1-4 mths indicate the number of months after emergence. Note, there is approximately a four-month gap between the 4mths and Mating categories. Female weights are for first year females only.

Table 3.4: Mean weights (\pm s.e.) for adult male and female *A. flavipes* in patch, strip and forest sites. *Male weights for patch and strips sites are combined. Letters indicate different sample groups from results of a *t*-test for males and a multiple comparisons test for females.

Site Type	Mean Weight Males	Mean Weight Females
Patch	57.8* (\pm 1.2) ^a	29.65 (\pm 0.7) ^a
Strip		28.14 (\pm 1.2) ^a
Forest	46.4 (\pm 1.6) ^b	24.15 (\pm 0.5) ^b

3.3.5 Movements

The mean distance moved between captures within a site was calculated for males and females for each trapping session (Figure 3.6). The overall mean distance moved by males was 82.67 m (± 11.51 s.e., $n=75$), which was significantly greater than that moved by females at 30.64 m (± 1.57 s.e., $n=265$) ($U=5649$, $P<0.001$). Movements tended to increase in distance during active times such as mating, particularly for males. Details of inter-patch and landscape movements are contained in Chapter 4, which show increases in movements for males, to up to 720 m. Home range movement information for lactating females is contained in Chapter 5, including movements into the landscape matrix.

3.4 Discussion

Autecological analysis revealed new information about *A. flavipes* in its southern range. This included the male-biased, natal dispersal strategy evidenced from recapture rates, which is the first record of such a strategy for this species. The only other published information on the dispersal behaviour of *A. flavipes* comes from studies conducted by Watt (1997) and Smith (1984) for north-eastern and southern Queensland populations, respectively. Results from these studies differ from that found here. Neither found a distinct juvenile dispersal period, with juvenile males remaining in their natal areas for up to six months. However, once dispersed, males moved further from their natal areas than females (Smith, 1984). The lack of data on the dispersal behaviour of *A. flavipes* indicates a gap in information for regions between southern Australia and Queensland.

Male-biased natal dispersal has also been recorded by Cockburn *et al.* (1985) for populations of *A. stuartii*. Cockburn *et al.* (1985) suggested that mothers initiated the dispersal, with the primary goal being inbreeding avoidance. A similar mechanism could exist amongst the *A. flavipes* populations of the present study, although this reasoning does not explain the difference in the timing of dispersal between regional populations of the species. Watt (1997) suggested that the extended stay of males in their natal area in north-eastern Queensland populations may allow refined kin recognition in the relatively low-density populations, thereby reducing the risk of inbreeding. Low population density may also call for a greater need for maternal

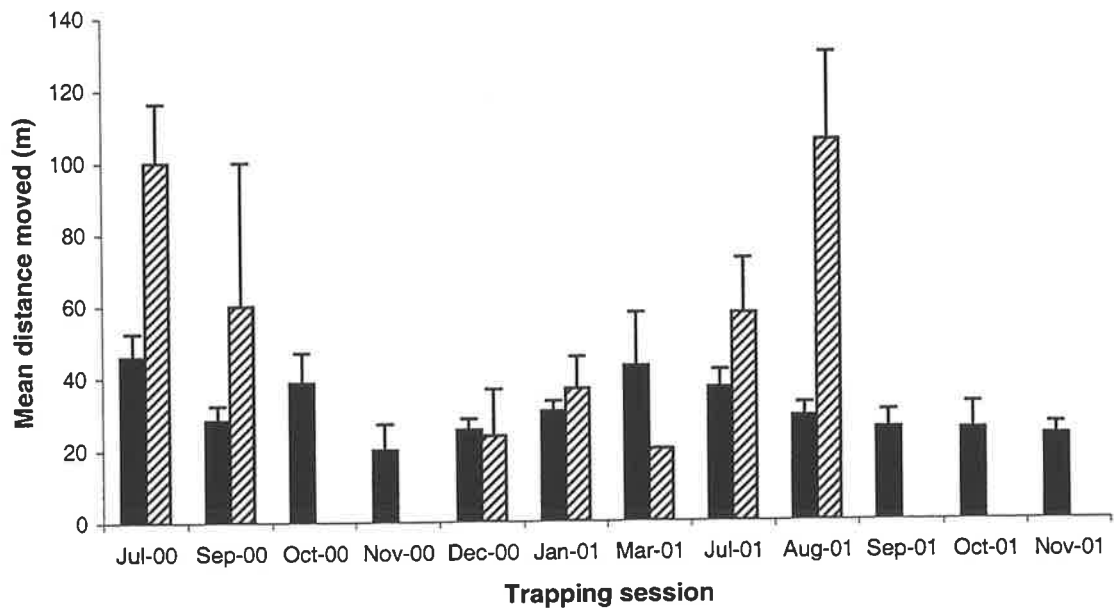


Figure 3.6: Mean distance moved between trap captures within a site for female (filled) and male (hatched) *A. flavipes*.

investment. As suggested by Lee *et al.* (1982), an increase in maternal investment can lead to higher juvenile survival. However, such responses by low-density populations would not explain the dispersal behaviour recorded by Smith (1984) where population densities were comparable to the present study (see below). Lee *et al.* (1982) postulated that differences found in the timing of dispersal between regional populations of *A. stuartii* may be the result of differences in the timing of insect declines; dispersal appeared to occur at the approximate time of declines in insect abundance. It is evident that breeding activity and parturition are timed so that young are nurtured during invertebrate peaks (Van Dyck, 1982). It may also follow that dispersal is triggered by declines in insect abundance, which would increase local competition for diminishing food resources. However, Cockburn *et al.* (1985) found that dispersal did not ultimately change the group size in a site (and therefore did not change competition levels), as unrelated males immigrated into the study area during the dispersal phase. The precise timing of immigration of non-related males could not be ascertained in the present study, but it would likely be towards the end of the dispersal phase. Overall, the dispersal behaviour for *A. flavipes* is considered to be male-biased, although the timing is delayed in northern regions. Inbreeding avoidance is considered a likely mechanism for dispersal, although the reason for differences in the timing between regions remains unclear.

Nesting habitat for *A. flavipes* has also not previously been documented. The majority of nest sites used by *A. flavipes* in the present study were in the crowns of *X. semiplana* with only 33% in tree cavities. In comparison, *A. stuartii* have been recorded nesting primarily in natural tree cavities, predominantly in stags (Cockburn and Lazenby-Cohen, 1992). It is not known whether there was a preference for *X. semiplana* over tree cavities in the present study, or a restriction in the availability of cavities forcing the animals to find an alternative. Greater sample sizes and an analysis of use versus availability are required to determine any preference for nesting habitat. Nest sites also tended to be located towards the edges of patches and strips (60% of nests; see Table 3.3). Again, this may have been a factor of nest-site availability in edges compared to interiors, but certainly indicates a tolerance of edge habitat by *A. flavipes*.

Breeding dates recorded in the present study were comparable with those found for populations of *A. flavipes* in tropical Queensland (Watt, 1997) and Canberra (Dickman,

1980). Peaks in male weights also corresponded with those recorded by Inns (1976) for captive individuals taken from the Mt Lofty Ranges in South Australia, indicating continuity of breeding times in this region. Populations in south-eastern Queensland, however, have been recorded breeding as early as mid June (Van Dyck, 1982) to as late as early September (Smith, 1984). Regional variation in breeding times has been documented elsewhere for *A. flavipes* and other *Antechinus* species (Lee *et al.*, 1977; Dickman, 1982; Van Dyck, 1982; Cockburn *et al.*, 1983; Watt, 1997). Such variation has been most successfully correlated with altitude and latitude (Dickman, 1982), as well as peaks in insect abundance (Lee *et al.*, 1977; Van Dyck, 1982). There have also been attempts to explain regional variations as products of sympatry and allopatry between *Antechinus* species (Dickman, 1982). In the present study, differences in breeding times were also evident between forest populations and those in patches and strips. The populations of *A. flavipes* that showed the greatest variation in breeding times were less than five kilometres apart and were not considered to be in different climatic or geographical regions. It is therefore unlikely that broadscale geographical or climatic factors, and associated differences in invertebrate peaks, were responsible for this variation. There may be specific local climatic or habitat factors that influenced the timing of mating within sites, although such factors are poorly understood (Van Dyck, 1982). Local conditions may have also influenced the variation in breeding times seen within forest sites between the 2000 to 2001 seasons. In these sites, parturition dates and mating times were estimated to be at least two weeks earlier in 2001 than 2000. Dickman (1980) detected a similar pattern in *A. flavipes* and *A. stuartii* populations in the ACT. Such variation in timing between seasons is not considered common (Lee *et al.*, 1977) and was only observed in the three forest sites of the present study.

Male die-off usually occurs before females give birth (Lee and Cockburn, 1985b). However, this was not the case for three males captured in September 2000 in the present study. Inns (1976) also found males present with post-parous females in a captive population. However, the constant supply of food and lower stress levels (compared to wild animals) was considered to be the cause of the absence of die-off in these animals (Inns, 1976). Smith (1984) noted a variable survival period for post-mating males in Queensland populations of *A. flavipes*, with one male still alive at the onset of parturition. However, no adult males were captured in the following two trapping sessions in this study. The males that survived until September in the present

study may have been spared the stress of mating, possibly being late weaners and/or unmated as adults.

Mean adult weights in the present study fell within the range of weights recorded for other *A. flavipes* populations (Table 3.5). Of note is the large difference between the present study animal weights and those measured by Inns (1976), which were also from populations in the Mt Lofty Ranges. However, values from Inns (1976) may be artificially high since these animals were held in captivity with a plentiful supply of food. Variability was present amongst mean weights of animals in the present study, with forest animals being significantly lighter than those from other sites. Similar differences in weight (and other factors) were detected by Bowers *et al.* (1996) in a study of meadow voles (*Microtus pennsylvanicus*) in an artificially fragmented landscape in Virginia, USA. Bowers *et al.* (1996) found that resident females in habitat fragments were larger in body size, had higher proportions of pregnancies and higher residency times than those in contiguous forest. Moreover, residents in patch edges had significantly larger home ranges, larger body sizes, longer residency times and higher pregnancy rates than those in patch interiors. The differences were attributed to an increase in habitat quality at edges (which was possibly caused by frequent mowing around fragments to maintain habitat isolation), as well a decrease in energy expenditure by edge animals due to fewer territorial boundaries to defend (Bowers and Dooley Jr, 1999). The patches and strips inhabited by *A. flavipes* in the present study are considered to consist mostly of 'edge' habitat, given the small size of the patches and suggestion that edge effects can reach up to 500 m or more from a forest edge (Andr n and Anglestam, 1988; Laurance, 1994; Sih *et al.*, 2000). Given this, habitat edges may have provided some of the positive edge effects suggested by Bowers and Dooley Jr (1999). Laurance (1994) noted *A. flavipes* as being 'edge-favouring', while Carthew and Kuback (2002) noted a significantly higher number of individual *A. flavipes* captured alongside (<5 m from) tracks 12-15 m wide than in forest sites located at least 200 m from any track. This possibly suggests that habitat edges provide more favourable conditions for *A. flavipes*. Although trapping grids in forest sites of the present study were located approximately 50 m from an edge and therefore possibly under the influence of edge effects, these sites had markedly lower edge exposure than patch or strip sites. A thorough analysis of the effects of habitat edges and interiors on

Table 3.5: Mean adult body weights reported for *A. flavipes* populations.

Population Location	Weight (g)		Source
	Male	Female	
NE Qld	68	39	Van Dyck (1982)
NE Qld	38	37	Watt (1997)
SE Qld	57	32	Smith (1984)
SE Qld	48	28	Van Dyck (1982)
Canberra, ACT	51	35	Dickman (1980)
Nangwarry, SA	45	32	Carthew and Kuback (2002)
Mt Lofty, SA	75	42	Inns (1976)
Mt Lofty, SA	55	28	Present study

Table 3.6: Trap success expressed as a percentage for studies on *A. flavipes*. Max indicates the maximum trap success recorded for a sampling period during a study.

Population Location	Trap Success (%)	Max	Source
NE Qld	3.3	~5	Watt (1997)
SE Qld	7.6	na	Smith (1984)
Central NSW	11	18	Goldingay and Denny (1986)
Canberra, ACT	2.5	na	Dickman (1980)
Nangwarry, SA	2.3	4.6	Carthew (1999)
Mt Lofty, SA	7	17	Present study

weights of *A. flavipes* was not possible in this study. Such analyses would be useful in future empirical studies on the effects of habitat fragmentation on local populations.

A comparison of trapping data with other populations of *A. flavipes* shows that the overall trap success for the present study fell within the range recorded for *A. flavipes* (Table 3.6). However, maximum success per trapping session (17%) and per site (29%) are possibly some of the highest figures recorded for this species. The mean density for resident females (1.95 per ha) was also comparable to those calculated by Smith (1984) using Known to be Alive estimates for populations in south-east Queensland. Smith (1984) found a mean density range of 1.3 to 2.52 females per ha over three seasons, with a maximum mean density of up to 3.4 per ha. Watt (1997), on the other hand found markedly lower densities for populations in north-eastern Queensland. Watt (1997) used the number of individuals per area trapped and calculated an overall population mean of 0.11 individuals per ha, with a range of 0.02 to 0.31. The only other density estimate found in the literature for southern populations of *A. flavipes* was 4.17 individuals per ha, for a population in Victoria studied by Reeckman in 1975 (cited in Dickman, 1980).

The examination of associations between population density and vegetation characteristics provided some insight into the factors influencing population density. The greatest association was found with the presence of tall *X. semiplana*. This is not surprising given that most *A. flavipes* nest sites were found in these plants. Other notable results included the negative, moderate to weak associations found between population density and species diversity and species richness respectively. These negative trends were likely to be a result of sites with higher diversity and species richness lacking in a presence of tall *X. semiplana* (see Table 2.2). However, all correlation results must be treated cautiously, as only simple vegetation surveys were used to obtain data and sample size (i.e. number of sites) was relatively low for reliable correlation estimates.

