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Spatial organisation and habitat selection patterns of three marsupial herbivores within a patchy forestry environment

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Chapter 1

General Introduction

1.1 Landscapes and Habitat Quality

Structural features of a landscape can be characterised by habitat types, patch size, patch shape and configuration (Franklin and Forman 1987). They affect the availability of resources, particularly food and shelter within an area. This, in turn, has an impact upon survival and reproduction, distribution and abundance of animals, and species diversity. Management regimes that alter landscapes can therefore have profound consequences on the ecological properties of an area and its fauna (Franklin and Forman 1987).

Whether modifications to the landscape enhance or hinder species is presumably related to the effect they have on habitat quality. Examples of species that have benefited from changes in land use due to forestry practices, include moose *Alces alces* in Scandinavia (Cederlund and Markgren 1987), and white-tailed deer *Odocoileus virginianus* in North America (Alverson *et al.* 1988). In England, roe deer *Capreolus capreolus* and fallow deer *Dama dama* have benefited from land-clearing for agricultural purposes and the forestry practice of coppicing woodlands (Kay 1993). In Australia, red kangaroos *Macropus rufus* have benefited from the conversion of semi-arid land into pastoral land, because of the increased abundance of waterholes (Landsberg *et al.* 1999). Other species, however, have been hindered by fragmentation of the natural landscape. For example, logging in Indonesia is detrimental to several primates, including the Bornean gibbon *Hylobates muelleri* and orangutan *Pongo pygmaeus* (Wilson and Wilson 1975).

Habitat quality can be defined as the availability of resources (primarily food and shelter), possibly balanced against the risk of predation, to local fauna. These aspects of quality may be difficult to measure directly. As a result, researchers often examine animals' patterns of habitat use and selection as indirect measures of habitat quality. Such measures are based on the assumption that animals are free to choose which habitats they use, and that animals will use habitats with the greatest resources to promote survival and reproduction. Since at least the first assumption is not always true, interpretation of habitat use and selection as surrogates of habitat quality, therefore requires some discretion.

1.2 Habitat Use and Selection by Herbivores

Habitat use describes the relative proportions of different habitats utilised by animals, and provides information on where animals find and use resources. It can be difficult to understand the significance of results of habitat use, however, in isolation of landscape composition. For example, results may show that an animal feeds in a certain habitat, but it cannot be determined whether this is because food is particularly abundant within that habitat, or because the animal has little alternative.

Researchers use habitat selection, as distinct from simply habitat use, to try to understand the relative importance of habitats. This involves examining the relative proportions of habitats used, in relation to what was potentially available to an animal within the landscape. If the proportion of use is greater than the proportion available, the habitat is selected for. Conversely, if the proportion of use is less than the proportion available, the habitat is selected against. Habitat selection, however, makes several assumptions which are important to evaluate before deciding whether results are biologically meaningful (Garshelis 2000). First, habitat selection relies on the researcher defining a realistic area as potentially

'available' to an animal (Manly *et al.* 1993). This may be difficult because we have little understanding of how animals perceive their environment (Powell 2000). For example, an animal may not visit an area either because it is familiar with its lack of resources and is avoiding it; or because it simply does not know that it exists, as it lies beyond its cognitive map of the local landscape (Powell 2000). The second assumption of habitat selection is that animals are free to use all areas within the designated 'available' area. This is often not the case, because interspecific interactions (aggression or predation), and intra-specific interactions (territorial defence), may result in some areas being off-limits to animals. The third assumption is that high quality habitats are limited and, therefore, are used selectively. However, if an important, high quality habitat is abundant, then it may appear to be used as available, or even selected against, when in fact it may be critical to the animal's survival (Garshelis 2000).

Another potential problem with studying habitat selection is that results vary in relation to the scale of the study (Wiens 1989). Herbivores interact with forage resources at several ecological scales. These levels can be presented in a hierarchical manner, as orders of selection, ranging from geographical distribution of the animal species to the selection of home-range, feeding site, plant species and plant-part consumed by individuals (Johnson 1980a, Senft *et al.* 1987). The hierarchy incorporates decreasing spatial and temporal ranges with each subsequent scale or level. For example, the geographic distribution of a species is linked to its evolutionary ancestor, climate patterns and distribution of vegetation, all of which operate on a geological time-scale. An animal's home-range, on the other hand, is usually based on one or more years of positional data. A feeding area may be defined over a year or a season (especially for migratory animals), or a day. Selection of vegetation consumed within a feeding area is related to its availability at the higher feeding area level, and can be examined over a range of time frames: a year, a season, a 24 hour period, or a foraging bout. Finally, plant-parts consumed can relate to a foraging bout or a mouthful of food. Not surprisingly then, patterns of selection are known to vary in relation to the scale of the study (Wiens 1989). Each scale contributes to the understanding an animals' ecology, but a multi-scale approach is required to accelerate our knowledge of the dynamics operating within a species. A minimum of three sequential scales within a hierarchy of decisions has been recommended (O'Neill 1989), because this enables constraining effects of higher levels, and repercussions at lower levels, to be identified.

Despite all of the above potential limitations and assumptions involved with using habitat selection to determine habitat quality, being able to determine whether animals are selecting for or against a habitat and/or resource is useful. If interpreted well, it can be used to increase our understanding of feeding ecology, and provide information on interactions between animals and their environment.

1.3 The Tasmanian Forestry Landscape and its Herbivores

Tasmania is an island state in south-eastern Australia. Within Tasmania, of the 1 523 000 ha of State forest land, 42 644 ha is managed as hardwood and softwood plantations (excluding privately owned tree farms) (Wells 2000). The three main tree species used in Tasmanian plantations are *Eucalyptus globulus*, which is native to the state; *E. nitens*, which has been introduced from mainland Australia; and *Pinus radiata*, which is a non-indigenous species. Establishment of these plantations has resulted in landscapes that are becoming fragmented and patchy in nature. This pattern of fragmentation will continue, as another 35 000 ha will be converted from native forest into plantations within the next 10 years (Wells 2000). It is not known, however, what impact this change in the landscape will have upon native fauna.

Within Tasmania there are four major species within the mammalian herbivore guild. These are the red-necked or Bennett's wallaby (*Macropus rufogriseus rufogriseus*), red-bellied pademelon (*Thylogale billardierii*), common brushtail possum (*Trichosurus vulpecula*), and common wombat (*Vombatus ursinus*). Despite the common status and wide distribution of

these herbivores (Rounsevell *et al.* 1991), there is only limited information on their ecology within either natural or man-modified Tasmanian environments.

The Bennett's wallaby is a sub-species of the mainland red-necked wallaby, while the red-bellied pademelon became extinct on mainland Australia and is now restricted to Tasmania and its large off-shore islands. There is anecdotal evidence to suggest that wallaby and pademelon populations have increased with patchiness of the landscape through land-clearing practices (Calaby 1991, Driessen and Hocking 1992). The two macropod species are essentially nocturnal and solitary, and are classed as grazers or mixed-feeders, eating mostly grasses and forbs (Sanson 1989, Calaby 1991, Johnson and Rose 1991, Sprent 1997). Population densities have never been estimated for the two macropod species within native habitats. Densities are available for wallabies and pademelons at one forestry plantation over a three month period (Statham 1983), and for a location where forest abutted pastoral land (Johnson 1977). Home-range size and shape was also determined for a small number of wallabies and pademelons at this pastoral site (Johnson 1978, Mooney and Johnson 1979).

The common brushtail possum also occurs on mainland Australia. Possums are nocturnal, solitary animals and are generally classed as arboreal folivores (How 1991, Flannery 1994). Tasmanian possums, however, spend more time at ground level than their mainland counterparts (Statham 1994). The effects of changes in land use practices on Tasmania's possums have not been identified. A mark-recapture study of possums in burnt commercial hardwood forests, suggested that food resources were more abundant within these areas 4-6 years after a fire, compared with areas burnt 40-60 years before (Hocking 1981). Hocking (1981) also estimated population density and home-range size using trapping data. These values were relatively consistent with Australian mainland studies (see reviews by Green 1984 and Kerle 1984). Hocking's (1981) trapping study was unable to determine the extent of use of hollow-bearing trees by possums for diurnal refuge. Gibbons and Lindermayer (1996) suggest that possums may be hindered by forestry practices because the old, large trees targeted for tree-felling, are the same hollow-bearing trees used by possums for diurnal shelter.

Common wombats also occur on the Australian mainland. Very little information is available on wombats' ecology within Tasmanian environments while studies of mainland populations have shown that these animals are mainly nocturnal, but may forage and bask during the day in winter (McIlroy 1991). They are grazers, and their main food is native grasses, but they also eat sedges, mat-rushes, and the roots of trees and shrubs (McIlroy 1991). High wombat densities have been associated with areas where forest abuts pastoral land (McIlroy 1991). The distribution of wombats has generally declined since European settlement, probably due to the presence of livestock which compete for food, and compact the soil which makes burrowing difficult.

A fifth herbivore species, the introduced European rabbit (*Oryctolagus cuniculus*), can also be an important species within the forestry environment due to its impact upon seedlings. Rabbits are grazers and prefer green grass and herbaceous dicots (Myers 1991). Very little is known, however, about the general ecology of rabbits within the Tasmanian environment (Guiler 1968, Coleman *et al.* 1997).

Whether fragmenting Tasmania's native forest, through plantation establishment, benefits or hinders herbivores is important information. Management of these species is difficult with limited quantitative information on how these animals use their environment.

1.4 Impact of Herbivores on Commercial Tasmanian Forests

At the same time as current forestry land management may be having an impact on these herbivore populations; three of the four species (wallabies, pademelons and possums) are also the major vertebrate "pest" species of the Tasmanian forestry industry. These native herbivores browse seedlings within native regeneration and planted tree farms until seedlings

reach about 1 m in height (1-2 years after planting). This browsing can decrease growth rate, change tree form (e.g. forked trunks), and reduce seedling survival (Coleman *et al.* 1997). With little information on how herbivore species utilise the forestry environment, and in the absence of monitoring herbivore populations, the current management practice used by forest companies to minimise browsing damage to seedlings involves poisoning local herbivore populations with 1080 (sodium monofluoroacetate), immediately before planting. This strategy is based on the assumption that fewer animals will cause less browsing damage. Forestry companies are presently under considerable political pressure to stop the poisoning of native fauna. Furthermore, forestry landscapes can be manipulated, for example, in terms of plantation shape, size and proximity to certain habitats. Non-lethal alternatives to poisoning may be achievable through these sorts of manipulations (e.g. Bulinski and McArthur 2000, O'Reilly-Wapstra *et al.* 2002, Scott *et al.* 2002). The development of non-lethal alternatives is difficult, however, without baseline ecological information on the major herbivore species that contribute to the damage.

1.5 Project Design

This thesis is an investigation into the density of the four main herbivore species, and the use and selection of habitats by these species, within a patchy forestry environment. Emphasis has been placed on the three species that cause damage to seedlings in forestry plantations. Results from this research will increase our understanding of some of the fundamental aspects of the feeding ecology of these species. At the same time, it will contribute to the general conservation of Tasmania's herbivore populations, and form an important biological basis for the development of non-lethal alternatives to present poisoning practices for managing browsing damage. Finally, some of the information is highly relevant to improving current lethal practices of reducing mammal browsing to commercial seedlings within plantations.

There were four main aspects to the project. These were to determine: (1) herbivore densities overall and at the habitat level; (2) herbivore patterns of habitat use; (3) herbivore patterns of habitat selection; and (4) use of food resources by herbivores within one specific habitat.

Herbivore densities were determined using modified line-transect sampling methods, which I developed for this study. This enabled methods usually applied to daytime surveys in open habitat, to be used in nocturnal surveys of densely vegetated habitats.

Patterns of habitat use and selection were determined from both a spotlighting study, and a radio-telemetry study of wallabies, pademelons and possums. Both methods were used to obtain this information because each method has some limitations, but the two techniques, used together, provide complimentary data. This complimentation is largely because animal surveys collect data at the population level, while radio-telemetry studies collect data at the individual level.

Advantages of animal surveys are that they potentially offer large samples, and animals are theoretically encountered at random. Results therefore reflect patterns of the entire population. However, there are four main disadvantages to this method. First, surveys often involve quick glimpses of animals seen at a distance. Information on sex and age are therefore usually unobtainable even though this is biologically important information. Second, the observer's presence can affect animals' movements. This may positively bias estimates if animals are attracted to the observer, or negatively bias estimates if animals flush in response to the observer. Third, it is often not possible to survey in densely vegetated habitats as it is not possible to see the study animals. This limits results to relatively open habitats, which is undesirable because densely vegetated habitats are often used as shelter during resting periods and are therefore biologically important. Fourth, surveying animals is not possible if inactive animals can not be seen. For example, possums that nest inside tree-hollows during the day, can only be surveyed at night when they are active.

In contrast, the advantages of radio-telemetry are that it provides very detailed information on individuals of known sex and body mass, data can be collected remotely (reducing human impact), and data can be collected irrespective of an animal's activity. There are three main disadvantages of this method. Firstly, sample size is often small due to limited resources (time and money). Secondly, animals may not have been selected at random, due to biases introduced when animals were caught for transmitter attachment (e.g. trapping or darting bias). This may restrict extrapolation of selection patterns of individuals to those of the species as a whole. Thirdly, data have an associated error, because an animal's locations are estimated from triangulated bearings.

Patterns of habitat use and selection were examined at three sequential spatio-temporal scales, within a hierarchy of decisions. These were: (1) location of home-range within the landscape (Johnson's [1980a] second-order selection); (2) feeding area within the home-range (Johnson's [1980a] third-order selection); and (3) vegetation consumed within the feeding area (Johnson's [1980a] fourth-order selection). The radio-telemetry study was used to examine Scales 1 and 2 at the individual animal level, and obtain daytime information on animals' behaviour. Spotlighting surveys were used to examine Scale 2 for the entire herbivore community, at the population level. These surveys were also used to estimate herbivore densities for the overall area and individual habitats. Fenced and unfenced vegetation plots, located within a highly used habitat (the young plantation), were monitored over time to examine Scale 3.

A particular applied aspect of this project involved examining how wallabies, pademelons and possums responded to the planting of commercial *Eucalyptus nitens* seedlings on a prepared site. To do this, I examined the use of this patchy forestry environment by these species from approximately 6 months before planting, until 8 months after planting. A multi-year approach was not taken in this case, primarily due to time constraints, but also because forestry operations that occur during plantation establishment (e.g. planting of seedlings and herbiciding) have a greater impact upon results than time, making comparisons with results from subsequent years of limited value.

In addition to the study outlined above, this thesis contains two chapters on the effects of a 1080 poisoning operation, which targeted wallabies, pademelons and possums at this site. This operation was carried out by forestry staff as part of normal forestry practices while preparing to re-plant the study site at the end of my research. The plantation was considered to have failed because of high mortality and low growth rates of surviving trees. This poisoning operation was not part of the original project, but provided a unique opportunity to study the effects of poisoning on a known herbivore population. This work is included within this thesis, because part of the process of developing non-lethal alternatives to poisoning, involves understanding effects of the present lethal control method.

1.6 Thesis Aims

The aims of this thesis were to:

- (1) quantify herbivore population densities at the overall and habitat scales,
- (2) describe spatial organisation of these three herbivore species at a range of temporal and spatial scales,
- (3) identify important habitats used and selected by wallabies, pademelons and possums,
- (4) understand the response of herbivores to the planting of commercial seedlings, and
- (5) quantify the impact of the herbivores on commercial seedlings and other vegetation, on a newly established plantation.

1.7 Thesis Structure

The work in relation to the aims above (Section 1.6) is presented in a descending order of scale, as follows: (1) spotlighting surveys of the herbivore community [population level]; (2) radio-tracking wallabies, pademelons and possums [individual level]; (3) monitoring vegetation plots on the young plantation [within habitat level]; and (4) monitoring commercial seedlings on the young plantation. Within each chapter, results of the ecological aspect of the work are described first, with management implications given at the end. Chapters were written as independent manuscripts which, therefore, may sometimes appear repetitive in their description of the study site and methods used. The contributions of co-authors are described within the relevant chapters and reprints of published papers are attached inside the back cover. References are listed at the end of the thesis.

A brief outline of each chapter follows. Chapter 2 describes the study site within the patchy forestry environment, and the three main study species (wallaby, pademelon and possum). Chapter 3 describes how standard line-transect sampling methods were modified for the requirements of this project (surveying in darkness and within densely vegetated habitats). Chapter 4 quantifies herbivore population densities, and describes patterns of habitat selection at the population level. Chapter 5 describes the effects of a 1080 operation on local herbivore densities. Chapter 6 documents the accuracy of the radio-telemetry system within this environment. Chapter 7 quantifies wallaby, pademelon and possum home-range size and shape. Chapter 8 examines herbivore patterns of habitat use and selection at the individual level. Chapter 9 describes the effects of the 1080 operation on radio-collared animals used in the biological study. Chapter 10 quantifies the effects of foraging herbivores on ground vegetation on the young plantation. Chapter 11 quantifies the effects of mammal browsing and other factors on the commercial *Eucalyptus nitens* seedlings on the young plantation. Chapter 12 provides a synthesis of results from previous chapters.

Chapter 2

Study Site & Animals

2.1 Study Site

The study site is located within the forest company Gunns' (formerly North Forest Products) 'Surrey Hills' Tree Farm Estate, in north-west Tasmania (41° 28' S, 145° 48' E). With an area of 1080 km², this estate represents Australia's largest eucalypt tree farm, and exploitation of this region began around 1950 (Orme 1971). Surrey Hills is a basalt plateau which is drained by several major rivers. It has an undulating topography, with elevation ranging between 520 and 670 m above sea level (Orme 1971). Soils are mainly krazonzems and deep yellow podzolics, derived from basalt (Orme 1971).

The study site itself has an elevation of 640 m a.s.l., with a relatively flat topography. Mean annual rainfall for this region is 2200 mm, with approximately 250 rain days a year (G. Holtz unpublished data). Frost and snow showers can be experienced at any time of the year, although heavier snow falls are restricted to late winter and early spring (Orme 1971). Snow falls are usually light, with snow melting within a few days of deposition. Mean minimum annual temperature is 4° C and mean annual maximum is 12° C (G. Holtz unpublished data). Generally, summers are mild, and winters cold with heavy frosts and occasional snow. During the course of the fieldstudy, temperatures ranged from -7° to 32° C (G. Holtz unpublished data).

As a result of the commercial forestry activities that have taken place at Surrey Hills, the environment is highly fragmented, with abrupt boundaries between patches of habitat (Plate 1). Five habitats dominated the study site: (1) a prepared site that was planted during the fieldstudy (Gunns' plantation D97LV102c; hereafter referred to as the 'young plantation'); (2) older plantations of *Eucalyptus nitens* (Gunns' plantations D92LV101a, D90LV101b, D91LV102a and D92LV102b; 5-7 years of age and approximately 5 m tall); (3) native forest (rainforest and wet eucalypt forest); (4) grassland; and (5) harvested uncleared land, consisting of scrub and fallen vegetation (Figure 2.1, Plates 2.2 and 2.3).

2.1.1 Geographic Information System

A geographic information system (GIS) was developed for the study site. The ArcView® GIS was based upon Gunns' GIS which contained spatial information on plantation boundaries, roads and waterways. Gunns' data for plantation boundaries and roads were collected using differential global positioning systems (GPS) and information digitized from colour aerial photographs (scale 1:10 000, J. Dick *pers. comm.*). Information on waterways had been digitized from the 1:25 000 Tasmanian topographic map series (Guildford, Pearse and Lea maps, numbers 3841, 3840 and 4040, respectively) (J. Dick *pers. comm.*). As Gunns had no spatial information on the native vegetation within the study site, this information was gathered using colour aerial photographs (scale 1: 6 200, C. Mann *pers. comm.*). Habitat boundaries were identified using a stereoscope, and digitized into the GIS. Vegetation was classed into three habitat types: native forest (tree canopy present), grassland (tree canopy absent) and uncleared harvested land. Where habitat boundaries were unclear from aerial photos, points were ground-checked using differential GPS and corrected accordingly. Data was georeferenced using six grid references. When extracting information from the GIS, a 'patch' was defined as a polygon of a single habitat type.

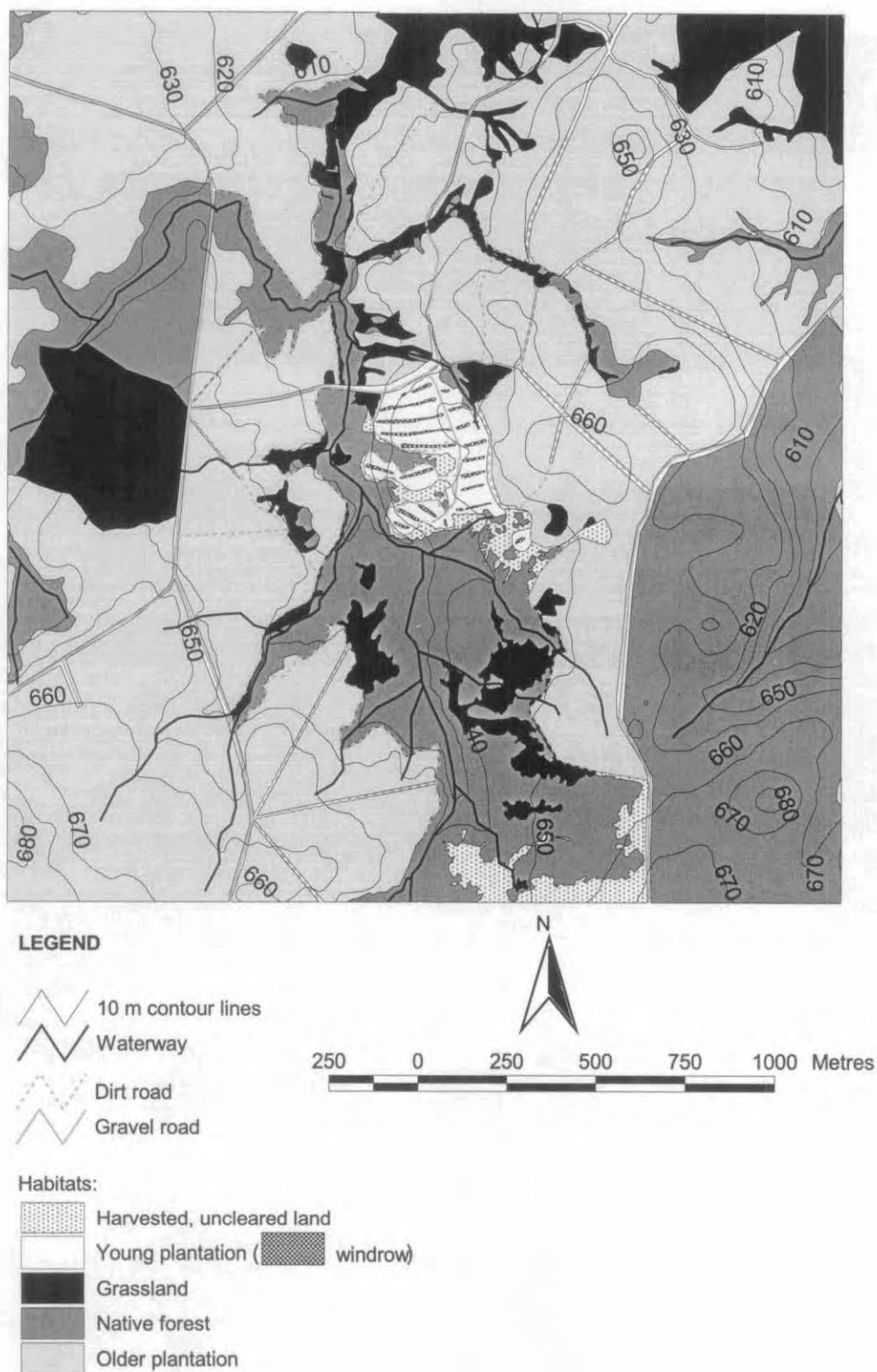


Figure 2.1 A map of the Surrey Hills study site, showing the location of the young plantation in the centre, and the fragmented nature of the other habitats.



Plate 2.1 The heterogeneous nature of the different habitat types within this fragmented forestry environment, and the abrupt boundaries between patches of habitat.

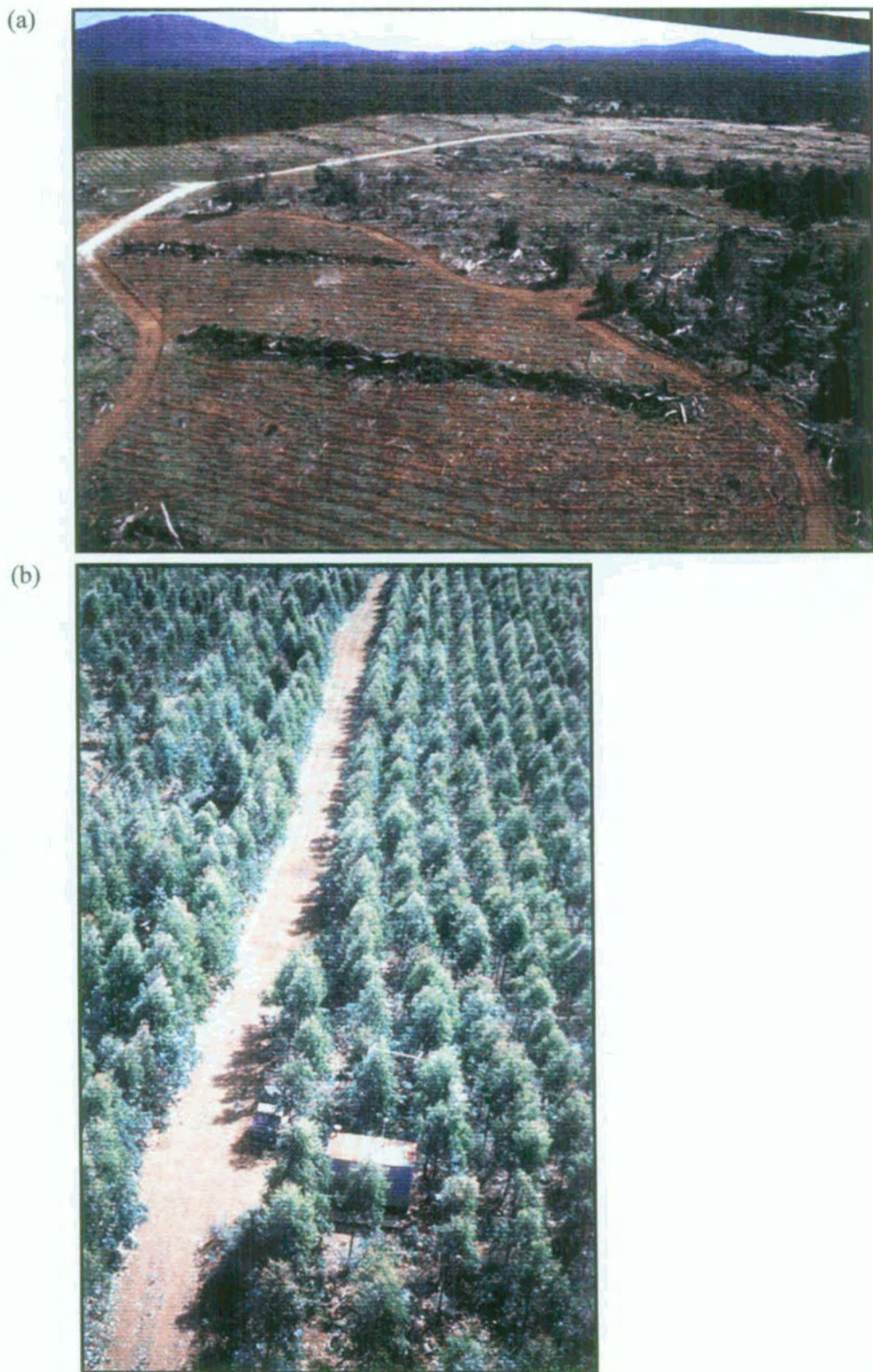


Plate 2.2 Aerial photos of (a) the young *Eucalyptus nitens* plantation and
(b) older *E. nitens* plantation.

(a)



(b)



(c)



Plate 2.3

Aerial photographs of (a) native forest, (b) grassland and (c) harvested uncleared land habitats.

2.1.2 Forestry operations on the young plantation

The young plantation was the focal point of the study site, and three forestry operations took place during the data collection period. These were: (1) the planting of seedlings, (2) herbiciding and (3) poisoning; all carried out by Gunns' staff. This section describes the history of these, and other forestry operations, that took place on the young plantation both before and during the data collection period. A time-line of these events is shown in Figure 2.2.

The young plantation had been selectively logged some time before clearing in March 1996. At the time of clearing, vegetation consisted of wet eucalypt forest and grassland. In the process of clearing, windrows (piles of wood debris) were created at 80 m intervals, running parallel across the entire plantation. Windrows were usually less than 2 m high and consisted of medium to large logs. In preparation of planting, the site was ripped with a bulldozer to create planting lines, and windrows were burnt to increase planting area, add nutrients into the soil in the form of ash, and reduce their attractiveness as shelter to herbivores.

Eucalyptus nitens seedlings (approximately 20 cm in height) were planted in November 1997, eight months into the study. Approximately 1100 seedlings were planted per hectare and the entire site was planted in a single day. Seedlings were container-stock, meaning they were grown in the nursery in pots and transferred directly into the field.

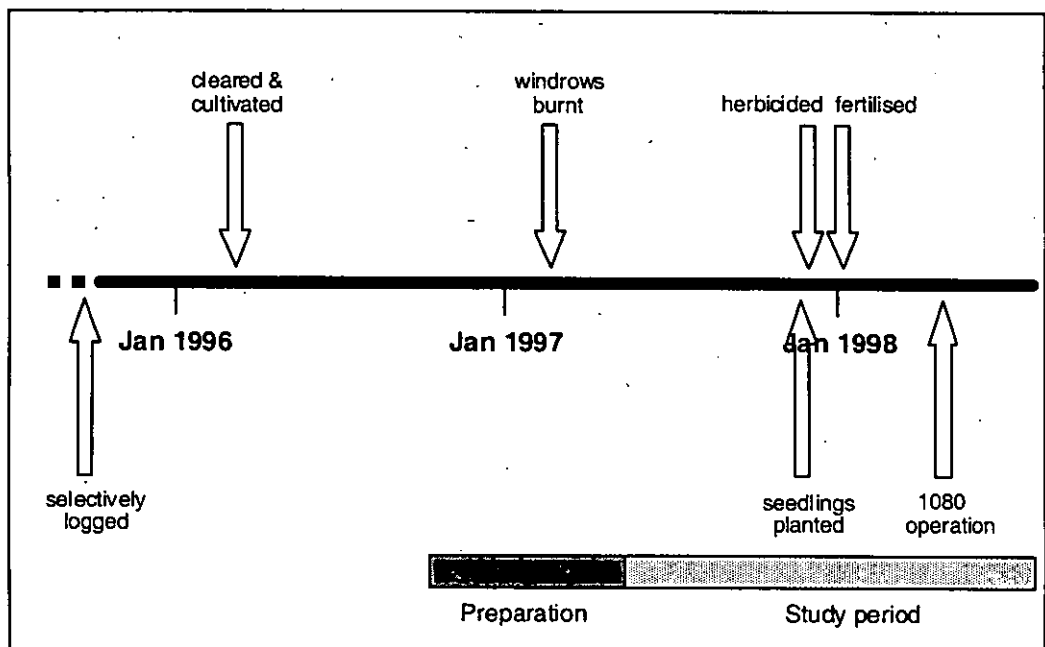


Figure 2.2 Time-line of forestry operations that took place on the young plantation over the course of the field study.

In contrast to common forestry practices, this site was not treated with 1080 (sodium monofluoroacetate) poison before planting. Compound 1080 is normally used to reduce local herbivore populations immediately before planting, as it is assumed that fewer herbivores will result in less browsing damage to seedlings. It was not used in the present study, however, as one of the main aims of this work was to investigate the relationship between herbivores and the forestry environment in the absence of poisoning. At present very little is understood about how these animals utilise the forestry environment, yet this information is critical for the development of non-lethal alternatives for reducing browsing damage to seedlings. Consequently, no poisoning operations were conducted within a 2 km radius of

the young plantation for the duration of the field study, including an eight month period before the start of data collection.

Common herbiciding practices were also modified, to meet the aims of this study. Herbiciding operations are generally considered necessary to reduce competition between seedlings and weeds for limited water and nutrients. Standard herbiciding practice at Surrey Hills was as follows. Herbiciding took place up to six weeks before planting, and two types of herbicide were administered during aerial spraying: Gesatop® and Roundup®. Gesatop® (Simazine) is a residual herbicide that is washed into the soil (up to 10 cm deep) and kills pre- and post- emergent vegetation. It is used to target broadleaf and grass weeds. Roundup® (Glyphosate) is a non-residual contact herbicide, used to kill a variety of weeds. Within 7-10 days of herbiciding, plants usually wilt and turn yellow. Plants would be expected to be brown, and essentially dead, at four weeks. It is common to plant eucalypt seedlings when other vegetation is still dying but not dead. During the present study, however, herbiciding was restricted to spot applications of Roundup®, one week after planting. All vegetation within a 0.5 m radius of each seedling was sprayed during this process, while seedlings were shielded by plastic containers. This operation was conducted after planting rather than before, due to logistical constraints relating to the availability of ground crew.

Spot herbiciding was used in preference to common herbiciding practices for two reasons. Firstly, blanket herbiciding would have killed the weed cover thereby introducing a confounding effect into the present study, whereby animals' response to the introduction of seedlings and simultaneous death to other forage, could not have been differentiated. Secondly, there is increasing pressure for forestry companies to reduce their use of herbicides, so in anticipation of this, it would be useful to investigate how alternative herbiciding regimes (such as spot-herbiciding) affect use of newly established plantations by herbivores.

Shortly after planting, seedlings were fertilised to maximise growth. Seedlings were given a spot application of slow-release Pivot® fertiliser (20N:18P:0K) six weeks after planting. Granules were placed 20-30 cm from the seedling and were washed into the ground by rain, within three weeks of application.

The final forestry operation conducted at this site was a 1080 poisoning program. This event was not carried out as part of this study, but as a normal procedure in preparation for replanting the young plantation. This provided a unique opportunity to document the effects of a 1080 operation on a population of known individuals. Consequently, data collection continued and this work is presented in Chapters 5 and 9, where the 1080 operation is described in detail.

2.2 Study Animals

In the following sections, a brief summary is given of the ecology of the three marsupial species that were the focus of this study.

2.2.1 Bennett's wallaby *Macropus rufogriseus rufogriseus*

The Bennett's wallaby is a medium-sized, native marsupial belonging to the Family Macropodidae (Plate 2.4). The species is sexually dimorphic with average male body mass around 20 kg, and female body mass around 14 kg (Calaby 1991). Bennett's wallabies occur throughout Tasmania and many of its larger offshore islands. The range of its closely related sub-species, the red-necked wallaby (*Macropus rufogriseus banksianus*), extends from south-eastern Queensland to south-eastern South Australia (Calaby 1991). The species is abundant throughout its range, and is one of the few marsupial species to have benefited from land-clearing (Calaby and Grigg 1989; Johnson *et al.* 1989). On the Australian

(a) Bennett's wallaby



(b) Red-bellied pademelon



(c) Brushtail possum



Plate 2.4

The three study species: (a) Bennett's wallaby (*Macropus rufogriseus rufogriseus*, photo courtesy of J.Bulinski), (b) red-bellied pademelon (*Thylogale billardierii*, photo courtesy of C.McArthur), and (c) brushtail possum (*Trichosurus vulpecula*, photo courtesy of C.McArthur).

mainland, *M. rufogriseus* occurs mainly in open sclerophyll forest, while in Tasmania it is found in a variety of habitats, including rainforest, open forest, heathlands and sedgeland (Statham 1983).

Macropus rufogriseus is a relatively solitary, nocturnal macropod. Animals forage predominantly at night, and spend most of the day sheltering in cover habitat within forest (Calaby 1991). Where bushland abuts pasture, this species may be highly abundant (Jarman *et al.* 1987), moving short distances to feed in open areas at night (Mooney and Johnson 1979). These animals are primarily grazers, feeding on monocotyledons such as grasses, and soft-leaved dicotyledons (Calaby 1991, Statham 1983, Jarman and Phillips 1989, Sprent 1997). Home-range size has been reported at around 100 ha in Tasmania (Mooney and Johnson 1979), and 16 ha on the mainland (Johnson 1987).

Bennett's wallabies' breeding patterns are quite different to those of the mainland sub-species. Bennett's wallabies have a well-defined breeding season with a peak of births occurring in February and March, while the mainland sub-species breeds throughout the year (Calaby 1991). As with other macropod species, *M. rufogriseus* gives birth to only one young at a time. Gestation takes around 30 days and pouch life approximately 280 days (Calaby 1991). Young continue to suckle, however, until 12 to 17 months of age (Calaby 1991). Females reach sexual maturity between 11 and 21 months of age, and males at around 12 months (Lee and Ward 1989).

2.2.2 Red-bellied pademelon *Thylogale billardierii*

The red-bellied pademelon is another native marsupial belonging to the Family Macropodidae (Plate 2.4). The species is sexually dimorphic, with average male body mass around 7 kg and female body mass around 4 kg (Johnson and Rose 1991). It is common and abundant throughout Tasmania and the larger islands of Bass Strait, but extinct on the mainland where it was formerly distributed in south-eastern South Australia and Victoria (Johnson and Rose 1991). Red-bellied pademelons occur in a variety of habitats but favour densely vegetated areas such as wet sclerophyll forest and rainforest, which provide suitable cover for daytime shelter (Green 1973, Johnson and Rose 1991). This species is also thought to have benefited from land clearing activities, as population densities are thought to have increased over time (Frith 1973, Driessen and Hocking 1992).

The pademelon is essentially nocturnal and forages predominantly at night (Johnson and Rose 1991). In areas where forest abuts cleared land, animals may move short distances onto open areas to feed at night (Johnson 1978). Pademelons are considered a mixed-feeder, with browsing grade dentition (Sanson 1989). Consequently, this species feeds on a wide range of plant species, including grasses, forbs, shrubs and browse (Statham 1983, Sprent 1997). Male pademelon home-range has been estimated at 156 ha and elongate in shape (Johnson 1978).

Breeding occurs throughout the year but peaks from April to June. Gestation takes around 30 days, and pouch life is approximately 200 days (Johnson and Rose 1991). Sexual maturity is reached at around 14 months (Johnson and Rose 1991).

2.2.3 Common brushtail possum *Trichosurus vulpecula*

The brushtail possum is a medium-sized native arboreal marsupial, belonging to the Family Phalangeridae (Plate 2.4). Average male body mass is between 2-4.5 kg, and female body mass between 1.5-3.5 kg (How 1991). Its range is extensive, and one of the largest for any Australian marsupial (Kerle 1984). Three sub-species are described. *Trichosurus vulpecula fuliginosus* is the endemic Tasmanian sub-species, and is abundant throughout the state (How 1991). Brushtail possums occur in most areas where there are trees, favouring open forests and woodland.

The brushtail possum is essentially nocturnal, foraging mostly at night and resting in dens during the day (How 1991). Dens usually comprise tree hollows or logs, which animals will defend from other possums (Flannery 1984). Possums forage both in the canopy and on the ground, although Tasmanian brushtails are mostly ground-feeders (Statham 1984). This species is a browser, feeding predominantly on soft and hard-leaved dicotyledons (Statham 1984, Fitzgerald 1984, Green 1984, Kerle 1984, Evans 1992, and Owen and Norton 1995). Their diet often includes *Eucalyptus* leaves, but individuals have limited intake as they can not completely detoxify tannins (Hume 1999). A wide range of other foodstuffs may also occasionally be eaten, these include fruit, buds, bark, insects and meat (Kerle 1984).

Possum home-range size varies between habitat types. Australian studies have reported home-ranges between 0.1-20.6 ha, while in New Zealand, where possums are introduced, populations have estimated ranges of 0.03-45.8 ha (see review by Green 1984). In some populations, males may establish territories by scent-marking with secretions produced from chin, chest and anal glands (Flannery 1994). Brushtail possums usually give birth to a single young during autumn (Hocking 1981). Gestation is around 18 days and pouch life around 4-5 months (How 1991). Another 1-2 months are spent suckling and riding on the mother's back before weaning is complete (Flannery 1994). Female sexual maturity is reached at around 23 months of age (Hocking 1981).

Chapter 3

Population Surveys: Spotlighting Methods

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Dr Colin Southwell helped with the design of the pilot study and the final sampling method. He also carried out the analysis on the differential detection probabilities of the macropods on the young plantation. Dr Clare McArthur provided financial and logistical support, and edited manuscripts.

3.1 Introduction

Having reliable information on the abundance of species is important for effective management of wildlife populations. In some situations, such as monitoring trends in abundance, an index of relative abundance is sufficient, provided the index is robust to variation in absolute abundance across time and space. In some circumstances, such as calculation of safe harvesting rates or grazing pressure, abundance indices are insufficient and estimates of absolute abundance are necessary. Within Australia, estimates of absolute abundance have been used to calculate safe harvesting rates for several species of macropod: red kangaroos (*M. rufus*, Caughley and Grigg 1981), eastern and western grey kangaroos (*M. giganteus* and *M. fuliginosus*, Southwell *et al.* 1995, Caughley and Grigg 1981), common wallaroos (*M. robustus*, Southwell *et al.* 1995) and whiptail wallabies (*M. parryi*, Southwell *et al.* 1995).

In Tasmania, the two most abundant species of macropod, Bennett's wallaby *M. rufogriseus rufogriseus* and red-bellied pademelon *T. billardieri*, conflict with human activities and are managed to reduce their impact on agricultural, grazing and forestry practices. To assess the impact of management, such as harvesting and poisoning, of these species, state-wide roadside counts are used to monitor broad-scale changes in relative abundance (Driessen and Hocking 1992). While estimates of absolute abundance at both large and small scales would be useful to estimate the actual size of the harvest or the local impact of agricultural or forestry operations, none are available.

Wallabies and pademelons have characteristics that make estimation of both relative and absolute abundance very difficult; they are relatively small, mostly or entirely nocturnal, shelter during daylight in heavy cover, and feed at night in habitats with both light and heavy cover (Johnson and Rose 1991, Calaby 1991). Heavy cover reduces sightability and is noisy to travel through, causing evasive movement of animals before they are detected, with a consequent underestimation of abundance. Occupation of habitats with both light and heavy cover when feeding at night can lead to confounding of both relative and absolute abundance estimates unless the estimation methods are robust to large changes in sightability. One way to accommodate changes in sightability is to use line-transect sampling. This method requires measurement of perpendicular distances to sighted animals. Usually a rangefinder and compass are used to estimate perpendicular distance, but it is impractical to use these tools at night when the animals are most active.

Studies aimed at evaluating ground survey methods for estimating the abundance of macropods have focused on large species that are crepuscular and occupy habitats with little

cover (Coulson and Raines 1985, Southwell 1994). Despite the problems outlined above, however, line-transect sampling offers the potential to estimate absolute abundance of wallabies and pademelon, as well as other species with similar characteristics. In this chapter, I describe modifications to the line-transect methods previously used for large macropods and discuss the perceived success of these modifications.

3.2 Materials and Methods

3.2.1 Study site

The 428 ha study area was located within the forestry company Gunns' 'Surrey Hills' Tree Farm, in north-west Tasmania (41° 28' S, 145° 48' E). Four habitats dominated the site: older *Eucalyptus nitens* plantation, native forest, grassland, and a prepared site that was planted during the fieldstudy (hereafter referred to as 'the young plantation'), see Figure 3.1.

Vegetation within older plantations consisted of *Eucalyptus nitens*, 5-7 years of age and approximately 5 m in height. Plantation trees were often 'bushy' due to low side branches, but leaves were located on branch tips and there was little understorey. Trees were 2.5 m apart along tree rows and 3 m apart between tree rows. Visibility was generally good along and across tree rows. Windrows (parallel lines of wood debris heaped < 2 m high) were present at 40-80 m intervals. Understorey species mainly consisted of rush (*Juncus pallidus*), bracken (*Pteridium esculentum*), native grasses (*Poa spp.*) and introduced herbaceous dicots. Older plantation was noisy to travel through due to the eucalypts' low side branches and dry leaves and twigs underfoot.

Vegetation within native forest consisted of interspersed patches of callidendrous rainforest (main species: *Nothofagus cunninghamii*) and wet eucalypt forest (main species: *E. delegatensis*). Visibility within callidendrous rainforest was generally very good. Patches of wet eucalypt forest were usually densely vegetated, with complex understorey. Travelling through native forest could be very quiet (rainforest) or extremely noisy (eucalypt forest). Streams were also frequently encountered, as most patches of native forest had been left as streamside reserves.

Vegetation in grassland mainly consisted of montane tussocks (*Poa spp.*) and pasture grasses. Minor plants included *J. pallidus*, *Gahnia grandis*, *Hakea spp.* and *E. delegatensis*. Tussock height ranged from 0.2-1.0 m. Visibility largely depended upon the particular grassland sampled, as some were less densely vegetated than others. Visibility at ground level was poor to fair. Travelling through grassland produced little noise.

Vegetation on the young plantation consisted of introduced herbaceous dicots, pasture grasses, native grasses and occasional shrubs (*Hakea* and *E. delegatensis* seedlings). Visibility on the young plantation was excellent but interrupted by windrows spaced at 40-80 m intervals. *Eucalyptus nitens* seedlings were planted eight months after the study began. Seedlings were approximately 20 cm tall and plants within a 0.5 m radius of seedlings were sprayed with a herbicide (Roundup®) one week after planting. Travelling on the young plantation produced little noise.

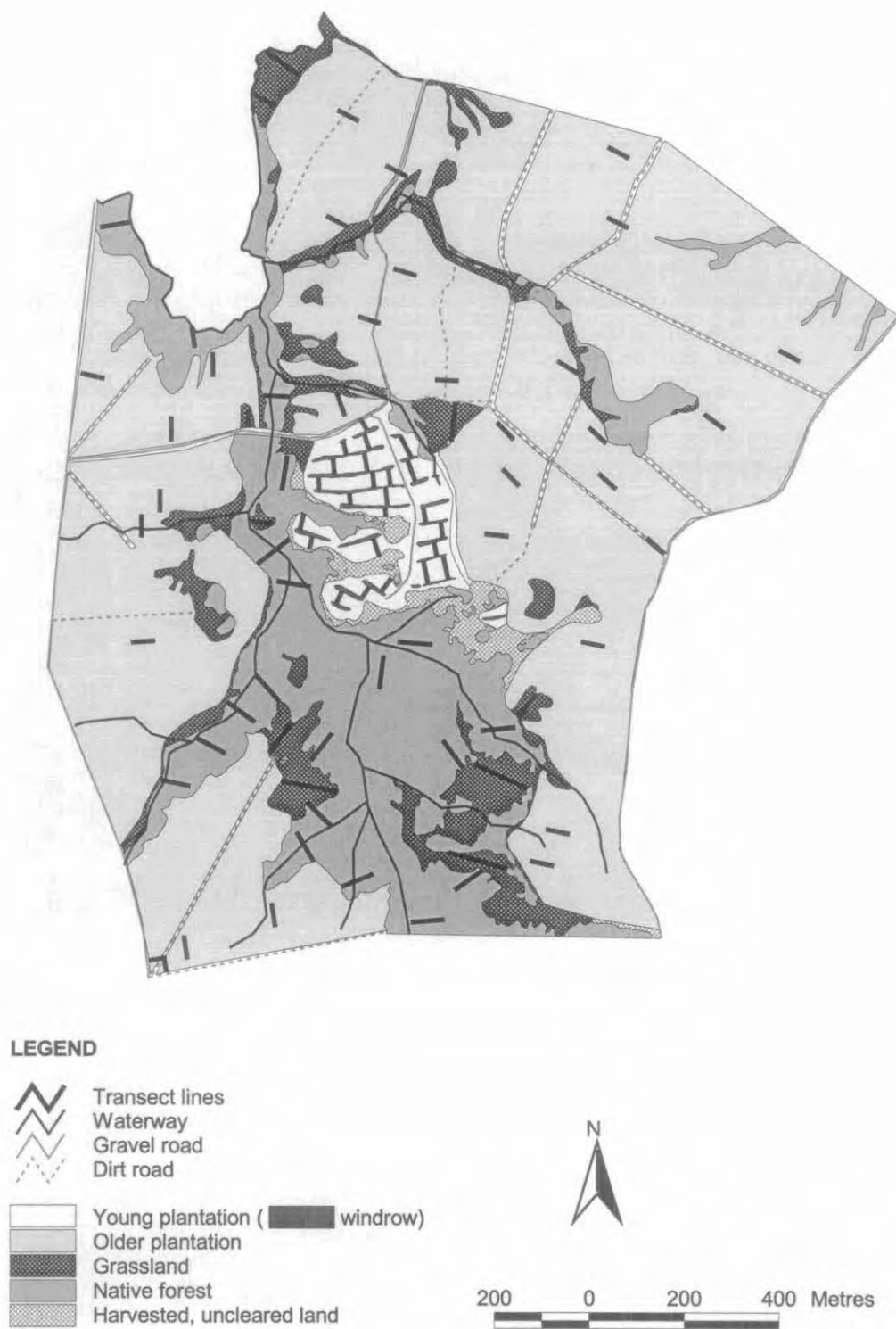


Figure 3.1 Map of the 428 ha study site showing locations of the permanent 84 transect lines location within the four main habitat types: young plantation, older plantation, grassland and native forest.

3.2.2 Transect lines

There was a total of 84 permanent transect lines (Figure 3.1). The number and length of transect lines varied between habitats due to differences in size, shape and number of patches of each habitat type (Table 3.1). Within each of the four habitat types, transect lines were located at random, except for the orientation. The young plantation was heavily sampled for its size, as this was the main habitat of interest. Spotlighting surveys were not conducted in the fifth habitat type (uncleared harvested land) due to extremely restricted visibility (often less than 1 m).

Table 3.1 Properties of transect lines in different habitats and the number of observations recorded for the macropod species during fieldtrips used in estimating probabilities of detection (Fieldtrips 1-11). Upper distances for each category are given in metres. Species/habitat combinations, where adequate sample sizes allowed line-transect analysis, are indicated with an asterisk.

	Young plantation	Grassland	Older plantation	Native forest
No. of transects	24	10	30	20
Length of each transect line (m)	40-70	60-180	70-80	80-100
Total length of transect line (km)	1.93	1.18	2.04	1.90
Upper distance for category 1	5	5	3	5
Upper distance for category 2	10	10	6	10
Upper distance for category 3	20	20	9	15
Upper distance for category 4	30	30	12	25
Upper distance for category 5	40	40	15	35
Upper distance for category 6	50	50	25	∞
Upper distance for category 7	70	∞	∞	
Upper distance for category 8	∞			
No. of wallabies sighted	226*	55*	5	4
No. of pademelons sighted	538*	244*	55*	14*

Transect lines were cleared in native forest to reduce noise produced while travelling. Cleared paths were only wide enough to accommodate one person, and obstacles such as large logs were left *in situ*. Footbridges (with animal barriers) and duck-boarding were constructed where necessary to enable quiet travel over waterways. In older plantations, a 70 m segment of tree row was used as the transect line. These trees were pruned of low side branches to reduce noise produced while travelling.

Pilot studies on the young plantation indicated potential problems with animals flushing in response to humans. To compensate for this, windrows were used to screen approaching observers, by locating transect lines perpendicular to windrows. Starting points were then approached from behind these structures. Additional steps taken to reduce the impact of humans in all habitats included familiarizing animals with spotlighting activities before the first data collection, and only using small headtorches while travelling between transect lines, when lights were pointed downwards so the bulb was not directly visible to animals facing head on.

Line-transect sampling uses data on the perpendicular distance of sighted animals from the transect line to derive a probability of detection function (Buckland *et al.* 1993). Perpendicular distance data may be measured exactly for each sighted animal (ungrouped data), or recorded as a distance category (grouped data). As previously discussed, it is impractical to use a rangefinder and compass at night to measure perpendicular distance, hence distance categories were used. Consequently, a permanent grid of markers was established for each transect line to identify distance categories adjacent to the transect line.

Pilot studies, conducted during the day and night, were important for determining the width of each distance category. Densely vegetated habitats required narrower categories and categories needed to be narrower closer to the line. Distance category markers were constructed from wooden garden stakes (1.75 m in height) and colour-coded road-side reflectors (de Neefe Signs Ltd., Tasmania), positioned at 10 m intervals along the transect line and varying distances away from the line (depending on habitat type, see Table 3.1). A minimum of six distance categories were used. Within older plantations, tree rows lying parallel to the transect line were used to mark grid lines.

3.2.3 Survey

Data were collected over 11 fieldtrips between May 1997 and July 1998. Each fieldtrip consisted of six nights spotlighting (2-3 nights apart), taking place over 17 days. All fieldtrips incorporated full-moon. Fieldtrips were approximately four weeks apart with the exception of Fieldtrips 5-6 and 9-10, which took place immediately before and after forestry operations (planting [Trips 5-6] and poisoning the young plantation with Compound 1080 (sodium monofluoroacetate) [Trips 9-11]).

A two-person team walking in single file at 1 km.h⁻¹ surveyed the line-transects at night. Both ground and canopy were searched for animals. Transect lines were divided into six groups of 14; one group was walked per night. Data collection commenced one hour after sunset and was completed within six hours.

In older plantations, native forest and the young plantation, only one person searched with a 100 W spotlight. The second person scribed and checked along the line with a headtorch while the first person was looking out to the sides. In grasslands, two people searched for animals with spotlights to compensate for the presence of low dense tussocks along the line. The front person focused on the line and out to the first two distance categories and the second person searched out to the sides, in distance categories 3-7.

For consistency of data collection, the same person (K. le Mar) was always the front person, and the primary identifier of animals. The second person and scribe, was one of two people that worked on the project for six months at a time. These two workers had received extensive training before their first fieldtrip.

Data for individual transect lines on the young plantation were collected from two positions: on top of the windrow, directly above the start of the line (for < 3 min.), followed by walking the transect line. The scribe remained on the windrow observing animals with a spotlight while the first person descended the windrow. When the first person re-commenced spotlighting, the scribe descended the windrow and joined the spotlihter. This system enabled animals' movements to be monitored during the front person's descent, so that animals initially recorded from the windrow were not re-recorded while walking the line. Data collected from the two positions were pooled.

Surveys were conducted in both clear and sub-optimal weather (light-medium rain and light snowfalls). Spotlighting in sub-optimal weather was not avoided as light rain was characteristic of the site, while medium rain and snowfalls would occasionally occur once surveying a transect line had already commenced. Under such circumstances, data collection only continued if sightability was still satisfactory to the furthest distance category and animals did not flush for shelter. Typically, local fauna were observed to continue feeding irrespective of weather conditions.

3.2.4 Data analysis

Data were pooled across transect lines within each habitat type. Histograms of the number of animals in each distance category in each habitat were examined for evidence of evasive movement before detection. Animals moving in response to spotlighters will typically flush away from the transect line. When this occurs, the histogram has a peak or spike at some distance away from zero (Buckland *et al.* 1993).

The use of markers for distance estimation resulted in grouped data and no further grouping was attempted in analysis. Data were truncated to strip half-width distances of 70 m for the young plantation, 50 m for grassland, 35 m for native forest and 21 m for older plantation. Strip half-width (w) is the width from the transect line to the outermost edge of the survey area on one side of the line, therefore, half of the total width sampled for a transect line. Animals sighted along the cleared paths in older plantation and native forest transect lines were excluded from the data sets as these features may have resulted in increased sightability along the line. For the first distance category, the closest data were taken from the edge of the track (approximately 0.15 m from the centre of the path).

Probabilities of detection were calculated from distance data using line-transect methodology (Buckland *et al.* 1993) and the computer program DISTANCE (Laake *et al.* 1994). Data were analysed for individuals rather than clusters (Buckland *et al.* 1993) as wallabies and pademelons are essentially solitary animals that may feed in close proximity but do not behave as a unit (Calaby 1991). Akaike's Information Criterion was used to select the best of four candidate line-transect models (key function/adjustment: uniform/cosine, uniform/polynomial, half-normal/hermite and hazard-rate/cosine) (Buckland *et al.* 1993).

Tests for differential detection probabilities between wallabies and pademelons within the same habitats, wallabies between two habitats and pademelons between four habitats, were conducted by using data sets with a common truncation distance (50 m). Differences in estimated detection probabilities were tested for significance using z-tests, where x_1 is the probability of detecting a species in habitat 1 and x_2 is the probability of detecting species in habitat 2. The respective standard errors are $s.e.(x_1)$ and $s.e.(x_2)$. Thus,

$$z = (x_1 - x_2) / \sqrt{\{s.e.(x_1)^2\} + \{s.e.(x_2)^2\}}$$

(Buckland *et al.* 1993). Because z has a normal distribution, results are significant at the 5 % confidence level where $z < -1.96$ or $z > 1.96$ (Buckland *et al.* 1993).

The bias and precision of strip-transect density estimates (\hat{D}_{ST}) for a range of strip half-widths (w) were assessed in relation to line-transect density estimates (\hat{D}_{LT}) for wallabies and pademelons on the young plantation. Line-transect density estimates should be more accurate than strip-transect density estimates because sightability is calculated and included in the equation for the former but not for the latter. Strip-transect density estimation involves trade-offs in bias and precision as w increases. Having a small w increases the likelihood that all animals will be seen, enabling the implicit assumption of perfect sightability within the strip to be met, but reduces the sample size and therefore reduces the precision. Increasing w should lead to better precision, but at the expense of increased negative bias (under-estimation of true density). Strip-transect density was estimated as

$$\hat{D}_{ST} = n/(2Lw)$$

where n = total number of animals seen, L = total transect length and w = transect half-width distance. The strip half-widths used were 10, 20, 30, 40, 50 and 70 m. The variance n was calculated using

$$\text{var}(n) = L \sum_{i=1}^k l_i \left(\frac{n_i}{l_i} - \frac{n}{L} \right)^2 / (k-1)$$

where n_i = number of animals seen on transect i , l_i = length of transect i , n = total number of animals, L = total transect length, and k = number of transects (Buckland *et al.* 1993). Line-transect density was estimated as

$$(\hat{D}_{LT}) = n \cdot \hat{f}(0) / 2Lw$$

where n is the number of animals seen within $w = 70$ m, L is total transect length, and $\hat{f}(0)$ is a sightability parameter derived from the perpendicular distances transects (Buckland *et al.* 1993). Values for $\hat{f}(0)$ were calculated from data pooled over time (Fieldtrips 1-11) for each species using the computer program DISTANCE (Laake *et al.* 1994), (Chapter 4). The variance of \hat{D}_{LT} was calculated using

$$\text{var}(\hat{D}_{LT}) = (\hat{D}_{LT})^2 \cdot \{[\text{cv}(n)]^2 + [\text{cv}\{\hat{f}(0)\}]^2\}$$

where cv is the coefficient of variation (Buckland *et al.* 1993). The percent relative bias (PRB) of strip-transect estimates in relation to line-transect estimates was calculated as

$$\text{PRB} = ((\hat{D}_{ST} - \hat{D}_{LT}) \cdot 100) / \hat{D}_{LT}$$

Precision was compared between strip- and line-transect estimates in a similar way, where percent relative cv (PRCV) was calculated as

$$\text{PRCV} = ((\text{cv}_{ST} - \text{cv}_{LT}) \cdot 100) / \text{cv}_{LT}$$

where cv_{ST} = strip-transect cv and cv_{LT} = line-transect cv (Buckland *et al.* 1993).

3.3 Results

Sample sizes were large enough to produce valid sighting histograms for pademelons in three habitats (young plantation, grassland and older plantation) and wallabies in two habitats (young plantation and grassland) (Table 3.1 and Figure 3.2). Few pademelons (14) were sighted in native forest but they are included here for indicative purposes.

Few animals were detected on cleared paths: two of 57 pademelons in older plantation and one of 15 pademelons in native forest. These sightings were removed from the data set before analysis.

The sighting histograms were shouldered (Figures 3.2f), relatively flat to Distance Category 4 (Figures 3.2a and 3.2e) or gradually stepped (Figures 3.2b-3.2d). These results indicate that animals were not moving away from the transect line in response to the observer. The shapes of the sighting histograms also indicated that the probability of detecting animals usually started to decline at 10 m, with the exception of wallabies on the young plantation (30 m).

Detection probabilities for pademelons and wallabies varied significantly between habitats (Table 3.2). On the young plantation, the probability of detecting wallabies was significantly higher than for pademelons. This pattern was reversed in grasslands. Detection probabilities also varied significantly within species between habitats. The probability of detecting pademelons was significantly different between all four habitats: detection was greatest on the young plantation, followed by grassland, older plantation and native forest (see Table 3.2).

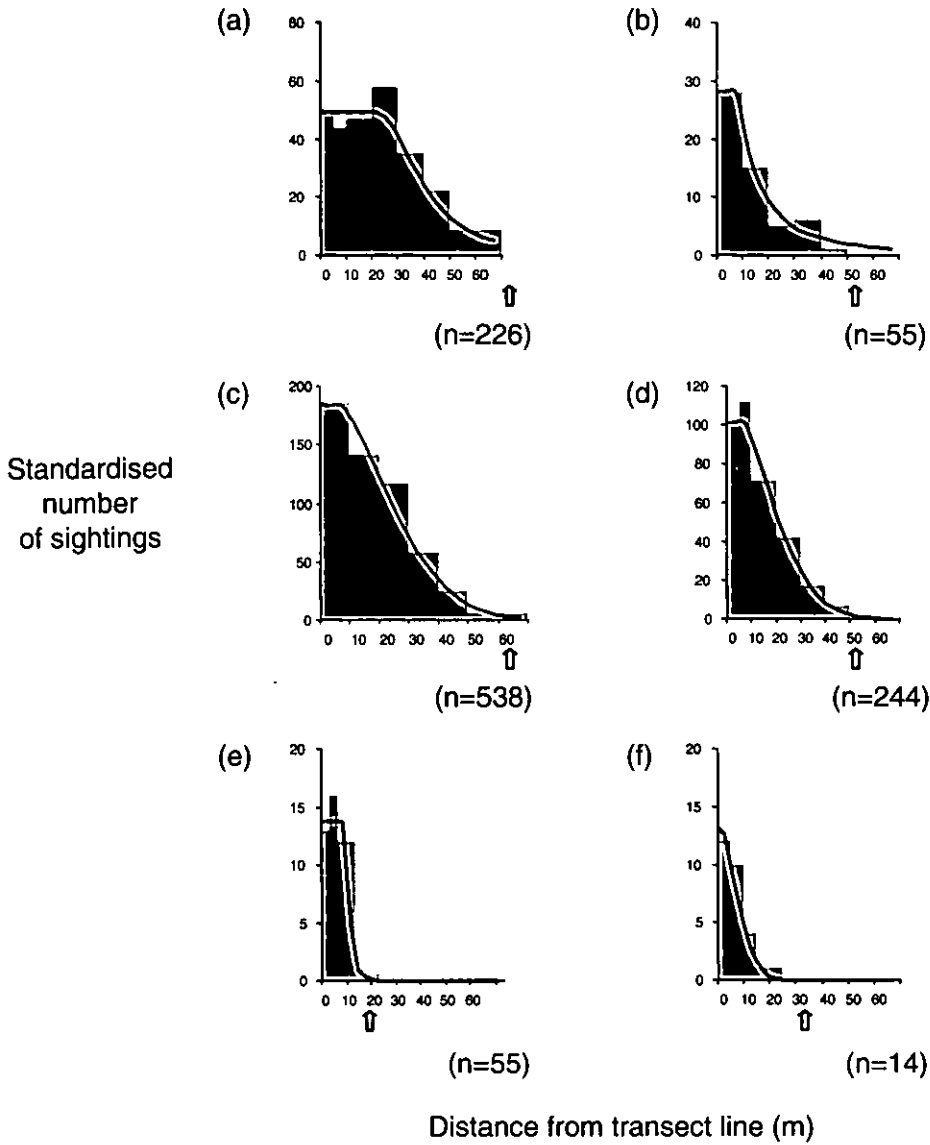


Figure 3.2 Sighting histograms and fitted detection functions for wallabies in two habitats and pademelons in four habitats (sampling periods 1-11): wallabies (a) on the young plantation, (b) in grassland, and pademelons (c) on the young plantation, (d) in grassland, (e) in older plantation and (f) native forest. Where appropriate, the number of sightings in a given habitat type were standardised using equivalent units of area for all distance categories. Actual total number of sightings (n) are given in parentheses. The arrow indicates the point of truncation.

Table 3.2 Estimated detection probabilities (% c.v.) for data truncated to 50 m. Superscripts indicate ranking of detection probabilities in decreasing order (a-f), all estimates were significantly different from each other ($P < 0.05$). (n.a. = not available)

Habitat	Wallabies	Pademelons
Young Plantation	0.8313 ^a (5.49 %)	0.5550 ^b (3.70 %)
Grassland	0.3802 ^d (18.33 %)	0.4589 ^c (5.02 %)
Older Plantation	n.a.	0.2358 ^e (7.21 %)
Native Forest	n.a.	0.2062 ^f (19.92 %)

Results for the percent relative bias (PRB) and percent relative cv (PRCV) are given in Table 3.3. Results showed that density estimates calculated using strip-transect analysis were negatively biased compared with line-transect density estimates. At small strip half-widths (10 and 20 m) there was a moderate negative bias for the estimated density of wallaby, but at moderate strip half-widths (30 m) there was almost no difference between strip- and line-transect density estimates (PRB = 0.1). As strip half-width increased past 30 m, PRB declined. The PRCV of strip-transect estimates decreased with increasing strip half-width. Results for pademelons showed that negative bias of strip-transect estimates increased with increasing strip half-width. This bias was also present at the narrowest strip half-width (10 m). The PRCV of strip-transect estimates decreased with increasing strip half-width.

Table 3.3 The percent relative bias (PRB) and percent relative cv (PRCV) results for data for wallabies and pademelons for the young plantation, using strip-transect analyses and varying strip half-widths compared with line-transect analysis.

Strip half-width (m)	Wallaby		Pademelon	
	PRB	PRCV	PRB	PRCV
10	-25.3	20.2	-23.8	13.4
20	-36.3	17.3	-39.0	13.1
30	-0.1	10.5	-47.6	3.98
40	-38.0	6.2	-54.5	2.4
50	-42.9	-0.9	-61.4	-0.8
70	-54.8	-4.2	-71.7	-3.03

3.4 Discussion

In using a method in previously untested environments and species, it is important to evaluate the results and sampling methods in relation to the assumptions of line-transect sampling (Buckland *et al.* 1993).

3.4.1 Assumption 1: Objects on the line are detected with a probability close to 1

Conceptually, this assumption can fail if an animal is on the transect line, does not move and is still not sighted. This usually occurs when the observer is in a different plane to the animals, for example, if an animal is in a burrow and the observer passes over it, if the animal is on the ground but the observer is in an aircraft far above it (e.g. Clancy *et al.*

1997), or if the animal is in a tree and the observer passes below it (e.g. Whitesides *et al.* 1988). Meeting this assumption is seldom a problem when surveying macropods on foot because if the animal is on the line and does not move, the observer will literally intercept it.

3.4.2 Assumption 2: Objects are detected at their initial location before any movement in response to the observer

Detecting animals at their initial location, before any movement in response to the observer, is a problem in both open and closed habitats. At this site, minimizing visual cues when on the young plantation and in grassland habitats was important, and reducing noise produced while travelling was critical within native forest and older plantation habitats. Noise produced while travelling in closed habitats is probably the greatest issue when surveying macropods in heavy cover.

The direction in which animals move before detection can determine whether or not this confounding effect is detected. If animals move perpendicular to the transect line, it can usually be detected from the sighting histograms, as peaks or spikes occur at some distance from the line (Buckland *et al.* 1993). If animals move parallel to the transect line however, this movement cannot be detected from the sighting histogram, but can result in an underestimate of density if undetected animals flush ahead of the observer, causing other undetected animals to also flush ahead, thereby having a snowballing effect on animals present (Buckland *et al.* 1993). This situation was reported in Southwell's (1994) evaluation work on the use of ground line-transect sampling to estimate macropod abundance for populations of known size, where a negative bias in density estimates occurred when density of (untamed) populations exceeded 30 animals.km⁻¹.

The shapes of sighting histograms produced during this study indicate that animals were not flushing perpendicular to transect lines in response to the observer. Within the closed habitats, noise produced by travelling was minimized by clearing transect lines and access paths. Older plantation transect lines and paths were also raked before the first fieldtrip and at subsequent four-monthly intervals to remove dry leaves and twigs. To reduce visual cues on the young plantation, windrows were used to screen observers approaching transect lines. Little could be done in grassland, limiting observers to being as quiet and cryptic as possible.

Although it is not possible to determine whether animals were flushing parallel to the transect lines, two strategies were used on the young plantation to reduce the chance of this happening. Firstly, transect lines were positioned perpendicular to windrows so these structures could be utilized as barriers to animals moving parallel to transect lines. Secondly, the use of a high point as the first data collecting station, enabled animals' locations to be recorded quickly and easily, before the animals had a chance to react.

The use of short lines (40–80 m) in three of the four habitat types reduced the chance of animals moving undetected by the observer. That is, short transect lines in some habitats could be scanned along their entire length at the beginning of each survey, and animals that flushed forward could be heard. Some transect lines in grassland, however, were long (100–180 m) and could not be completely scanned at the start of each survey. Observers could also be seen approaching the start of the transect line and while conducting the survey. As a result, density estimates for this habitat may underestimate true density.

3.4.3 Assumption 3: Perpendicular distances are measured accurately

The grid markers were easily seen and differentiated and, combined with small distances between markers (approximately 10 m), enabled animals to be quickly and accurately placed

into distance categories. The shapes of sighting histograms produced during this study suggest that appropriate widths had been used.

The use of narrow, clearly marked distance categories rather than collecting actual perpendicular distance data eliminated many problems associated with data heaping (Buckland *et al.* 1993). Data heaping can occur with distance categories if markers are not clearly visible and the observer needs to decide which category an animal is in. The small distance between grid markers, however, prevented this from happening, and results from the sighting histograms indicate that data heaping did not take place.

Overall, the sampling methods used during this study largely fulfilled the assumptions of line-transect sampling. Steps were taken, wherever possible, to ensure that animals on the transect line were seen with a probability close to, or equal to, 1.0, and we were satisfied that this assumption was generally met. Additionally, animals did not display evasive (perpendicular) movement before detection in response to the observer, and distance data were measured accurately. It is possible, however, that evasive movement parallel to the transect line may have occurred within grasslands, and this could result in an underestimate of true density for this habitat type.