The increase in density detected in most sites between 2000 and 2001 suggests a higher survival rate of juveniles emerging in 2000 than in 1999. A possible explanation for this is the increase in rainfall recorded for 2000 (1011 mm) from 1999 (867 mm), potentially resulting in a more favourable growing season for plants and invertebrates.

Such conditions would increase the food resources available for juvenile *A. flavipes* emerging in the summer of 2000/2001. Investigation into the relationships between seasonal conditions and invertebrate abundance and availability for *A. flavipes* would be required to further this discussion.

Results of autecological analysis have revealed inter- and intra-population variations, which have usually been noted on a larger geographical and regional scale for *Antechinus*. These results may be simply due to natural variations within and between populations, or they may prove to be valuable in explaining the effects of habitat fragmentation on these populations.

CHAPTER 4: PRESENCE, BREEDING ACTIVITY AND INTER-PATCH MOVEMENTS

4.1 Introduction

In many areas of Australia, small mammal species have been detected in small habitat remnants, set in potentially inhospitable agricultural landscapes. However, little is understood regarding their ability to persist and breed in small remnants, their mobility between remnants, or the suitability of the landscape matrix for movement. Examining the movement of animals between remnants is of particular importance in determining their ability to form a metapopulation amongst local populations. Such a process has been recognised as a key step in alleviating the potential effects of habitat fragmentation (Merriam, 1991), described in Chapter 2. Movement between patches will be influenced by a species' mobility as well as the type of matrix surrounding the remnants (Bennett, 1998). The matrix often consists of agricultural land, which can present a barrier to the movement of animals (Bennett, 1990, Goosem and Marsh, 1997). However, this is not always the case. Species recorded using agricultural land for movement and/or habitat in Australia include the euro (*Macropus robustus*) in the wheatbelt of Western Australia (Arnold *et al.*, 1993), the bush rat (*Rattus fuscipes*) in pine plantations of north-eastern Victoria and New South Wales (Barnett *et al.*, 1977; Suckling and Heislars, 1978) and the sugar glider (*Petaurus breviceps*) at one site in open paddock in Willung, Victoria (Suckling, 1984).

This Chapter presents results of an investigation into the population structure and maintenance of *A. flavipes* in the fragmented landscape studied. This was carried out by examining: (1) their presence and breeding activity in remnant patches and strips, (2) their ability to move amongst patches and strips and (3) the possible types of matrix used for movement.

4.2 Materials and Methods

Thirteen small patch and strip sites (sites 1 to 12 and Add), as described in Chapter 2, were selected for this phase of the study (data for forest sites are excluded). Sites 1 to

12 were used to examine the presence and breeding activity of *A. flavipes*. Presence is defined here as the occurrence of at least one capture of *A. flavipes*, and breeding activity is defined as the presence of pregnant females, post-parous females, or newly emerged juveniles. Sites 1 to 9 and Add (ten sites in total) were used to examine inter-patch movements. The 10 sites were in three distinct groups (see Figure 2.1). Distances between groups ranged from 2.7 km to 5.7 km, with the distances between sites within a group ranging from 80 m to 700 m. This latter range is within known distances moved by *A. flavipes* (Carthew, 1999). The landscape matrix between study sites included exotic pine plantation, open paddock, swampy (largely treeless) waterway, and narrow, linear strips of native vegetation.

Sampling was carried out as described in Chapter 2. During sampling in December 2000, every effort was made to capture newly emerged juveniles. However, this proved difficult as the juvenile emergence time differed slightly between sites, and juvenile males, in particular, did not stay in the area of emergence for long. Examination of the weights and the sex ratio of juveniles captured (assuming a 1:1 sex ratio of weaned young), indicated that the time of emergence was sampled for five of the 10 sites, sampled towards the end of emergence for two sites and missed for three sites.

Sites were sampled for three or four consecutive nights during the breeding season and for five consecutive nights during juvenile emergence and dispersal (weather permitting). All sites were sampled within a 12-day period during each session.

4.3 Results

From the thirteen sites surveyed, a total of 646 captures of 172 individual *A. flavipes* were made from 9,406 trap nights. Figure 4.1 shows the number of individual *A. flavipes* of each age class captured during each trapping session. *Antechinus flavipes* was detected in all sites surveyed. The species was detected in 10 of the 12 sites surveyed for presence, in 80-100% of trapping sessions. At the remaining two sites (6 and 7), *A. flavipes* was detected in 65% and 25% of trapping sessions respectively. Breeding activity was also observed in all sites, with nine of the 12 showing breeding activity in both the 2000 and 2001 breeding seasons. Of the three sites for which

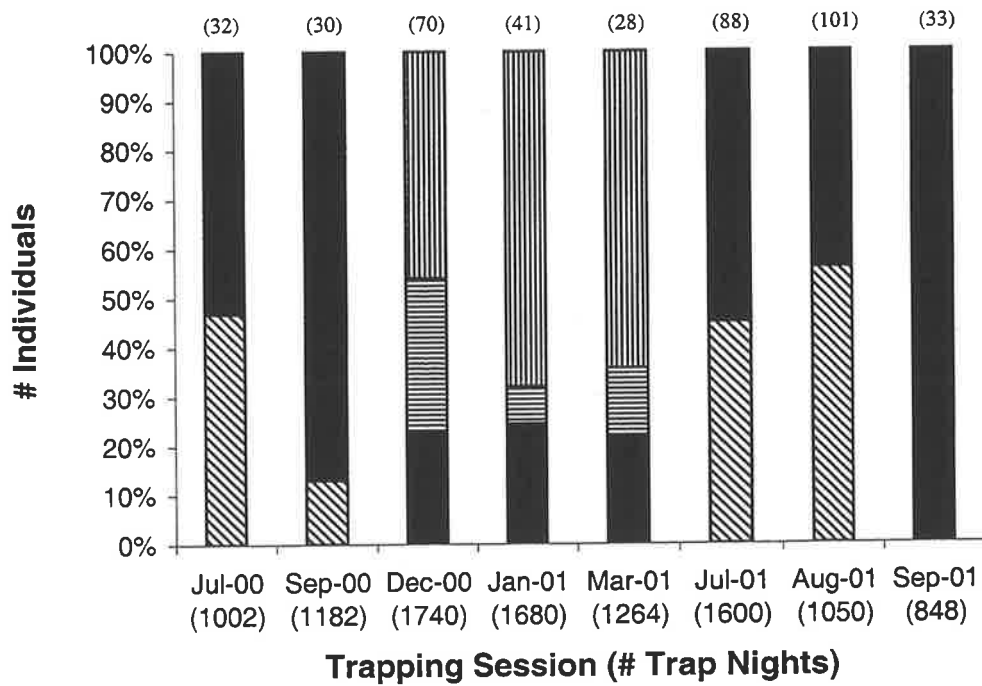


Figure 4.1: The percentage of adult female (filled), adult male (diagonal hatch), juvenile female (vertical hatch) and juvenile male (horizontal hatch) *A. flavipes* captured during each trapping session. Numbers of individuals are given in parentheses above columns. Numbers of trap nights are given in parentheses for each session. Note that July, August and September sessions in 2001 included three sites not surveyed during other sessions.

breeding activity was detected in only one season, two sites did not have trapping completed for the 2001 season, due to fox disturbance (Table 4.1).

Sampling conducted to investigate inter-patch movements during juvenile emergence and dispersal and during the 2001 mating season resulted in the capture of 50 females and 58 males. From December 2000 to September 2001, a total of 287 recaptures resulted in the detection of nine inter-patch movements, all within site groups. One movement was made by a juvenile female between December 2000 and January 2001, with the remainder by adult males during the 2001 mating season. The mean linear distance moved between sites was 458 m (± 75.3 s.e.), with a range of 80 m to 720 m (Table 4.2). The actual landscape types traversed by individuals are not known. However all individuals had to traverse habitat other than native vegetation, such as clearings, pine plantation or roads, in order to move between remnants. Clearings ranged from 6 m to 30 m in width, roads were 12 m to 32 m wide, open paddock areas were 200 m to 320 m wide and pine plantations were between 200 m and 500 m wide.

Other movements through the landscape can be inferred from recapture results. The overall recapture rate was 81.6% for females and 35.4% for males. Recapture rates over time were relatively consistent and markedly higher for females than for males, particularly during the time of emergence and dispersal (see Figure 3.3). In August 2001 the recapture rate for males increased markedly. However, this was due to the influx of new males captured in July 2001 (see Figure 3.4), rather than recaptures of males marked during emergence in December 2000. Recapture patterns imply male-biased dispersal of juveniles out of their natal patches (those in which they were first marked) into the surrounding landscape, and the movement of new males through the landscape into study patches.

4.4 Discussion

It is evident from the literature that a reduction in habitat area and the associated effects on habitat structure can result in a reduction in species diversity, abundance or changes in composition (Bennett, 1987; Saunders, 1990; Dunstan and Fox, 1996; Bolger *et al.* 1997; Estades and Temple, 1999; Knight and Fox, 2000). However,

Table 4.1: Number of trapping sessions, percentage of sessions detecting presence of *A. flavipes* and number of seasons detecting breeding activity, per site. * Trapping during the 2001 breeding season not completed due to fox disturbance.

Site	No. trapping sessions	% sessions detected	No. seasons with breeding activity
1	8	100	2
2	8	100	2
3	8	100	2
4	6	85	2
5	6	100	2
6	6	65	1
7	4	25	1*
8	6	100	2
9	5	80	1*
10	5	80	2
11	5	100	2
12	5	80	2

Table 4.2: Details of inter-patch movements detected via capture-recapture. * Same individual.

Sex	Time	Site		Linear Dist. (m)	Available landscape matrix
		From	To		
Female	Dispersal	2	6	280	Exotic pine, clearing, road
Male	Mating	7	6	80	Clearing, road
Male	Mating	1	8	320	Exotic pine, paddock
Male	Mating*	1	8	260	Exotic pine, paddock
Male	Mating	1	Add	660	Exotic pine, swampy waterway
Male	Mating*	8	5	640	Exotic pine, native vegetation
Male	Mating	8	3	720	Exotic pine, swampy waterway
Male	Mating	1	Add	620	Exotic pine, native vegetation
Male	Mating	8	3	540	Exotic pine, swampy waterway

despite the chronic reduction in habitat in the study area, the small size of the study patches and the number of years since land clearance, *A. flavipes* was detected in all 12 study sites, with most sites detecting *A. flavipes* in 80-100% of trapping sessions. Furthermore, up to 89% of patch and strip sites contained higher population densities than two of the three forest sites (see Chapter 3). Breeding activity was also detected in all study sites during at least one breeding season, demonstrating that the patches served as breeding habitat and not just as refuge sites for transient individuals.

The presence of *A. flavipes* in small habitat patches has been recorded in other studies. For example, Laurance (1994) and Bentley *et al.* (2000) recorded *A. flavipes* in remnant patches between 1.4 and 26 ha in north-eastern Queensland and south-eastern Queensland respectively. The patch size threshold for *A. flavipes* appears to be quite small, with presence detected in discrete habitat patches 1.3 ha in size. As a comparison, *A. stuartii* has been detected in remnants as small as 1 ha (Dunstan and Fox, 1996), although it was absent from remnants ranging from 1 to 10 ha in size where *A. flavipes* was present (Bentley *et al.*, 2000). It is also considered more likely to be detected in larger remnants than smaller (Lindenmayer *et al.*, 1999).

The detection of *A. flavipes* in linear strips has also been previously reported. Van der Ree (2001, *pers. comm.*) detected a breeding population of *A. flavipes* in linear roadside vegetation in Victoria. The roadside strips were 20 to 40 m wide and adjacent to open paddock and regenerating remnant woodland. Linear habitat is not always suitable for small mammal species. Parameters such as habitat width may affect some species, such as root voles (*Microtus oeconomus*; Andreassen *et al.*, 1996), while foraging behaviour or social structure may deter others (such as Leadbeater's possum, *Gymnobelideus leadbeateri*) from inhabiting linear patches despite the presence of favourable habitat within them (Lindenmayer *et al.*, 1993). In the present study, the effect of strip width, patch size and other factors associated with small remnants could not be tested. However, survey results suggest that patches of at least 1.3 ha in size, strips of 20 to 30 m in width with a high degree of connectivity to larger fragments, all of appropriate habitat structure, are suitable for occupation and breeding by *A. flavipes*.

The differences in presence and breeding activity observed between seasons and sampling sessions for some sites may have been due to a number of factors, including dispersal and mortality. Also, the sites in which *A. flavipes* was not detected in all sessions contained relatively small resident populations (1 to 2 resident females), potentially lowering the probability of detection (in the absence of males) compared to more densely populated sites. Trap shyness may have played a part, as there were times when an individual remained un-detected for a number of sampling sessions (up to four) before entering the traps again. Breeding activity remained relatively constant between seasons for all sites except one (excluding the two sites where fox disturbance interrupted sampling). At this site, a total of three females and three males were captured, however none were recaptured during the mating season of 2001. The lack of detection of breeding activity in this site may have been due to a change in habitat conditions or the death or dispersal of animals before the onset of the mating season.