3.4.4 Differences in sightability within species and between habitats

Differential sightability between species and/or habitats has been reported by many authors (Johns 1985, Whiteside *et al.* 1988, Coulson 1993, Grigg *et al.* 1997, Clancy *et al.* 1997). Differences between species can be generalized as follows: larger, less cryptic, reactive animals, especially those in groups, are more easily seen and visible at greater distances from transect lines than smaller, cryptic, solitary or stationary animals. Additionally, differences in sightability between habitats can be generalized as being greater in more open habitats with little or low understorey, and lower in more densely vegetated habitats with tall understorey.

While such differences were expected within this study, the direction of differences did not always confer with our pre-survey expectations. For example, we expected sighting probability to be greater for wallabies than for pademelons in all habitats because of their larger body size, but this was not the case in grassland. The grassland results are thought to reflect differences in the two species' responses to the observer, and the difficulty with distinguishing the pelage and silhouette of wallabies from tussock grass. Pademelons have a relatively dark pelage and once caught in the spotlight beam tended to stop feeding, look towards the spotlight and then flush to the nearest habitat offering shelter (native forest or older plantation). In contrast, wallabies are a paler colour and often did not respond to the observer. That is, once caught in the spotlighting beam animals did not always respond by lifting their head and looking towards the light. If wallabies were within approximately 30 m of the transect line, they tended to move a short distance before resuming feeding, while some animals fed throughout the survey. This was problematic in grassland as the vegetation was a similar colour and shape (round) to the flanks of a wallaby.

In addition, results from truncated data sets with varying width distances for wallabies and pademelons on the young plantation, suggest that sightability can also decline at vastly different rates for two species within a single habitat. This result is not surprising, theoretically; however, the rapid decline of sightability of pademelons with increasing width distance was significant, and had not been identified during pilot studies.

Such unexpected findings on sightability provide strong support for the use of line-transect sampling over strip-transect sampling, as the latter requires a judgement on the sightability before starting the survey. The large differences in sightability between species and habitats also indicate that counts, uncorrected for sightability, will not provide an abundance index that is robust across species or habitats.

3.4.5 Sample size

To accurately estimate sightability using line-transect sampling, 60-80 sightings are recommended (Buckland *et al.* 1993). These sample sizes were not achieved for wallabies or pademelons in the closed habitats (native forest and older plantation) or for wallabies in grassland. Results for the sighting histograms for these species within these habitat types did, however, meet the shape criteria required for line-transect analysis. Consequently, despite small sample sizes these data are presented here for indicative purposes. In addition, an attempt was made in this study to use line-transect sampling for brushtail possums (*Trichosurus vulpecula*) and European rabbits (*Oryctolagus cuniculus*), two other species considered to be pests by the forestry industry and currently managed in the absence of accurate population monitoring. Too few sightings were recorded to enable data to be analysed using line-transect analyses (data not shown). Consequently, future studies incorporating wallabies, pademelons, possums or rabbits in these respective habitat types may require greater effort than used here. This could be achieved by either increasing transect length or the number of replicates.

3.4.6 Logistics

Installing the grid markers was labour-intensive and expensive, taking two people four months to complete the task and requiring over 5000 reflectors and stakes. This investment of time, energy and resources would be appropriate for long-term studies but impractical for short-term projects.

Restricting the number of distance categories used in a study to five or six would reduce materials required for grids but still allow enough degrees of freedom for statistical analysis. Fewer stakes (longer grid cells) could also be used in two habitats to reduce the materials required; cells could be placed at 20 m intervals on young plantations and older plantations. Within the more densely vegetated habitats (native forest and grassland), however, 20 m intervals may be too large for accurate data collection.

3.4.7 Management implications

Line-transect sampling, as described here, is unlikely to be economically feasible for routine use in forestry operations. The results from this study, however, could be used to improve estimates made by strip-transect sampling on newly established young plantations, on the condition that general sampling methods are consistent with those used in this study. That is, the same method for data collection would be used but in the absence of a grid system, so animals are recorded as present only within the defined strip, without information on their exact perpendicular distance. Consequently, data collection would still require a two-person team (one to spotlight and one to record) to survey on foot at night, walking at approximately 1 km.h^{-1} , using transect lines situated perpendicular to windrows (located at random throughout the site) and collecting data from both the top of windrows and by walking the line. Additionally, confounding effects from human impact would also need to be minimized by approaching the site as quietly as possible, and remaining quiet and cryptic while surveying and travelling between transect lines.

Assuming this condition can be met, we recommend using a strip half-width distance of 30 m when strip-transect sampling for wallabies and pademelons on a newly established plantation. The use of a 30 m strip half-width distance accurately estimated the density of wallabies on the young plantation ($\text{PRB} = 0.1$) and was relatively precise ($\text{PRCV} = 10 \%$). The use of a larger strip half-width resulted in density being underestimated and little gain in precision. It is interesting to note that there was a moderate negative bias at small strip half-

width distances (10 and 20 m) for wallabies on the young plantation. This result was counter to theoretical expectation and we suggest that it was attributable to random variation from small sample size. A strip half-width of 30 m also represented a compromise distance for pademelons. The use of any width distance caused density to be underestimated, while precision improved at distances greater than 20 m. There was little gain in precision, however, by increasing strip half-width distance beyond 30 m. Consequently, we recommend $w = 30$ m, as the cv is only 4 % higher than for a line-transect estimate.

Chapter 4

Population Surveys: Density & Habitat Selection

4.1 Introduction

Fundamental to any ecological study is information on species-specific abundance, usually measured as population density. Within Tasmania, population densities of herbivore species are largely unknown. This situation most likely reflects the nocturnal behaviour of these herbivores, and their tendency to utilise densely vegetated habitats (Chapter 3). Where density data is available, results are limited to pademelons, wallabies and possums on plantations and pastoral land, over short time periods (months) (Johnson 1977, Statham 1983). Consequently, one of the major objectives of this chapter was to provide estimates of population densities of the major herbivores within the Tasmanian environment.

A second objective was to determine patterns of habitat selection at the population level for each species. On a hierarchical scale of habitat selection, patterns determined by this chapter fall between species' distribution and the locations of animals' home-ranges. This represents selection at a relatively large scale. The specific aims of this chapter were to:

- (a) determine population density of each species at a range of spatial and temporal scales,
- (b) examine patterns of habitat selection by the major herbivore species at this site, and
- (c) monitor densities and patterns of habitat use for each species, before and after planting seedlings on the young plantation.

4.2 Materials and Methods

The 428 ha study site is described in Chapter 3. Four habitats dominated the site: (1) older plantation, 274.7 ha; (2) native forest, 89.1 ha; (3) grassland, 38.4 ha; and (4) the young plantation, 17.8 ha. The fifth habitat, harvested uncleared land (7.8 ha), was not used in this study due to low visibility (Chapter 3).

4.2.1 Data collection

Fieldwork began in May 1997 and was completed in April 1998. Each sampling period consisted of six nights spotlighting, 2-3 nights apart, over 17 days. There were eight sampling periods. Five were conducted before seedlings were planted on the young plantation, and the remaining three took place after planting. Sampling periods were approximately four weeks apart with the exception of times either side of planting, which were conducted as close together as possible (less than two weeks apart).

Due to the nocturnal feeding behaviour of the herbivore species, and their tendency to use both open and closed habitats at night, conventional line-transect sampling methods were inadequate. Consequently, surveying methods were modified as described in Chapter 3 to enable line-transect sampling to be used successfully at this site. Briefly, walked line-transect counts were conducted along 84 permanent transect lines, located at random within the four habitats. There were 24 transect lines on the young plantation; 30 in older plantation; 20 in native forest and 10 in grassland (see Figure 3.1). Because only one young plantation was available within this study area, statistical inferences are limited to this site only. Additionally, several steps were taken to maximise independence of data from transect

lines, and hence, avoid problems with pseudoreplication in this habitat. Firstly, the site was surveyed using many short transect lines rather than a few long transect lines. Secondly, transect lines were distributed at random throughout the habitat but with a minimal distance of 50 m between transect lines.

4.2.2 Line-transect analysis

Species' densities were calculated for each transect line, each sampling period, using line-transect methodology (Buckland *et al.* 1993) and the program DISTANCE (Laake *et al.* 1994) when there were sufficient sightings to produce valid sighting histograms (Buckland *et al.* 1993), and strip-transect analysis when there were too few sightings (Buckland *et al.* 1993).

To accurately estimate the sightability parameter, 60-80 sightings are recommended (Buckland *et al.* 1993). This sample size was not achieved during individual sampling periods for any species in any habitat. Consequently, this parameter was calculated from data pooled over time for each species in each habitat type, on the assumption that variation in sightability across time was likely to be trivial relative to variation between species and habitats.

4.2.3 Overall density

Population stability in the entire 428 ha study area was examined for each species by graphing overall density (\hat{D}_T) over time. Overall density was calculated for each sampling period, from the estimated densities for the four separate habitats and the proportion of habitat available as:

$$\hat{D}_T = \sum [\hat{D}_i \cdot (A_i / A)] \text{ and } SE(\bar{x} \hat{D}_T) = \sqrt{[\sum \{(A_i / A)^2 \cdot \text{var}(\hat{D}_i)\}]}$$

where \hat{D}_i is the mean density in habitat *i*, A_i is the area of habitat *i* and A is the area of the entire study site (G. McPherson *pers. comm.*).

4.2.4 Biomass density

Biomass density was estimated using species overall density, multiplied by average species body mass. Possum body mass was taken from trapped animals used in the telemetry study (Chapter 7). Body mass estimates for wallabies, pademelons, rabbits and wombats were taken from Strahan (1991).

4.2.5 Habitat selection

If animals use habitats non-selectively, then population density will be consistent between different habitat types. If animals are selective, however, density can vary significantly between habitats. Consequently, population density of each species was compared between habitats to examine patterns of habitat selection. Mean density within a habitat was calculated using the mean density for each transect line in that habitat, pooled from all sampling periods. Density data generally had a poisson distribution. Only pademelon data could be normalised. Pademelon data were log transformed before a general linear model

(GLM) was used to examine the effect of habitat on density. Where results were significant, Tukey-Kramer tests were used for pairwise comparisons between habitats. Data for wallabies, possums, rabbits and wombats could not be normalized due to the predominance of zeros within data sets. Consequently, Kruskal-Wallis tests were used to detect any significant differences in mean density between habitats within each species. Pairwise comparisons between rank sums were made using the Q statistic (Zar 1996), with $\alpha=0.008$ after a Bonferroni adjustment for multiple comparisons. Statistical tests were performed using the computer program SAS/STATS (SAS 1989).

4.2.6 Temporal change

Species' patterns of habitat selection were examined for temporal change. Sampling periods were used as replicates of time, but could not be tested with a repeated measures GLM because data were not normally distributed. Consequently, data were graphed and assessed visually.

4.2.7 Macropod distribution on the young plantation

By collecting survey data as the presence of animals within grid cells, data could also be used to model species' spatial distribution within habitats. Analysis was restricted to macropod distribution on the young plantation because data were abundant for these species, and this was the habitat of greatest interest. The presence of pademelons and wallabies on the young plantation was modelled in relation to environmental features, using logistic regression. The environmental features used were distance to: nearest edge of the young plantation, windrow, grassland, older plantation, native forest, harvested uncleared land and waterways. The spatial unit of analysis consisted of grid cells formed by stakes located away from and along the transect lines. Only those cells for which line transect analysis inferred perfect sightability (i.e. cells within the shoulder of the estimated detection function: for pademelons cells ≤ 10 m either side of the transect line; for wallabies cells ≤ 30 m [Chapter 3]), were used for analysis.

Before modelling, the data were assessed visually to determine the form of the relationship (linear or curvilinear) between the response variable and each predictor variable. Scatter-plots were constructed of sorted and grouped data for ease of interpretation. The binary response (presence/absence) was forced into a frequency format by sorting values for each predictor variable in ascending order and grouping data into blocks that contained approximately equal numbers of samples. Within each block, the proportion of cells with animals present and the mean predictor variable values were calculated. Logits of the response variable were used in this process, as linear logistic relations are easier to recognise visually than sigmoidal ones. To enable logits to be calculated, it was necessary to eliminate zeros from the proportion of presence data set, consequently, 0.1 was added to the proportion value before transformation [$\log\{(p + 0.1)/(1 - (p + 0.1))\}$, where p is the proportion of blocks with animals present]. Results for each block were then plotted.

Probabilities of the presence of wallabies and pademelons on the young plantation were estimated using the logistic procedure (PROC LOGISTIC) in SAS (SAS 1989). A Score statistic, -2 Log Likelihood values, and re-scaled maximised R^2 values were used to estimate the joint significance of the predictive variables; concordance was used to measure the predictive ability of the derived model for each species (SAS 1989). The models were not tested further, as low maximised R^2 values suggested they had relatively poor explanatory power.

4.2.8 Herbivore densities before and after planting

Species densities on the young plantation were compared immediately before and after planting of *E. nitens* seedlings, to see if animals responded to this new potential food source. A Wilcoxon signed rank test was used to compare species' densities between Sampling Periods 5 and 6. The null hypothesis was that planting did not affect densities on the young plantation. Statistical tests were performed using the Univariate procedure (PROC UNIVARIATE) in SAS/STATS (SAS 1989).

4.3 Results

Adequate sample sizes and valid sighting histograms enabled line-transect analysis for pademelons in three habitats (young plantation, grassland and older plantation), wallabies in two habitats (young plantation and grassland) and wombats in one habitat (young plantation) (see Chapter 3). The other species/habitat combinations were analyzed using strip-transect analysis.

4.3.1 Overall density

Pademelons and wallabies dominated this environment on both a numerical and biomass basis (Table 4.1). Possums, wombats and rabbits were present in very low densities (Table 4.1). Overall densities of the five herbivore species appeared relatively stable throughout the study period (Figure 4.1).

Table 4.1 The overall density (animals.ha⁻¹ ± s.e.) and biomass density (kg.ha⁻¹) for each species throughout the entire study area (428 ha).

Species	Overall Density	Biomass Density
Pademelon	1.46 ± 0.75	7.93
Wallaby	0.26 ± 0.22	4.41
Possum	0.04 ± 0.07	0.16
Wombat	0.06 ± 0.08	1.66
Rabbit	0.02 ± 0.06	0.03

4.3.2 Habitat selection

Mean density of each species within each habitat is shown in Table 4.2. Results from the GLM on average pademelon density in the four habitats showed a significant habitat effect ($F_{3,80}=57.85$, $P=0.0001$). Kruskal-Wallis tests showed significant differences in density between habitats for wallabies ($\chi^2=48.298$, d.f.=3, $P=0.0001$), possums ($\chi^2=11.072$, d.f.=3, $P=0.0113$), rabbits ($\chi^2=18.713$, d.f.=3, $P=0.0003$) and wombats ($\chi^2=35.381$, d.f.=3, $P=0.0001$). Multiple comparison tests showed the same trend for pademelons, wallabies and wombats: significantly higher densities on the young plantation and within grasslands than within older plantation and native forest (Table 4.2). Possum densities were significantly higher in grassland than older plantation. Rabbit densities were significantly higher on the young plantation than in older plantation and native forest habitats (Table 4.2).

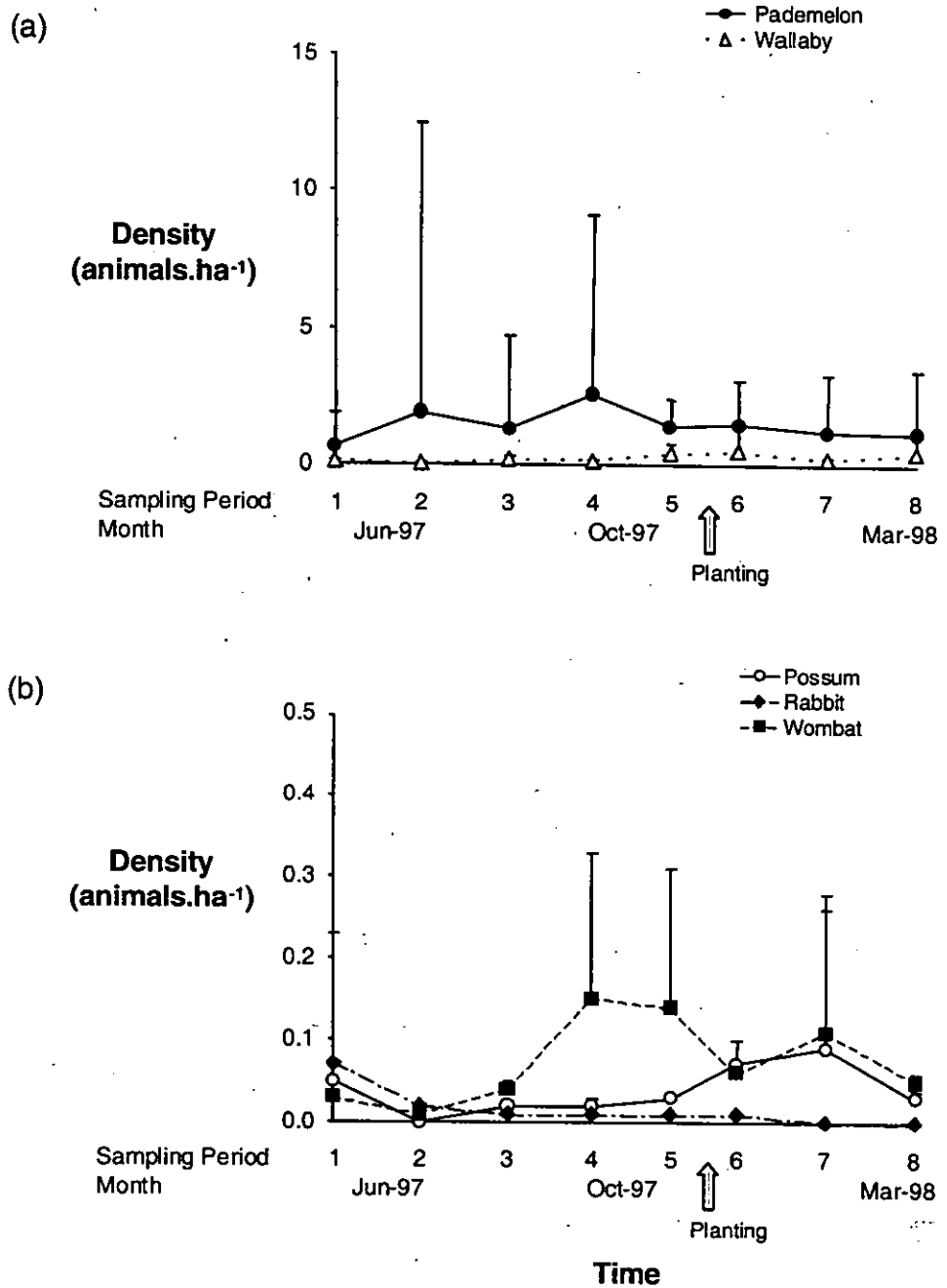


Figure 4.1 Overall densities (+s.e.) of (a) pademelons and wallabies, and (b) possums, rabbits and wombats, throughout the entire 428 ha study area, over the 11 month study period.

Table 4.2 Mean density of each species within each habitat, using data from May 1997 – March 1998 (Sampling Periods 1-8). Results calculated using line-transect analysis are indicated with an asterisk. Superscripts show comparisons between habitats: within each species, different letters are significantly different ($\alpha = 0.008$).

Species	Young Plantation	Grassland	Older Plantation	Native Forest
Pademelon	$8.66 \pm 1.21^{*a}$	$4.91 \pm 1.24^{*a}$	$0.86 \pm 0.21^{*b}$	0.36 ± 0.13^b
Wallaby	$1.96 \pm 0.39^{*a}$	$1.37 \pm 0.55^{*a}$	0.07 ± 0.04^b	0.04 ± 0.03^b
Possum	0.10 ± 0.04^{ab}	0.13 ± 0.05^a	0.01 ± 0.01^b	0.07 ± 0.04^{ab}
Wombat	$0.64 \pm 0.13^{*a}$	0.21 ± 0.07^a	0.03 ± 0.02^b	0.00 ± 0.00^b
Rabbit	0.10 ± 0.03^a	0.04 ± 0.03^{ab}	0.01 ± 0.01^b	0.00 ± 0.00^b

Densities of pademelons and wallabies within the four habitats over time are shown in Figure 4.2. Densities were consistently high on the young plantation and within grassland, and low in older plantation and native forest. Pademelon densities appeared to peak on the young plantation in Sampling Period 4 (September) ($16.2 \text{ animals.ha}^{-1}$) and were high in grassland in Sampling Periods 3-6 (November/December) ($8.5 \text{ animals.ha}^{-1}$).

4.3.3 Distribution of macropods on the young plantation

Pademelons were recorded in 29 % of their 218 sampling units. Wallabies were recorded in only 10 % of their 652 sampling units. Scatterplots (not shown) suggested a curvilinear relationship between the presence of pademelons and wallabies with distance to older plantation. Stepwise selection was therefore performed using a quadratic term for this variable and linear terms for the other six predictor variables.

No significant relationship was detected for any variable for pademelons. Three predictor variables were significant for wallabies: young plantation edge ($P = 0.000$), distance to grassland ($P = 0.000$) and distance to waterways ($P = 0.007$). The relationship was positive for distance to young plantation edge, and negative for distance to grassland and waterways. The combined effect of the three variables on the probability of seeing wallabies was significant ($-2\text{LOGL} = 44.539$, d.f. = 3, $P = 0.000$). The resulting predictive logistic regression equation for wallabies on the young plantation was

$$\text{Presence} = \exp(-1.0198 + 0.0222.B - 0.0084.G - 0.0090.W) / [1 + \exp(-1.0198 + 0.0222.B - 0.0084.G - 0.0090.W)]$$

where B = distance to nearest young plantation edge, G = distance to nearest grassland and W = distance to nearest waterway. The model explained only 14.0 % of the variation in presence of wallabies on the young plantation. Of the three predictive variables, distance to young plantation edge explained 10.3 %, distance to grassland 2.7 % and distance to waterways 1.4 % of the variation. The confidence interval displacement value ($c = 0.75$) associated with the model suggests that its predictive power was satisfactory (Lemeshow and Le Gall 1994).

4.3.4 Densities of animals before and after planting

Densities of pademelons and wallabies within the four habitats pre- and post- planting of *Eucalyptus* seedlings on the young plantation are shown in Figure 4.2. Results from the Wilcoxon signed ranks tests showed no significant difference in densities of herbivore

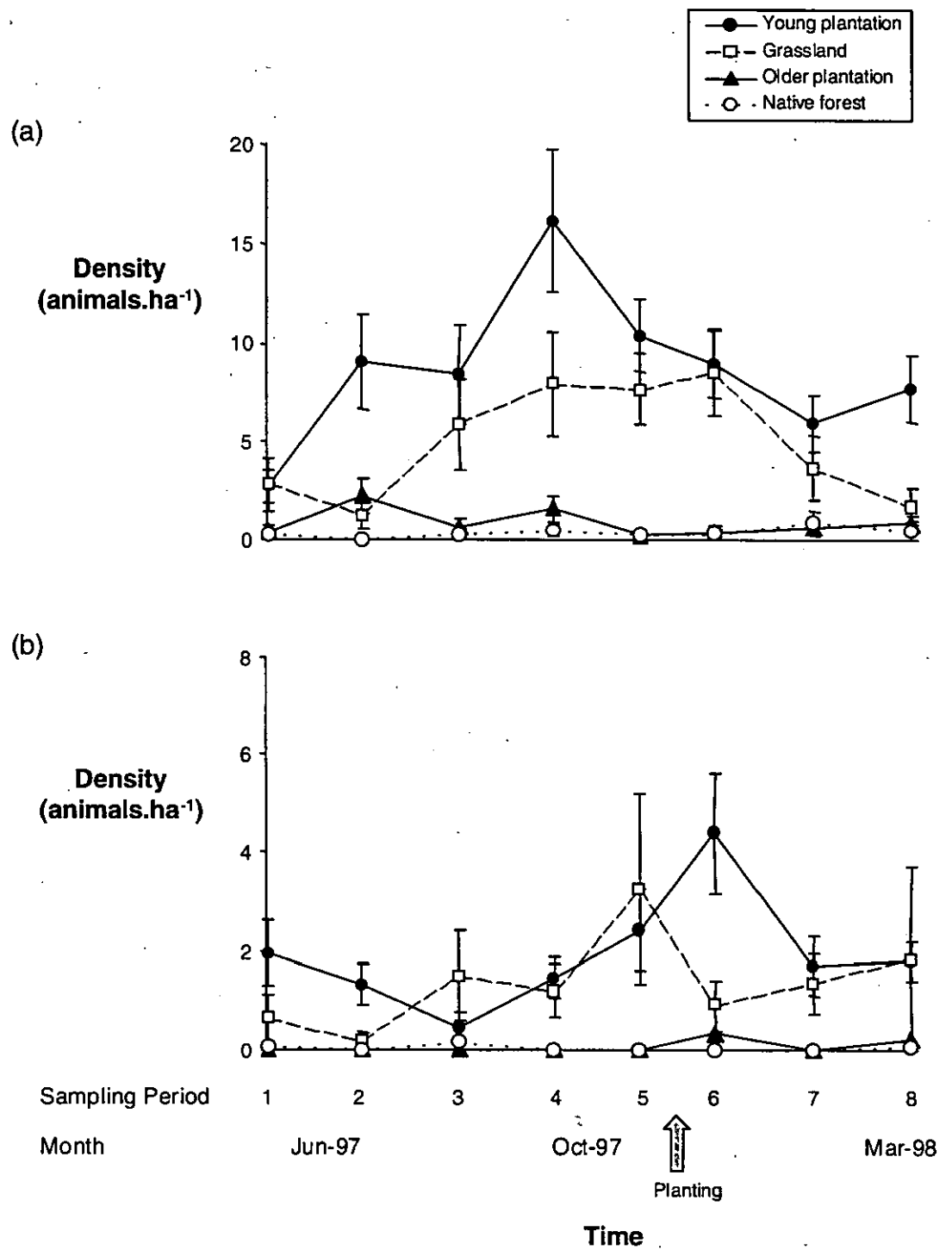


Figure 4.2 Densities (mean \pm s.e.) of (a) pademelons and (b) wallabies within the four habitats over the 11 month study.

species between these periods: pademelon $P = 0.436$; wallaby $P = 0.135$; possum $P = 1.000$; rabbit $P = 1.000$ and wombat $P = 0.984$.

4.4 Discussion

4.4.1 Overall density

Overall density results plotted over time, showed that herbivore populations were relatively stable throughout the study period. Consequently, any changes detected for species' patterns of habitat use were likely to reflect changes in animals' behaviour, rather than a sudden influx or emigration.

Within this fragmented forestry environment, pademelons were the most abundant species, followed by wallaby (less than one-fifth the size of the pademelon population). Possums, wombats and rabbits were present in relatively low abundance. Comparisons of biomass density between the two macropod species showed that pademelons were the dominant consumer at this site (1 kg.ha⁻¹ pademelon: 0.56 kg.ha⁻¹ wallaby).

The scarcity of density data for herbivore species, within Tasmania, prevents comparisons between this forestry site and natural environments. However, where possible, comparisons with other studies are made below.

There are no density estimates for red-bellied pademelons in natural environments in Tasmania, or on mainland Australia before they became extinct. Johnson (1977) has reported pademelon densities of 0.8 and 17.3 animals.ha⁻¹ for two adjacent transect lines, surveyed on the same nights at a forest/pasture site in Tasmania. The accuracy of these values is difficult to evaluate however, because (1) only two transect lines were used; (2) transect lines were roads and therefore, were a potential source of bias (eg. some species are attracted to roads while others avoid them); (3) surveys were conducted more than once a night, introducing pseudoreplication into the data set; and (4) the method of analysis is outdated (right angle sighting distances). Despite these factors however, Johnson's values are consistent with range extremes determined in the present study.

Comparisons of overall density of wallabies (0.26 animals.ha⁻¹) with densities in natural environments suggest that values are high in the present study. Although density estimates are unavailable for native forest in Tasmania, estimates of 0.01-0.035 and 0.06-0.15 animals.ha⁻¹ have been reported for red-necked wallabies (*M. r. banksianus*) in south-eastern mainland Australia (Southwell *et al.* [1995], and Robertshaw and Harden [1986], respectively). Density estimates within human-modified environments, where grazed pastoral land abuts native forest, have reported higher densities than those found in the present study. A density of 0.529 animals.ha⁻¹ has been reported at Wallaby Creek in New South Wales (Southwell 1994), with a peak of 8.0 animals.ha⁻¹ in pasture habitat (Jarmen *et al.* 1987). These extremely high densities were thought to reflect the productivity of the pastoral land, which red-necked wallabies used as a feeding ground (Jarmen *et al.* 1987). The high densities of *M. rufogriseus* within forest/pasture and forest/plantation environments may reflect the availability of food and shelter habitats, side by side, which is advantageous to many macropod species, as discussed below.

Comparisons of overall possum density (0.04 animals.ha⁻¹) with densities in natural environments suggest that values are extremely low in the present study. Hocking (1981) estimated 0.31 possums.ha⁻¹ from trapping surveys conducted in 80-year-old Tasmanian forest. How (1972) estimated 0.44 animals.ha⁻¹ from trapping surveys within *Eucalyptus* forest in south-eastern mainland Australia. These densities are low, however, when compared with estimates of up to 25 animals.ha⁻¹ from trapping surveys of feral populations in New Zealand (Coleman *et al.* 1980). Low possum population densities within the present study may be a function of limited resources, such as food or shelter. For example, den sites

may have been limited within the forestry environment, reflecting land-clearing practices that remove too many hollow-bearing trees (Gibbons and Lindenmayer 1996). Sixty-four percent of the study area was older plantation (5-7 years old), which lacked tree hollows. This will be discussed further in Chapter 12.

There is very little information available on the density of wombats in natural environments. Wombat density on the young plantation ($0.64 \text{ animals.ha}^{-1}$), was similar to that reported by Rishworth *et al.* (1995) within a one year old pine plantation (*Pinus radiata*) that lacked windrows (about $0.8 \text{ animals.ha}^{-1}$). McIlroy (1991) has reported that wombat densities are particularly high ($0.3 \text{ animals.ha}^{-1}$) in areas where native forest abuts pastoral land. Hence, wombats (like macropods), appear advantaged by fragmented landscapes where food and shelter habitats lie side by side. It is unclear, therefore, why the patchy environment within the present study did not support a higher wombat population. Food may have been limited at this site, as discussed below.

Overall rabbit density ($0.02 \text{ animals.ha}^{-1}$) appeared very low at this site. Densities of rabbits in Tasmania are unknown (Coleman *et al.* 1997). Generally low densities in north-western Tasmania have been attributed to unfavourable climate and vegetation (Guiler 1968).

4.4.2 Habitat selection

Pademelons, wallabies and wombats selected habitats based upon their feeding niche (discussed in detail in Sections 1.3 and 2.2). All three species had highest densities in young plantation followed by grassland, and the dominant vegetation within these habitats were grasses and herbaceous dicots. Avoidance of older plantation and native forest by pademelons, wallabies and wombats reflects the lack of understorey within these habitats. Older plantations in particular often represented a monoculture, as the ground comprised bare soil. Hence, older plantations may represent a biological 'desert' to these species.

It is interesting to note that the young plantation and grassland comprised only 13 % of the total study area, but 56 % of the pademelon population, 80 % of the wallaby population and 72 % of the wombat population utilised these habitats at night. Of this 13 % of selected habitat, the young plantation represented only 4 %, in which 25 %, 32 % and 43 % of the pademelon, wallaby and wombat populations foraged at night, respectively. This suggests that these two habitats, and the young plantation in particular, were extremely important feeding areas for these species.

Although macropods used older plantation and native forest habitats less than the young plantation and grassland habitats at night, these may be important habitats during the day. Incidental observations of the young plantation and grassland suggest they are essentially empty during the day. This result is consistent with other macropod studies, where animals selected cover habitat during the day and open feeding areas at night, for example, red-necked pademelon *T. thetis* (Johnson 1991), Tamar wallaby *M. eugenii* (Smith 1991), and black-striped wallaby *M. dorsalis* (Kirkpatrick 1991). Consequently, the relatively high macropod densities within the present study may relate to the fragmented nature of this forestry environment. With highly heterogeneous habitats lying side by side, and abrupt boundaries between habitats so ecotones or transitional zones are absent, complimentary resources such as shelter and feeding sites occur within close proximity. Hence, this landscape mosaic is probably ideal for macropods, and theoretically, capable of supporting high population densities. Similar patterns of high species abundance resulting from human-modified landscapes has been reported for a range of other species, including deer in Great Britain (Hannen and Whelan 1989), white-tailed deer *Odocoileus virginianus* in North America (Alverson *et al.* 1988), moose *Alces alces* in Norway (Hjeljord *et al.* 1990), and roe deer *Capreolus capreolus* in Scandinavia (Hansson 1979). Daytime movements of pademelons and wallabies determined using radio-telemetry, will be discussed in Chapter 7.

Possums' patterns of habitat selection did not reflect their feeding niche (discussed in detail in Section 2.2). Possum densities were highest in grassland and lowest in older plantation. These results were unexpected as possums are folivores and grasslands contained very little browse. Possums seen on the grassland transects often appeared close to (or in) very old, native *Eucalytus* trees, despite the rarity of trees within this habitat. It was not possible to detect whether these animals were feeding or moving in and out of dens sites, but if den sites were limited within the broader forestry environment, this may explain the relatively high possum density within grasslands. Low possum density in older plantation also contradicts this species' feeding niche, as this habitat is highly abundant in foliage. Possums were only seen on the ground within this habitat; no possum was ever seen in a *Eucalypt nitens* tree, or feeding on *E. nitens* foliage. Therefore, older plantations may have been avoided by possums at night because they offered little shelter and unpalatable foliage despite its abundance.

Rabbits' patterns of habitat selection were consistent with their feeding niche (discussed in detail in Sections 1.3). Density was highest on the young plantation followed by grassland, while older plantation and native forest were avoided. Again this reflects the lack of potential food available to animals within these latter habitats.

4.4.3 Distribution of macropods on the young plantation

The areas in which animals forage within a habitat can depend on the spatial distribution of vegetation, competition within and between animal species, and movement patterns associated with attempting to forage efficiently (Bell 1991). Distribution of vegetation on the young plantation was likely to be the most important factor influencing the location of pademelons and wallabies within this habitat. I was unable to investigate this with my data. However, it is possible to explore the relationships between environmental features and aspects of predation risk, and intra- and inter-specific competition that may also have influenced the location of animals.

The association between wallaby location on the young plantation and distance to grasslands and waterways were so minor as to question their biological significance. The effect of distance from young plantation edge, however, explained enough variation (10 %) in location to warrant discussion. I suggest the negative relationship between wallaby presence and plantation edge can be understood in terms of risks of predation, and inter-specific competition. Two scenarios are possible:

Scenario 1

In Scenario One, pademelons and wallabies do not interact and they respond to predation risk in different ways. Pademelons are distributed evenly throughout the young plantation because there is equal risk of predation for this species at both the young plantation edge and young plantation interior (Figure 4.3). In this case, windrows provide the same effective shelter from predators as the surrounding forest. I did observe that when disturbed, pademelons often sought shelter in windrows rather than travelling to neighbouring habitat. Statham (1983) also found that pademelons only travelled up to 100 m from forest edge when using a plantation without windrows.

In this scenario, wallabies are distributed towards the young plantation interior because they are at lower risk of predation away from forest habitat, and therefore from the young plantation edge. This represents a different predator avoidance strategy to pademelons. wallaby is a relatively large species and can out-run its predators, consequently, this species relies on early detection to escape predation, rather than crypsis (Jarman 1991).

Although this scenario is appealing in its simplicity, and has some supporting evidence, it is inconsistent with several observations (see below).

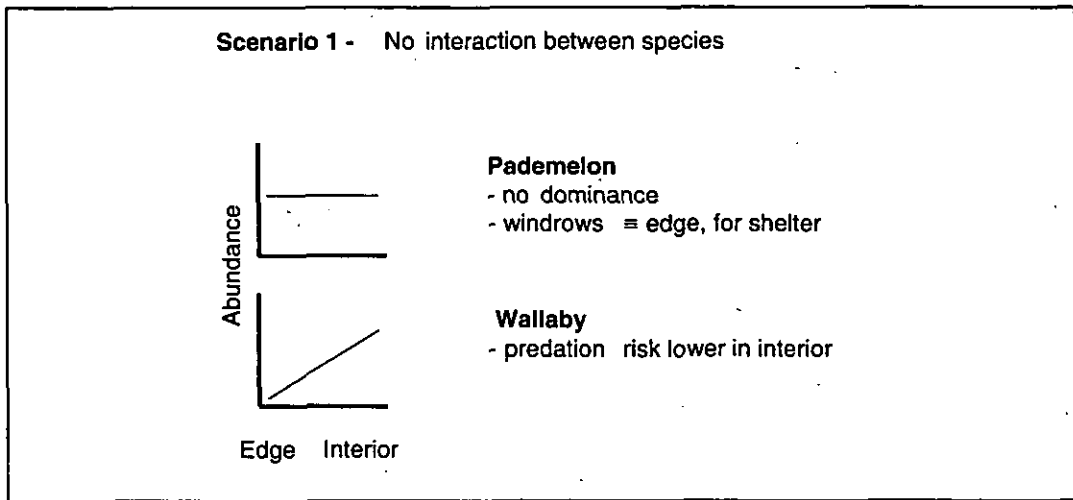


Figure 4.3 A diagrammatic representation of Scenario 1 for explaining the spatial distribution of pademelons and wallabies on the young plantation.

Scenario 2

In Scenario Two, pademelons prefer the young plantation edge because risk of predation is lower nearer forest habitat. Windrows provide secondary, less effective shelter from predators. Male pademelons are dominant to females and sub-adults. Pademelons are also dominant to wallabies and the state of the pademelon population (high or low density) influences the spatial distribution of wallabies on the young plantation.

When pademelon density is high, as suggested for the present study, dominant (large male) pademelons occupy and territorially defend feeding areas near the young plantation edge. Subordinate animals (young male and female pademelons, and wallabies) are displaced to the young plantation interior (Figure 4.4). This has been described as an ideal-despotic distribution (Fretwell 1972), and in this case could result in no net edge effect for pademelons, but an edge effect for wallabies.

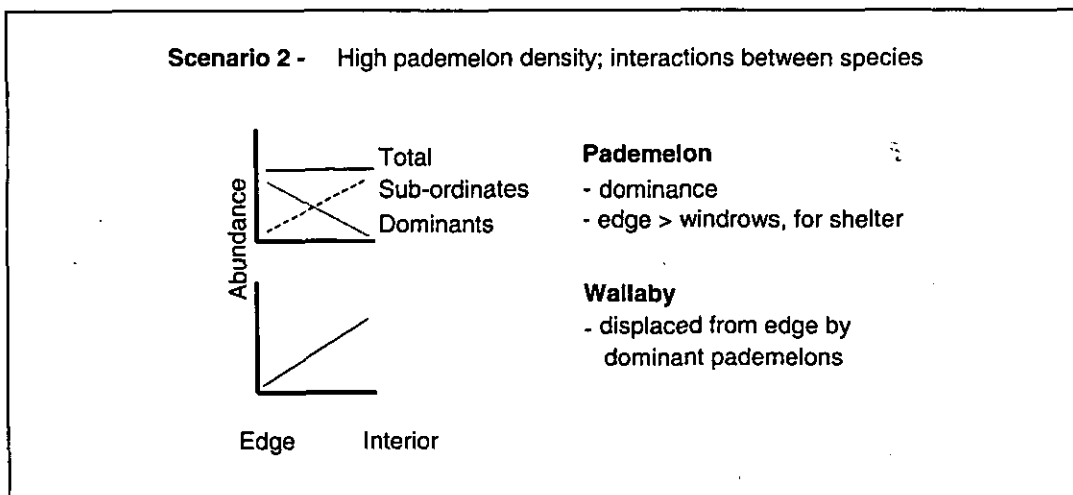


Figure 4.4 A diagrammatic representation of Scenario 2 for explaining the spatial distribution of pademelons and wallabies on the young plantation during high pademelon density.

In contrast, when pademelon density is low, an edge effect for pademelons would be apparent because the majority of animals could use the edge without invoking aggressive interactions. Wallabies may or may not show an edge effect, depending upon how they perceive the risk of predation throughout the young plantation (Figure 4.5).

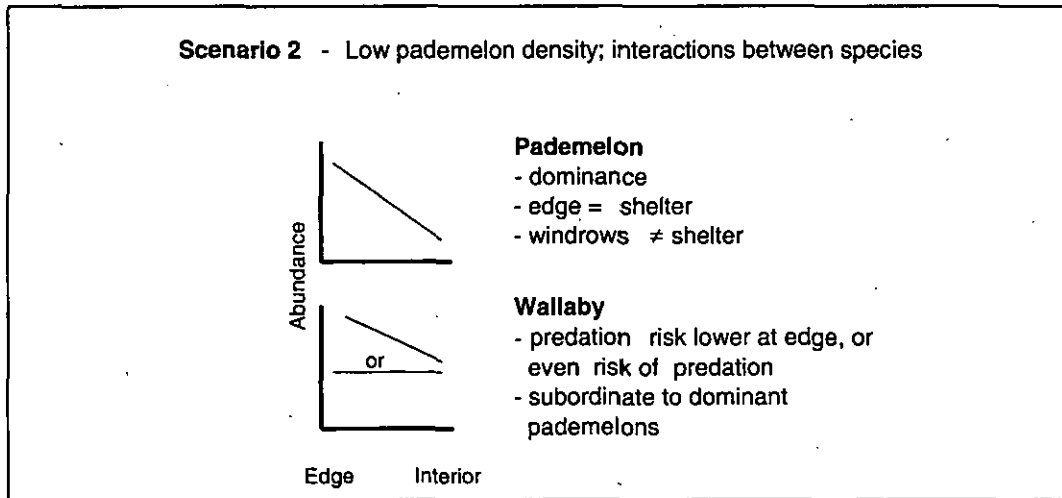


Figure 4.5 A diagrammatic representation of Scenario 2 for explaining the spatial distribution of pademelons and wallabies on the young plantation during low pademelon density.

In support of this more complicated scenario, pademelons have been observed aggressively defending bait stations and displacing wallabies (N. Marsh *pers. comm.*), despite the fact that pademelons are the smaller species. Furthermore, results from a poisoning operation on this site at the end of the study showed a highly biased male sex ratio for pademelons killed (80 % male) plus a low kill-rate for wallabies (Chapters 5). As the majority of poisoned bait had been distributed along the young plantation perimeter, these kill statistics are consistent with effective targeting of male pademelons but ineffective targeting of subordinate pademelons and wallabies. This in turn could reflect their different exposures to bait due to their respective distributions on the young plantation.

4.4.4 Herbivore densities before and after planting

Pademelon and wallaby densities did not change in response to planting eucalypt seedlings on the young plantation. Consequently, *E. nitens* seedlings did not appear to attract animals onto the site. This result is consistent with captive animal trials, which have shown that *E. nitens* is not a preferred food of pademelons (Procter 1998, McArthur *et al.* 2000). Wallabies preference for *E. nitens* is not known. Bulinski (1999a) also found no correlation between macropod pellet abundance and damage to *E. nitens* seedlings across 32 Tasmanian plantations, suggesting that the abundance of macropods between sites was not determined by feeding on *E. nitens* seedlings. Browsing damage to seedlings and the consumption of ground vegetation by herbivores on the young plantation, will be discussed in Chapters 10 and 11.

Chapter 5

Population Surveys: 1080-Poisoning Operation

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5.1 Introduction

Vertebrate browsing damage has long been recognized as reducing productivity in eucalypt plantations (Mollison 1960, Gilbert 1961, Cremer 1969, Montague 1996). Browsing damage can lower plantation productivity by reducing seedling growth and survival and by promoting the development of multiple leaders (Wilkinson and Neilson 1995, Montague 1996, Bulinski and McArthur 1999). Within Tasmania, management of browsing damage commonly involves reducing local herbivore populations immediately before seedlings are planted and sometimes after planting while they are still small enough to be vulnerable. These control measures involve poisoning with '1080' (sodium monofluoroacetate) or shooting animals and are based on the assumption that fewer animals will result in less damage. As animals targeted by the poisoning or shooting operations include native species, and operations are conducted without population monitoring, there is public concern over the use of these practices. Consequently, quantitative data on the effects of these methods are important, but such data are rare and are contained mainly within unpublished reports.

The only two previous studies in the scientific literature investigating the effects of 1080-poisoning programs on herbivore populations in commercial forestry in Australia have used indices of relative abundance to monitor differences in herbivore numbers between poisoned and unpoisoned plantations. Bulinski (2000) used scat surveys to compare a range of poisoned and unpoisoned plantations, and Marsh (1998) used strip transect surveys collected along tracks to compare one poisoned and one unpoisoned plantation. While these methods provide some information on patterns of abundance within species, they do not enable accurate estimates of population sizes, and comparisons between species are not valid (Southwell 1989). In addition, both studies only monitored herbivore numbers on the targeted plantations; species' abundance in the surrounding environment was not investigated, and Bulinski (2000) did not collect pre-poisoning data. Hence, the present study aimed to (1) obtain estimates of absolute density data before and after poisoning on a plantation, which could then be compared between species, and (2) determine changes in densities both on the poisoned plantation and in the surrounding environment. Five species of herbivore were monitored: the four target species (the red-bellied pademelon, the Bennett's wallaby, the common brushtail possum, and the introduced European rabbit) and a non-target species, the common wombat. Wombats are not known to browse seedlings but they are abundant within the herbivore community in forestry environments and have been killed incidentally during previous 1080 operations (McIlroy 1982, Triggs 1996).

5.2 Materials and Methods

5.2.1 Study site

The 428 ha study area is described in Chapter 3. Four habitats dominated this site: (1) a young plantation; (2) older plantations of *E. nitens*; (3) grassland; and (4) native forest. The young plantation was planted with *E. nitens* seedlings in November 1997, three months before this study began. It had not been treated with 1080 poison or herbicide before planting, but vegetation within a 0.5 m radius of individual seedlings was treated with Roundup® herbicide approximately one week after the seedlings were planted.

5.2.2 1080 poisoning operation

Three weeks before 1080 poisoning, bait stations were established at 10 m intervals along the young plantation's centre and boundary, and along nearby firebreaks. Fresh chopped carrot was placed at bait stations on five occasions (referred to as 'free-feeds'), 3-6 days apart, over a period of 17 days in April 1998. Fifty kg of bait was distributed on and around the young plantation during each of the first two free-feeds; 60 kg was used in each of the last three free-feeds. Sixty kg of poisoned bait (chopped carrot mixed with 0.014 % 1080 in liquid solution and blue dye) was distributed at the bait stations (referred to as the 'bait-line' in Figure 5.1) three days after the final free-feed. Bait stations were checked for remaining poisoned bait 24 hours after distribution.

5.2.3 Density estimates

Nocturnal line-transect sampling was used to monitor changes in species' density over time. Data were collected during four sampling periods: (1) before free-feeding, (2) during free-feeding, (3) immediately post-poisoning, and (4) six weeks post-poisoning. Each sampling period consisted of six nights spotlighting, 2-3 nights apart, over 17 days. Sampling methods are described in detail Chapter 3. In brief, 84 permanent transect lines were located within four habitats: 24 on the young plantation, 30 in older plantations, 10 in grassland and 20 in native forest. Transect lines were surveyed at night by a two-person team walking in single file at 1 km.h⁻¹, using a 100 W spotlight. Both ground and canopy were searched for animals. Perpendicular distance to sighted animals was recorded using established distance categories marked by wooden stakes and reflectors. Data collection began one hour after sunset and was completed within six hours.

5.2.4 Data analysis

Probabilities of detection were calculated from distance data using line-transect methodology (Buckland *et al.* 1993) and the computer program DISTANCE (Laake *et al.* 1994). To accurately estimate a sightability parameter ($\hat{f}(0)$) for a given species in a particular habitat type, 60-80 sightings are recommended (Buckland *et al.* 1993). This sample size was not achieved during individual sampling periods for any species in any habitat. Consequently, $\hat{f}(0)$ was calculated from data collected during the present study, pooled with data from a longer 11-month study immediately preceding this one, at this site. Data were pooled over time for each species in each habitat type, on the assumption that variation in sightability

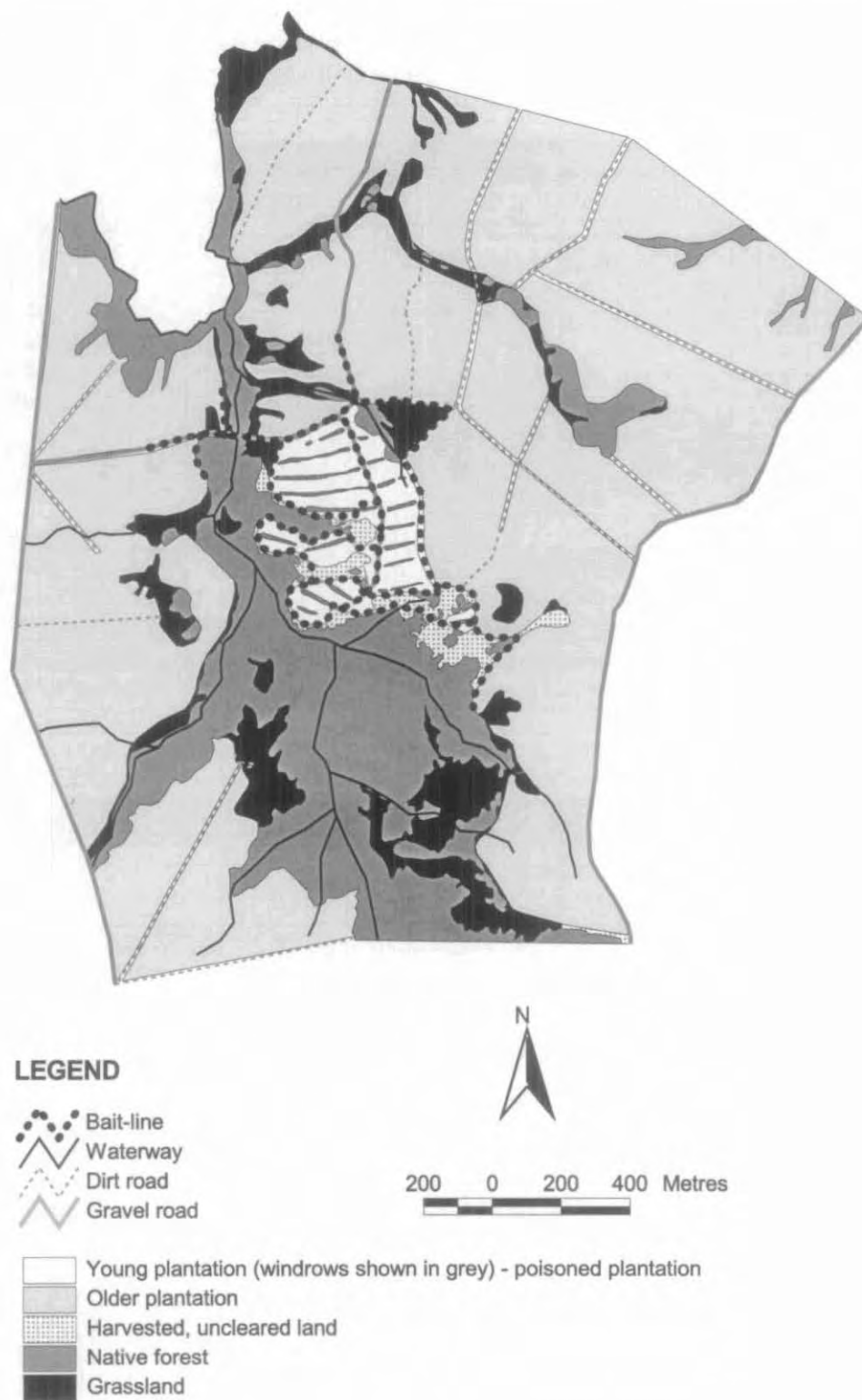


Figure 5.1 Map of the 428 ha study site, showing the four dominant habitats and the location of the bait-line.

across time was likely to be small relative to variation between species and habitats. The use of markers for distance estimation resulted in grouped data and no further grouping was attempted. Data were truncated to strip half-width distances of 70 m for the young plantation, 50 m for grassland, 35 m for native forest and 21 m for older plantation. Strip half-width (w) is the width from the transect line to the outermost edge of the survey area on one side of the line, and therefore half of the total width sampled for a transect line. Data were analyzed for individuals rather than clusters (Buckland *et al.* 1993), as the five species consist of essentially solitary animals that may feed in close proximity but do not behave as a unit (Calaby 1991, Johnson and Rose 1991).

Akaike's Information Criterion was used to select the best of four candidate line-transect models (key function/adjustment: uniform/cosine, uniform/polynomial, half-normal/hermite and hazard-rate/cosine) (Buckland *et al.* 1993). Where too few sightings were collected to produce valid sighting histograms for line-transect analysis, densities were estimated using strip transect analysis (Buckland *et al.* 1993).

5.2.5 Overall density

Population stability in the entire 428 ha study area was examined for each species by graphing overall density (\hat{D}_T) over time. Overall density was calculated for each sampling period, from the estimated densities for the four separate habitats and the proportion of habitat available as:

$$\hat{D}_T = \sum [\hat{D}_i \cdot (A_i / A)] \text{ and } SE_{(\bar{x})} \hat{D}_T = \sqrt{[\sum \{(A_i / A)^2 \cdot \text{var}(\hat{D}_i)\}]}$$

where A is the area of the entire study site (G. McPherson, *pers. comm.*).

5.2.6 Habitat use

Species' mean densities within the four habitats were calculated for each sampling period and examined for patterns within and between habitats over time. Density data could not be normalized by transformation, due to the predominance of zeros (absence) within data sets. Differences in density estimates within habitats between sampling periods of interest were therefore compared with the null hypothesis of no difference, using the non-parametric Wilcoxon Signed Ranks test, with $\alpha = 0.012$ after Bonferroni adjustment for multiple comparisons. Statistical analyses were conducted using the univariate procedure in SAS (SAS 1990).

5.3 Results

Line-transect analysis was possible for pademelons in three habitats (young plantation, grassland and older plantations), wallabies in two habitats (young plantation and grassland) and wombats in one habitat (young plantation). The other species/habitat combinations were analyzed using strip-transect analysis. Rabbits were only observed in one habitat (young plantation), while wombats were observed in three habitats (no sightings in native forest).

Overall density results showed that pademelons were the dominant species within this environment, followed by wallabies, wombats, possums and rabbits (Table 5.1).

Table 5.1 Species' densities (\pm s.e.) at a overall scale and on the young plantation before free-feeding.

Species	Overall density (\pm s.e.) (animals.ha ⁻¹)	Density (\pm s.e.) on the young plantation (animals.ha ⁻¹)
Pademelon	1.12 \pm 1.51	7.73 \pm 1.70
Wallaby	0.41 \pm 0.79	1.83 \pm 0.40
Wombat	0.05 \pm 0.11	1.07 \pm 0.53
Possum	0.03 \pm 0.09	0.00 \pm 0.00
Rabbit	0.00 \pm 0.00	0.11 \pm 0.07

The five species' populations were relatively stable throughout the study (Figure 5.2).

Before free-feeding the ranking of species' densities on the young plantation was consistent with the overall pattern (Table 5.1). Estimated densities of pademelons and rabbits on the young plantation did not change significantly during, compared with before, free-feeding (Figure 5.3)($S = 13$, d.f. = 23, $P = 0.528$; $S = 0.00$, d.f. = 23, $P = 1.00$, respectively).

Estimated densities of wallabies and wombats on the young plantation tended to be lower during, compared with before, free-feeding but the differences were not significant (Figure 5.3) ($S = 25.4$, d.f. = 23, $P = 0.116$; $S = 5$, d.f. = 23, $P = 0.25$, respectively). No possums were observed on the young plantation throughout this period.

All poisoned bait was taken within 24 hours of distribution. Pademelon density declined significantly between pre-free-feeding and immediately post-poisoning on the young plantation ($S = 65$, d.f. = 23, $P = 0.000$) (Figure 5.3a), but not in other habitats ($P \geq 0.34$, in all cases). Wallaby density on the young plantation also declined significantly during this period ($S = 32$, d.f. = 23, $P = 0.008$) (Figure 5.3b), while density in other habitats remained unchanged ($P \geq 0.5$, in all cases). Densities of possums, rabbits and wombats on the young plantation did not change significantly following poisoning compared with before free-feeding ($P = 0.5$, $P = 0.5$ and $P = 1.0$, respectively) (Figures 5.3c-e).

Six weeks after poisoning, pademelon density on the young plantation had increased compared with immediately post-poisoning ($S = -47.5$, d.f. = 23, $P = 0.001$), but was still significantly lower than before free-feeding ($S = 61$, d.f. = 23, $P = 0.012$). Pademelon density in other habitats were unchanged throughout this period ($P > 0.1$, in all cases) (Figure 5.3a). Six weeks after poisoning, wallaby density on the young plantation was again similar to before free-feeding ($S = -22$, d.f. = 23, $P = 0.268$). Wallaby density in other habitats did not change significantly during this period ($P \geq 0.5$) (Figure 5.3b). No significant changes were detected for possum, rabbit or wombat densities on the young plantation between immediately post-poisoning and six weeks after poisoning ($P = 0.5$, $P = 1.0$ and $P = 0.9$, respectively) (Figure 5.3c-e).

5.4 Discussion

Data presented here provide the first quantitative information on absolute changes in animal densities in relation to a poisoning operation in Tasmanian forests. Ideally we would have compared a large random sample of poisoned and unpoisoned plantations, or a single closely matched unpoisoned plantation as a control (as discussed in Hurlbert (1984) and Stewart-Oaten *et al.* (1986)). Neither situation was logistically possible. Equivalent difficulties in obtaining between-site replication are inherent in many large scale ecological studies (for example, Efford *et al.* (2000) and Pople *et al.* (2000)). We can be confident of results of density changes in all such studies, provided the methods are satisfactory. Generalizations about effects, in our case about the effects of 1080, however, can only be built up by

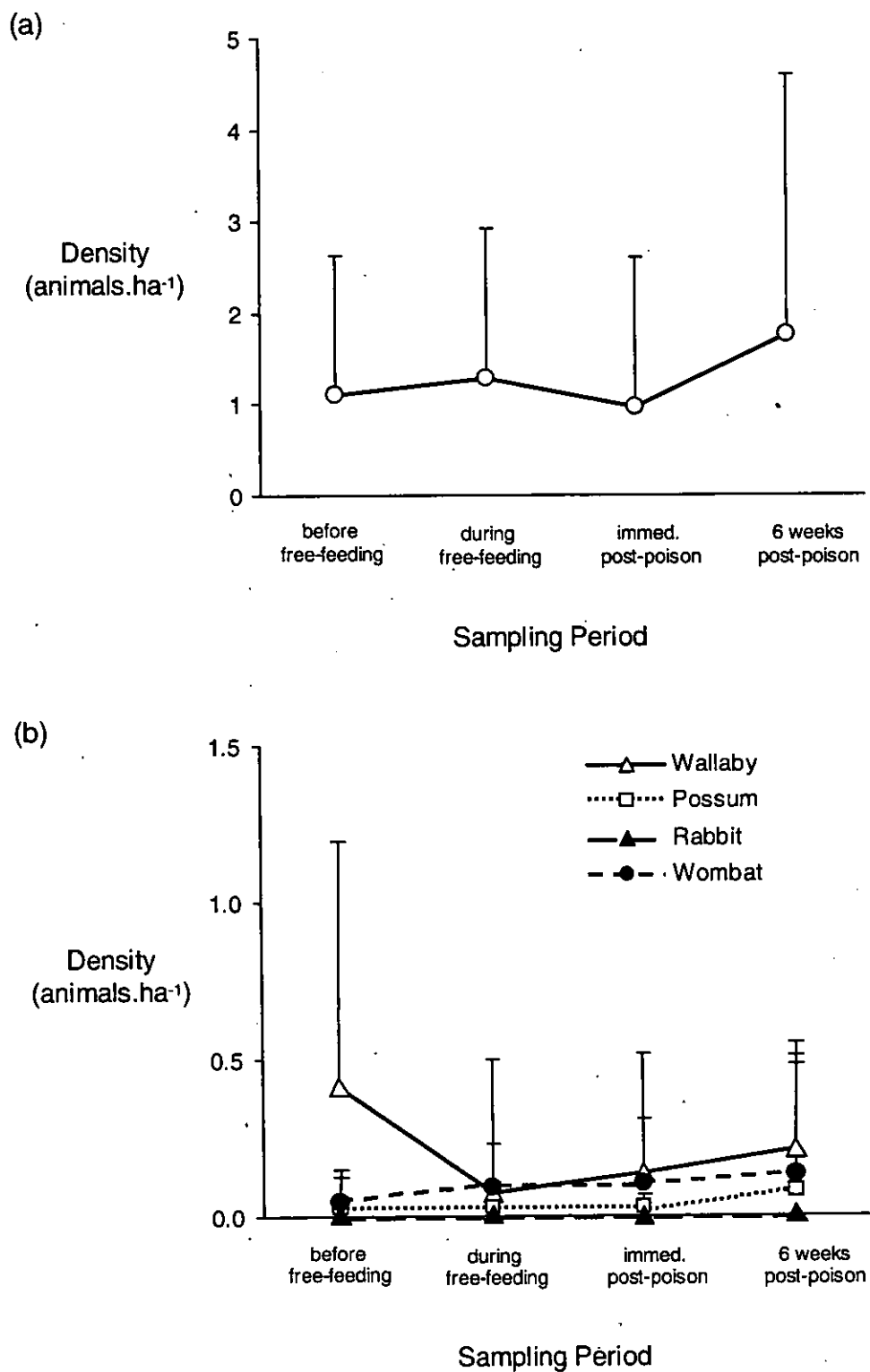


Figure 5.2 The estimated overall densities (\pm s.e.) of (a) pademelons and (b) wallabies, possums, rabbits and wombats before free-feeding, during free-feeding, immediately post-poisoning, and six weeks post-poisoning.

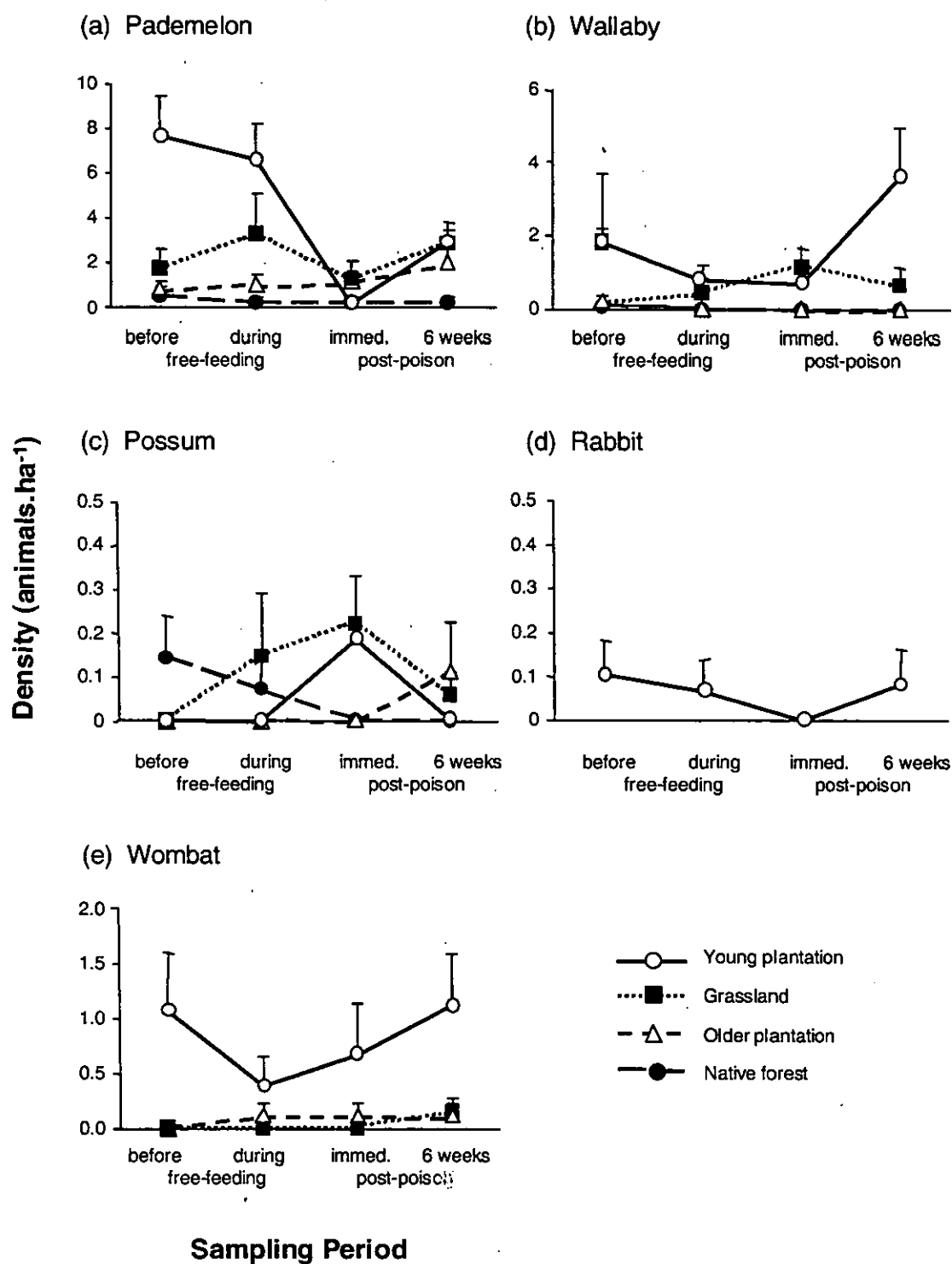


Figure 5.3 The estimated densities (mean \pm s.e.) of (a) pademelons, (b) wallabies, (c) possums, (d) rabbits and (e) wombats in four habitats (young plantation, grassland, older plantation and native forest) before free-feeding, during free-feeding, immediately post-poisoning and six weeks post-poisoning.

comparing single site results with results from other studies, if available. Where possible, we do this in the following sections.

5.4.1 Pademelons

The relative stability of the overall density of pademelons over time suggests that any effects of the poisoning operation were localised. Free-feeding did not draw new pademelons into the region during free-feeding. There was also no shift in densities between habitats within the region, indicating that animals were not attracted onto the young plantation from surrounding habitat. This suggests that only animals within the local population near the bait-line were targeted during this operation. Poisoning effectively targeted pademelons on the young plantation. The 98 % decrease in density following poisoning was assumed to reflect animal deaths and not a general population trend due to other factors, because densities in other habitats did not change at this time, and carcasses had been found on site ($n=25$, Chapter 9). This result is similar to findings from Marsh (1998) and Bulinski (2000). Marsh (1998) found estimated pademelon densities were 97 % lower on one poisoned plantation compared with a nearby unpoisoned plantation. Bulinski (2000) found that at 80 days after planting, cumulative pademelon scat density on poisoned plantations was lower than on unpoisoned plantations.

A very high kill-rate for pademelons may be related to two factors. First, pademelons have a relatively low tolerance to 1080 compared with the other herbivores present (McIlroy 1982). Second, pademelons may dominate bait stations, restricting other species from accessing bait. Nocturnal filming at bait stations has shown that pademelons aggressively defend bait stations from wallabies (N. Marsh *pers. comm.*). The possible dominance of pademelons over wallabies was unexpected, given that pademelons are the smaller species (Calaby 1991, Johnson and Rose 1991). This potential dominance requires further investigation because it has important implications for the effectiveness of 1080 at targetting wallabies, and the subsequent impact on browsing damage (discussed below).

By six weeks after poisoning, pademelon density on the young plantation had increased but was still lower than before free-feeding. The long-term implications of this trend are not known, but will depend upon the source of the density increase. If the increase represents neighbouring animals extending their home-range as a result of a vacuum effect, as previously reported for brushtail possums (Efford *et al.* 2000), then poisoning may reduce pademelon density in the long term. If, however, the increase represents dispersing pademelons migrating into the area, density on the young plantation may continue to increase until it reaches pre-poisoning levels. This situation would not reduce pademelon density in the long term, but it may create a window of time in which seedlings are relatively free of browsing damage. Consequently, the source of the density increase on the young plantation following poisoning is worthy of investigation as it has important implications for managing damage to seedlings, and site utilization by other species (see below).

5.4.2 Wallabies

As with pademelons, overall densities results for wallabies suggested that any effects of the poisoning operation were localised. Free-feeding appeared to cause a slight but not significant decline in wallaby density on the young plantation. If this trend is real, it may reflect a change in foraging behaviour at this time, rather than a true density decline. Wallabies appeared to be active earlier in the day than usual during this period (K. le Mar *pers. obs.*), and hence fewer animals may have been active during spotlighting. Wallaby density declined 60 % from before free-feeding to immediately post-poisoning on the young plantation. However, because the declining trend was already apparent during free-feeding,

it is not clear whether the poisoning operation was totally responsible for this change. Furthermore, only three carcasses were found on the young plantation after poisoning (Chapter 9). Mixed results have been reported for the effectiveness of 1080-poisoning on reducing wallaby abundance on plantations. Marsh (1998) reported a 75 % lower density on one poisoned plantation than one unpoisoned plantation, while Bulinski (2000) found no significant difference in cumulative scat density at 80 days between poisoned and unpoisoned plantations.

Six weeks after poisoning, wallaby density increased again on the young plantation, presumably reflecting movement of individuals from surrounding habitat. We hypothesize that this influx was triggered by the sustained lower (albeit increasing) density of the more dominant pademelon.

The impact of replacing many pademelons with some pademelons and some wallabies is currently unknown. *Eucalyptus nitens* seedlings are not a preferred food item for pademelons (Procter 1998, McArthur *et al.* 2000). Preferences for plantation species by wallabies have not been investigated, but are clearly important before the relative impact of these two species can be determined.

5.4.3 Possums and rabbits

Densities of the two other target species (possums and rabbits) were not reduced significantly by the 1080-poisoning operation. However, their densities were extremely low, which may have precluded any decline from being detected. No change in possum and rabbit density due to poisoning, however, is consistent with Marsh's (1998) and Bulinski's (2000) results. Seven possum carcasses and one rabbit carcass were found after poisoning (Chapter 9).

5.4.4 Wombats

Density changes for wombats on the young plantation followed a similar pattern to wallabies, but no trends were significant. On this basis, the 1080-poisoning operation did not cause a detectable decline in wombat density either on the treated plantation, or within the larger region. No other studies have quantified population changes in similar situations, so that no comparisons are possible at this population level. No wombat carcasses were found after poisoning (Chapter 9), although it cannot be inferred that no animals were killed, since carcasses may have been in burrows.