There were a number of recorded movements between remnants by *A. flavipes* in this landscape, with one movement made by a juvenile female during dispersal and eight by adult males during the 2001 mating season. These results suggest that *A. flavipes* is physically capable of entering the matrix and forming a metapopulation in this landscape. Although only one inter-patch movement was detected during the juvenile dispersal phase, results of recapture rates of juveniles suggest that there was an almost complete movement of males out of their natal patches at this time. Only three of the 23 males marked during juvenile dispersal were recaptured. This low recapture rate for males was not due to new males occupying traps at this time, as new male captures were also low. There is also no evidence to suggest, from this or other studies conducted on *Antechinus* species, that males avoid traps during times of dispersal (Smith, 1984; Cockburn *et al.*, 1985; Goldingay and Denny, 1986; Watt, 1997). It is suggested, then, that the majority of juvenile males dispersed into the surrounding matrix, and either settled in other areas of suitable habitat outside the study area, or died in the attempt. Given that a high number of unmarked males were detected in the study patches during the 2001 breeding season, which most likely originating from areas outside of the study sites, it is likely that at least a proportion of animals marked at emergence made successful movements to other habitat patches. It may be possible that they settled within study sites but remained undetected until the breeding season,

although this is considered unlikely for the majority of males. Nearby pine plantations may have harboured individuals, as found by Barnett *et al.* (1977) and Suckling and Heislars (1978) for *A. stuartii* and *A. swainsonii*. These populations were found in stands of young pine (8 to 22 yrs) with their persistence considered to be dependent on the presence of suitable vegetation along creeks. In the present study, the majority of pine plantations surrounding the sites were between 10 and 28 years old, with some containing creeks lined with native vegetation. Pine areas were not sampled during this study due to logistical constraints, although sampling in these areas would prove valuable in determining whether or not pine plantations were able to support populations of *A. flavipes*.

According to Laurance (1994), a species that can tolerate the matrix will be more successful in persisting in habitat fragments than those species that avoid the matrix. While agricultural landscapes can prove to be a barrier to the movement of some species, results from this study suggest that *A. flavipes* was able to tolerate and use the agricultural matrix for movement between patches, and not just during times of juvenile dispersal. Laurance (1994) also suggested that *A. flavipes* was able to move through and tolerate an agricultural matrix to survive in fragments of rainforest in the southern Atherton Tableland of north-eastern Queensland. Similarly, Dickman (1991) reported that *A. flavipes* had been recorded foraging in an open paddock, some 100m from the nearest tree. Much of the matrix between sites in the present study consisted of exotic pine plantations, which was traversed by some *A. flavipes*. Pine is also suggested to provide connectivity between populations of *A. stuartii* in fragmented habitat in New South Wales (Lindenmayer *et al.*, 1999), where the probability of occurrence was found not to differ between remnants and forests. It is suggested that the ability of *A. flavipes* to move through an agricultural matrix is due to the species' tolerance of the matrix, the dispersal behaviour exhibited (at least) by males, and the distances between patches (i.e. less than 700 m) being within the movement capability of the species. An increase in distance between sites may affect the mobility of the species and/or their survival during inter-patch movements.

The movement behaviour shown by male and female *A. flavipes* in this study has implications for the persistence of the species in fragmented habitat. Most importantly, movement between patches allows the formation of a metapopulation,

permitting the species to maintain the demographic parameters of local populations. Immigrants entering local populations provide population replenishment, otherwise known as the 'rescue effect' (Brown and Kodric-Brown, 1977). Such replenishment appeared to be largely carried out by males leading up to the mating period, which was vital as there was a distinct lack of male captures immediately after the dispersal events of December and January. The contribution to demographic replenishment by females would be limited, due to their philopatric nature (evidenced from recapture patterns). Although one female dispersed up to 280m from its natal patch, the overall frequency of movements and the maximum distance moved during dispersal is likely to be less for females than for males, as shown in other studies of *Antechinus* (Smith, 1984; Cockburn *et al.*, 1985). It is suggested that the primary role of females in the demographic maintenance of local populations is to persist and breed in the patches. The complete disappearance of females from a remnant may mean the permanent extinction of a breeding population in that remnant, unless a source of migrants is within the movement distance for females.

It appears that the metapopulation size and the corresponding habitat area utilised has been adequate to ensure the persistence of *A. flavipes* in this landscape over the past 50 years. Assuming that the total metapopulation habitat area does not decrease further, it is suggested that the long-term persistence of *A. flavipes* in this fragmented landscape will firstly be determined by the ability of females to maintain a presence in small patches and secondly by the ability of unrelated males to move into the patches to breed. In turn, these factors will be affected by the maintenance of distances between sites and the maintenance of the landscape matrix. Results of this study illustrate the importance of recognising the occurrence of metapopulations in a fragmented landscape. When forming management goals for species in fragmented landscapes, the total area that the metapopulation inhabits should be considered, not just individual habitat patches. Similarly, when considering population size for viability analysis, the total metapopulation size should be considered as well as the degree of interaction between patches.

CHAPTER 5: HOME RANGE OF *Antechinus flavipes* IN A FRAGMENTED LANDSCAPE.

5.1 Introduction

Small areas of habitat, such as forest fragments, can be limiting with regards to food resources, nesting resources and territorial or home range space. Resource limitations, combined with a species' behavioural or social requirements can affect their persistence in small areas of habitat. Lindenmayer *et al.* (1993) suggested that the social structure, home range size and shape requirements and foraging behaviour of Leadbeater's possum (*Gymnobelideus leadbeateri*) contributed to the absence of this species from linear areas of high quality habitat. These linear fragments did not appear to meet their behavioural and resource needs and the species did not appear able to change its behaviour to suit. Where home range space becomes limiting, an adaptation of home range behaviour may be required to allow successful habitation.

The study patches in the southern Mt Lofty Ranges were shown to contain functioning populations of *Antechinus flavipes*, which were able to breed and wean young successfully, irrespective of the size and shape of the patch (see Chapter 4). Chapter 5 describes the use of space by *A. flavipes* in six study sites. In particular, it examines evidence of any change in home range behaviour in response to habitat restriction. The home range behaviour of lactating females was chosen for this study as lactation is considered one of the most resource intensive periods of an antechinus' life cycle (Lee and Cockburn, 1985b). Therefore, behaviour by lactating females is likely to be sensitive to habitat restriction. The main aims of this study were to (1) examine home range area of lactating females in restricted (patches and strips) and unrestricted (forest) sites, (2) determine the extent of home range overlap between lactating females in patches/strips and in forests, (3) detect the use of the surrounding landscape as a resource supplement and (4) conduct a preliminary investigation into home range area in relation to true habitat area and local population density. If *A. flavipes* is able to respond to habitat restriction, the expected outcome will be a decrease in home range area and/or an increase in home range overlap.

5.2 Materials and Methods

Home range was investigated using radio-telemetry (radio-tracking). Tracking sessions occurred in October and November 2000 with three sites included in each session, and then repeated in October and November 2001. Reliability of results from the 2000 sessions was limited due to differences in reproductive stages of tracked females and a lack of accuracy testing of observations. These sessions were treated as pilots. The bulk of this Chapter describes the results from the 2001 sessions. The timing of tracking for the 2001 sessions ensured that target females at each site were lactating with nest young.

Home range has been defined as the area most frequently used by an animal (Andreassen *et al.*, 1993), implying the exclusion of non-typical movements. The concept of home range is usually used to infer the total area traversed during an animal's lifetime. However, it is recognized that animals can vary their home range size and location over time, requiring refinement of the definition to make it relevant to a particular project (White and Garrott, 1990; Andreassen *et al.*, 1993; Otis and White, 1999). Considering this, home range is defined here as the area most frequently used by lactating females with nest young, during evening and night-time activity.

5.2.1 Site Selection

Sites 1 and 2 (discrete patches) and 5 and 10 (linear strips) were selected as representative restricted habitat sites, and F1 and F3 as unrestricted forest sites (see Figure 2.1). The linear strips were directly connected to larger remnants of native vegetation at one end and surrounded on either side by exotic pine plantation and/or open paddock. The discrete patches were surrounded by pine plantations on three sides and open paddock on the fourth. Sampling areas for the two sites in contiguous forest began at least 50 m in from an edge of the forest and 15 m in from a road or track clearing that dissected the forest. In 2001, the six sites each contained between four and nine lactating females, of which three (per site) would be fitted with transmitters for examination of home range.

5.2.2 Trapping

Trapping was used to trap target females, as described in Chapter 2. Each site contained two lines of seven trapping stations, with two traps set per station. Traps were set for three nights at the beginning of each session. Captured females were identified, marked by clipping fur, weighed and fitted (where possible) with single stage, LT2 radio transmitters (Titley's Electronics, NSW). Transmitters were attached by a nylon cable tie around the neck and weighed approximately 1 g. All females fitted with transmitters weighed in excess of 20 g. Transmitters had a battery life of up to 10 days and a signal range of up to 100 m, depending on terrain. Traps were also set for up to four nights at the end of each session to recover transmitters.

5.2.3 Triangulation

Radio-tracking observations were made via triangulation, by two observers using 3-pronged hand-held Yagi antennas and RX5 and Biotel RX3 hand-held receivers (Biotelemetry Tracking, Australia). Observations were made from fixed tracking stations, with seven to 13 tracking stations established at each site in cleared areas adjacent to the sites, or within forest adjacent to trap lines (forest sites only). Stations were 50 m apart and at least 5 m from the edge of the vegetation (except in forest sites). A spread of 50 m was estimated as the optimum distance between stations to obtain a right angle at the intersection of observers' bearings when a target animal was at the centre of the patch. This was done in an effort to minimize location error (Springer, 1979), although it should be noted that not all bearings were expected to result in a right angle intersection. When taking observations, the choice of stations ensured that the target animal was between observers. Bearings to the target animal were taken simultaneously. Observations were taken between the hours of 4.30 pm and 3.00 am, which comprised part of the observed and documented activity period for *A. flavipes* (Wardell-Johnson, 1986; Marchesan, D. 2001, *pers. obs*). Day-time locations were also taken for each animal to determine nesting sites and to detect any day-time activity.

Sampling error and bias associated with the triangulation method of radio-tracking should be calculated (Springer, 1979; Saltz, 1985), particularly when working in small

habitat patches (Saltz, 1994). Bias, precision (pooled standard deviation of sampling variance) and location error were estimated by conducting test observations within three of the sites used in the study. A transmitter was set in a total of 12 different locations amongst the three sites with each observer using a total of six stations for the 12 locations. Maximum distances between observers and transmitter ranged from 35 m to 65 m (mean 46.19, \pm 0.9 s.e., $n=12$). Each location was measured twice by each observer, giving a total of 48 observations. After each location was observed, the true bearing was taken. Mean bias (in degrees) was calculated from all observations, while the mean precision angle was calculated from the 24 sets of repeated measures. The mean location error was calculated on the 12 true bearings, using a method adapted from MacDonald and Amlaner (1980):

$$\text{Location error} = \max(d1, d2) * \tan \alpha$$

where $\max(d1, d2)$ is the longest of the two observation distances between observer and the target and α is the precision angle (in this case, mean precision angle). This method differs from that preferred by some (eg. Springer, 1979; Saltz, 1985) but does provide a simplified and relatively conservative measure of error.

5.2.4 Analysis

Home range analysis was conducted using ArcView GIS 3.1 (Environmental Systems Research Institute, 1992) in conjunction with the Animal Movement SA v 2.04 extension (Hooge *et al.*, 1999). The Kernel Home Range (KHR) estimator was used to determine home range area, shape and location. The KHR estimator is a non-parametric model with no assumptions regarding the shape and distribution of the home range. It is based on that described by Worton (1989), with the option of an ad hoc calculation or least squares cross validation (LSCV) for the smoothing parameter based on Silverman (1986). The LSCV was used in this study to obtain the smoothing parameter, as recommended by Seaman *et al* (1999) and Hooge *et al* (1999). Another popular home range estimator is the minimum convex polygon (MCP). This estimator calculates a minimum area polygon by connecting the outer-most observation locations (White and Garrott, 1990). An MCP output produces a total area figure but does not produce an

estimate of the frequency, or density of use; it assumes equal use over all space within the polygon. The MCP estimator is also highly sensitive to sample number (White and Garrott, 1990), which together with the limits in output made it inadequate for home range analysis in this study.

The KHR output consists of user specified, utilization-distribution isopleths. These are contours that display the probability of density of locations (Worton, 1989). Specified isopleths were the 95%, 85%, 75%, 65%, 55%, 45% and 35% probability contours. Home range area was taken from the 95% isopleth. Core home range area was estimated by plotting the isopleth area against isopleth value and identifying the point at which there is a relatively significant change in the slope of the curve. The corresponding isopleth was read as the estimated boundary of the home range core (Harris *et al.*, 1990). Home range areas were compared between individuals and tested for significant difference between site types (ie. restricted and unrestricted), using a *t*-test. Home range area was also plotted against a measure of population density using true habitat area (as described in Chapter 3). Results from the KHR were also used to examine the degree of home range overlap within sites and between site types. Trap capture locations for other lactating females resident in the study sites were plotted onto the home range estimates to supplement home range overlap data. These trap capture data were taken from the radio-tracking sessions and for the September 2001 round of trapping.

A rectified aerial photograph was overlaid onto home range plots to determine if home range areas or observations extended out of the study patches and into the surrounding matrix. The calculated location error was taken into consideration when analysing these results.

5.2.5 *Number of Observations*

The number of observations recorded per animal should be maximised to decrease bias and variance, allowing a more accurate analysis of home range (Seaman *et al.*, 1999). Observation number as well as the duration of the sampling period also should be adequate to allow an animal to express its entire home range (Otis and White, 1999). Seaman *et al.* (1999) proposed a minimum of 30, but preferably ≥ 50 observations be

obtained per animal, for home range studies using kernel estimators. Given this, a target of 40 to 50 observations was set for this study. The extent of home range expression was then tested *post hoc* by graphing home range area for the 95% utilization distribution isopleth against observation number (Harris *et al.*, 1990; Comport *et al.*, 1996; Hansteen *et al.*, 1997; Otis and White, 1999). The presence of an asymptote in the curve would suggest adequate home range expression. Results of such an analysis will not differentiate between the effects of sample duration and observation number, but it will give some insight into the extent of home range expression.

5.2.6 Autocorrelation

Autocorrelation occurs when the time between observations is too short to ensure that “an animal’s position...at time $t + k$ is not a function of its position at time t ”, resulting in dependence amongst successive observations (Swihart and Slade, 1985, p. 1176). While autocorrelation generally increases with a decrease in time interval between observations, cyclic behaviour can also produce autocorrelated data (Swihart and Slade, 1985). It has been argued that a lack of independence amongst successive observations violates the assumptions of statistical home range estimators (Swihart and Slade, 1985; Worton, 1989). However, a more recent consensus is that non-parametric home range estimators (such as kernel estimators) can be used successfully on successive, dependent observations (Swihart and Slade, 1997; DeSolla *et al.*, 1999; Otis and White, 1999). This is true provided that (1) the total sampling period is long enough for an individual to express its entire home range, (2) the time interval between observations remains relatively constant and (3) an individual’s home range is treated as one experimental unit. Given all of this, sampling was designed to maintain a high number of observations and allow the use of potentially dependent, successive data. The total sampling period was restricted by the battery life of the transmitters to seven nights (allowing for three nights of trapping to collar animals). However, the interval between observations remained relatively constant at 1-1½ hours, which was the time taken to complete one round of tracking. Each home range was treated as one experimental unit when comparing home ranges over all sites, and each site was treated as one experimental unit when comparing home ranges between site types. The assumption with this design is that the break between nightly sessions is not considered a sampling interval.

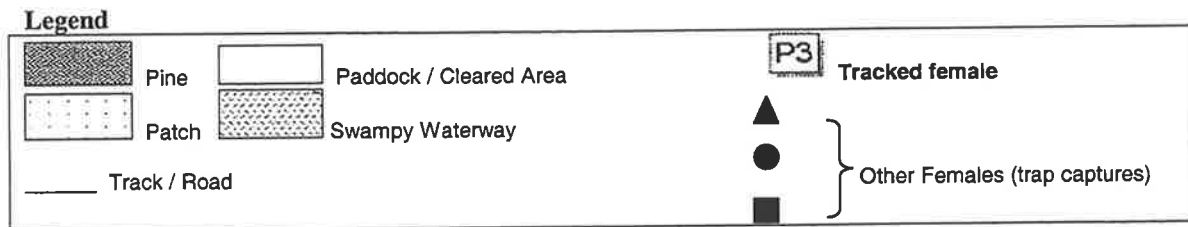
5.3 Results

5.3.1 Home Range

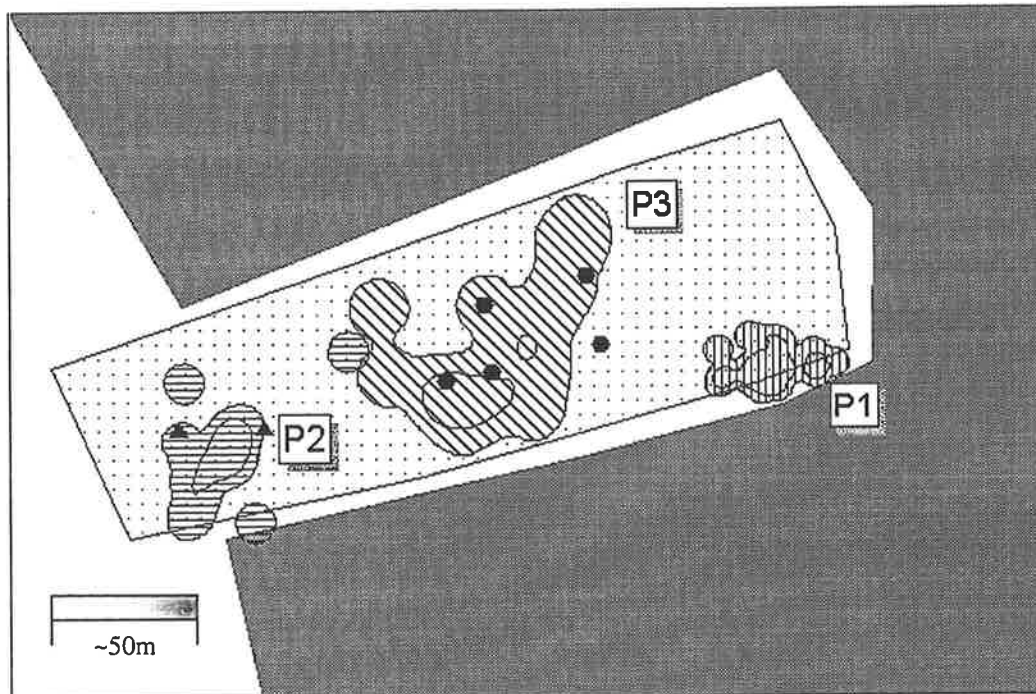
Of the 18 females targeted, 16 were captured and 12 successfully fitted with transmitters. All but one transmitter were later recovered. Data were collected over seven consecutive nights in each radio-tracking session, with an average of 28.9 (± 1.5 s.e) observations collected per female (range 20 to 37). Home range estimates (95% isopleth and the core isopleth for each female) are shown in Figure 5.1. Overall, mean home range area was 0.28 ha (Table 5.1) with a mean home range area for restricted habitat females at 0.27 ha (± 0.07 s.e., $n=9$), and 0.31 ha (± 0.16 s.e., $n=3$) for unrestricted habitat females. Core areas were calculated from the core boundary isopleth estimated for each animal. These isopleths ranged from the 45% to 75% isopleth. The mean core area was 0.06 ha (Table 5.1), with a mean core area for restricted habitat females at 0.05 ha (± 0.01 s.e.), and 0.09 ha (± 0.05 s.e.) for unrestricted habitat females. Results of a *t*-test showed no significant difference between home range area for unrestricted and restricted sites ($t = 0.44$, 10 d.f., $P = 0.673$). However, the sample size for unrestricted sites was <5 , too small for a reliable *t*-test. An examination of scatter plots graphing home range area and true habitat area, and home range area and density of resident females, showed no associations (Pearson's $r = -0.2$ and -0.11 respectively).

Of the 12 females tracked, 10 occurred in sites with other tracked females, enabling analysis of home range overlap. Four of these females showed a small amount of home range overlap (1% to 3%) with the majority of overlap occurring within the 85-95% isopleths. Only one female had the core of her home range encroached by another female, and then only slightly (Figure 5.1a). Trap capture locations for other females known to be resident and lactating (but not tracked) were plotted over the home range estimates to determine other incidents of home range overlap (see Figure 5.1). Results show five of the 12 other females were captured at least once within the home range estimate of tracked females. Four females were plotted within the 85-95% isopleths (Figures 5.1a, b, d and e), and one within the core of a home range (Figure 5.1a). An examination of the trap capture history of the overlapping tracked and other females

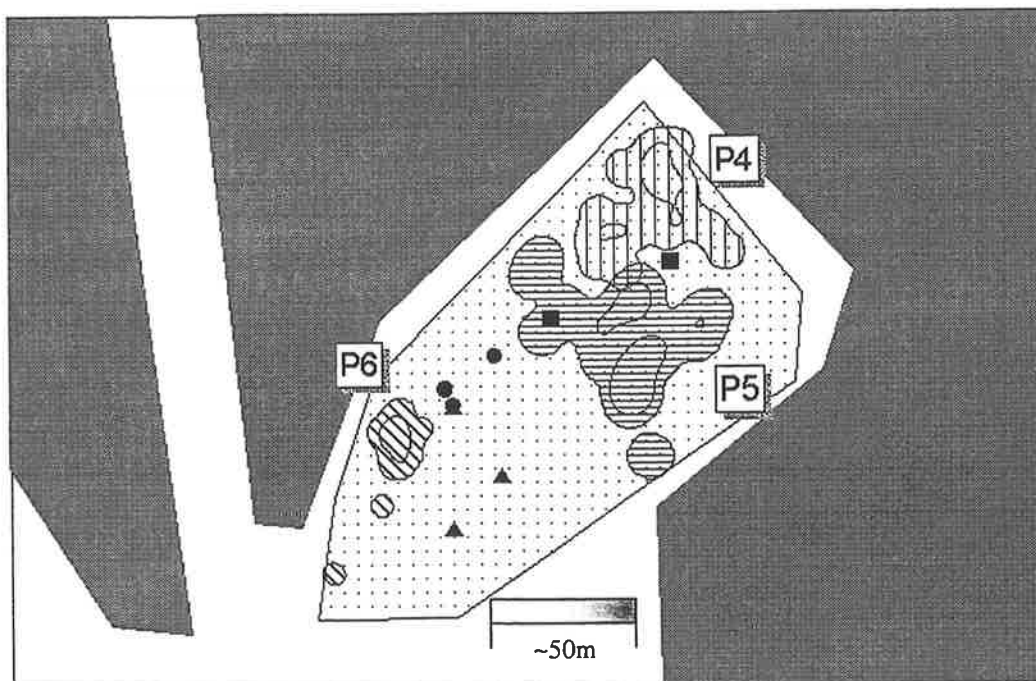
Figure 5.1: Home range estimates showing the 95% isopleth (outer boundaries) and core areas (inner boundaries) for tracked females within (a) site 1, (b) site 2, (c) site 10, (d) site 5, (e) site F1 and (g) site F3.

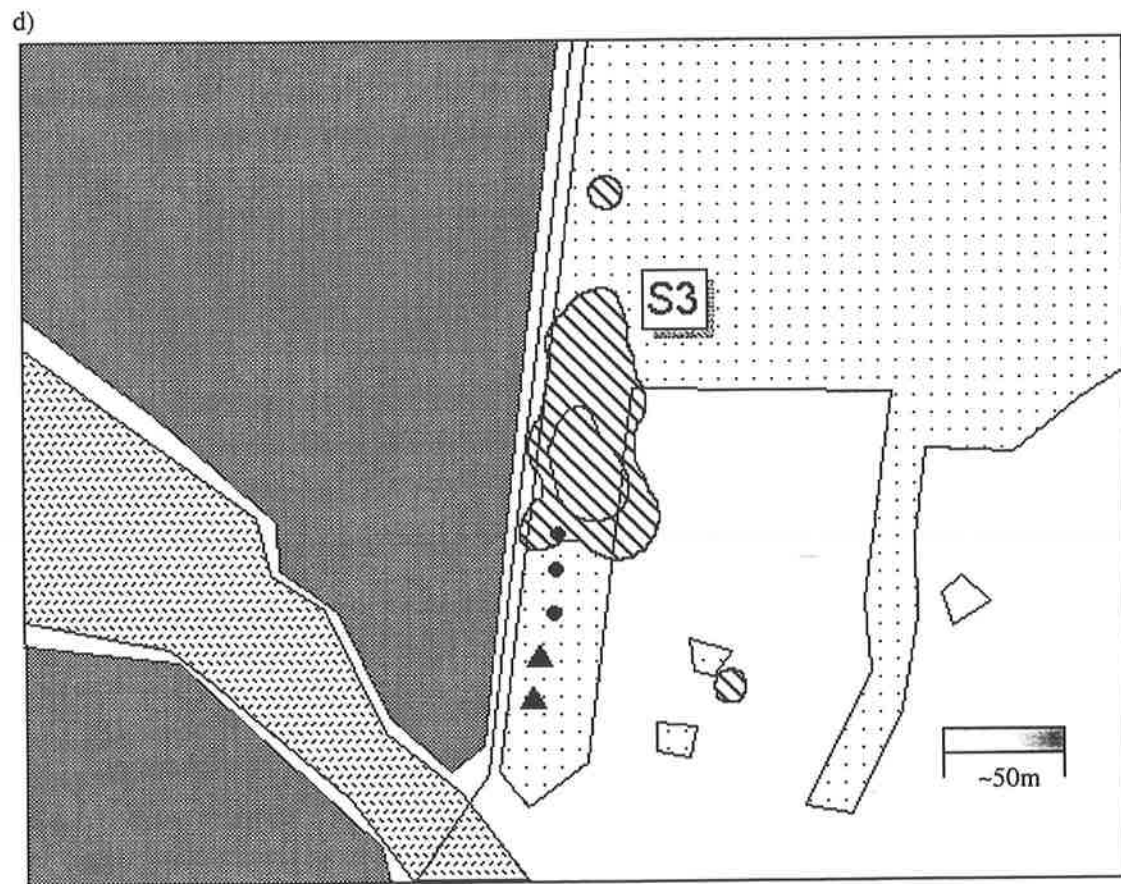
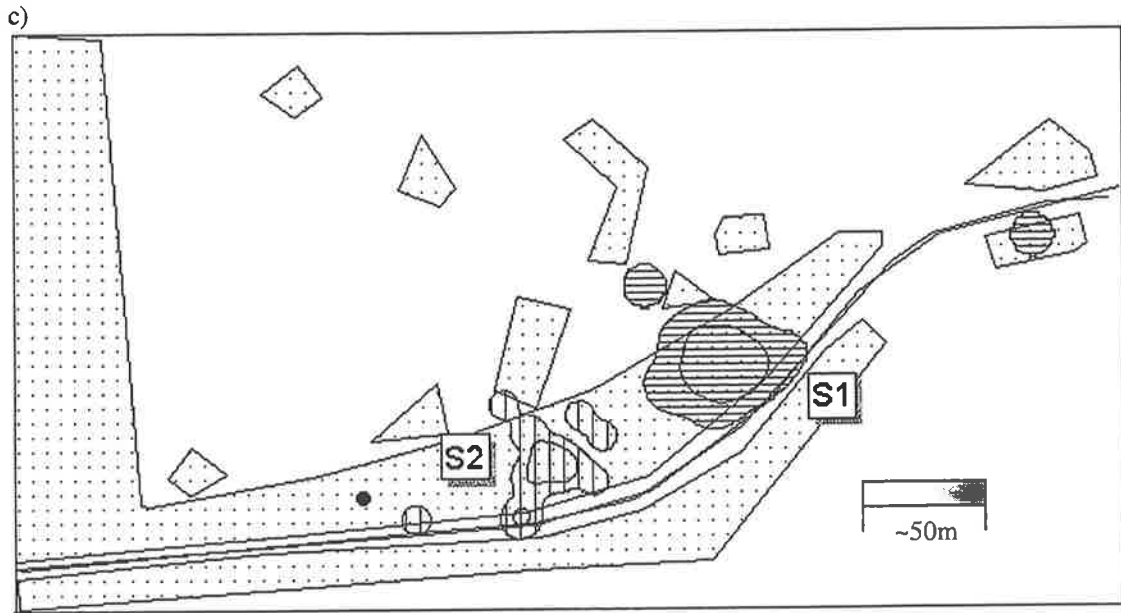


a)