5.5 Conclusion

Although statistical differences in densities cannot alone infer that the 1080 operation caused any changes, it is reasonable from a biological basis (including the presence of carcasses) to conclude that 1080 was responsible for reducing local macropod populations during this study. Our results indicated a large effect of 1080 on the local pademelon population and at least some effect on the wallabies. The fact that these patterns are similar to those found by Bulinski (2000) and Marsh (1998) using other methods for estimating animal densities on other plantations, suggests that these patterns may be reasonably common.

Although the proximate aim of any 1080 operation in forestry is to reduce animal numbers, the ultimate aim is obviously to reduce browsing damage to seedlings. This study raises a number of questions that need to be investigated in this regard. These include the potential long-term effects of poisoning on animal numbers and species interactions. Both of these

factors influence population dynamics between species that forage on a plantation, and ultimately may affect the amount of browsing damage that occurs.

Chapter 6

Radio-Telemetry: Accuracy of the System

6.1 Introduction

Animals' movements can be detected remotely by putting radio-collars on individuals and estimating their locations from bearings taken at known sites. Positional data collected in this manner are not exact however, because true bearings to animals are unknown (White and Garrot 1990). Consequently, it is useful to quantify the accuracy of the telemetry system, before interpreting animal movement data (Saltz and Alkon 1985).

Telemetry system accuracy is usually quantified in one or more of the following ways: (1) bearing, (2) linear and (3) areal error (Heezen and Tester 1967, Saltz and White 1990, Zimmerman and Powell 1995). Bearing error is used to check the precision of the receiving stations, while linear and areal errors describe the accuracy of animals' estimated locations (Zimmerman and Powell 1995). Linear error describes accuracy in terms of distance, which is important for analysing how far animals travel, or their rate of movement (Saltz and White 1990). Areal error provides error in units of area, which is important when analysing animals' patterns of habitat use (Saltz and White 1990). If areal error is to be used in habitat selection studies however, acceptable error size for detecting selection will depend upon the patchiness of the environment, hence the need to also report average habitat patch size for a given study site (Saltz 1994).

Bearing accuracy, linear and areal errors can all be estimated from a single field trial. This involves placing transmitters at known sites around the study area, and estimating their location using the radio-tracking method. Resulting positional data are then compared with transmitters' known locations, to quantify error or accuracy (White and Garrot 1990, Zimmerman and Powell 1995).

Several reviews have documented how few published articles report telemetry error (e.g., Saltz 1994 and Simpkins and Hubert 1998), but many authors still fail to test or report system accuracy (e.g., Myserud *et al.* 1999, Brøseth and Pedersen 2000, and Johnson *et al.* 2000). Furthermore, if authors do report telemetry error, they often fail to adequately describe how the accuracy tests were conducted, and in particular, the number of test points used (e.g., Priddel *et al.* 1988, Palomares *et al.* 2000, and Revilla *et al.* 2000). It has also been noted that authors may provide error in units inappropriate to the study (Saltz and White 1990, Saltz 1994). For example, studies examining habitat selection should report error in units of area, rather than distance (e.g., Palomares *et al.* 2000, and Revilla *et al.* 2000). The aim of this chapter was to quantify the bearing accuracy, linear error and areal error of the radio-tracking system that will be used in Chapter 7 to investigate animal movement data.

6.2 Materials and Methods

6.2.1 Test points

Fieldwork was conducted over a four-day period in March 1998. One hundred and sixty-three test points were used, each consisting of a transmitter taped (≈ 1 m high) to a wooden stake, hammered into the ground. This height represented the approximate neck height of a Bennett's wallaby; one of the species used in the Chapter 7. Test points were located at

predetermined sites that represented a range of directions and distances from towers, vegetation types and topographic features (Lee *et al.* 1985) (Figure 6.1). Test points were also positioned at least 15 m inside habitat boundaries, because this distance was thought adequate to produce signal absorption indicative of the habitat type.

Correct test point locations were mapped manually, within an ArcView® GIS (ESRI 1996). The overall accuracy of the mapped data was conservatively estimated at ± 30 m. Errors for test points within two habitats (young plantation and uncleared harvested land), however, would have been less than ± 30 m, as they were mapped using measurements (distance and magnetic bearing) from GPS points (accuracy ± 1 m).

6.2.2 Radio-tracking system

Each single-stage radio-transmitter (Sirtrack Electronics, New Zealand) emitted a unique frequency between 150-152 MHz, enabling individual devices to be recognized. Transmitter signals were detected from each receiving station using an Automated Telemetry Services receiver and a directional, null-peak design antenna mounted on a 6 m metal frame, housed within a caravan. Three receiving stations ('towers') were located approximately 1.9 km apart, in an equilateral triangle around the centre of the study site as recommended by White (1985) (Figure 6.1). This tower arrangement had been tested during pilot studies and appeared to represent the optimal design for a three-tower system at this site. Locations of towers were known to an accuracy of ± 1 m. Bearings were read to the nearest 0.5° from a compass rosette that rotated with the mast supporting the antenna. Mast orientation was checked at the start of each day. Individual transmitters were tracked simultaneously by a tower operator in each of the three receiving stations. A fourth person was in radio contact with the three tower operators to coordinate tracking activities and map bearings as they were collected, in order to eliminate back-bearings (MacDonald and Amlaner 1979). Positional data were collected as magnetic bearings from each of the three towers to the individual transmitter. Approximately six minutes were spent on each transmitter, as this reflected the time spent following a radio-collared animal in Chapter 7. In addition, a control transmitter was also relocated each hour to monitor any variations in radio-waves due to weather, equipment problems or human error. All three radio-trackers had extensive experience with operating radio-tracking equipment and collecting data at this site.

Only ten transmitters were available during the course of this work, consequently two people were present within the study site, moving transmitters between test points while trackers collected data on other (stationary) collars. The ground crew was in radio contact with the mapper to ensure that data had been collected for each test point before transmitters were moved to subsequent locations.

Only locations estimated in the null antenna phase were used in the analysis. Radio-tracking data were prepared for analysis using three steps: (1) bearings were converted from magnetic north to grid north, (2) data were corrected for individual tower bias, and (3) data were checked for intersecting bearings, using the software package Location Of A Signal (LOAS®, ESS 1999). Data with three intersecting bearings were triangulated using the Maximum Likelihood Estimator in LOAS® (ESS 1999). Data with only two bearings, either because only two out of three locations intersected, or because the third tower could not detect a null signal, were biangulated using the Best Biangulation Estimator in LOAS® (ESS 1999).

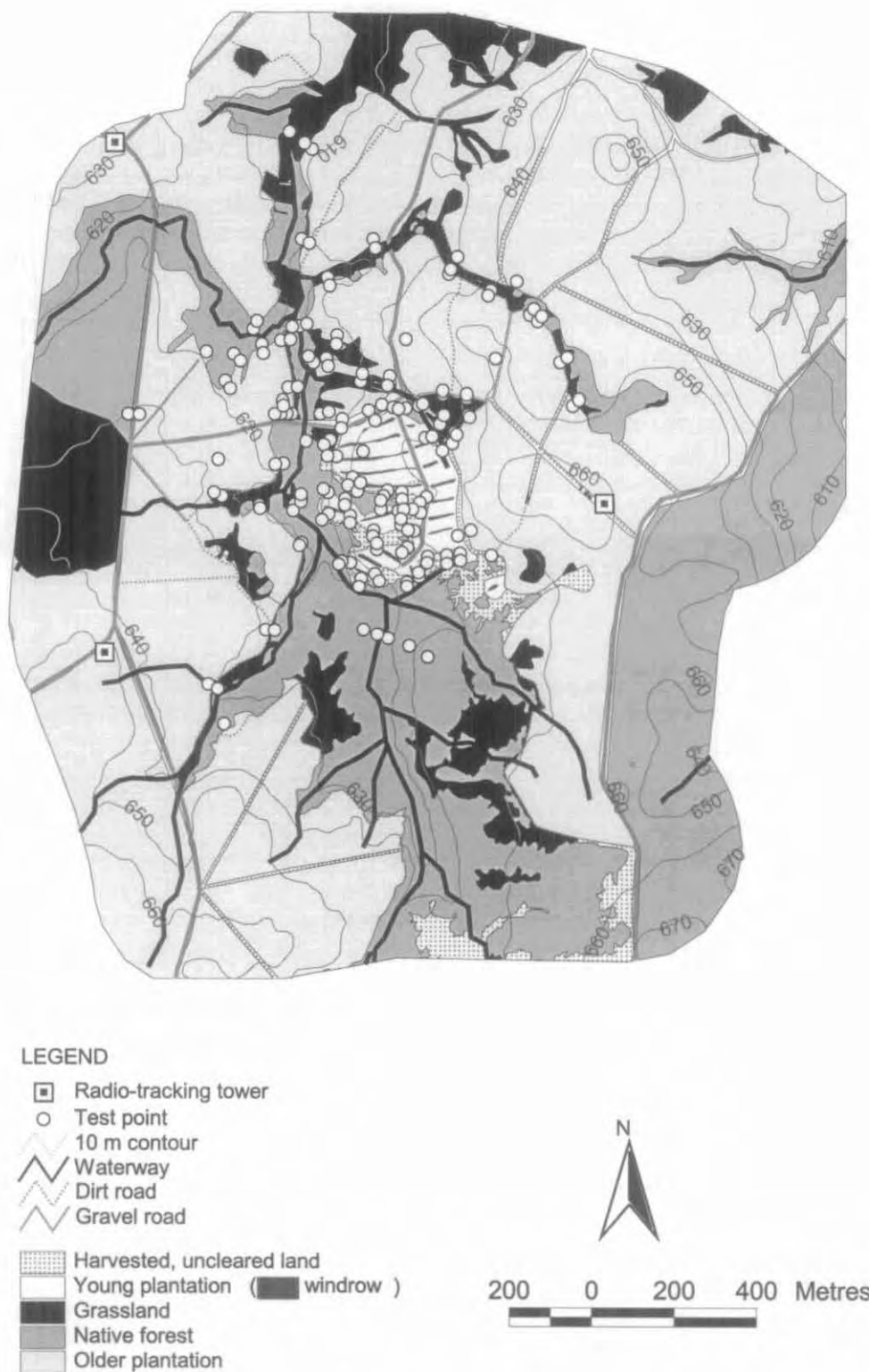


Figure 6.1 Topographic map of the 770 ha radio-tracking study site, showing the locations of 163 test points used to quantify telemetry system error.

6.2.3 Calculating error

Three sources of error were examined for the radio-tracking system. First, precision of the bearings were calculated from differences between the correct bearing from individual towers to test points, and the radio-tracked bearing to test points (White and Garrott 1990). Second, linear errors were calculated from the distances between correct test point locations and estimated test point locations from triangulated bearings (White and Garrott 1990). Authors often report mean linear error with standard error, but this is not strictly correct as linear error data is usually not normally distributed (Zimmerman and Powell 1995). Consequently, we normalized data (log transformation), calculated mean linear error and standard error, and then backtransformed results. Third, the mean areal error was calculated as the size of the average 95 % confidence circle centred over each point estimate. This value was calculated using Error Area (E_A) = πL^2 , where radius (r) is the standard deviation of linear errors multiplied by 1.96 (z statistic) (Zimmerman and Powell 1995). The standard deviation used here was the average of the upper and lower standard deviations from the backtransformed linear error data.

6.3 Results

Eighty-nine percent of transmitters were detected by three towers. Ten percent were detected by two towers, and 1 % was detected by only one tower. Mean tower bearing error (\pm s.e.) was as follows: Tower 1, $-1.05^\circ \pm 0.32^\circ$ ($n=159$); Tower 2, $0.21^\circ \pm 0.31^\circ$ ($n=161$); and Tower 3, $-1.04^\circ \pm 0.35^\circ$ ($n=148$). Histograms of bearing errors for each tower are shown in Figure 6.2.

Radio-tracked bearings for nine of the 144 test points detected by all three towers, were found to have two non-intersecting bearings and were biangulated. Bearings for the remaining 135 test points could be triangulated to estimate transmitter locations. Backtransformed mean linear error (\pm s.d.) of the triangulation system, for both the entire study site and individual habitats, are shown in Table 6.1.

Table 6.1 Backtransformed mean linear error (\pm s.d.) for the entire study site and individual habitats from triangulated data, and the mean linear error (\pm s.d.) for biangulated data.

Data	Habitat	Number of test points	Mean linear error (m)	- s.d. (m)	+ s.d. (m)
(1) Triangulated	Entire study site	135	66	35	72
	Young plantation	25	61	33	72
	Older plantation	40	69	35	70
	Native forest	42	75	42	95
	Grassland	13	65	35	74
	Harvested land	15	45	16	26
(2) Biangulated	Entire study site	26	72	34	153

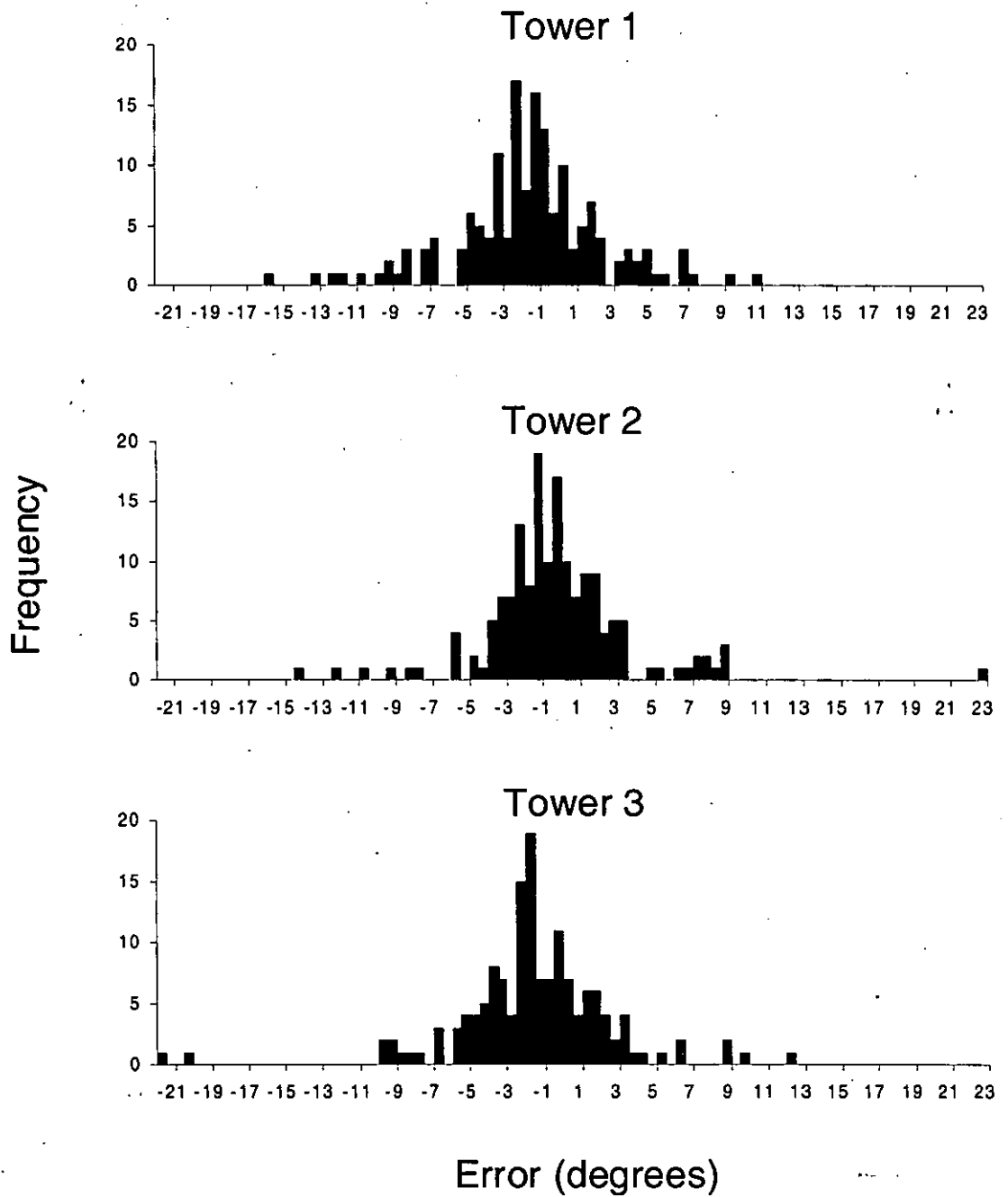


Figure 6.2

Frequency histograms of the bearing errors, calculated as the difference between actual bearings and radio-tracked bearings to individual test points.

The average standard deviation for the backtransformed mean linear error of triangulated data for the entire study site, was 53 m ($n=135$). Using this value, areal error (95 % confidence circle) for the telemetry system was 3.45 ha. Areal error for the five habitats (using their respective backtransformed mean linear error), are given in Table 6.2. The backtransformed mean linear error (\pm s.d.) for the biangulated data is shown in Table 6.1. The average standard deviation for backtransformed mean linear error data was 60 m ($n=26$). Using this value, areal error for biangulated data was 4.28 ha.

Table 6.2 Areal error for the entire study site and individual habitats from triangulated data.

Habitat	Areal error (ha)
Young plantation	3.34
Older plantation	3.31
Native forest	5.67
Grassland	3.56
Harvested land	0.53

Patch size varied considerably within the 770 ha study area. Habitat patch statistics are shown in Table 6.3. The largest patches comprised older plantation. The smallest patches comprised a single tree derived from native forest, usually in a larger patch of harvested uncleared land.

Table 6.3 Descriptive statistics on habitat patch sizes within the 770 ha study site.

Habitat	Total area (ha)	Number of patches	Mean patch size (ha)	Median patch size (ha)	Range (ha)
Young plantation	18.0	2	9	n.a.	2.0 – 16.0
Older plantation	435.1	4	108.8	106.7	1.5 – 232.1
Native forest	221.8	28	7.9	0.2	<0.1 – 132.4
Grassland	77.9	28	2.8	1.0	0.1 – 20.1
Harvested land	14.0	4	3.5	3.1	1.0 – 6.7

6.4 Discussion

The bearing accuracy of the telemetry system was similar to other studies; for example, Laundré *et al.* (1987), Pyke and O'Conner (1990), Taylor (1993), Chamberlain and Leopold (2000), and Relyea *et al.* (2000). Results for individual tower error were then used (Chapter 7) to correct bearings collected from radio-collared animals at this site, thereby reducing one source of error within the data.

Linear error for the telemetry system was similar to other studies, by representing < 5 % of the average distance between towers. For example, Pyke and O'Conner (1990) 3 % [mean linear error 12 m, towers 400 m apart]; Slade *et al.* (1965, in Pyke and O'Conner 1990) 10 % [50 m, 500 m apart]; and Priddell (1983, in Pyke and O'Conner 1990) 2 % [210 m, 9.5 km apart].

Areal error varied between habitats and reflected vegetation density. Error was greatest within native forest, probably as a result of signals being absorbed by the vegetation. Error was particularly low within harvested uncleared land, which was somewhat surprising. Although this habitat lacked canopy cover, wood debris was piled high at ground level, which was anticipated to absorb or reflect radio-waves, thereby causing high error. Consequently, although areal error was relatively consistent with vegetation density, testing

telemetry system accuracy within all habitats likely to be encountered by study animals is valuable for interpreting subsequent analyses involving animal movement data. Only a single error value will be applied when analysing animals' patterns of habitat selection in Chapter 7, however, because the true location of these animals at any one time were not known, and hence could not be assigned to a single habitat type.

Areal error for the telemetry system was similar to Chamberlain and Leopold's (2000) reported error. Patch size however, varied considerably between the two studies. Within Chamberlain and Leopold's (2000) study, minimum patch size was 5 ha compared with the present study's <0.1 ha. Nams (1989) has reported that areal error can be 1.5 times mean habitat patch size, and still provide sufficient power to be used in habitat selection analyses to detect animals' patterns of habitat use. Consequently, in the present study, the average acceptable patch size was 2.30 ha, and this size requirement was met by all five habitats. Furthermore, large proportions of each habitat comprised individual patches that exceeded 2.30 ha: 100 % of older plantation, 98 % of native forest, 89 % of young plantation, 83 % of harvested uncleared land, and 77 % of grassland. It should, therefore, be possible to detect selective use in most parts of these habitats. Patch shape for native forest, harvested uncleared land and grassland, however, was not necessarily round. This can be problematic when the error centred over each point estimate is circular (Nams 1989), but nothing could be done to overcome this issue.

Errors associated with biangulated data were larger than those estimated from triangulated data. This result was expected because the third bearing makes the system more precise. Biangulated data were therefore only used where triangulation was not possible within the biological data set (Chapter 7).

Theoretically, telemetry system accuracy at this site could have been improved in a number of ways. Firstly, positioning towers closer together generally improves reception (Kenward 1987). Due to the topography of the site, however, relocating towers would have involved moving them from local highpoints into gullies. In turn, this would have reduced reception as well as making human access to the towers difficult. Pilot studies conducted during the eight month preparation time clearly showed that the tower locations used were optimal for this site. Secondly, increasing tower height generally improves reception (Kenward 1987). This was not possible, however, as taller towers would have made the caravans structurally unsound. Radio-tracking in the absence of caravans would have been impossible given the inclement weather at this site, and the length of the radio-tracking sessions. Thirdly, increasing the number of receiving stations generally improves the accuracy of telemetry data (Kenward 1987). Additional fixed stations would have been prohibitively expensive, while a mobile station (for example, using a hand-held antenna from a car) could have introduced confounding effects on animal movements from human activities.

Accuracy tests on telemetry systems can be negatively biased if workers are aware of the purpose of the study (Mills and Knowlton 1989). Blind testing is therefore recommended. This was not possible during the present study, due to logistical constraints. Results presented here, however, were thought to reflect normal system error because unlike Mills and Knowlton's (1989) workers, tower operators were used to frequent random testing, even when they were tracking radio-collared animals.

In conclusion, considering the scale of the accuracy test and the magnitude of the linear and areal errors, the telemetry system can be considered adequately accurate for investigating patterns of animals' movements and habitat use in Chapter 7.

Chapter 7

Radio-Telemetry: Home-Range

7.1 Introduction

Knowledge of a species' utilisation of space is important in understanding its ecology. Studies of space-use often focus on the biological concept of the home-range, where 'home-range' is defined as the area in which an animal carries out its normal activities (for example, travelling, foraging, resting, searching for mates and caring for young), during a specified time period (Burt 1943). Measuring animals' home-range size, shape and patterns of utilisation is important in studies of population density, habitat selection and resource distribution, foraging behaviour and social interactions between individuals (Harris *et al.* 1990).

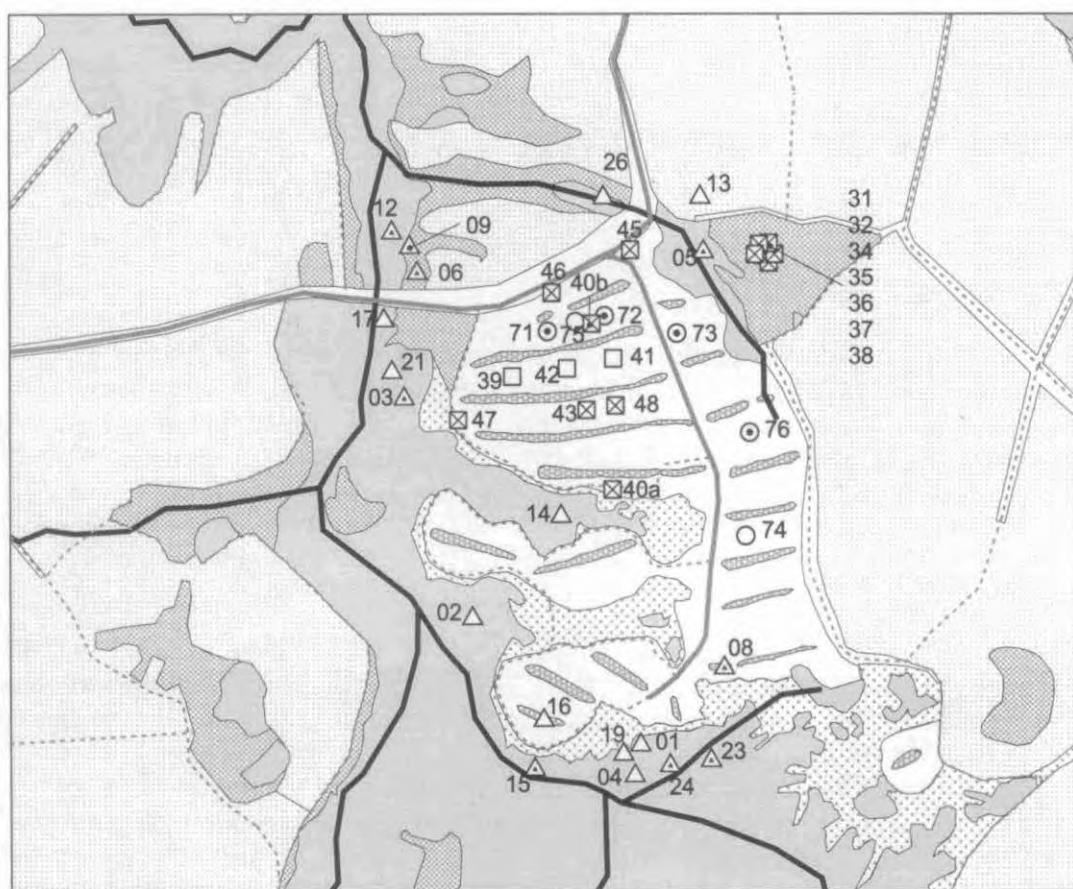
Information on Bennett's wallaby and red-bellied pademelon home-range are scarce (e.g., Johnson [1978] and Mooney and Johnson [1979]). Brushtail possums have been well studied on the mainland, but very little data is available for Tasmanian populations, and studies have shown that results are dependent upon forest type (see review by Green 1984). Hence, mainland possum data cannot necessarily be extrapolated to the Tasmanian, and more particularly, a forestry environment. Due to the scarcity of data on wallaby, pademelon and possum home-range within Tasmanian environments, a radio-telemetry study was conducted at Surrey Hills. The aim was to provide information on inter and intra-specific differences in home-range size and shape.

Materials and Methods

7.2.1 Capture of animals

Wallabies, pademelons and possums were caught on and around the young plantation, as this was the main habitat of interest. Pademelons were caught by trapping and darting animals. The trap was a large fenced area (20 m diameter, fenced with netting 1.8 m high), located on a grassland approximately 75 m north-east of the young plantation. Animals entered through a gate at the base of the netting. Animals within the trap were detected using night vision goggles, whereupon the gate was closed manually by a cord release mechanism from a distance of approximately 70 m. Animals were then caught using a small hand net. Only male pademelons entered the large trap, consequently, dart-gunning animals on the young plantation was employed to catch female pademelons and additional males. It is interesting to note that about 80 % of pademelons darted on the young plantation were male.

Wallabies also evaded the large trap, and smaller wallaby traps (based on Pollock and Montague's [1991] design), located near windrows on the young plantation and in native forest near the plantation edge. Hence, wallabies were also caught using a dart-gun on the young plantation. Possums were caught using Mascot cage traps and small wallaby traps (Pollock and Montague 1991) on the young plantation, and within nearby native forest. Possums were also inadvertently caught in the smaller wallaby traps. Animals' sites of capture are shown in Figure 7.1.



LEGEND

- Female Wallaby
- ⊙ Male Wallaby
- Female Pademelon
- ⊠ Male Pademelon
- △ Female Possum
- ⊠ Male Possum
- Waterway
- == Gravel Road
- - - Dirt Road
- Young Plantation
- ▨ Windrow
- ▨ Older Plantation
- ▨ Grassland
- Native Forest
- ▨ Harvested uncleared land

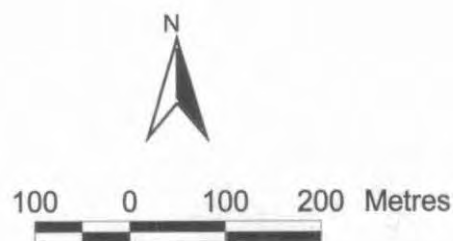


Figure 7.1 Sites of capture for wallabies, pademelons and possums radio-collared during this study. Numbers represent animals' unique tattoo.

7.2.2 Radio-telemetry

Animals were held in hessian sacks and lightly anaesthetised with Zoletil® (tiletamine hydrochloride and zolazepam hydrochloride) during radio-collar attachment (7 mg/kg for dart-gunned macropods, 4 mg/kg for trapped pademelons and 2 mg/kg for possums). Animals were released at the sight of capture within 6 hrs of being caught. Between collaring and release, animals were housed either inside the field hut or the vehicle. Wallabies and pademelons were also given an ear-tag, and all animals were given a unique ear tattoo.

In total, 17 pademelons (14 male and 3 female), six wallabies (4 male and 2 adult female) and 19 possums (9 male and 10 female) were fitted with a single-stage radio-transmitter (Sirtrack Electronics, New Zealand) attached to a leather collar. Each transmitter emitted a unique frequency between 150-152 MHz.

Transmitter signals were detected from receiving stations using an Automated Telemetry Services receiver and a directional, null-peak design antenna mounted on a 6 m metal frame, housed within a caravan. Three receiving stations ('towers') were located approximately 1.9 km apart, in an equilateral triangle around the centre of the study site on local high points (see Chapter 6).

The two female wallabies and three female pademelons all had very small, unfurred, pouch young when they were caught. The reproductive status of female possums was highly variable depending upon the time of year when they were caught. The most common situations were very small, unfurred pouch young and recently evacuated pouches with an active mammary gland.

7.2.3 Data collection

Radio-tracking data were collected during the eight spotlighting sampling periods (May 1997 - March 1998) but on alternative nights to spotlighting. Five sampling periods were conducted before *E. nitens* seedlings were planted on the young plantation in November 1997. The remaining three sampling periods took place after planting. Sampling periods were approximately four weeks apart with the exception of those surrounding planting, which were as close together as possible (less than 2 weeks apart).

Each radio-tracking sampling period comprised nine 12-hour data collection sessions conducted over 17 days. Sessions were classed as 'dark' or 'light'. Dark sessions commenced one hour before sunset and ended 12 hours later. Light sessions consisted of the 12 hours not sampled during the dark sessions. Consequently, the data sessions did not truly represent darkness and daylight. Over winter, darkness extended beyond the 12-hour 'dark' sampling session and in summer, sunrise took place during a 'dark' session. Twelve hour sessions were used throughout the study because, overall, this provided 24-hour data during a given fieldtrip, and it was logistically impossible to radio-track for more than 12 hours at a time, which would have been required for nocturnal sessions during winter. Six dark sessions and three light sessions were conducted each sampling period. Emphasis was placed on dark sessions, as wallabies, pademelons and possums are essentially nocturnal (Strahan 1991).

During each 12 hour data collection session, 5-6 animals and at least one control point, were located every 45 minutes. The control transmitters were located at known points on the young plantation, approximately 1 m above ground level to represent wallaby neck height. The use of these control transmitters enabled any variations in radio-waves due to weather, equipment problems and human error (fatigue), to be monitored within every 45 minute interval. In order to reduce human bias, workers collecting the telemetry data were unaware of which device represented the control (Mills and Knowlton 1989), although they were aware of the testing procedure. Additionally, unknown to the workers, two control points were used from time to time to check this testing system. Approximately 5-6 minutes were

spent tracking each device, enabling workers to have a 10 minute break every 35 minutes. Precision of re-locating animals at 45 minute intervals compared with continuous data sets, is described in Appendix A.

Locations of the towers were known to ± 1 m. Bearings were read to the nearest 0.5° from a compass rosette that rotated with the mast supporting the antenna. Transmitters attached to animals were tracked simultaneously by a tower operator in each of the three receiving stations. A fourth person was in radio contact with the three tower operators to coordinate tracking activities and map bearings as they were collected, to eliminate back-bearings (MacDonald and Amlaner 1979). Position data were collected as magnetic bearings from each of the three towers to the animal.

7.2.4 GIS

The ArcView® GIS described in Chapter 2 was used throughout this study. Five habitat types were recognised: (1) young *Eucalyptus nitens* plantation [young plantation]; (2) 5-7 year old *E. nitens* plantations [older plantation]; (3) native forest; (4) grassland and (5) harvested, uncleared land.

7.2.5 Data analysis

Magnetic bearings were adjusted for control point data, converted to grid north by adding 12.3° (the required conversion for data collected on a 1987 map of the Pearse region), and corrected for individual tower bias (Chapter 5). Before triangulation, data were checked for non-intersecting bearings using the software package Location Of A Signal (LOAS®). Data with three intersecting bearings were triangulated using the Maximum Likelihood Estimator in LOAS® (ESS 1999). If two of the three bearings were parallel, or only two null signals had been collected, data were biangulated using the Best Biangulation Estimator in LOAS® (ESS 1999). Once x and y coordinates were estimated for each data point, the files were imported into ArcView®, mapped and analysed.

Home-range was calculated using two estimators as recommended by Harris *et al.* (1990) and Powell (2000). The first estimator was the minimum convex polygon (MCP). The MCP draws the smallest convex polygon possible containing all of an animal's known, or estimated, locations (Hayne 1949). The advantage of the MCP is that it is the oldest, simplest and most commonly used method, thereby enabling direct comparisons between studies. It also defines a limit to the area that the animal can visit, and therefore the habitat types that it can encounter (Kenward 1992). The main disadvantage of the MCP is that it only provides information on the boundary of the home-range and ignores the internal areas of greatest use, which are important biologically (Hayne 1949). The MCP is also strongly influenced by small sample sizes, and is sensitive to extreme data points, which can greatly inflate home-range size and include areas that may never actually be visited by the animal (Kenward 1987, Powell 2000).

The second estimator was the fixed-Kernel method (KE). The KE method uses probability density functions to identify areas of concentrated use. The least squares cross-validation smoothing parameter was also used, as recommended by Seaman *et al.* (1999) and Hooze and Eichenlaub (1997). A 95 % isopleth was used for calculating home-range and a 50 % isopleth for calculating animals' core area.

Home-ranges were estimated using the software program 'Animal Movement' (Hooze and Eichenlaub 1997) for animals with relatively complete data sets (six animals per species, Table 7.1). Home-range analyses may have contained statistically auto-correlated data although fixes were considered to be biologically independent, as a time period of 45

minutes was sufficient to allow any radio-collared animal to traverse the entirety of its home-range. Like Powell (1987) and Goodrich and Buskirk (1998), the problems of serial auto-correlation were considered unimportant because individual movements were likely to depend upon past experience and knowledge of resources within the home-range.

Table 7.1 Twelve hour dark (■) and light session (□) data sets, used in the analysis of animals' movements, over the 11 month study period. Asterisks indicate the six animals from each species used in home-range analysis.