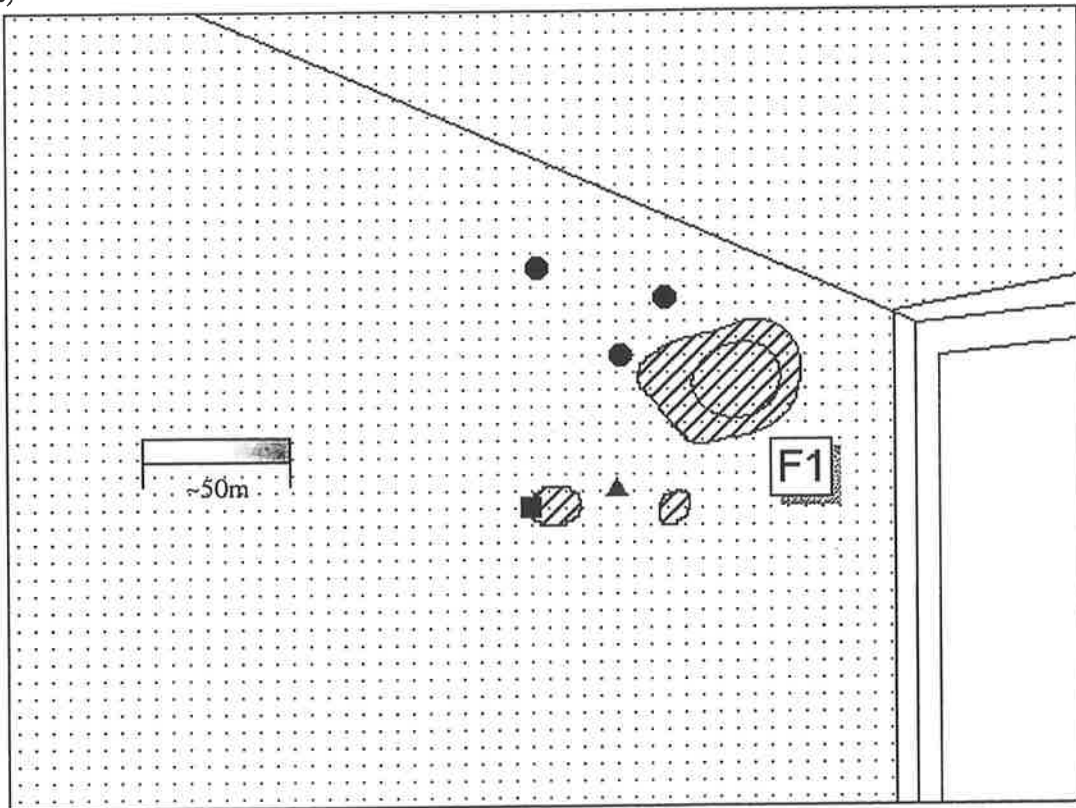


b)





e)



f)

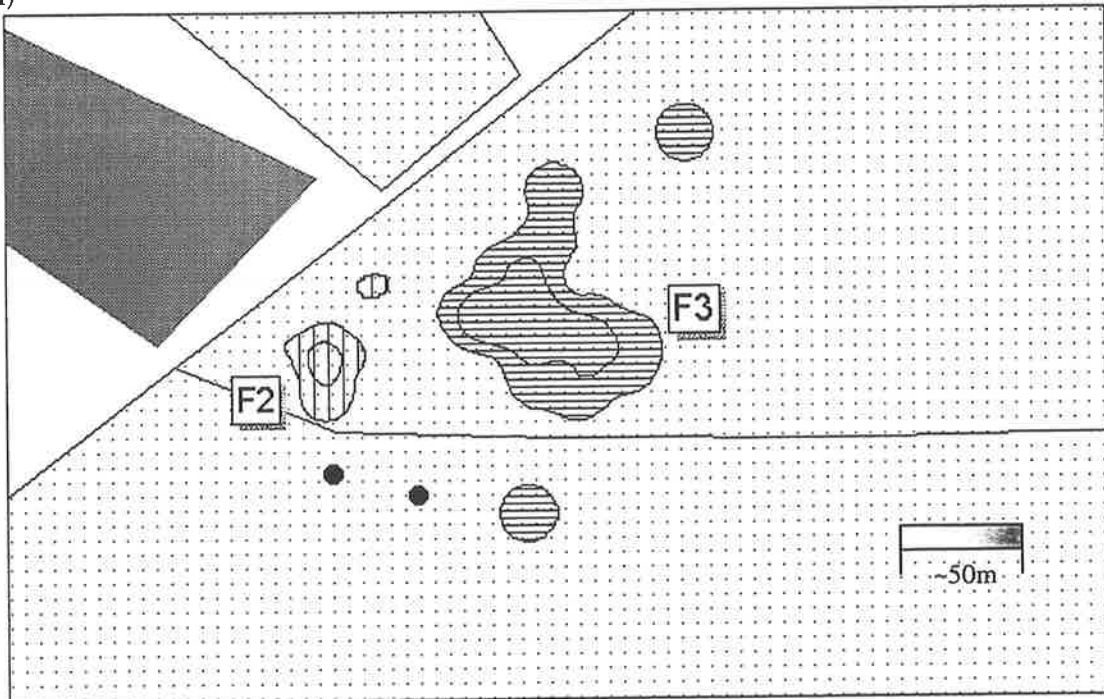


Table 5.1: The number of observations, area in hectares for the 95% isopleth (95% KHR) and the core (Core KHR), with isopleths defining the core given for each female.

Female	No. obs	95% KHR Area (ha)	Core KHR	
			Isopleth	Area (ha)
P 1	27	0.10	65%	0.04
P 2	20	0.17	65%	0.03
P 3	25	0.60	55%	0.08
P 4	27	0.15	55%	0.02
P 5	28	0.23	65%	0.05
P 6	24	0.04	65%	0.01
S 1	28	0.29	65%	0.06
S 2	28	0.16	65%	0.03
S 3	37	0.66	75%	0.15
F 1	36	0.20	65%	0.06
F 2	36	0.11	45%	0.02
F 3	31	0.63	65%	0.19
Mean (\pm s.e.)	28.9 (1.5)	0.28 (0.06)	-	0.06 (0.02)

shows a relatively high amount of overlap in capture locations. This suggests that the females are possibly related; mother and daughter or siblings. A comparison of home range overlap between site types and in relation to effective habitat area was not attempted, due to the lower than expected number of sites that contained at least two tracked females. However, visual analysis of home range estimates and captures of other females shows that the denser sites (1, 2, 5 and F1) tended to have a higher degree of overlap.

Home range plots of females in discrete patches (sites 1 and 2) did not normally extend out of the patch into the surrounding matrix. A few observations were plotted in open paddock (4 for female P2). However, these were omitted from the home range estimate as the tracking stations used for these observations resulted in a highly acute angle, which produces a high location error (Saltz, 1985). This situation could not be avoided, as movement to a more appropriate station would have 'spooked' the animal back to her nest (Marchesan, D. 2001, *pers. obs.*). Indeed, most observations for P2 were from stations that produced a highly acute observation angle. Thus, the home range estimate for this female should be treated with caution.

Individuals in the linear strips were also occasionally recorded in areas outside the main strip of native vegetation. Observations were made in adjacent paddocks amongst isolated trees (1 for female S1, 1 for female S2 and 1 for female S3) and along a small roadside patch of vegetation (1 for female S1). It is unlikely that these observations were due to location error, as observation angles were relatively close to the optimum of 90° . Furthermore, when the location error to two standard deviations (see below) was applied using the observation distance to the estimated location, three observations remained outside of the habitat strips. Also, the observation locations were close to scattered trees in the paddocks, which would be suitable for foraging.

Day-time radio-tracking showed that seven of the 12 females made movements during the day, with 24% of all day-time observations made away from nest sites. The location of nest sites were identified as those trees and yaccas which had repeated day-time observations (described in Chapter 3). Faecal mounds were also present at the base of some nest trees and yaccas.

5.3.2 Accuracy Test

Mean bias was calculated at +0.14 degrees (± 13.1 , 1 s.d., $n=48$) and mean precision at ± 5.5 degrees ($n=24$). The maximum observation distance (distance from observer to transmitter, or max [d1,d2]) ranged from 35 m to 65 m (mean 48.1, ± 2.7 s.e., $n=12$), producing a location error of 4.64 m for 1 s.d. or 9.28 m for 2 s.d., at the mean observation distance.

Although mean bias was minimal, the standard deviation was quite large. The conditions in which the tests were done were not ideal (high winds) and in retrospect were not representative of conditions experienced whilst tracking at night. Because of this, a second analysis was carried out on observations collected during night-time tracking, on two transmitters removed by animals and left in nests at two sites. A total of 12 and 13 repeated measures were taken for the first and second transmitter respectively. Mean bias was calculated on these data at -1.08 degrees (± 6.03 , 1 s.d., $n=25$) and mean precision at ± 5.75 ($n=2$). This equated to a location error of 4.86 m for 1 s.d. or 9.72 m for 2 s.d. at the above mean observation distance. Radio-tracking data were adjusted to the bias calculated on the second set of test data while the mean precision was used in the calculation of location error to verify the likely location of observations plotted in the matrix.

The number of observations obtained per animal (20 to 37) was lower than the target set (40 to 50). To determine whether or not home ranges were adequately expressed, observations were graphed against cumulative area for the 95% isopleth. In addition, capture locations of the target females for September to November 2001 were compared with home range estimates. It was considered that observations for nine of the 12 animals produced an asymptote in the area/observation number curve (Figure 5.2), although seven of the 12 had been captured at least once outside the home range plot (P1, P2, P3, P4, P5, F1 and F3). Five of these animals had produced an asymptote. Of the three animals that did not produce an asymptote, two had the lowest numbers of observations (20, 24).

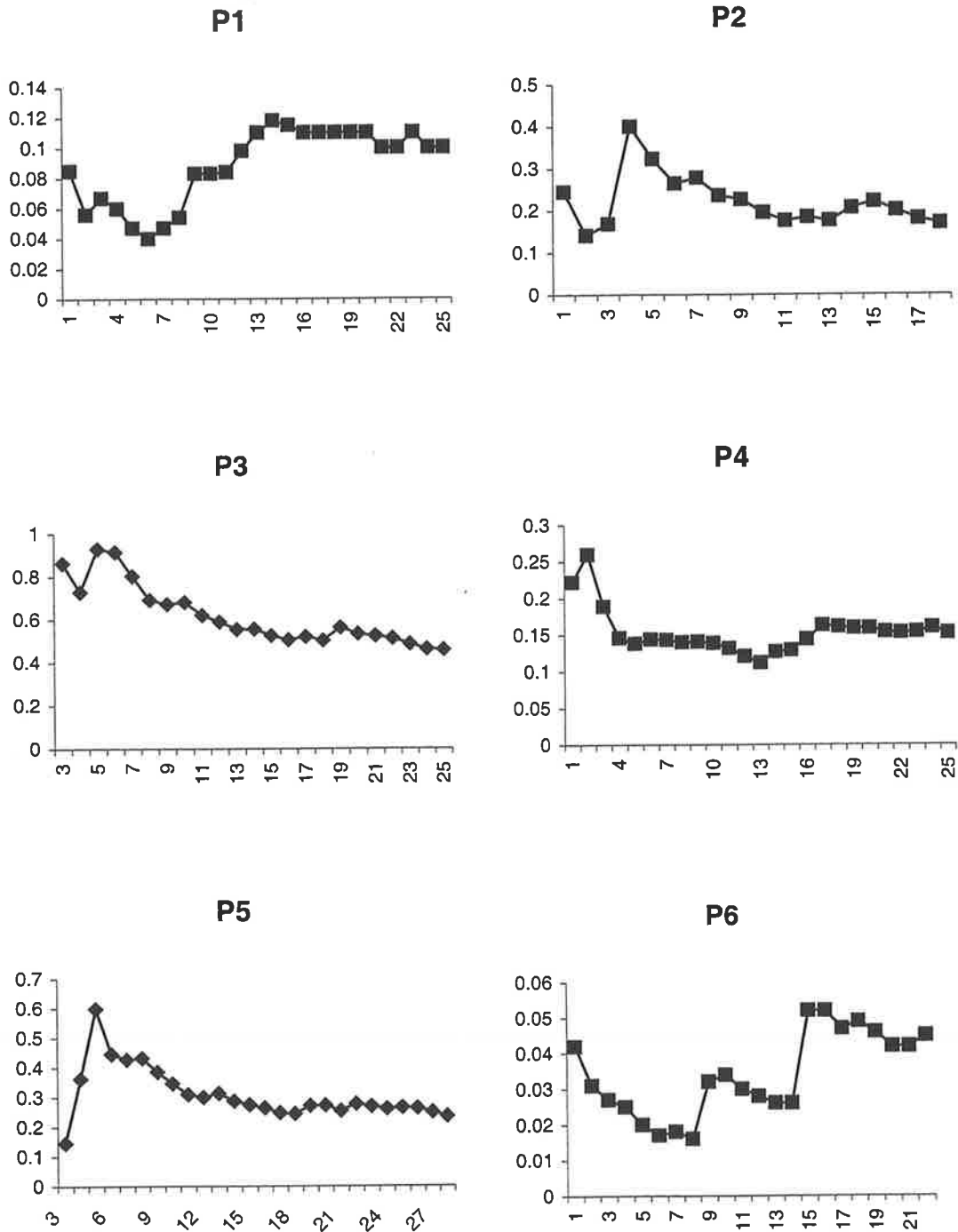


Figure 5.2a: Plots showing home range area in ha for the 95% isopleth (Y-axis) against the number of observations (X-axis), for each female in patch sites. It is considered that females P1, P2, P3, P4 and P5 produced asymptotes.