Species	Sex	ID	Sampling Period							
			1	2	3	4	5	6	7	8
Wallaby	Male	71 *	■	■	■	■	■	■	■	■
		72 *	■	■	■	■	■	■	■	■
		73 *	■	■	■	■	■	■	■	■
		76 *	■	■	■	■	■	■	■	■
	Female	74 *	■	■	■	■	■	■	■	■
Pademelon	Male	31 *	■	■	■	■	■	■	■	■
		32	■	■	■	■	■	■	■	■
		35	■	■	■	■	■	■	■	■
		36	■	■	■	■	■	■	■	■
		37	■	■	■	■	■	■	■	■
		38 *	■	■	■	■	■	■	■	■
		40a	■	■	■	■	■	■	■	■
		40b	■	■	■	■	■	■	■	■
		43 *	■	■	■	■	■	■	■	■
		45	■	■	■	■	■	■	■	■
		46	■	■	■	■	■	■	■	■
		47	■	■	■	■	■	■	■	■
		48	■	■	■	■	■	■	■	■
	Female	39 *	■	■	■	■	■	■	■	■
Possums	Male	03 *	■	■	■	■	■	■	■	■
		05	■	■	■	■	■	■	■	■
		06 *	■	■	■	■	■	■	■	■
		09	■	■	■	■	■	■	■	■
		12	■	■	■	■	■	■	■	■
		15	■	■	■	■	■	■	■	■
		23	■	■	■	■	■	■	■	■
		24	■	■	■	■	■	■	■	■
	Female	01 *	■	■	■	■	■	■	■	■
		02 *	■	■	■	■	■	■	■	■
		04	■	■	■	■	■	■	■	■
		08 *	■	■	■	■	■	■	■	■
		13	■	■	■	■	■	■	■	■
		14	■	■	■	■	■	■	■	■

Inter- and intraspecific differences in home-range size were tested with a GLM (PROC GLM) using SAS (SAS 1990). Before modelling, data were tested for normal distribution by plotting residuals. MCP and 95 % KE estimates were normalised using square-root transformations. Tukey-Kramer tests were used to make multiple comparisons within and between species.

For each species, the percentage overlap of animals' home-ranges (95 % and 50 % KE) were calculated using ArcView®. Where home-ranges between two animals overlapped, the proportional area of overlap was based on the smaller home-range.

7.3 Results

7.3.1 Data set

Relatively complete data sets were collected for all six wallabies, but only six of the 17 pademelons and six of the 19 possums (Table 7.1). The incomplete data set for pademelons resulted from four animals dying during the study and one animal removing its collar. Evidence for the four animals dying included finding damaged devices, 5-9 months after attachment, where both the epoxy resin casing and leather collars had clearly been chewed by carnivores. Two of these collars were also found near piles of fur and shards of bone. Another collar was found on top of a log (> 1 m above ground). It is not known whether these pademelons died and were then consumed by scavenging carnivores, or whether the animals were killed by predators. Either way, the species responsible for damaging the transmitters were the Tasmanian devil (*Sarcophilus harrisii*) and/or the spotted-tailed quoll (*Dasyurus viverrinus*). Both species are known to kill pademelons (Jones and Barmuta 1998), and had been seen within the study site (*pers. obs.*). The collars themselves were considered highly unlikely to have caused increased mortality, as they represented a very small proportion of the animals' body mass and, due to their shape and location, would not have interfered with animals' normal activities. The fifth pademelon was thought to have removed its transmitter, as the collar was found intact but the leather appeared to have stretched over time.

Incomplete data sets for possums were due to battery problems. Devices had an expected lifespan of 16.9 months (Sirtrack, New Zealand), but signals became unstable between 2-13 months (average of 8 months), making data collection difficult. This problem was recognised early on, consequently a large effort was invested in trapping new animals between Sampling Periods 2 and 7 to increase sample size, and re-trap previously collared animals to attach new devices. This proved only moderately successful, however, as possums appeared trap-shy at this site, despite an extensive array of baits being utilised. In addition to predator and equipment problems, one small male possum and one large male pademelon dispersed during the study period. These animals were excluded from the analyses.

7.3.2 Home-range

The MCP, 95 % KE and 50 % KE home-ranges are shown for individuals in Table 7.2. MCP home-ranges generally gave the largest values, as expected. Locations of each animal's KE home-range are shown in Figure 7.2. The 95 % KE home-ranges are best described as amoeboid in shape, with 1-4 discrete segments. The 50 % KE core areas were generally located within the centre of the 95 % KE, with 1-2 discrete segments. Strong asymmetry was detected, however, for possum female 01.

Home-range data are summarised by species and sex in Table 7.3. Results from the GLM showed significant species, sex and species*sex effects for most variables (Table 7.4). Multiple comparison tests showed that wallabies had larger MCP home-ranges than pademelons ($P=0.0176$), and larger 95% KE and 50% KE home-ranges than both pademelons ($P=0.0002$ and 0.0004), and possums (and $P=0.0067$ and 0.0071). Intersexual differences in home-range were only detected for wallabies (MCP: $P=0.0210$, 95% KE: $P=0.0001$, and 50% KE: $P=0.0013$). Male pademelons tended to have larger home-ranges than females, but this was not significant at the 0.05 level (95 % KE: $P=0.0821$).

Within each species, there was a high degree of overlap between the 95 % KE home-ranges of individuals, particularly for male wallabies and for male possums (both > 60%) (Table 7.5). Core areas (50 % KE) had much less overlap (Table 7.5).

Table 7.2 Estimated home-ranges (ha) for individual wallabies, pademelons and possums calculated using minimum convex polygon (MCP) and fixed-kernel (KE) methods. n=number of locations used in analysis

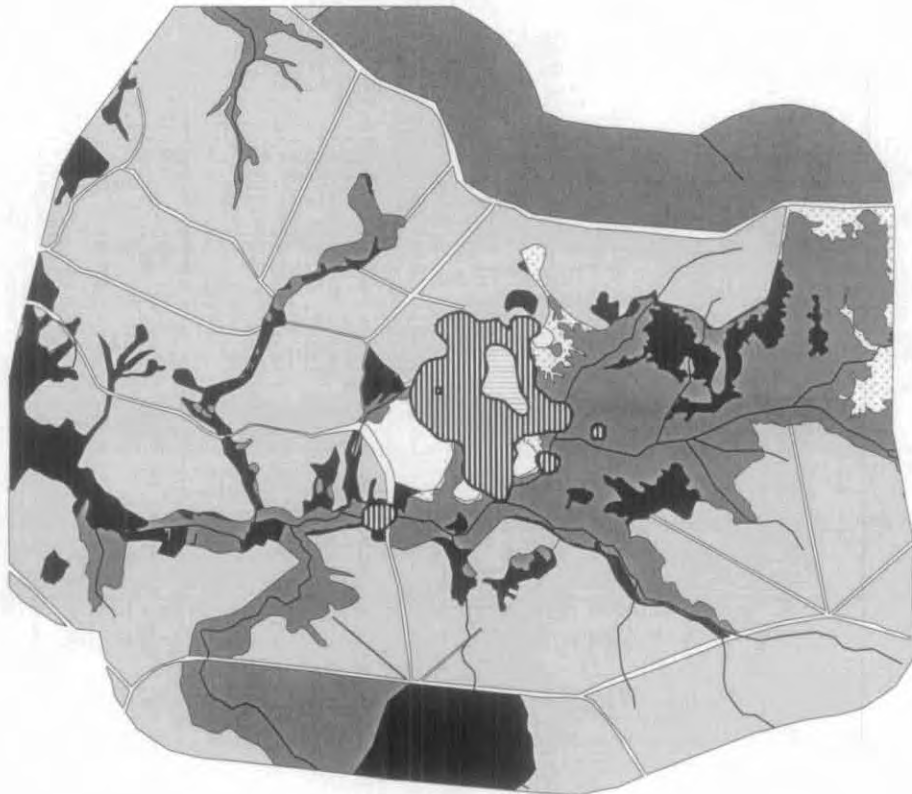
Species	Sex	ID	n	MCP	95% KE	50% KE
Wallaby	M	71	223	54.76	52.86	10.65
	M	72	236	64.73	56.33	13.13
	M	73	205	105.14	59.34	5.74
	M	76	244	78.97	74.90	14.63
	F	74	240	47.78	24.76	2.73
	F	75	256	18.35	19.19	3.77
Pademelon	M	31	240	17.00	17.81	2.25
	M	38	220	31.85	17.81	2.25
	M	43	144	40.28	29.05	4.14
	F	39	240	17.96	12.70	2.43
	F	41	158	20.30	12.19	2.05
	F	42	157	6.10	5.61	0.66
Possum	M	3	191	20.21	16.22	0.85
	M	6	158	26.03	19.82	3.63
	F	1	166	64.22	28.57	2.16
	F	2	139	40.25	12.63	2.62
	F	8	197	63.03	15.13	2.67
	F	17	128	20.97	8.44	0.95

Table 7.3 Average male and female body mass (\pm s.d.) and home-range size (\pm s.e.) estimated for wallaby, pademelon and possum, using minimum convex polygon (MCP) and fixed-kernel (KE) methods. Letters indicate significant differences between species ($\alpha=0.05$ after Tukey-Kramer adjustment).

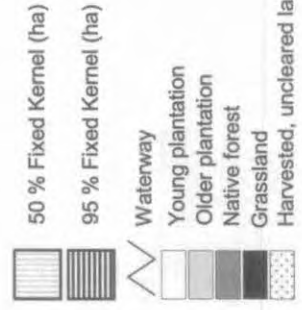
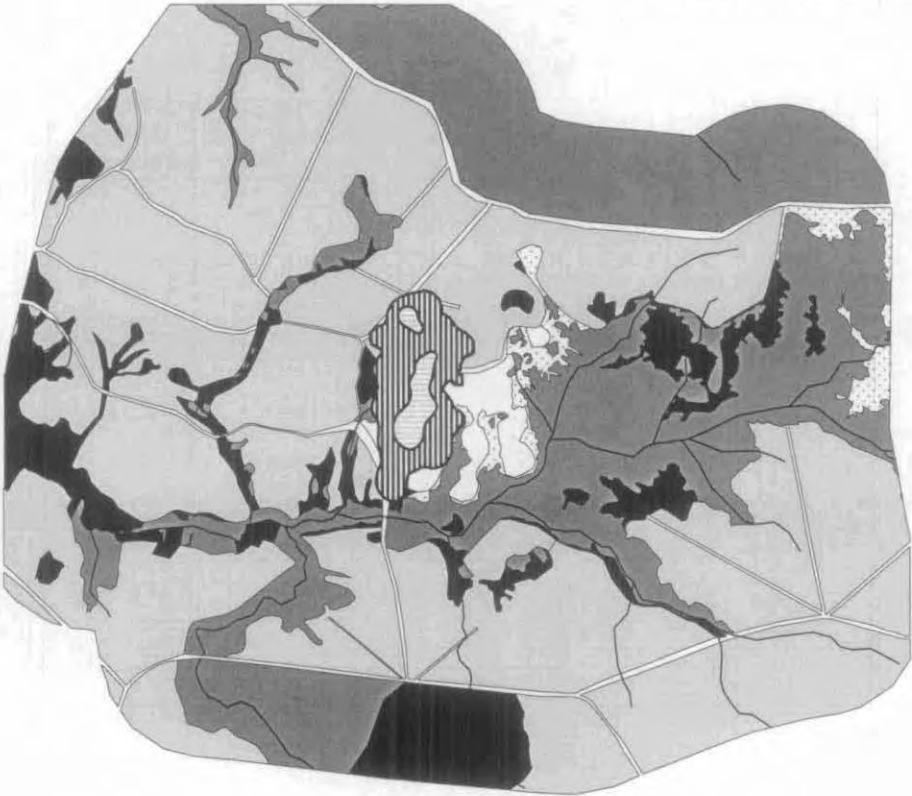
Species	Group	n	body mass	100 %	95 %	50 %
			at capture			(core)
			(kg)	MCP (ha)	KE (ha)	KE (ha)
Wallaby	Spp average	6	20.0 \pm 7.6	61.6 \pm 12.0 ^a	41.4 \pm 3.2 ^a	7.1 \pm 1.0 ^a
	Males	4	24.7 \pm 2.7	71.9 \pm 11.3	60.9 \pm 3.7	11.0 \pm 1.1
	Females	2	10.6 \pm 0.1	33.1 \pm 14.7	22.0 \pm 5.2	3.3 \pm 1.5
Pademelon	Spp average	6	5.5 \pm 1.4	22.3 \pm 4.9 ^b	15.9 \pm 3.0 ^b	2.3 \pm 0.9 ^b
	Males	3	5.8 \pm 2.2	29.7 \pm 6.8	21.6 \pm 4.2	2.9 \pm 1.2
	Females	3	5.2 \pm 0.3	14.8 \pm 4.4	10.2 \pm 4.2	1.7 \pm 1.2
Possum	Spp average	6	4.0 \pm 0.5	39.1 \pm 8.3 ^{ab}	17.1 \pm 3.2 ^b	2.2 \pm 0.9 ^b
	Males	2	4.5 \pm 0.3	23.1 \pm 2.9	18.0 \pm 5.2	2.2 \pm 1.5
	Females	4	3.8 \pm 0.3	47.1 \pm 10.3	16.2 \pm 3.7	2.1 \pm 1.1

Figure 7.2. Locations of the 95 % and 50 % fixed kernel home-ranges for six wallabies, six pademelons and six possums, within the 770 ha study area.

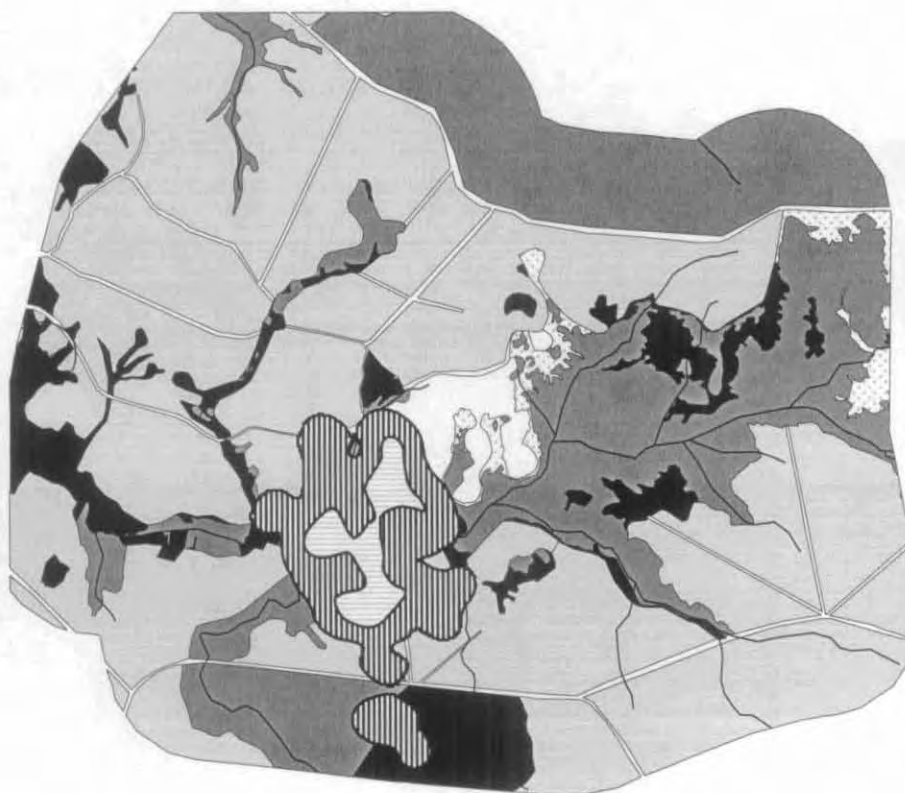
Wallaby: Female 74



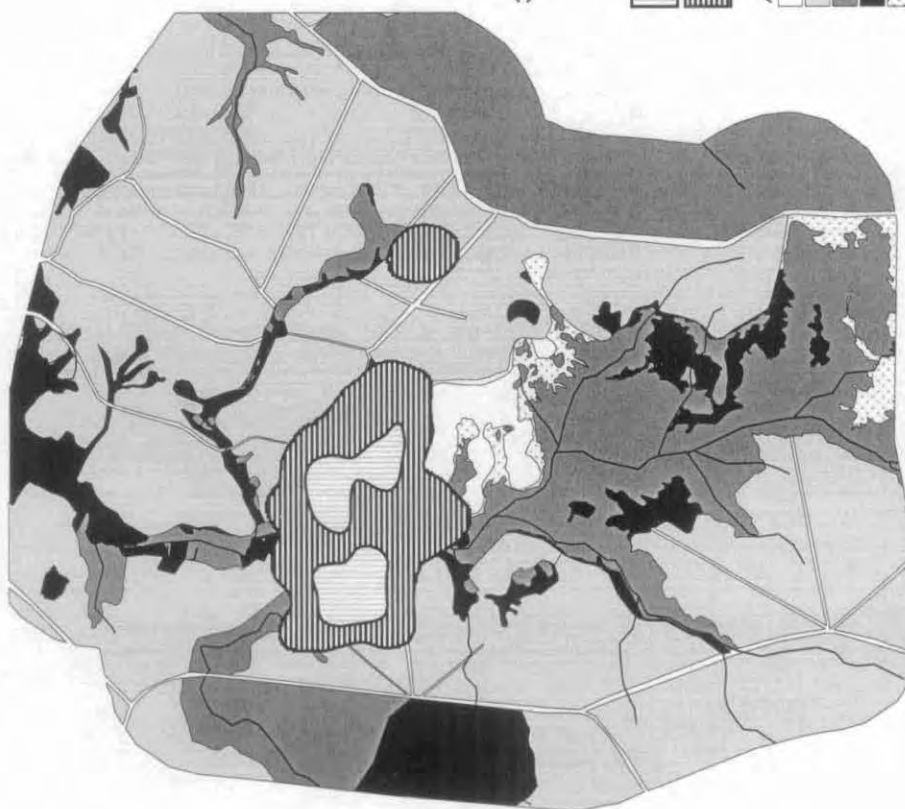
Wallaby: Female 75



Wallaby: Male 71



Wallaby: Male 72



300 0 300 600 Metres

50 % Fixed Kernel (ha)

95 % Fixed Kernel (ha)

Waterway

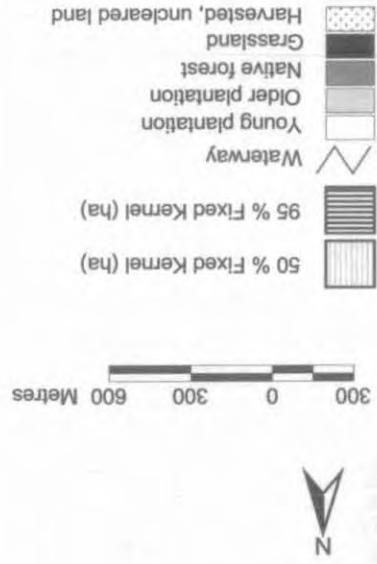
Young plantation

Older plantation

Native forest

Grassland

Harvested, uncleared land



Wallaby: Male 76



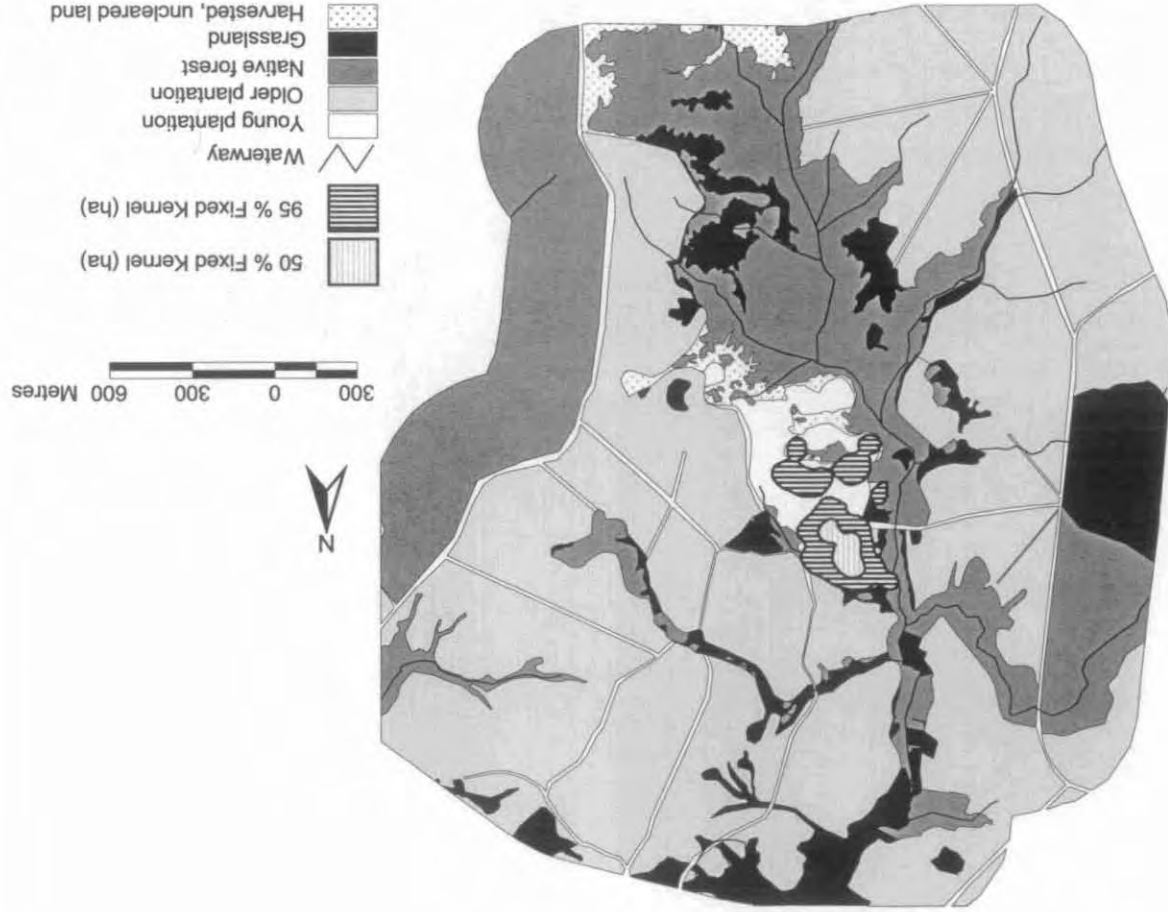
Wallaby: Male 73



Pademelon: Female 39



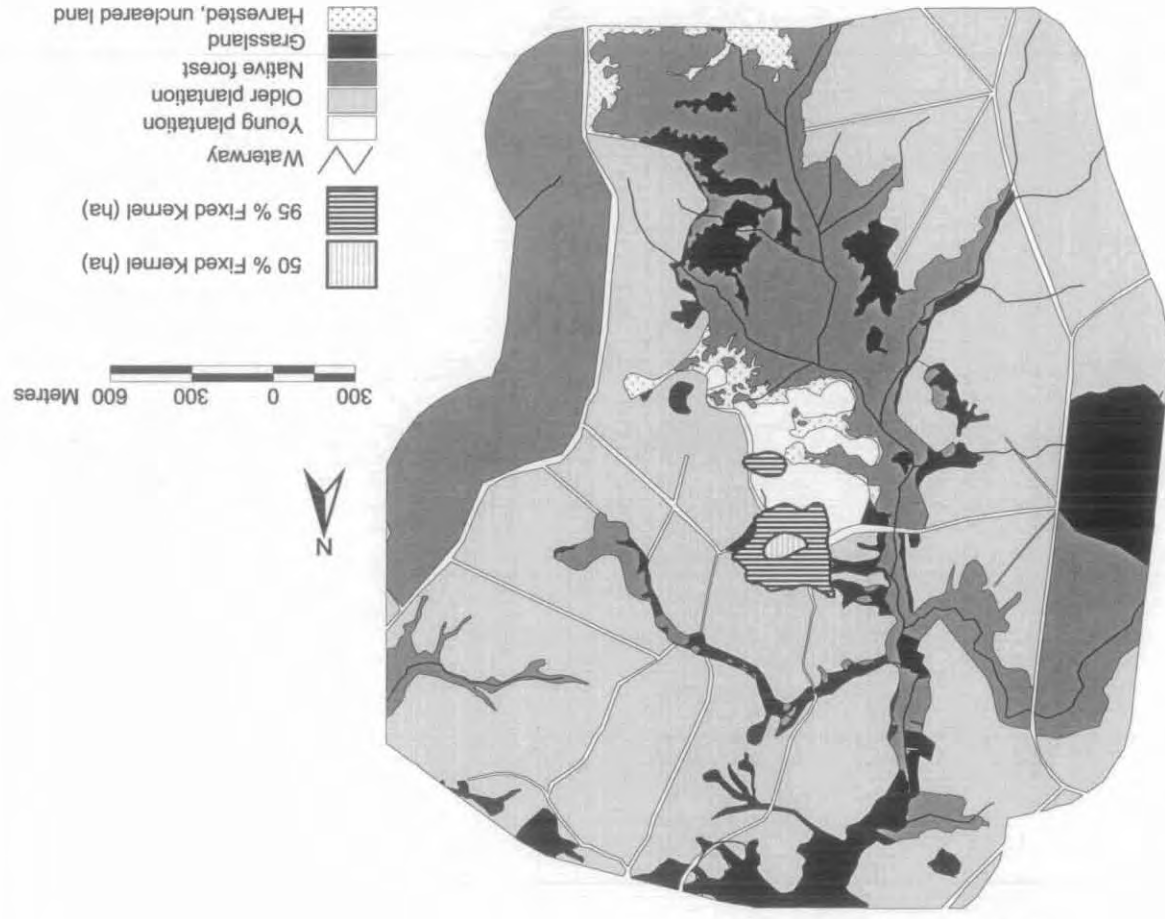
Pademelon: Female 41



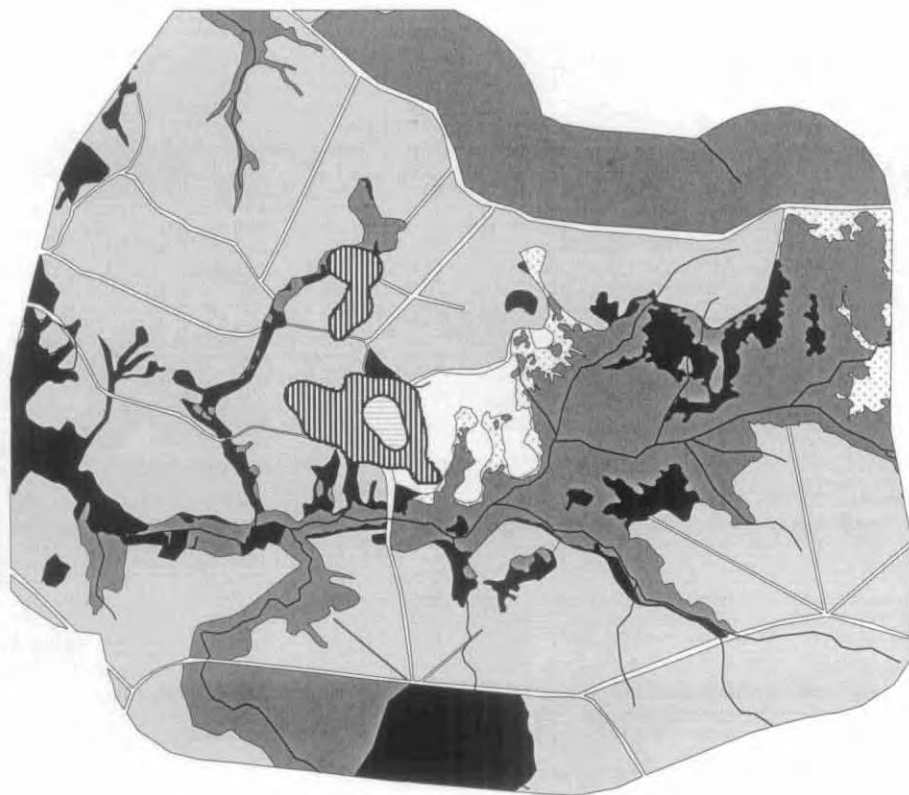
Pademelon: Female 42



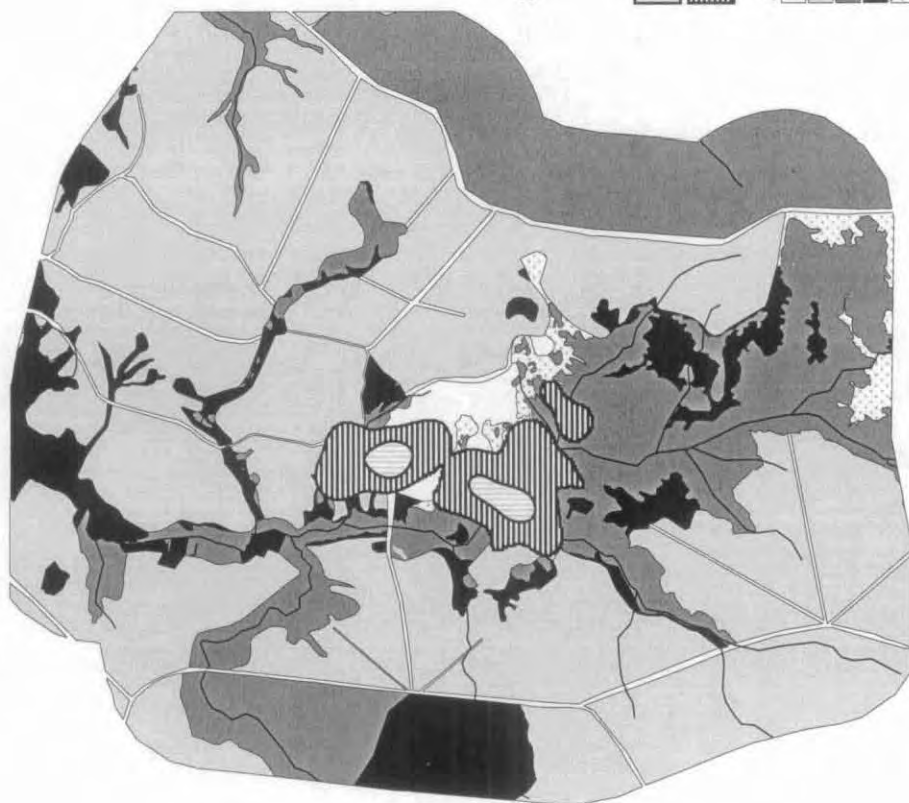
Pademelon: Male 31



Pademelon: Male 38



Pademelon: Male 43



300 0 300 600 Metres

50 % Fixed Kernel (ha)

95 % Fixed Kernel (ha)

Waterway

Young plantation

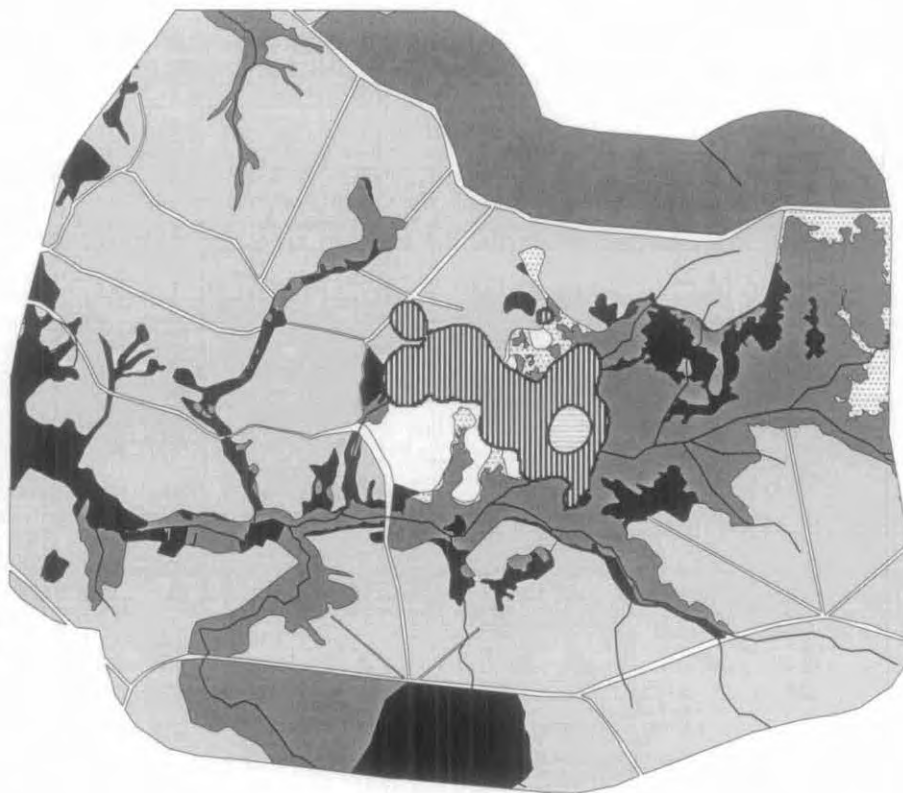
Older plantation

Native forest

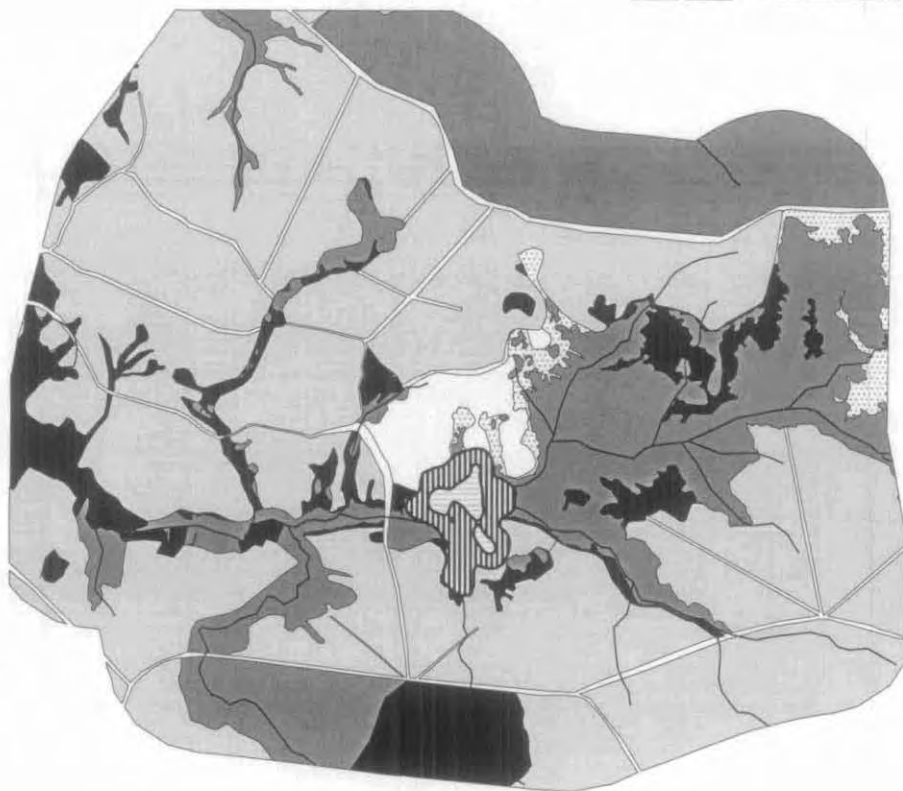
Grassland

Harvested, uncleared land

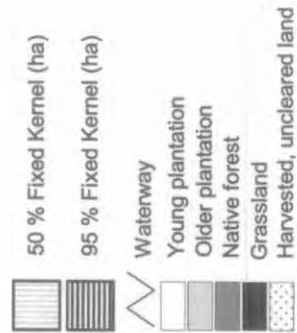
Possum: Female 01



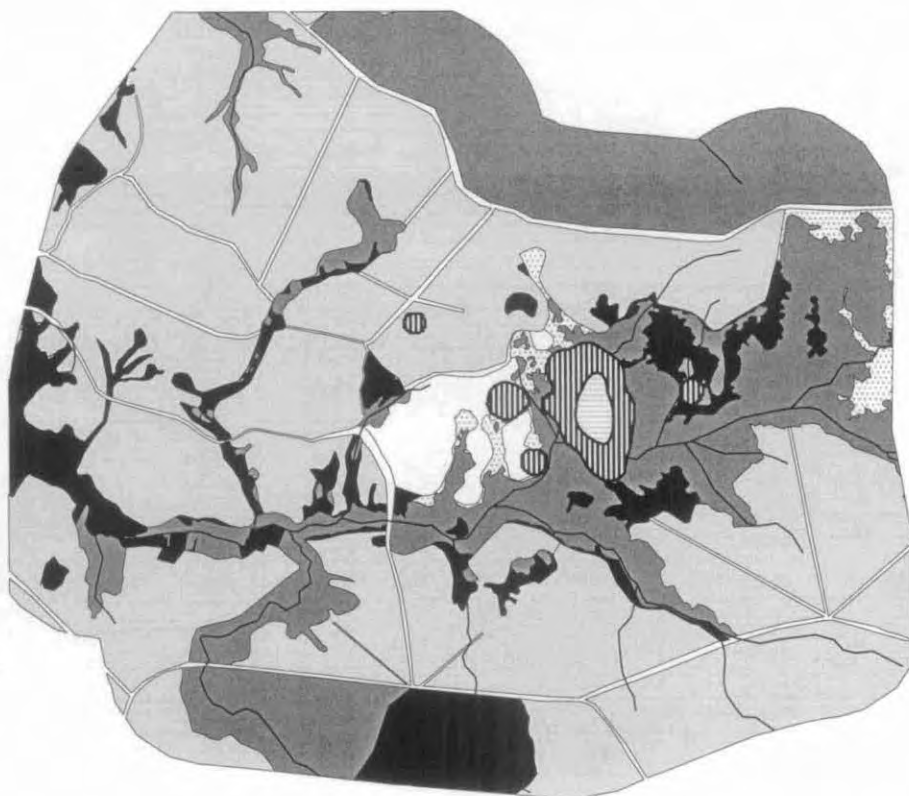
Possum: Female 02



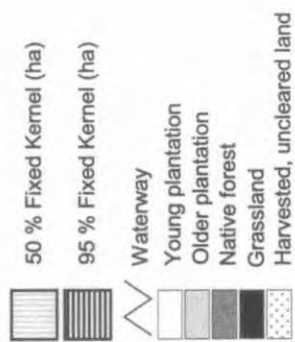
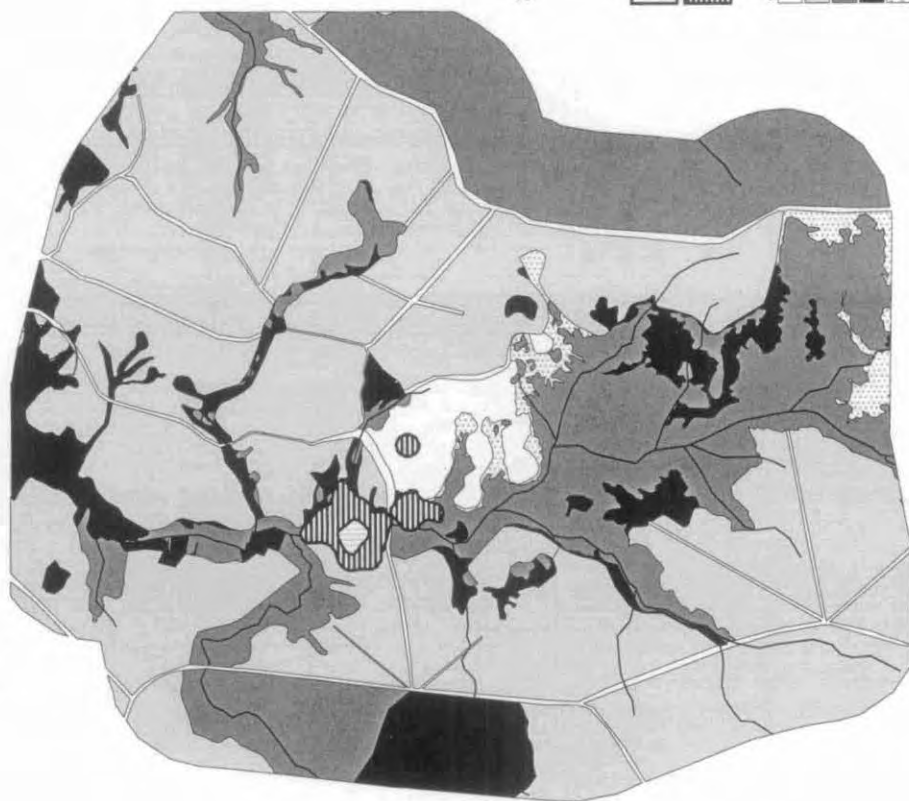
300 0 300 600 Metres