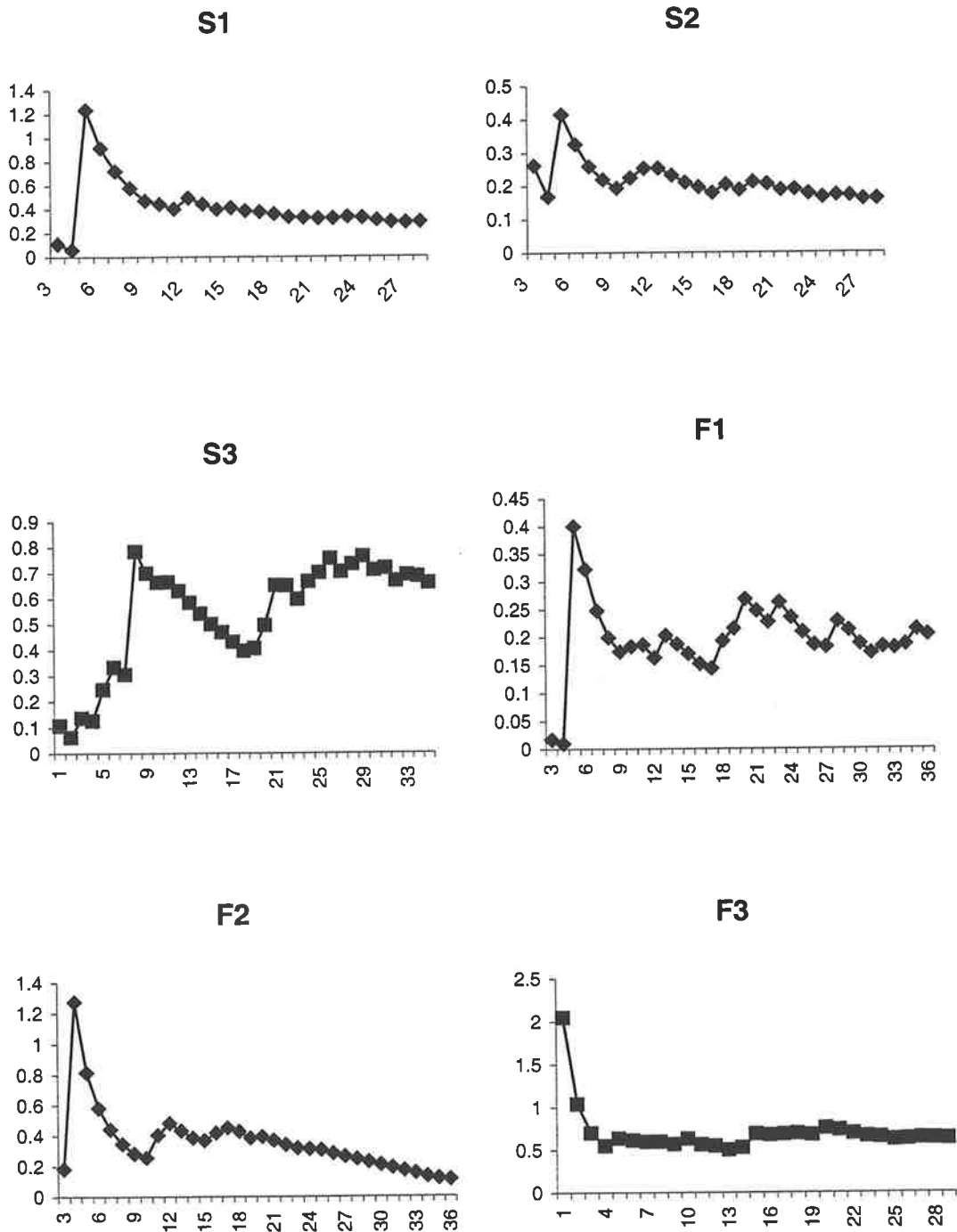


Figure 5.2b: Plots showing home range area in ha for the 95% isopleth (Y-axis) against the number of observations (X-axis), for each female in strip and forest sites. It is considered that females S1, S2, F2 and F3 produced asymptotes.

5.4 Discussion

Home range area and the degree of overlap had not previously been studied for *A. flavipes*. The only information found on home range for *A. flavipes* was in a study by Smith (1984). He used the average distance moved between successive captures (AvD) as an index of home range for *A. flavipes* populations in southern Queensland, and recorded the AvD for females during the period from October to January as 48.2 m. As a comparison, the AvD for October to January in this study was 29.53 m (± 2.71 s.e.). The results of home range area in the present study can be compared with Lazenby-Cohen and Cockburn (1991). They reported a mean foraging home range area of 0.38 ha for female *A. stuartii* which they considered to be stable throughout the year, and which is larger than that recorded for *A. flavipes* in the present study (0.28 ha). However, the home range area reported by Lazenby-Cohen and Cockburn (1991) was calculated using the minimum convex polygon estimate, and used as few as eight observations per animal. Such methodology limits the validity of comparisons with the mean home range estimate calculated in the present study. Moreover, the kernel home range estimates in the current study may have been affected by the lower than recommended number of observations obtained for each animal. Although only three animals did not produce an asymptote in area/observation graphs (thereby suggesting an inadequate number of observations or sampling duration), five of the nine that did produce an asymptote had been trapped outside of the 95% isopleth. This indicates the possibility that not all home ranges were fully expressed for the time period sampled.

Changes in home range behaviour in response to habitat restriction had also not been previously studied for *A. flavipes*, or indeed any other *Antechinus* species. Although results give an indication of home range area and some indication of overlap, they are generally inconclusive regarding *A. flavipes*' ability to change home range behaviour in response to habitat restriction. Results showed no significant difference in home range area between females in restricted (patches and strips) and unrestricted (forest) habitat, although it is acknowledged that sample sizes were too low to produce reliable results. Moreover, there were no correlations between home range area and either site population density or true habitat area. Therefore, results provide no clear evidence to suggest that *A. flavipes* did or will adapt home range size in response to limiting space resources.

Home range overlap was minimal amongst tracked females, with home ranges remaining largely exclusive. However, an overlay of trap captures for other females known to be resident in the study sites showed some degree of overlap. The animals involved in this overlap were possibly related as determined by their overlapping trap-capture location history. Related female *Antechinus* have been shown to live in close proximity (Cockburn *et al.*, 1985) and even nest communally (Cockburn and Lazenby-Cohen, 1992). Much of this behaviour has been recorded outside of lactation times, although Cockburn and Lazenby-Cohen (1992) recorded one *A. stuartii* mother and daughter pair sharing a nest during lactation. No other information was found regarding home range overlap in other *Antechinus* species. Comparisons of home range overlap between restricted and unrestricted habitat sites, and sites of varying habitat area, were difficult due to the low number of sites from which home range overlap could be reliably measured. However, a visual comparison indicated that denser sites may have greater degrees of home range overlap. This could not be quantified and would require further investigation to confirm the observation. Current results indicate that *A. flavipes* is able to at least tolerate home range overlap during lactation. Adaptation of home range behaviour in response to population density has been noted in other small mammal species, such as the greater glider (*Petauroides volans minor*). Comport *et al.* (1996) observed a significant amount of male home range overlap within a high-density population of the greater glider in south-eastern Queensland, which had not been recorded in lower density populations.

The use of the habitat matrix was detected for females in linear strips, with three observations located outside the strips, after correcting for location error. Such movements by animals into the matrix are not entirely surprising, as *A. flavipes* has been shown to tolerate an agricultural matrix (Chapter 4; Dickman, 1991). It is likely that these individuals moved into adjacent paddocks to forage in the scattered messmate stringybarks (*E. obliqua*), or amongst the exotic grasses of the paddock itself. Dunning *et al.* (1992) describes this process as 'supplementation', or the use of suitable resources between habitat patches. Such resource supplementation may assist populations of *A. flavipes* existing in habitat patches with restricted space and food resources and relatively high population density (such as site 5). The day-time activity observed during this study may have also been a response to a need to maximise foraging time in order to nurse young. Such activity was not recorded for an *A. flavipes* female in the

south-west of Western Australia (Wardell-Johnson, 1986), which was monitored for a number of 24hr periods. However, monitoring was carried out during December, after young had been weaned. Day-time captures have been recorded for other *A. flavipes* populations in southern Australia, particularly during the mating season (Carthew, S. 2001 *pers. comm.*; Petersen, B. 2001 *pers. comm.*).

Antechinus flavipes has been described as a species that is more adaptable than other members of the genus (Van Dyck, 1982) and is known to occupy a diverse range of habitats and tolerate an agricultural matrix. With such apparent adaptive qualities, it is feasible that *A. flavipes* might be able to adapt or change its home range behaviour to cope with limited space resources. This possibility would help explain the species' persistence in small areas of habitat within the fragmented landscape examined in this study. However, results provided only limited evidence for such adaptations. To better assess the relationship between habitat area, population density, home range area and home range overlap, additional radio-tracking work needs to be conducted on both male and female *A. flavipes*, during different lifecycle stages and in a greater number of restricted and unrestricted habitat sites.

CHAPTER 6: GENERAL DISCUSSION & IMPLICATIONS FOR CONSERVATION

6.1 Introduction

The potential threats of habitat fragmentation to the long-term persistence of species have received the attention of ecologists and conservationists world-wide.

Discussions and investigations have concerned the potential effects of habitat fragmentation (Gilpin, 1987; Andrén and Angelstam, 1988; Lacy, 1992; Arnold and Weeldenburg, 1998), potential remedies, including the role of wildlife corridors and landscape connectivity (Fahrig and Merriam, 1985; Bennett, 1990; Saunders and Hobbs, 1991; Hobbs, 1992; Simberloff *et al.*, 1992; Lindenmayer and Nix, 1993; Bennet *et al.*, 1994; Bennett, 1998) and the importance of the establishment and maintenance of metapopulations (Hanski and Gilpin, 1991; Lindenmayer and Lacy, 1995; Sarre, 1995; Hanski *et al.*, 1996). The effects of habitat fragmentation evidenced from empirical studies (Bennett, 1987; Saunders, 1990; Leung *et al.*, 1993; Bolger *et al.*, 1997) and expected from simulation models (Fahrig and Merriam, 1985; Lacy and Lindenmayer, 1995; Lindenmayer and Lacy, 1995; Tischendorf and Wissel, 1997) warrant an investigation into the ramifications of fragmentation in local areas. This is particularly true where habitat fragments have become a major part of natural system, such as in the Mt Lofty Ranges of South Australia.

Local studies on the effects of habitat fragmentation are important, as effects can often be species and landscape specific (Sarre *et al.*, 1995; Bennett, 1998; Delin and Andrén, 1999). Some species have been shown to suffer greatly from habitat fragmentation (Saunders, 1990), be deterred from entering agricultural landscapes (Downes *et al.*, 1997a) or avoid linear habitat (Lindenmayer *et al.*, 1993). However, results from the present study do not indicate such effects on the populations of *Antechinus flavipes* in this landscape. *A. flavipes* were found to be present and successfully breeding within small patches and linear habitat. They also displayed a tolerance of an agricultural matrix which allowed them to move between habitat patches and make foraging trips to scattered trees in open paddock. It appears that the metapopulations sampled here have successfully persisted in this landscape over the

past 50 years since land clearance, or at least have overcome the obstacles of habitat fragmentation to survive and reproduce in a 50 year old fragmented landscape. Moreover, when comparing population densities over the two seasons sampled, up to 89% of habitat patches and strips recorded higher densities than two of the three sites sampled in contiguous forest. Bentley *et al.* (2000) also found a greater abundance of *A. flavipes* in fragments than in forest sites.

6.2 Autecology and a fragmented landscape

The success of *A. flavipes* in the fragmented landscape of the present study can be attributed to a number of factors. Firstly, the general autecology of the species may provide advantages. The species has been described as adaptable (Van Dyck, 1982) and is considered a floristic generalist (Bentley *et al.*, 2000). It is also known to inhabit a wide variety of habitat types (Strahan, 1995). The tolerance of the agricultural matrix observed in the present study and noted by others (Dickman, 1991; Laurance, 1994) is consistent with the adaptable qualities of *A. flavipes* suggested by Van Dyck and others. In comparison, *A. stuartii* is considered to be less adaptable than *A. flavipes* (Van Dyck, 1982), has been described as hollow-dependent (Lindenmayer *et al.*, 1999) and was found to be absent in remnant patches within an Araucarian vine forest where *A. flavipes* was present (Bentley *et al.*, 2000). Other behavioural adaptations, in response to habitat restriction or other changes brought about by habitat fragmentation, would be important for persistence in a fragmented system. Evidence of such adaptations, such as a reduction in home range area or increase in home range overlap, was largely inconclusive in the present study. However, such behaviour would help to explain the persistence of *A. flavipes* in the relatively small habitat patches sampled in this study.

The life history strategy of the species, described by Lee *et al.* (1982), includes the complete die-off of males after mating. Such a strategy may assist in the persistence of small populations in small habitat patches, as it leaves more resources for lactating females and emerging young. The same sized carnivorous animal without male-die off in its life history may not fare as well in similar sized habitat patches. The marked dispersal behaviour shown by male *A. flavipes* (and known for other populations of *Antechinus*) allowed the establishment of a metapopulation and local population

replenishment. Such behaviour was also noted by Lindenmayer *et al.* (1999) as a possible explanation for the continued presence of *A. stuartii* in some areas of remnant vegetation, embedded in exotic pine and relatively close to larger areas of forest, in the Tumut region of southern NSW. The dispersal ability demonstrated by *A. flavipes* would complement their tolerance of the landscape matrix, allowing successful movements between habitat patches.

Antechinus flavipes has also been described as 'edge favouring' (Laurance, 1994), which could potentially be advantageous in the habitat remnants of the present study. Given the small size of habitat patches and evidence from other studies that suggest edge effects can reach up to 500 m (see Chapter 3), it is suggested that the habitat patches in the present study consisted mostly of 'edge' habitat. Habitat edges often differ from interiors in structure, species diversity and climatic conditions (Laurance and Yensen, 1991). Edges may negatively affect forest-dependent species (Lidicker, 1999; Sih *et al.*, 2000) but can provide preferred or complementary resources for other species (Laurance, 1994; Tufto *et al.*, 1996). The higher body weights, reproductive output and home range size of edge-dwelling meadow voles in Virginia, was partly attributed to improved habitat quality at fragment edges (Bowers and Dooley Jr, 1999). This may also be the case for *A. flavipes* in the present study, where body weights for patch animals were significantly higher than those in forest sites. The effect of edges on vegetation structure and insect abundance in the stringybark forests of this study need to be investigated to better explain *A. flavipes* success in edges and small fragments. Habitat edges may not always be favourable, particularly in relation to predation pressure (Andrén and Angelstam, 1988) and exotic species invasion (Laurance and Yensen, 1991; Lidicker, 1999). During the present study exotic species such as weeds and vertebrate pests were evident in most sites. However the predation pressure evident through fox presence did not cause a complete demise of the *A. flavipes* populations. This could be attributed to the scansorial nature of the species, which helps to avoid some or much of the predation risk on the ground.

6.3 Landscape Structure and Movement

Besides autecological aspects, landscape structure is another factor likely to contribute to the success of *A. flavipes* in this fragmented landscape. Landscape structure has been considered one of the most important factors in understanding the effects of habitat fragmentation on populations and in determining the success of such populations (Fahrig and Merriam, 1985; Claridge and Lindenmayer, 1994; Fahrig and Merriam, 1994; Bennett, 1998). One aspect of landscape structure is the distance between habitat patches. Between-site distances in the present study were within known dispersal distances recorded for *A. flavipes* (ie. less than 1 km), which allowed successful movements between sites. If distances were greater, movements may have been inhibited or the success of movements may have been affected. The critical between-site distance for a successful inter-patch movement is not known for *A. flavipes* and could not be measured in the present study.

The matrix type found within a landscape can also influence the movement ability of species and may affect the critical distance for successful movement. Although the preferred matrix type for movement could not be ascertained in this study, individuals had to pass through clearings up to 30 m wide, exotic pine plantations up to 500 m wide, roads or native strips of vegetation. They may have also entered open paddock. Such landscape types appear amenable to movements by *A. flavipes*. The particular features of the matrix allowing successful inter-patch movements are not clear, but perhaps a combination of open areas and cover (even though exotic pine) are adequate. The areas of scattered native vegetation within the matrix may have assisted in the survival of individuals during movement, and may have also increased the critical distance between sites. The habitat matrix can have a major influence on the persistence of species within habitat patches in relation to present and future resources (Lindenmayer, 1994). For example, the presence of scattered *Eucalyptus* trees in open paddocks in the present study allowed the supplementation of food resources. Such supplementation may have been critical for the continued persistence of the *A. flavipes* in the more restricted habitat areas, such as linear strips. The general habitat quality in the study sites would also be critical in allowing the successful persistence, breeding and weaning of young.

6.4 Continued Threats

Despite the occurrence and breeding activity recorded for *A. flavipes* in this study, populations remain vulnerable to the potential effects of habitat fragmentation. As suggested by Hanski *et al.* (1996), there is a possibility that populations are adjusting to the 'post-fragmentation equilibrium', which may in fact be extinction. Given the small sizes of local populations, demographic stochasticity (random births, deaths, dispersal, migration) can still have a major effect on the persistence of individuals in patches. This is particularly true for females, which remained largely philopatric and have potentially lower dispersal ability than males. Therefore, the extinction of females within a patch may mean the permanent extinction of a local population unless a source of female migrants is within dispersal distance from the patch. The degree of movement observed between most patches would probably assist in maintaining the genetic health of local populations. However, little can be ascertained about the genetic health of the metapopulation. To do this, the extent of the metapopulation needs to be established, as does the level of dispersal between neighbouring metapopulations and the inbreeding tolerance of the species. Environmental stochasticity continues to remain a threat, particularly fire and poor seasons affecting invertebrate populations. Also, the landscape in which these populations occur are not completely within reserved areas and so are still vulnerable to land management practices and associated effects, such as vegetation clearance, disturbance, erosion and exotic pest invasion.

The current landscape system should continue to function well for the *A. flavipes* populations examined in this study, provided the landscape structure does not change to the detriment of local populations. An important note to consider here is the management of the habitat patches involved, and the fact that metapopulations cross jurisdictional boundaries with regards to land management. *A. flavipes* was found to occupy habitat on a number of different land tenure types, including State Government, Crown, private and Local Council property. Land managers need to be aware of the need to integrate management activities in relation to wildlife conservation and at least understand the impact that their land management practices have on the wildlife systems that cross jurisdictional boundaries.

6.5 Future Studies

Many questions still remain regarding the persistence of our native flora and fauna in fragmented landscapes across all land tenure types in South Australia. To further the information base, future work could include an examination of other species (including threatened or endangered species) in fragmented landscapes, the effectiveness of inter-patch movements for the maintenance of genetic fitness of metapopulations, preferences for matrix type for movements between patches, the effects of distance between habitat patches in maintaining a metapopulation, the extent of habitation in the matrix (particularly pine plantations) and continued investigation into the ability of species to modify behaviour in order to cope with fragmented habitat. Ultimately, an examination of the importance of a metapopulation to the regional persistence of the species would be valuable. The results of this study begin to address the paucity of data on the persistence of *A. flavipes* in fragmented landscapes in South Australia and provide further reference to the response of our small, native mammals to habitat fragmentation Australia-wide.

APPENDIX 1

a) Table showing structural classification of vegetation, from Specht (1972), p28.

Life form and height of tallest stratum*	Projective foliage cover of tallest stratum*			
	Dense (70-100%)§	Med-dense (30-70%)	Sparse (10-30%)	Very sparse** (<10%)
†Trees >30 m	Tall closed-forest*	Tall open-forest	Tall woodland	Tall open-woodland
†Trees 10-30 m	Closed-forest*	Open-forest	Woodland	Open-woodland
†Trees 5-10 m	Low closed-forest*	Low open-forest	Low woodland	Low open-woodland
†Shrubs 2-8 m	Closed-scrub	Open-scrub	Tall shrubland	Tall open-shrubland
†Shrubs 0-2 m	Closed-heath	Open-heath	Low shrubland	Low open-shrubland
Hummock grasses 0-2 m	-	-	Hummock grassland	Open hummock grassland
Herbs (including moss, ferns, hemicryptophytes, geophytes, therophytes, hydrophytes, helophytes)	Closed-herbland‡- (1) Closed-tussock grassland (2) Closed-grassland (3) Closed-herbfield (4) Closed sedgeland (5) Closed-fermland (6) Closed-mossland	Herbland‡- (1) Tussock grassland (2) Grassland (3) Herbfield (4) Sedgeland (5) Fermland (6) Mossland	Open-herbland‡- (1) Open-tussock grassland (2) Open-grassland (3) Open-herbfield (4) Open-sedgeland (5) Open-fermland (6) Open-mossland	Ephemeral herbland

† A tree is defined as a woody plant more than 5 m tall, usually with a single stem.

A shrub is a woody plant less than 8 m tall, frequently with many stems arising at or near the base.

* Isolated trees (emergents) may project from the canopy of some communities. In some closed-forests, emergent *Araucaria*, *Acacia*, or *Eucalyptus* species may be so frequent that the resultant structural form may be classified better as an open-forest.

** Some ecologists prefer to ignore scattered trees and shrubs, equivalent to emergents in a predominately grassland, heath or shrubland formation.

‡ Appropriate names for the community will depend on the nature of the dominant herb.

§ The above values are based on the percentage number of leaf interceptions by random vertical point quadrats; stereoscopic profiling, developed by Howard (1970), would give higher values.

Reference

Howard, J. A., (1970). Stereoscopic profiling and the photogrammetric description of woody vegetation. *Australian Geographer*, 11: 359-372.

b) Descriptions of codes used in Table 2.2. Extract from Stokes *et al.* (1998).

See attached.

Structural Type (Vegetation Association Description)

F	Forest
W	Woodland
K	Mallee
S	Shrubland
P	Mat plants
H	Hummock grassland
G	Grassland (tussock)
V	Sedgeland
J	Herbland
X	Fernland
PL	Plantation
BU	Built-up
BG	Bare ground

Explanation:

The overstorey (i.e. the tallest stratum that has a cover $\geq 5\%$) of each structural type is dominated by plants of characteristic life form, as follows:

Forest - Dominated by trees*, cover $>30\%$.

Woodland - Dominated by trees*, cover $\geq 5\%$ and $<30\%$.

Mallee - Dominated by mallees, i.e. plants from genus *Eucalyptus*, multi-stemmed, arising from lignotuber.

Shrubland - Dominated by shrubs, i.e. plants that are woody, perennial, with foliage occupying all or part of total plant height, and with multiple stems and branches arising from a rootstock or very short common trunk. Generally $<5\text{m}$ tall. Includes genus *Xanthorrhoea*.

Mat plants - Herbaceous or woody plants of prostrate habit, with major stems growing along the ground. Rarely exceed 10 cm in height. Examples of mat plants are *Kunzea pomifera*, *Myoporum parvifolium*, *Carpobrotus rossi* and *Mimulus repens*.

Hummock grassland - Dominated by plants from the genera *Triodia* and *Plectrachne* (Graminae).

Grassland (tussock) - Dominated by genera from family Graminae (Poaceae), other than hummock grasses.

Sedgeland - Dominated by sedges or sedge-like plants, i.e. plants that are herbaceous, usually perennial, erect, generally tufted, arising from stolons, tubers, bulbs, rhizomes or seeds. Leaf sheath never split. Includes Juncaceae, Restionaceae, Typhaceae and Xyridaceae; also includes genera *Lomandra* and *Dianella* (Lilliaceae), *Lepidosperma* and *Gahnia* (Cyperaceae).

Herbland - Dominated by herbs, i.e. plants that are herbaceous or slightly woody, annual or sometimes perennial, erect or creepers, rarely exceeding 0.5m in height.

Fernland - Dominated by ferns and/or fern allies, i.e. non-vascular cryptograms of classes Filicopsida and Lycopsidea. This category includes *Ophioglossum* spp., *Lycopodium* spp., *Selaginella* spp. and *Isoetes* spp.

Plantation - Dominated by plants that have been deliberately planted, either with indigenous or non-indigenous species; includes revegetation using direct seeding and planting of seedlings. "Understorey" may be remnant indigenous vegetation.

Built-up - Buildings, pavement, parks and gardens, property frontages, etc.

Bare Ground - Areas such as quarry / borrow pits (or scraps) where no to little vegetation is apparent and the dominant cover is bare ground.

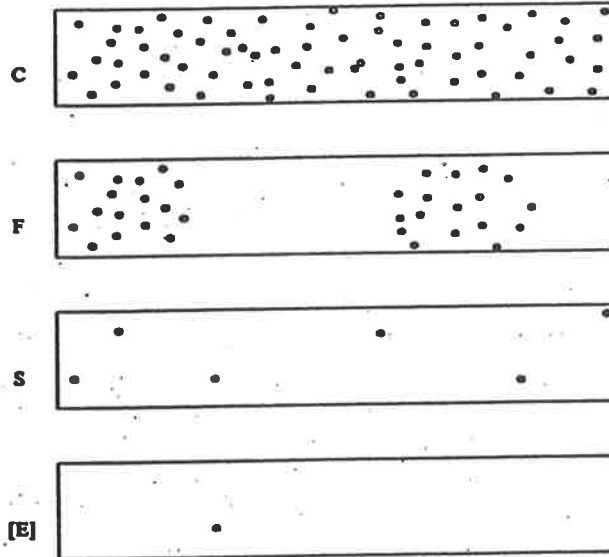
*Trees - woody; perennial; erect; canopy raised well above the ground. Depth of canopy is usually less than or equal to two thirds of the total tree height. Single stemmed, or if multi-stemmed, fewer than five individual trunks resulting from branching of a single short trunk, that is not a mallee-like lignotuber. Height usually $>2\text{m}$.

Ref: Adapted from Forward, L.R., and Robinson, A.C. (eds) (1996). *A Biological Survey of the South Olary Plains South Australia, 1991-1992*. Department of Environment and Natural Resources, South Australia.