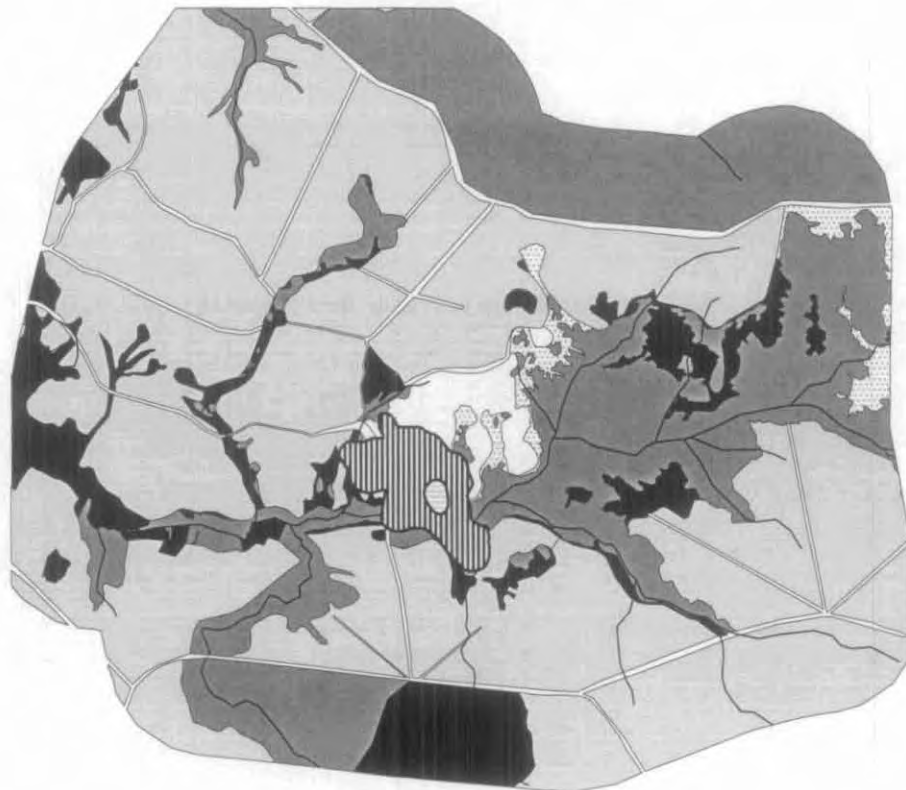
Possum: Female 08



Possum: Female 17



Possum: Male 03



Possum: Male 06

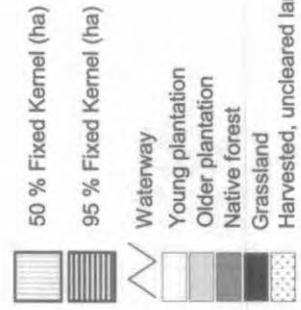
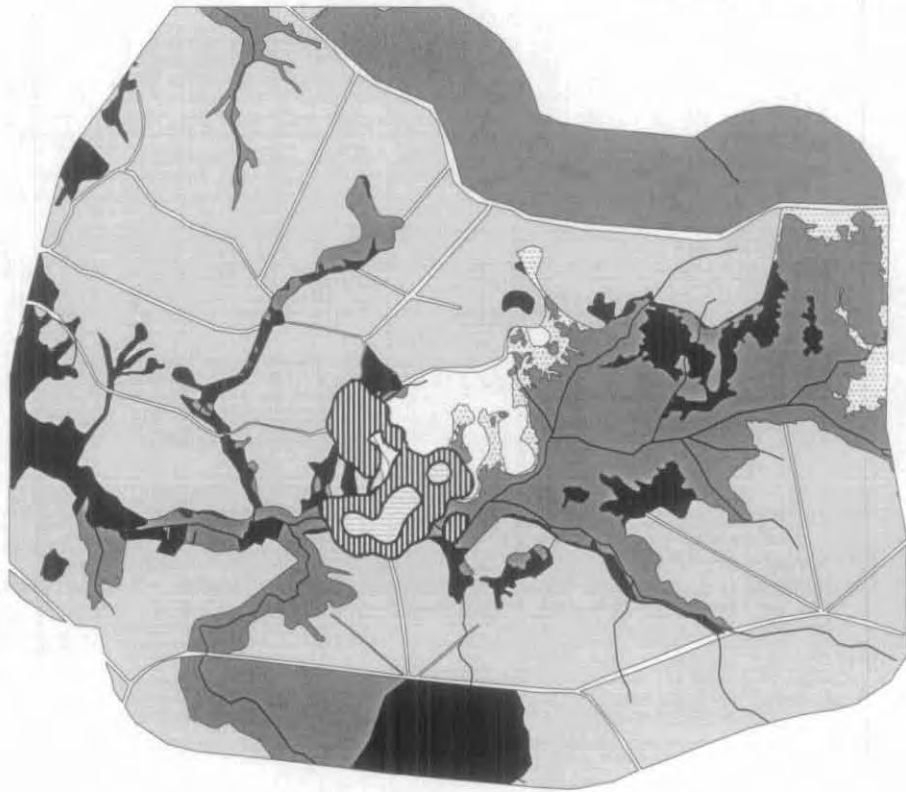


Table 7.4 Results from the GLM showing species, sex and species*sex interactions for wallabies, pademelons and possums. Asterisk indicates significance ($\alpha=0.05$).

Home-Range	Factor	d.f.	Type III SS	F	P
MCP	species	2	18.617	5.29	0.0225*
	sex	1	3.519	2.00	0.1826
	species*sex	2	17.557	4.99	0.0265*
95 % KE	species	2	2265.404	20.96	0.0001*
	sex	1	1252.805	23.18	0.0004*
	species*sex	2	992.538	9.18	0.0038*
50 % KE	species	2	87.360	9.37	0.0035*
	sex	1	38.171	8.19	0.0143*
	species*sex	2	46.500	4.99	0.0265*

Table 7.5 The proportional area (mean % overlap \pm s.e.) of overlapping 95 % KE home-range and 50 % KE core areas, for the three species. Sample size (n) refers to number of comparisons made.

Estimate	Group	Wallabies	Pademelons	Possums
95 % KE	Males	62.3 \pm 3.4 (n=6)	32.9 \pm 6.5 (n=3)	65.2 (n=1)
	Females	32.0 (n=1)	26.9 \pm 9.6 (n=3)	12.5 \pm 13.0 (n=6)
	b/w sexes	34.0 \pm 9.5 (n=8)	37.6 \pm 11.2 (n=9)	23.6 \pm 10.0 (n=8)
50 % KE	Males	4.6 \pm 1.2 (n=6)	2.3 \pm 1.2 (n=3)	0.3 (n=1)
	Females	0 (n=1)	0.0 \pm 0.0 (n=3)	0.5 \pm 0.5 (n=6)
	b/w sexes	0.2 \pm 0.1 (n=8)	2.1 \pm 1.2 (n=9)	1.4 \pm 1.0 (n=8)

7.4 Discussion

Data sets used to calculate home-range contained between 128 and 256 fixes, depending upon the animal involved. The use of unequal data sets is not ideal, however, examination of data described in Table 7.2 shows that within species, home-range size was not positively correlated with sample size.

7.4.1 Macropods

The home-range was larger for wallabies than pademelons. This result was presumably related to their difference in body mass, and hence, energetic requirements. A positive relationship between body mass and home-range size has been documented for mammals (McNab 1963), herbivores (Harestad and Bunnell 1979), and possibly macropods (Norbury *et al.* 1989).

Female wallaby MCP home-range was considerably smaller within the present study (33 ha), than that reported by Mooney and Johnson (1979) (130 ha). This difference could reflect methods used, and/or differences in habitat quality between the two study sites. Mooney and Johnson's (1979) study may have overestimated home-range size, as biangulated bearings were not collected simultaneously. Consequently, animals may have moved between fixes, resulting in inaccurate locations. The time delay and overall accuracy of Mooney and Johnson's (1979) telemetry system was not reported, making it difficult to assess the accuracy of their results (see Chapter 6).

If Mooney and Johnson's (1979) methods did not overestimate home-range size, smaller values within the present study may reflect differences in habitat quality, and/or habitat spatial organisation. Mooney and Johnson (1979) reported extremely large and elongate (> 1 km long) home-ranges, in response to animals travelling from diurnal forest refuge to nocturnal pastoral feeding grounds. In contrast, home-ranges estimated in the present study were considerably smaller and rounder in shape. The patchy mosaic landscape at Surrey Hills may therefore be an to advantage macropods, as habitats appear to be relatively productive, and food and shelter are available over a small spatial scale.

Due to the scarcity of data on the spatial organisation of wallabies within Tasmania, comparisons with the Australian mainland subspecies are useful. Johnson's (1987) study on *M. r. banksianus* at a forest/pasture site in New South Wales, reported smaller home-ranges than those estimated in the present study. This could be attributed to either of the following factors. Firstly, Johnson (1987) probably underestimated home-range size because he used only diurnal locations. Secondly, Johnson's study site contained areas of improved pasture which were highly productive (Jarman *et al.* 1987), and high food abundance is likely to be associated with small home-range.

Male pademelon MCP home-range was remarkably smaller within the present study (30 ha), than estimates reported by Johnson (1977) (156 ha) for the forest/pasture site described above. This difference could reflect an overestimation of home-range by Johnson (1977), due to the methods he used for estimating animals' locations. Firstly, Johnson (1977) biangulated bearings that were not collected simultaneously (see above). Secondly, he collected positional data by radio-tracking animals on foot. This method can affect animals' movements, if they flush in response to the observer's presence (see Chapter 3). Flushing increases animal's daily movements, thereby resulting in an overestimation of home-range size, and possibly changing home-range shape. If Johnson's (1977) study did not over-estimate home-range due to the methods used, small pademelon home-range within the present study may reflect habitat quality and habitat spatial organisation at this site, as discussed above for wallaby.

For the Tasmanian pademelon, a comparable mainland telemetry study comes from Johnson's (1980b) work on the red-necked pademelon *T. thetis*, at a forest/pasture site in New South Wales. Johnson's (1980b) home-range estimates were similar to those found in the present study. The accuracy of these home-range estimates are difficult to interpret, however, due to several shortcomings of the study. Firstly, animal's locations were estimated from biangulated bearings that were collected up to 15 minutes apart (Johnson 1980b). Consequently, it was unlikely that animals had not moved between fixes. Additionally, sample size was small (usually 2 animals [1 of each sex] were followed over a given time period), and data were collected on different animals in different seasons and years (Johnson 1980b).

7.4.2 Possums

Possum MCP home-ranges within the present study (male: 23.1 ha, female: 47.1 ha) were considerably larger than those reported by Hocking (1981) in 80-year-old Tasmanian rainforest (male: 7.0 ha and female: 4.2 ha). This difference may reflect poorer quality habitat at Surrey Hills, for example, limited den sites and/or food, as also suggested in Chapter 4 based on spotlighting data. Possum home-range within the present study was similar to estimates for an introduced population in New Zealand (male: 24.6 ha, range: 2.5-65.0; female: 18.3, range: 4.2-45.8) (Green and Coleman unpublished data, cited in Green 1984). It is unclear, however, whether or not the New Zealand population was recovering from a lethal control operation, which may have affected home-range size.

It should be noted that Hocking (1981) used trapping methods to estimate home-range size, and trapping has been shown to underestimate values determined by radio-telemetry methods

by 50 %, over a one year period (Ward 1978). However, even if Hocking's (1981) values are doubled, his home-range estimates are still substantially smaller than those reported within the present study. This suggests that methods alone were not responsible for the differences in home-range reported for these two sites.

7.4.3 Intersexual differences in home-range

Larger home-range for male wallabies and male pademelons compared with females is consistent with most other studies of macropods (Croft 1989). The same patterns was found, for example, in red-necked pademelons *T. thetis* (Johnson 1980b), bridled nailtail wallabies *Onychogalae fraenata* (Evans 1996), whiptail wallabies *M. parryi* (Kaufman 1974), common wallaroos *M. robustus erubescens* (Clancy and Croft 1989), eastern grey kangaroos *M. giganteus* (Jaremovic and Croft 1987) and western grey kangaroos *M. fuliginosus* (Arnold *et al.* 1992). Intersexual differences in home-range size are thought to reflect species' sexual dimorphism and polygynous mating system. Males are argued to have larger home-ranges so they can overlap with as many females as possible, in an attempt to increase their number of sexual encounters (Croft 1989). This concept is supported by the present study, because male's home-ranges overlapped with nearly all radio-collared females.

Larger male than female home-range for wallaby was inconsistent with findings from Mooney and Johnson (1979). However, Mooney and Johnson (1979) collected data for only one male, and he was probably a sub-adult. He weighed only 14 kg, approximately half the mass of a dominant adult male. Consequently, Mooney and Johnson's (1979) results are of minimal value.

Male and female possum home-range were of a similar size within the present study. This is inconsistent with previous studies, for example, Dunnet (1964), Ward (1978), Hocking (1981), and Hynes (1999). Results for the population at Surrey Hills could be an artefact of low male sample size ($n=2$). Additionally, female home-range at this site may be exceptionally large. Comparisons between male and female MCP and 95 % KE home-ranges suggest that although they have similar areas of 'high use', females occasionally forayed beyond these areas, and these movements greatly inflated their MCP estimate (Table 7.2). The reason for females' forays is unclear, but may reflect exploration for food.

Population densities and food abundance are fundamental to home-range size, shape and patterns of utilisation. This relationship, and temporal change within these variables, will be discussed in Chapter 12.

Chapter 8

Radio-Telemetry: Habitat Use and Selection

8.1 Introduction

Within fragmented environments, patches may vary in relation to foraging opportunities and risks of predation. A foraging herbivore should therefore use habitats non-randomly, or selectively, to maximise survival. Animals' patterns of habitat selection may be based upon foraging and/or predator-avoidance decisions. Results can also vary with the scale of the study (Wiens 1989). For example, woodland caribou *Rangifer tarandus* select habitats based upon predator avoidance at the larger scale (seasonal range), but food abundance at the smaller scale (daily area) (Rettie and Messier 2000). Female yellow-headed blackbirds select habitats base upon food abundance at the larger scale (breeding range) and predator avoidance at the smaller scale (nest location) (Orians and Wittenberger 1991). This chapter examines habitat utilisation and selection patterns of wallabies, pademelons and possums, at a range of temporal and spatial scales. At the larger scale, I examined choice of home-range (Johnson's [1980a] second-order selection), and at the smaller scale, choice of nocturnal and diurnal habitats within the home-range (Johnson's [1980a] third-order selection). Nocturnal activities were associated with foraging, while diurnal activities were associated largely with resting.

8.2 Materials and Methods

The collection of radio-telemetry data is described in Chapter 7.

8.2.1 Habitat Use

Species' patterns of habitat use within 12-hour dark and light sessions were graphed across time (45 minute intervals). Data were pooled across sexes within species. Habitat use was quantified as the proportion of each habitat type present within buffered areas, around animals' individual locations. Buffers of 3.45 ha, constructed as circles with a 105 m radius centred over each location, were used. This represented the error area associated with the telemetry system at this site (Chapter 6). The use of buffered points has been recommended by Rettie and McLoughlin (1999) for analysing patterns of habitat use within patchy environments where habitat boundaries are abundant. The use of buffered points better reflects the accuracy and precision of telemetry data than use of unbuffered points, although it reduces apparent precision, because estimated locations are often assigned a number of habitat types. Ignoring buffers, however, falsely represents (underestimates) the error of any system.

Data, collected from an individual over a given 12-hour session, were only used if there were at least 12 locations from the attempted 16 fixes. When analysing the dark data, an individual's data were only used if there were more than two 12-hour sessions available. This restriction was not applied to the light data, as data were less abundant for this time period.

8.2.2 Habitat selection

Habitat selection was examined at two spatial/temporal scales. In the first analysis, the composition of animals' home-ranges were compared with the availability of habitats within the entire study area. The second analysis examined whether animals used different habitats within their home-range during night and day. The first analysis therefore represented selection over the complete 11 months of the study, while the second analysis investigated habitat selection in each sampling period (≈ 6 weekly intervals).

(1) Location of home-ranges within the study area

This comparison determined whether home-ranges occurred randomly within the study area, or whether animals actively selected (or avoided) some habitats. Analysis was based on Manly *et al.*'s (1993) methods (design II, sampling protocol A). The 'available' area was the 770 ha study site. This study area was based on the 428 ha area used for the spotlighting surveys (Chapter 3), buffered by 370 m (where vegetation data was available), to encompass all telemetry locations. The relative proportions of the five habitat types within the 770 ha area were consistent with those within the 428 ha spotlighting study area. Hence, the use of the 770 ha area as the 'available' area enabled results from this radio-tracking study to be directly compared with those from the spotlighting study (Chapter 4). The areas 'used' by animals were their MCP home-ranges, calculated using dark and light data collected over the 11 month study period (Chapter 7). Home-ranges were buffered by 105 m to incorporate radio-tracking error, and habitat composition was determined using ArcView® GIS.

The advantages and disadvantages of using MCP and 95 % KE home-ranges have been discussed in Section 7.2.5. MCP home-ranges were used in preference to 95 % KE home-ranges within this analysis, for consistency with the subsequent analysis which examined habitat selection during night versus day periods. That is, it was important to use the entire data set, rather than the KE data set, which has removed the outermost 5 % of locations during this subsequent analysis, as these locations may represent data of biological importance, for example, the location of possum den sites.

For each animal, a resource selection index (\hat{w}_i) was calculated for the five habitat types using Manly *et al.* (1993). As a habitat type must be greater than zero for selection ratios to be calculated (otherwise there is a problem of division by zero), zero values were assigned a value of 0.01 % during analyses. Habitat preference and avoidance were calculated using the selection ratio determined for the species and Bonferroni adjusted 95 % confidence intervals. Where values for a habitat were greater than 1, the habitat was selected for; and where values were less than 1, the habitat was selected against.

(2) Night and day areas within an animal's home-range

These comparisons determined whether activities linked to the light cycle required animals to occupy different habitats: night was assumed to relate to foraging activities, while animals were thought to rest during the day. Analysis was based on Manly *et al.* (1993) [Design III, sampling protocol A]. During this analysis, the 'available' area for an animal was its home-range (calculated above), and 'used' areas were the animal's buffered locations, collected during a single session. As data needed to reflect the light cycle, 'dark' and 'light' data sets were sub-sampled to reflect 'night' and 'day'. Hence, night data comprised telemetry locations collected during dark sessions between sunset and sunrise or the end of the session (whichever occurred first), while day data comprised telemetry locations collected during light sessions following sunrise.

Selection ratios for each species were calculated using \hat{w}_i (Manly *et al.* 1993). Habitat preference and avoidance were calculated using the selection ratio determined for the species and Bonferroni adjusted 95 % confidence intervals. Again, where values for a habitat were greater than 1, the habitat was selected for; and where values were less than 1, the habitat was selected against.

8.2.3 Before and after planting

Animals' patterns of habitat use and selection were compared immediately before and after planting *E. nitens* seedlings on the young plantation. This would indicate if animals changed their foraging behaviour in response to the newly available food source.

8.3 Results

8.3.1 Habitat use

Data sets use in this investigation are shown in Table 7.1. Graphs of species' patterns of habitat use throughout night and day, over time, are shown in Figure 8.1. Species' proportional use of open habitats (young plantation and grassland), during night and day are summarised in Table 8.1. Wallabies and pademelons used open habitats (young plantation and grassland) more at night, than during the day. Possums spent most of their time in native forest, irrespective of the light cycle. Possums also appeared to use grassland more during the day than night.

Table 8.1 Percentage probability of use (mean \pm s.d.) of open habitats by wallabies, pademelons and possums, during night and day over the entire study period. n=8 sampling periods

Species	Night	Day
Wallaby	48 \pm 14	10 \pm 3
Pademelon	55 \pm 8	29 \pm 7
Possum	16 \pm 4	13 \pm 7

8.3.2 Habitat selection

(1) Home-range within study area

Within the 770 ha study area, the home-ranges of all three species showed strong patterns of habitat selection. Patterns were also similar between species (Table 8.2). Young plantation was highly selected for by all three species, followed by harvested uncleared land. Older plantation was selected against by all three species, and native forest selected against by wallaby.

Table 8.2 Habitat selection ratios (\hat{w}_i) \pm s.e. from the comparison between the proportions of habitat available within the study area, to those within animals' home-ranges. Asterisk signifies selected for, † selected against.

Habitat	Wallaby	Pademelon	Possum
Young plantation	$7.25 \pm 0.67^*$	$11.94 \pm 0.43^*$	$8.09 \pm 0.40^*$
Harvested, uncleared land	$1.70 \pm 0.24^*$	$3.12 \pm 0.39^*$	$2.10 \pm 0.19^*$
Grassland	0.96 ± 0.08	0.99 ± 0.07	$1.20 \pm 0.06^*$
Older plantation	$0.85 \pm 0.04^\dagger$	$0.55 \pm 0.06^\dagger$	$0.61 \pm 0.02^\dagger$
Native forest	$0.76 \pm 0.06^\dagger$	0.87 ± 0.09	1.04 ± 0.05

(2) Nocturnal and diurnal areas within home-range

Light-cycle or activity strongly affected use of the home-range by macropods (Table 8.3). During the night, wallabies and pademelons selected for young plantation. Both species selected against older plantation, but pademelons also selected against native forest and harvested uncleared land. During the day, both species reversed this pattern by selecting against young plantation and selecting for previously avoided habitats; wallabies selected for older plantation, while pademelons selected for native forest. In contrast, possums' patterns of habitat use were consistent over the light-cycle. Native forest was selected for both night and day, while all other habitats were selected against (Table 8.3).

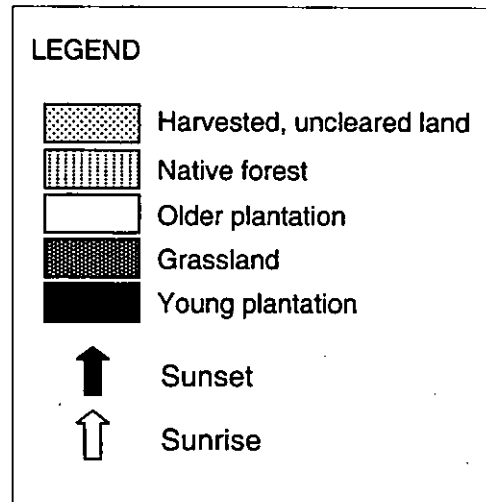
Table 8.3 Habitat selection ratios (\hat{w}_i) \pm s.e. from the comparison between the proportions of habitat available within animals' home-ranges, to those used by animals during night and day.
* signifies selected for, † signifies selected against

Time	Habitat	Wallaby	Pademelon	Possum
Night	Young plantation	$2.31 \pm 0.07^*$	$1.79 \pm 0.14^*$	$0.73 \pm 0.05^\dagger$
	Harvested, uncleared land	1.24 ± 0.14	$0.68 \pm 0.09^\dagger$	$0.38 \pm 0.02^\dagger$
	Grassland	1.02 ± 0.10	0.76 ± 0.12	$0.17 \pm 0.02^\dagger$
	Older plantation	$0.59 \pm 0.04^\dagger$	$0.62 \pm 0.05^\dagger$	$0.58 \pm 0.06^\dagger$
	Native forest	0.83 ± 0.09	$0.79 \pm 0.07^\dagger$	$2.04 \pm 0.10^*$
Day	Young plantation	$0.57 \pm 0.11^\dagger$	$0.73 \pm 0.07^\dagger$	$0.51 \pm 0.07^\dagger$
	Harvested, uncleared land	0.73 ± 0.17	$0.20 \pm 0.01^\dagger$	$0.63 \pm 0.07^\dagger$
	Grassland	0.81 ± 0.08	$0.31 \pm 0.04^\dagger$	$0.69 \pm 0.08^\dagger$
	Older plantation	$1.25 \pm 0.06^*$	0.95 ± 0.06	$0.55 \pm 0.08^\dagger$
	Native forest	0.94 ± 0.12	$1.89 \pm 0.14^*$	$1.97 \pm 0.10^*$

Patterns of habitat selection by males and females are shown in Table 8.4. Patterns for female macropods were consistent with those for the species overall. Patterns for male macropods were less clearcut. Although male wallabies selected for young plantation during the night, they showed no habitat selection during the day. Male pademelons appeared even less selective. They did not select for any habitat types during the night or day, although habitats were ranked in a similar order to those calculated for the species overall. Male and female possums showed similar patterns of habitat selection: native forest was selected for both night and day, while other habitats were generally selected against.

Species' patterns of habitat selection during each sampling period are shown in Figures 8.2-8.4. Results showed that the three species' patterns of habitat selection were relatively consistent over time. Wallabies, however, selected for harvested uncleared land at night, in Sampling Periods 7 and 8 (Figure 8.2). Results also showed that wallabies' selection for the

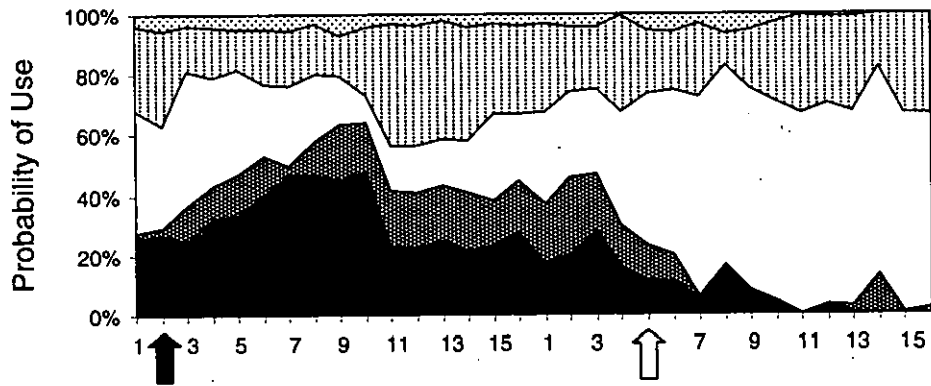
Figure 8.1 Patterns of habitat use by (a) Bennett's wallabies, (b) pademelons and (c) possums for each fix during dark and light sessions, within each sampling period.



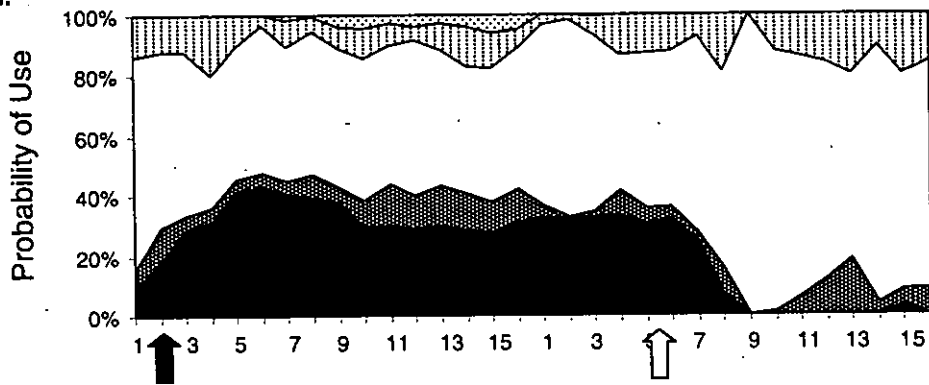
(a) Wallaby

Sampling period

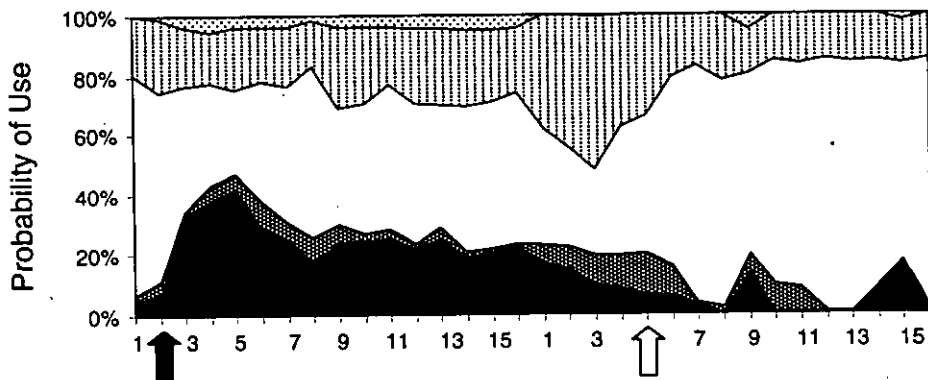
1. May '97



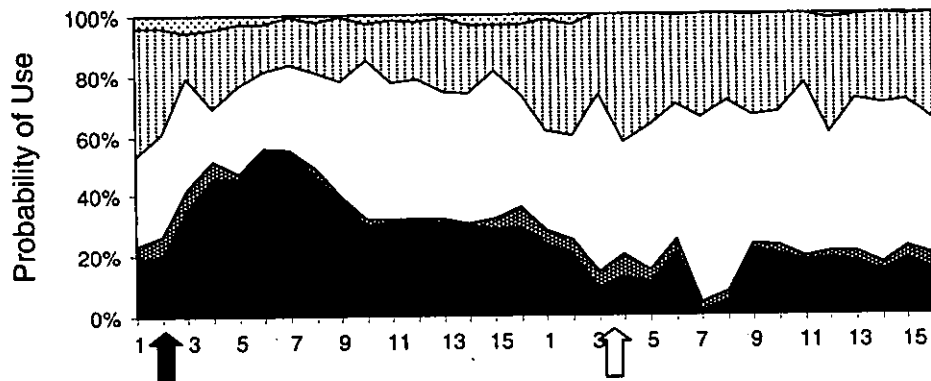
2. Jun./Jul.



3. Aug.



4. Sep.

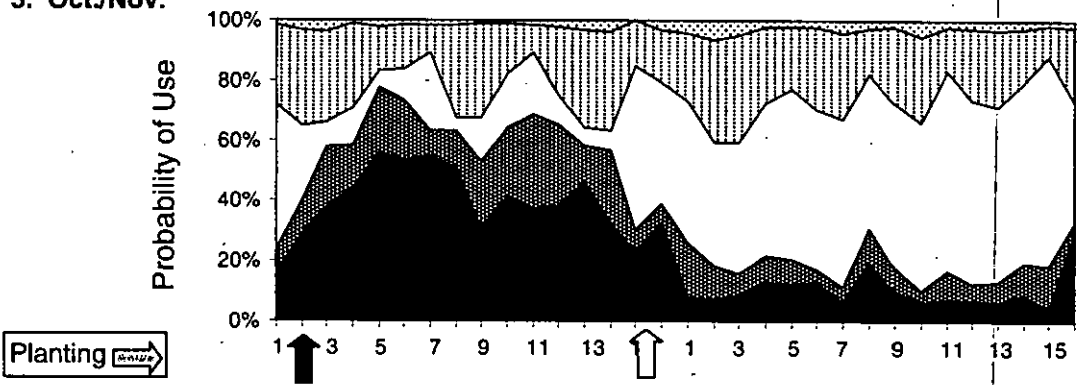


Fix within sampling period

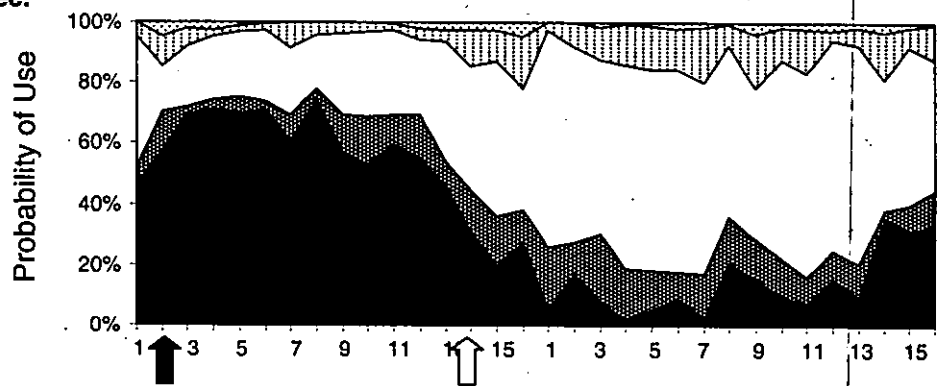
(a) Wallaby cont.