Density / Distribution (Vegetation Association Description)

C	Continuous
F	Fragmented
S	Scattered

Explanation



C - Continuous - a continuous, approximately uniform coverage of individuals in the overstorey, with little sign of human-induced thinning or fragmentation by human-induced "clearings".

F - Fragmented - patches of continuous overstorey interspersed with human-induced breaks or "clearings"; clearings and/or overstorey patches are too short to warrant definition of separate segments; indicates human-induced within segment variability (patchiness) of overstorey distribution. Note that the understorey may be uniform or may change coincidentally with overstorey.

S - Scattered - consistent scattering of remnant overstorey individuals, although extensive clearing by humans has occurred. This requires an assessment of whether the density is natural or human-induced - naturally very open communities should be recorded as continuous because their openness has not been caused by clearance or other human disturbances.

[E - Emergent species - if the tallest layer has <5% cover then it should be considered to be an emergent stratum, with the overstorey defined as the tallest stratum that has a cover of $\geq 5\%$. If an emergent stratum is present, record a species code for the dominant species in the stratum (refer to Section 5.1.11). Note that this very low density of individuals in the stratum may be natural or the result of extensive clearance.]

*The cut-off value of 5% is the standard used in the vegetation component of the Biological Survey of South Australia to define an emergent stratum. As a rule of thumb for estimating 5% cover, this represents 1/20th of the road reserve occupied by tree/mallee/shrub canopy, which could mean one individual with a five-metre diameter canopy every 100m, or a 10-metre diameter individual every 200m, or a 20-metre diameter individual every 400m.

Understorey Type (Vegetation Association Description)

LT	Low trees
SC	Shrubs - chenopod / semi-succulent
SH	Shrubs - heath
SO	Shrubs - other native
SE	Shrubs - exotic (alien)
PN	Mat plants - native
PE	Mat plants - exotic (alien)
H	Hummock grasses
GN	Grasses - native tussock
GE	Grasses - exotic (alien)
VN	Sedges - native
VE	Sedges - exotic (alien)
JN	Herbs - native
JE	Herbs - exotic (alien)
CN	Vine / climber / creeper - native
CE	Vine / climber / creeper - exotic (alien)
X	Ferns
B	Bare ground / litter
XX	Other

Explanation

General note: "Native" and "Exotic" as used in these codes and definitions should be considered synonymous with "indigenous" and "alien" respectively as used in the rest of this manual and defined in the Glossary (Appendix 1). "Native" and "Exotic" are used in these codes for reasons of data integrity within the Roadside Vegetation Database.

Low trees - Tree layer (refer to definition of "trees" in Appendix 3.1) below the overstorey and generally above the level of a shrub understorey; e.g. *Acacia pycnantha* or *Allocasuarina verticillata* under *Eucalyptus viminalis* Open Forest/Woodland in the Adelaide Hills. Refer to diagram below.



Shrubs - chenopod / semi-succulent - Indigenous species from the family Chenopodiaceae; (the term "semi-succulent" is from Specht 1972).

Shrubs - heath - Indigenous sclerophyllous shrubs, usually less than 2m tall, commonly with ericoid leaves, generally with denser cover eg. >30% (adapted from McDonald et al, 1990 and Specht, 1972).

Shrubs - other native - native shrubs other than chenopod and heath; includes *Melaleuca* spp., *Acacia* spp., *Xanthorrhoea* spp.

Shrubs - exotic - Alien shrub species.

Mat plants - native - Refer to definition of mat plant in Appendix 3.1.

Mat plants - exotic - Refer to definition of mat plant in Appendix 3.1.

Grasses - hummock - Native grasses from the genera *Triodia* and *Plectrachne*.

Grasses - native tussock - Native plants from family Graminae (Poaceae), other than hummock grasses.

Grasses - exotic - Alien grass species from family Poaceae (Graminae).

Sedges - native - Refer to definition of sedge in Appendix 3.1; includes genera *Lomandra* and *Dianella* (Liliaceae), and *Lepidosperma* and *Gahnia* (Cyperaceae).

Sedges - exotic - Refer to definition of sedge in Appendix 3.1; includes *Juncus acutus* and *J. articulatus*.

Herbs - native - Refer to definition of herb in Appendix 3.1.

Herbs - exotic - Refer to definition of herb in Appendix 3.1; includes *Echium plantagineum* (Salvation Jane).

Vine / climber / creeper - native - Climbing, twining, winding or scrambling plants; excludes mat plants which have their major stems growing along the ground; examples are *Billardiera* spp., *Cassythia* spp., *Hardenbergia* spp.

Vine / climber / creeper - exotic:- Includes *Myrsiphyllum asparagoides* (Bridal Creeper).

Ferns - Refer to definition of fern in Appendix 3.1.

Other - Use this category to flag any understorey structure that does not fit into any of the defined categories. Record a description and possible definition, to be discussed with database administrators and users regarding a possible new category to be added to the database.

Condition Of Understorey

1	Excellent	Very little or no sign of alien vegetation in the understorey*; resembles probable pre-European condition
2	Good	High proportion of native species and native cover in the understorey*; reasonable representation of probable pre-European vegetation
3	Moderate	Substantial invasion of aliens, but native understorey* persists; for example, may be a low proportion of native species and high native cover, or high proportion of native species and low native cover
4	Poor	The understorey* consists predominantly of alien species, although a small number of natives persist
5	Very poor	The understorey* consists only of alien species

*Or all strata if upper and lower strata are difficult to distinguish (refer to Section 5.1.7 for explanation).

Disturbances

AT	Access track through vegetation
BH	Bee Hives
BN	Burning of vegetation
BQ	Borrow / Quarry Pit
CA	Campsite
CL	Clearing
CR	Coppice regrowth
DR	Drains
EA	Earthmoving / Earthworks
FP	Firebreak - ploughed
FS	Firebreak - slashed
LP	Lopping
PF	Property frontage
PL	Plantings
PW	Power Lines
RA	Active rabbit warren
RD	Rubbish Dumping
SL	Slashing
SP	Stockpile (road materials)
SR	Spraying
TE	Telecommunications easement
WS	Wayside stop (parking bay)
WP	Water pipeline

Explanations:

(Many of these codes are self explanatory, although a few need clarification.)

AT - access track - access track running longitudinally along the road reserve rather than latitudinally across the road reserve; thus tracks, roads, laneways etc into properties should not be recorded.

BU - burning - this may include large scale wildfire burns, small scale burning off of understorey or litter and branches, or burnt fuel breaks.

CR - coppice regrowth - the sprouting of many trunks when the main trunk of a tree or mallee is lopped near the base. (Contrast with LP - lopping).

LP - lopping - the cutting of limbs and branches to lower the height of the canopy; typically under powerlines.

PF - property frontage - may include lawns, plantings, driveways, etc.; too short to justify a separate "Built Up" segment.

PL - plantings - deliberate plantings, either of alien or locally indigenous species; either too short to justify a separate "Plantation" segment, or the plantings are scattered and/or small.

SL - slashing - includes rough slashing and close mowing.

SP - stockpile - road construction materials, e.g. gravel dumps.

APPENDIX 2 – Plant species recorded in each site. * Rare species.

SPECIES	SITE															
	1	2	3	4	5	6	7	8	9	10	11	12	Add	F1	F2	F3
Native Species																
Canopy																
<i>Eucalyptus fasciculosa</i>		✓					✓									
<i>Eucalyptus obliqua</i>	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
<i>Eucalyptus baxteri</i>												✓				
<i>Eucalyptus rubida*</i>																✓
Sub-canopy																
<i>Exocarpos cupressiformis</i>		✓		✓✓		✓			✓						✓	
<i>Acacia melanoxylon</i>	✓			✓	✓			✓	✓	✓		✓				
<i>Acacia pycnantha</i>				✓		✓	✓		✓	✓	✓					
Shrub high																
<i>Xanthorrhoea semiplana</i>	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
<i>Hakea rostrata</i>	✓						✓			✓	✓	✓				
<i>Banksia marginata</i>										✓	✓	✓				
<i>Pultenaea daphnoides</i>	✓			✓				✓		✓				✓	✓	✓
<i>Ixodia achillaeoides</i>															✓	
<i>Allocasuarina striata</i>											✓	✓				
<i>Leptospermum sp</i>								✓			✓	✓				
<i>Hakea carinata</i>												✓				
<i>Olearia ramulosa</i>			✓	✓		✓	✓			✓	✓	✓			✓	✓
<i>Acacia retinoides var retinoides (swamp variant)</i>										✓						
<i>Logania recurva</i>				✓												
<i>Olearia teretifolia</i>			✓	✓		✓	✓			✓	✓			✓	✓	
<i>Pultenaea involucrata</i>				✓												
<i>Leptospermum myrsenoides</i>				✓												
Shrub low																
<i>Pteridium esculentum</i>	✓	✓	✓		✓	✓	✓	✓	✓				✓	✓	✓	✓
<i>Dianella revoluta</i>				✓		✓	✓			✓	✓					
<i>Acacia myrtifolia</i>	✓		✓	✓		✓		✓		✓	✓			✓	✓	✓
<i>Platylobium obtusangulum</i>	✓	✓		✓		✓				✓	✓	✓				
<i>Leucopogon virgatus</i>			✓							✓	✓					
<i>Goodenia ovata</i>			✓					✓			✓		✓	✓	✓	✓
<i>Pimelea stricta</i>			✓												✓	✓
<i>Daviesia leptophylla</i>	✓		✓			✓	✓				✓		✓			
<i>Senecio hispidulus</i>			✓					✓				✓				
<i>Daviesia ulcifolia</i>				✓							✓	✓				
<i>Spyridium thymifolium</i>	✓			✓												
<i>Solanum laciniatum</i>												✓				
<i>Pultenaea scabra*</i>			✓													
<i>Grevillea lavandulacae</i>				✓												

Native Species	1	2	3	4	5	6	7	8	9	10	11	12	Add	F1	F2	F3
Ground cover																
<i>Platysace heterophylla</i>												✓				
<i>Stellaria palustris*</i>								✓								
<i>Burchardia umbellata</i>				✓												
<i>Thelymitra sp</i>	✓		✓	✓								✓	✓			
<i>Lobelia gibbosa</i>																✓
<i>Dipodium punctatum</i>			✓					✓					✓	✓	✓	✓
<i>Cassytha glabella</i>						✓	✓				✓	✓				
<i>Hibbertia riparia</i>				✓		✓	✓			✓	✓	✓	✓	✓	✓	✓
<i>Billardiera cymosa</i>	✓	✓		✓						✓						✓
<i>Helichrysum bracteatum</i>						✓										
<i>Gonocarpus tetragynus</i>		✓		✓				✓			✓	✓				
<i>Arthropodium strictum</i>		✓	✓			✓										
<i>Wahlenbergia luteola</i>		✓		✓											✓	✓
<i>Oxalis perennans</i>				✓												✓
<i>Billardiera bignoniacea</i>			✓	✓				✓			✓		✓		✓	
<i>Acrotriche depressa</i>						✓	✓									
<i>Clematis microphylla</i>														✓		
<i>Astroloma striata</i>				✓												
<i>Isopogon ceratophyllus</i>				✓							✓	✓				
<i>Geranium retrorsum</i>												✓				
<i>Baeckea ramosissima</i>				✓								✓				
<i>Ranunculus lappaceus</i>								✓								
<i>Acaena novae-zelandiae</i>					✓					✓	✓	✓	✓	✓	✓	
<i>Viola sieberiana</i>	✓	✓	✓	✓				✓					✓	✓		
<i>Scaevola albida</i>																✓
<i>Pultenaea pedunculata</i>												✓				
Maidenhair fern								✓								
<i>Thysanotus patersonii</i>				✓												
<i>Acrotriche serrulata</i>				✓												
<i>Tricoryne tenella</i>				✓												
<i>Kennedia prostrata</i>				✓												
<i>Diuris maculata</i>						✓										
<i>Calochilus robertsonii</i>		✓														
<i>Pterostylis nutans</i>	✓	✓				✓										
<i>Pterostylis sp</i>	✓	✓				✓										
<i>Acianthus caudatus</i>	✓	✓												✓		✓
<i>Glossodia major</i>	✓															
<i>Corybas despectans</i>	✓					✓										
<i>Caladenia carnea</i>										✓						
<i>Diuris longifolia</i>				✓												
Grasses/Sedges																
<i>Stipa sp</i>							✓									
<i>Juncus pallidum</i>			✓					✓				✓				
<i>Lepidosperma sp</i>		✓				✓		✓			✓	✓		✓		✓
<i>Danthonia sp</i>																✓

	1	2	3	4	5	6	7	8	9	10	11	12	Add	F1	F2	F3
Weed Species																
Canopy																
<i>Pinus radiata</i>	✓	✓		✓	✓	✓	✓	✓	✓	✓	✓	✓			✓	
Sub canopy																
Ash Tree						✓										
Shrub high																
<i>Genista monspessulana</i>					✓			✓	✓		✓					✓
Crab apple									✓							
Shrub low																
<i>Ulex europaeus</i>		✓	✓	✓		✓	✓	✓	✓		✓		✓			
<i>Solanum nigrum</i>									✓				✓			
<i>Rubus sp</i>									✓							
Ground cover																
<i>Hedera helix</i>																
<i>Vinca major</i>						✓										
<i>Ranunculus pachycarpus</i>													✓			
<i>Plantago sp</i>																
<i>Hypochoeris radicata</i>	✓		✓	✓	✓			✓	✓	✓	✓	✓	✓		✓	✓
<i>Scabiosa atropurpurea</i>					✓	✓	✓			✓	✓					
Thistle spp					✓				✓							
Grasses																
<i>Briza major?</i>		✓				✓										
<i>Phalaris sp</i>	✓		✓		✓			✓		✓	✓	✓	✓			

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