Sampling period

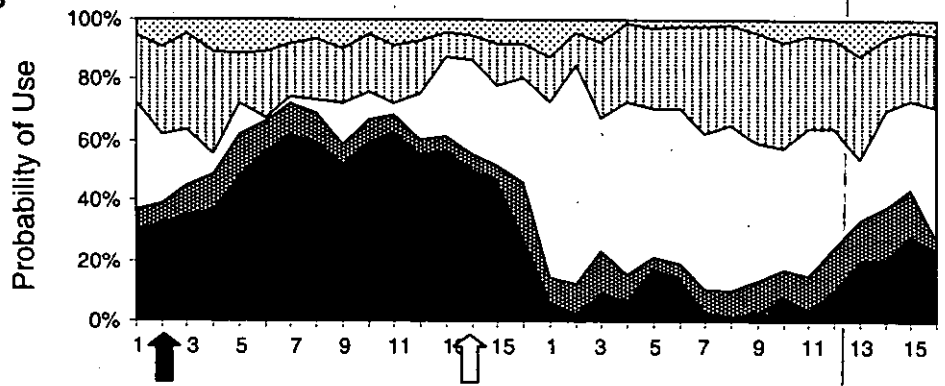
5. Oct./Nov.



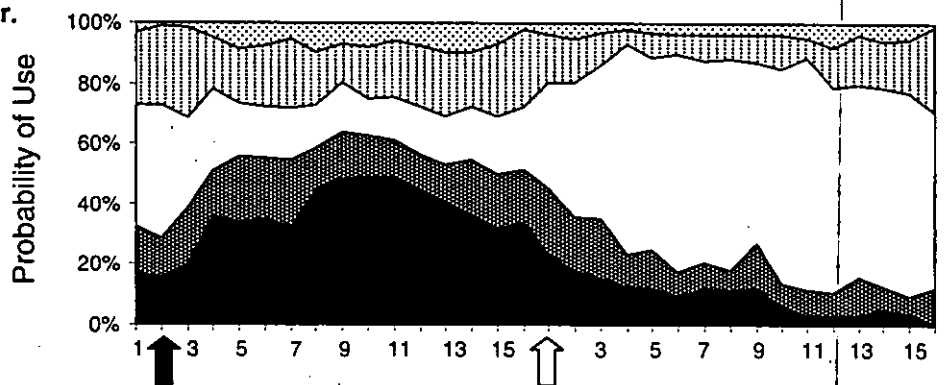
6. Nov./Dec.



7. Jan. '98



8. Feb./Mar.

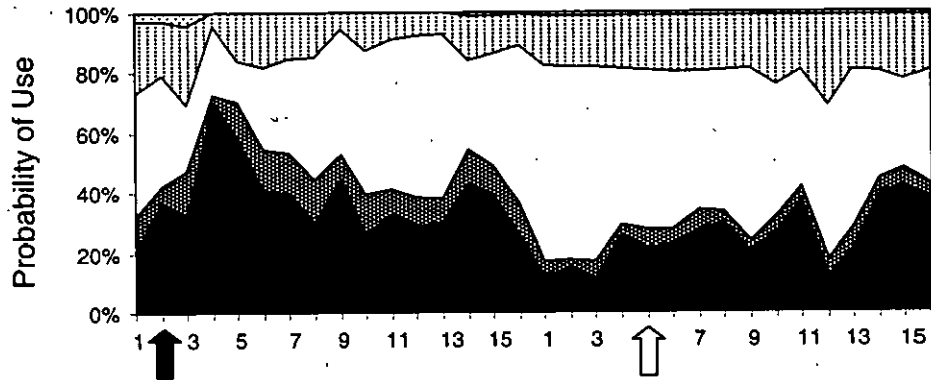


Fix within sampling period

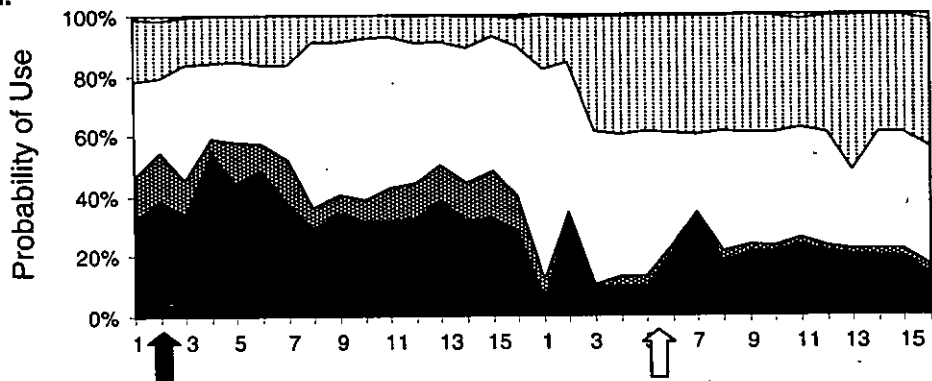
(b) Pademelon

Sampling period

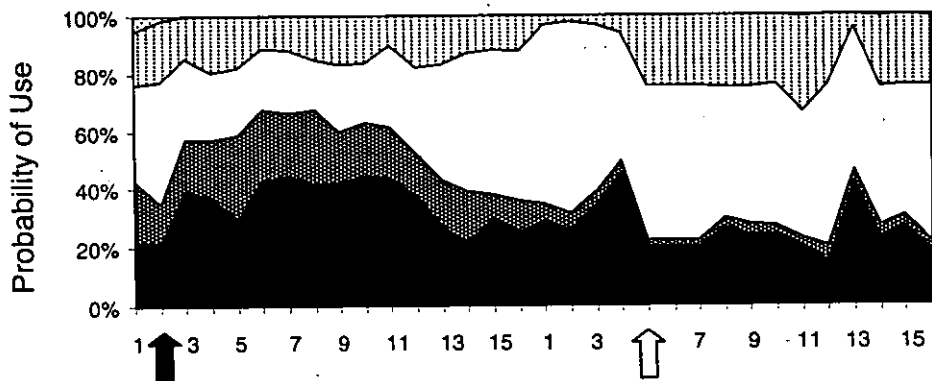
1. May '97



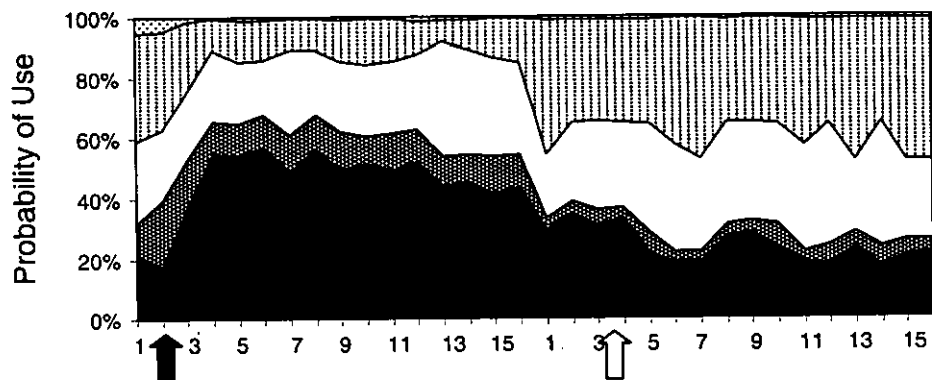
2. Jun./Jul.



3. Aug.



4. Sep.

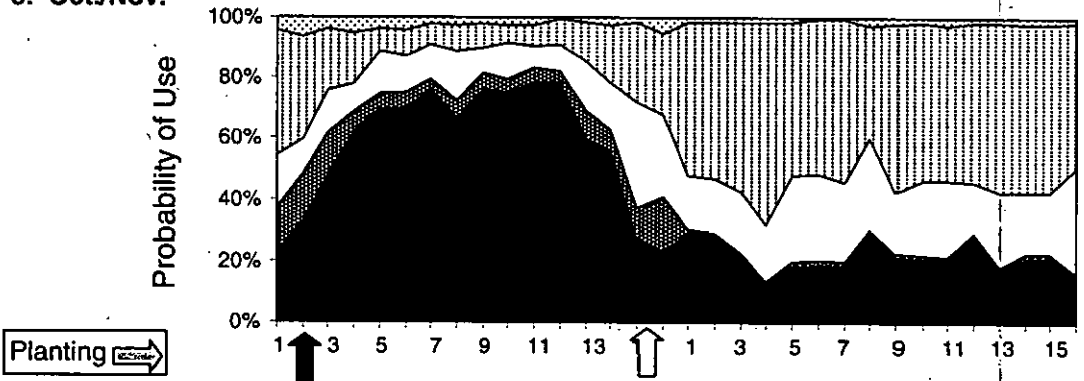


Fix within sampling period

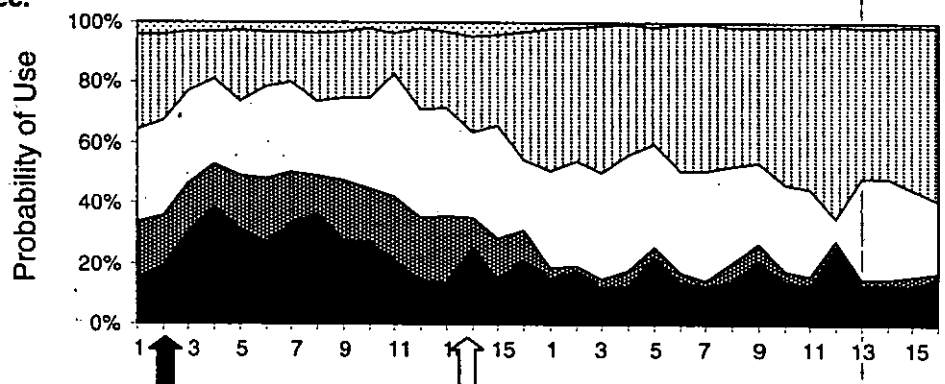
(b) Pademelon cont.

Sampling period

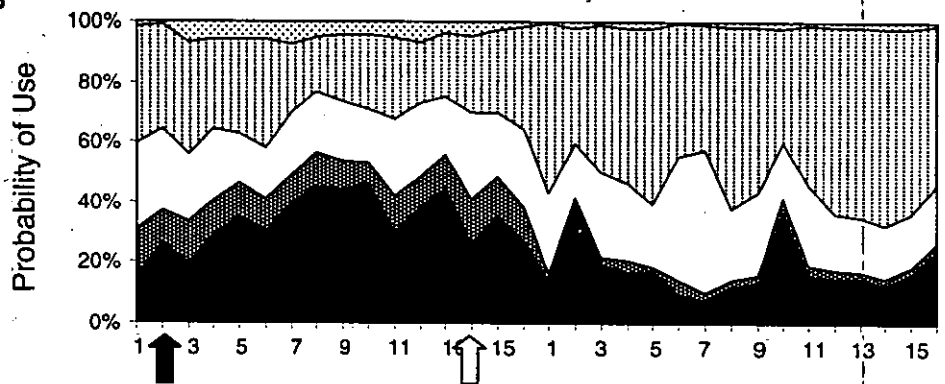
5. Oct./Nov.



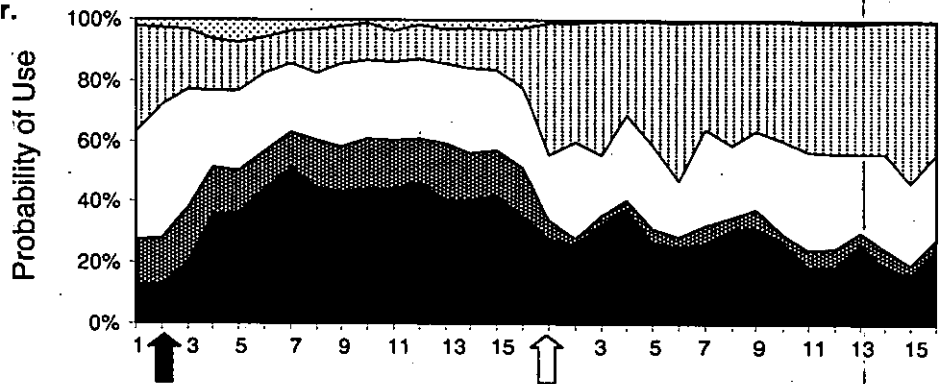
6. Nov./Dec.



7. Jan. '98



8. Feb./Mar.

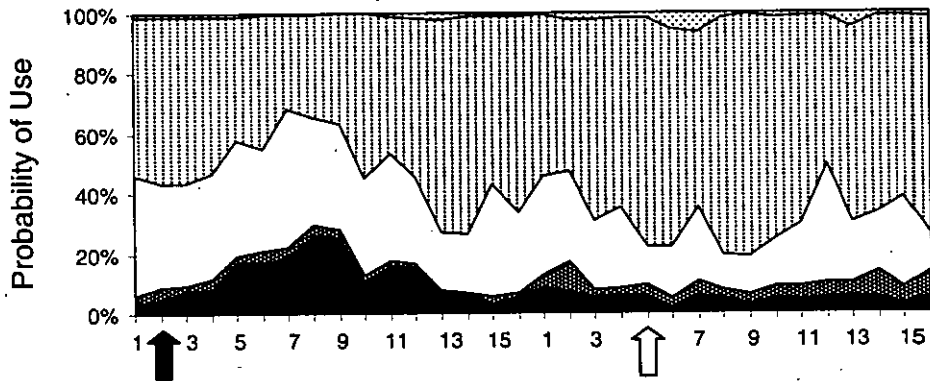


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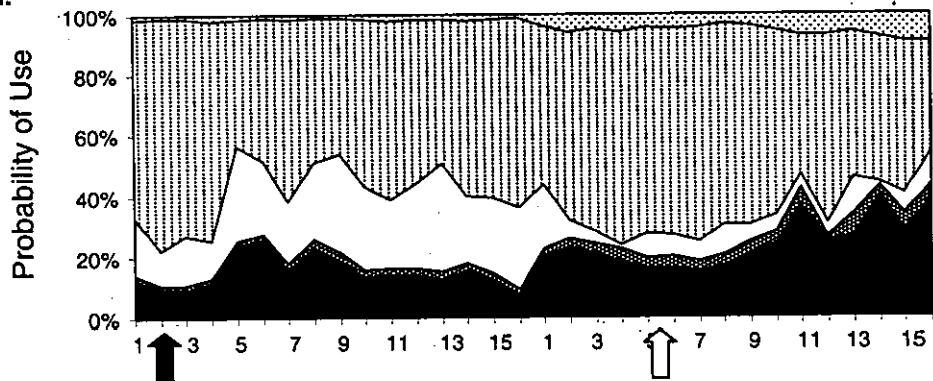
(c) Possum

Sampling period

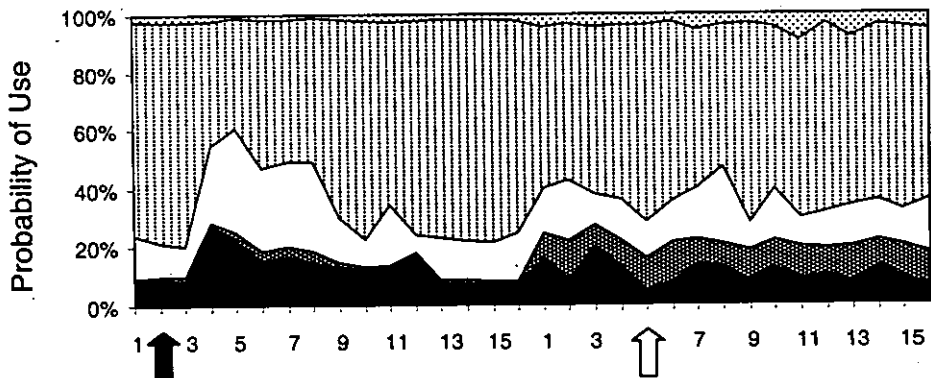
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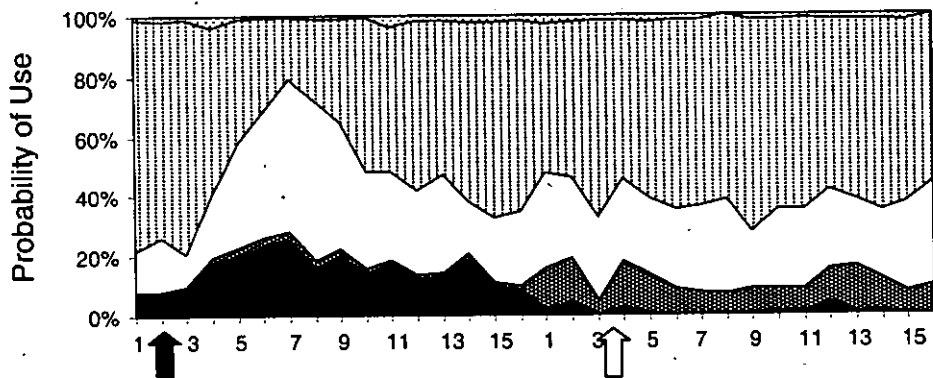
2. Jun/Jul.



3. Aug.



4. Sep.

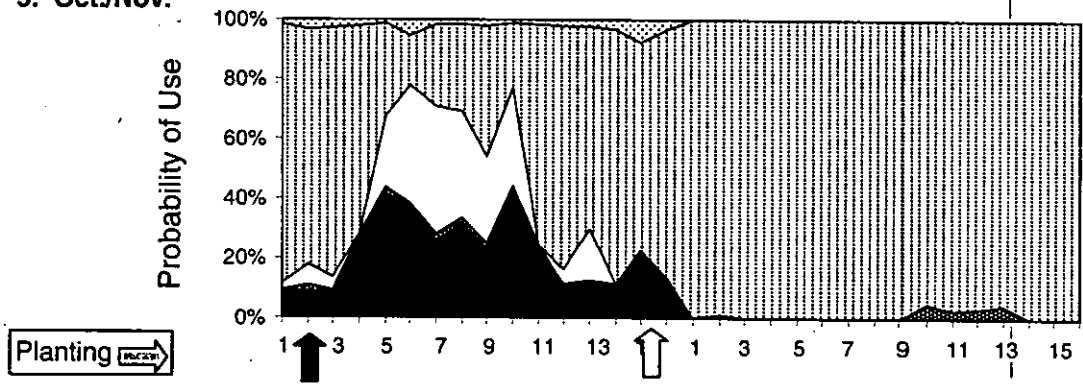


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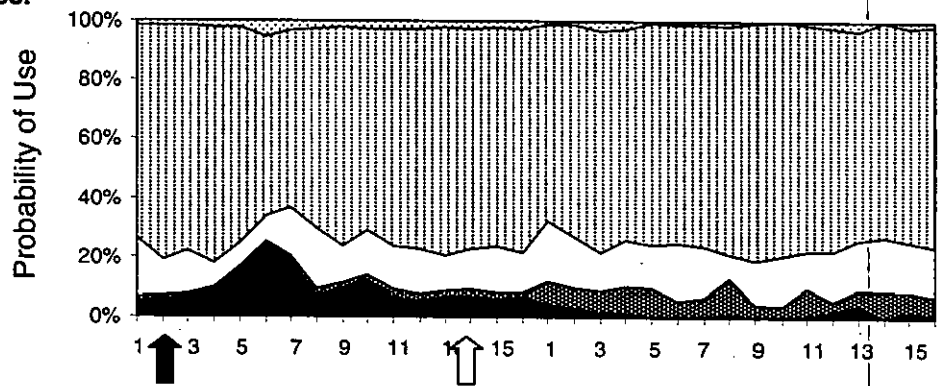
(c) Possum cont.

Sampling period

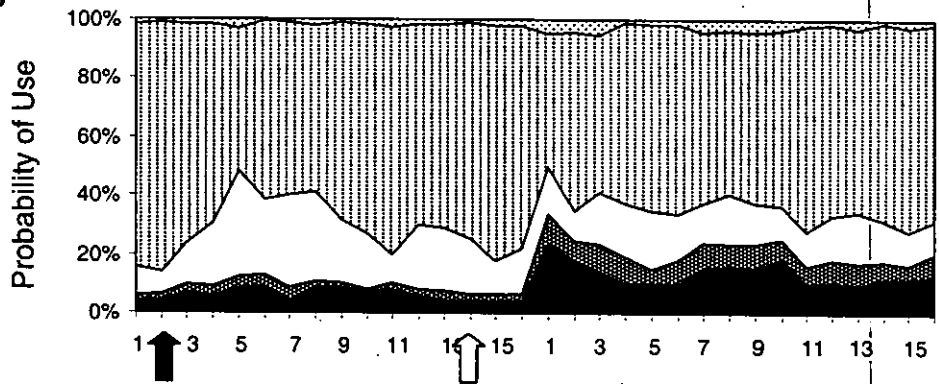
5. Oct./Nov.



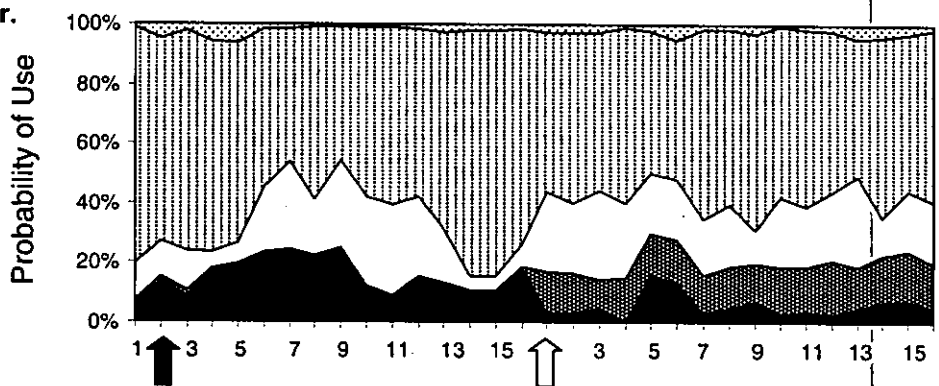
6. Nov./Dec.



7. Jan. '98



8. Feb./Mar.



Fix within sampling period

young plantation at night peaked in Sampling Period 6 (immediately after planting). Pademelons and possums showed no change in patterns of habitat selection following planting on the young plantation (Figures 8.3 and 8.4).

Table 8.4 Habitat selection ratios ($\hat{w}_i \pm \text{s.e.}$) for males and females, from the comparison between the proportions of habitat available within animals' home-ranges, to those used during night and day. Sample sizes were as follows: wallabies: males $n=4$, females $n=2$; pademelons: males $n=3$, females $n=3$; possums: males $n=2$, females $n=4$.
* signifies selected for, † signifies selected against

Sex	Time	Habitat	Wallaby	Pademelon	Possum
Males	Night	Young plantation	$1.94 \pm 0.07^*$	1.87 ± 0.38	0.91 ± 0.11
		Harvested land	0.95 ± 0.07	$0.45 \pm 0.13^\dagger$	$0.40 \pm 0.13^\dagger$
		Grassland	1.29 ± 0.12	1.00 ± 0.25	$0.16 \pm 0.04^\dagger$
		Older plantation	$0.71 \pm 0.02^\dagger$	$0.61 \pm 0.06^\dagger$	$0.49 \pm 0.20^\dagger$
		Native forest	1.10 ± 0.09	0.98 ± 0.24	$2.26 \pm 0.08^*$
	Day	Young plantation	$0.16 \pm 0.03^\dagger$	1.09 ± 0.16	0.80 ± 0.13
		Harvested land	$0.19 \pm 0.05^\dagger$	$0.23 \pm 0.06^\dagger$	1.05 ± 0.11
		Grassland	0.94 ± 0.14	$0.44 \pm 0.01^\dagger$	1.00 ± 0.00
		Older plantation	1.09 ± 0.07	1.09 ± 0.05	$0.69 \pm 0.02^\dagger$
		Native forest	1.34 ± 0.15	1.07 ± 0.14	$1.58 \pm 0.12^*$
Females	Night	Young plantation	$2.61 \pm 0.06^*$	$1.73 \pm 0.24^*$	$0.58 \pm 0.07^\dagger$
		Harvested land	1.42 ± 0.41	0.78 ± 0.17	$0.38 \pm 0.03^\dagger$
		Grassland	$0.19 \pm 0.12^\dagger$	0.51 ± 0.21	$0.18 \pm 0.03^\dagger$
		Older plantation	$0.22 \pm 0.09^\dagger$	0.65 ± 0.27	$0.62 \pm 0.09^\dagger$
		Native forest	$0.38 \pm 0.04^\dagger$	0.67 ± 0.13	$1.96 \pm 0.18^*$
	Day	Young plantation	0.84 ± 0.38	$0.46 \pm 0.02^\dagger$	$0.23 \pm 0.11^\dagger$
		Harvested land	1.01 ± 0.41	$0.19 \pm 0.02^\dagger$	$0.49 \pm 0.08^\dagger$
		Grassland	0.50 ± 0.21	$0.19 \pm 0.10^\dagger$	$0.42 \pm 0.13^\dagger$
		Older plantation	$1.67 \pm 0.01^*$	0.58 ± 0.21	$0.47 \pm 0.15^\dagger$
		Native forest	$0.37 \pm 0.04^\dagger$	$2.30 \pm 0.19^*$	$2.14 \pm 0.15^*$

8.4 Discussion

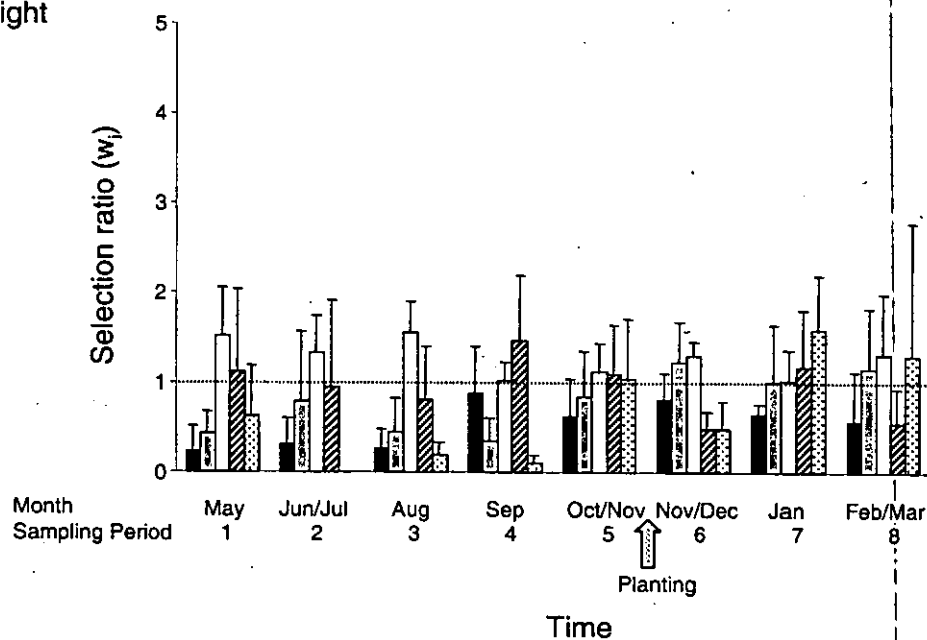
8.4.1 Habitat use

Wallabies, pademelons and possums showed clear patterns of habitat use. The two macropod species used open habitat (young plantation and grassland) more during the night, than during the day. This partitioning of home-range into nocturnal feeding areas and daytime shelter habitat are consistent with results for other macropod species (Chapter 4).

Possums primarily used native forest throughout the light cycle. This suggests that animals were both feeding and resting within this habitat. This result is consistent with possums feeding niche of foliovore, and their daytime use of hollow-bearing trees.

It is interesting to note that wallabies appeared to become more active at dawn and dusk in open habitats over summer, when night length was at its shortest. This suggests that animals may require a certain amount of time to conduct their foraging, and in summer this extends beyond the period of darkness. This result is inconsistent with other studies where macropods fed over a shorter period in summer in response to heat stress during the day, for example, eastern grey kangaroos *M. giganteus* (Clarke *et al.* 1989), red kangaroos *M. rufus* (Priddel 1986) and western grey kangaroos *M. fuliginosus* (Priddel 1986). Perhaps with Tasmania's cooler summer climate wallabies within the present study were able to be more

(a) Night



(b) Day

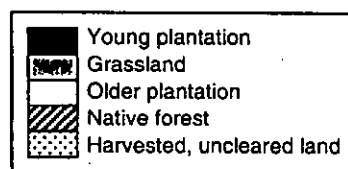
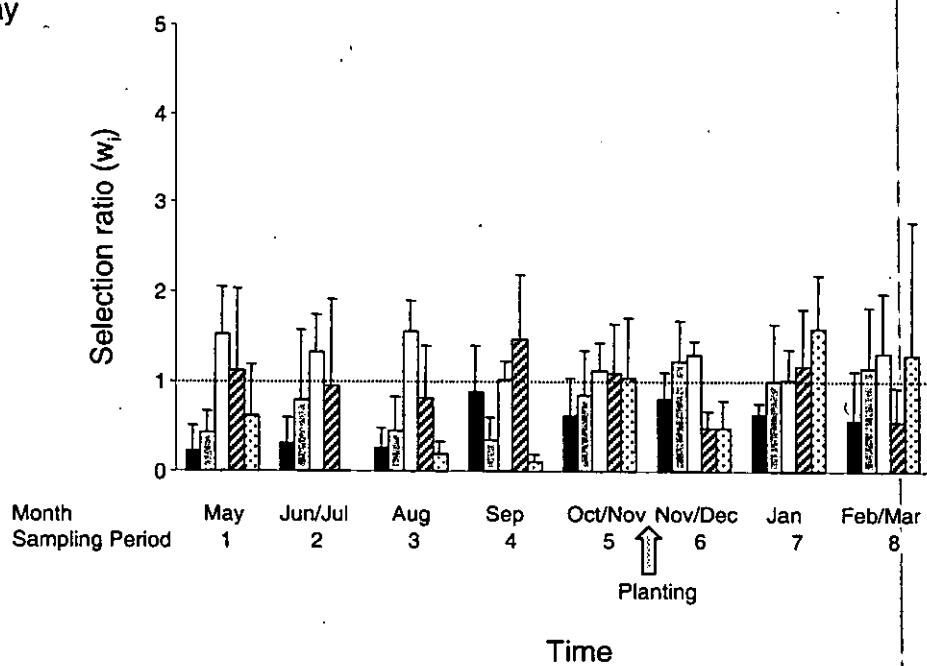


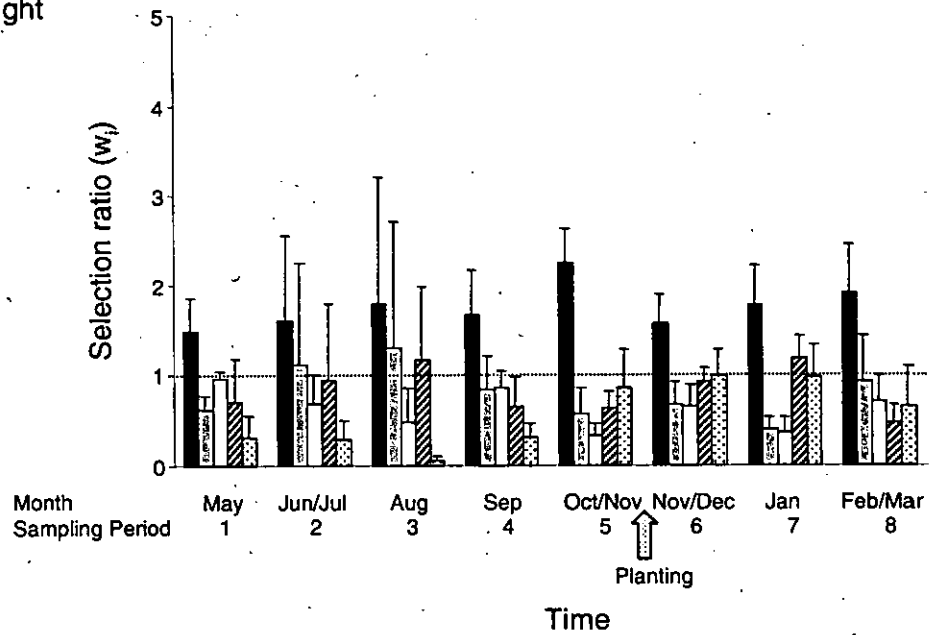
Figure 8.2

Patterns of habitat use (mean proportion of use + 95% C.I.) within wallabies' home-ranges during night and day, over time.

..... indicates no selection

above the no selection line = selected for; below = selected against

(a) Night



(b) Day

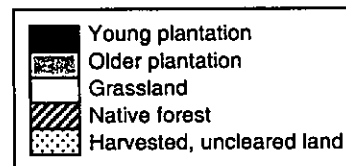
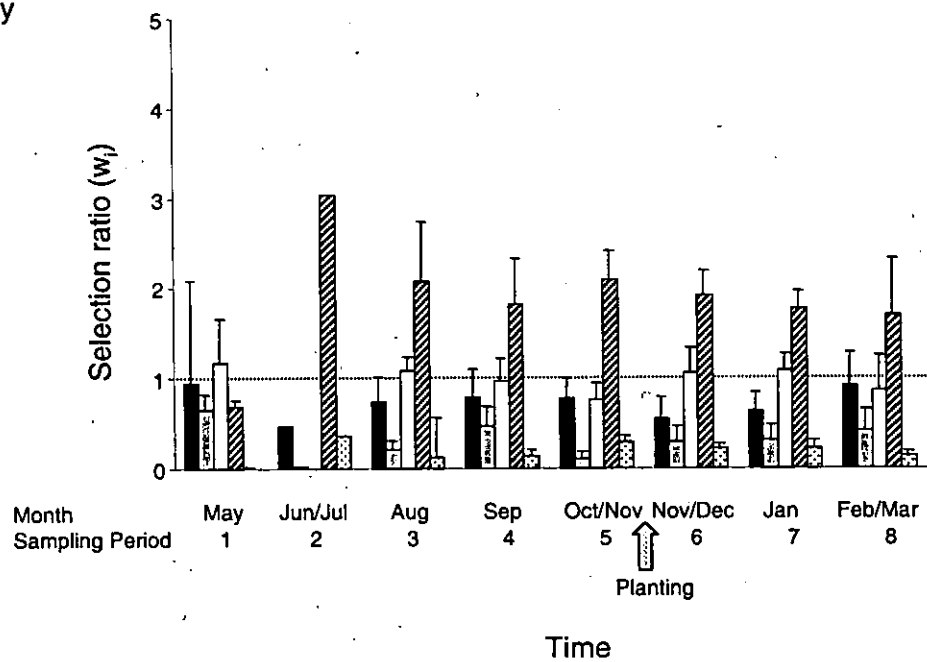


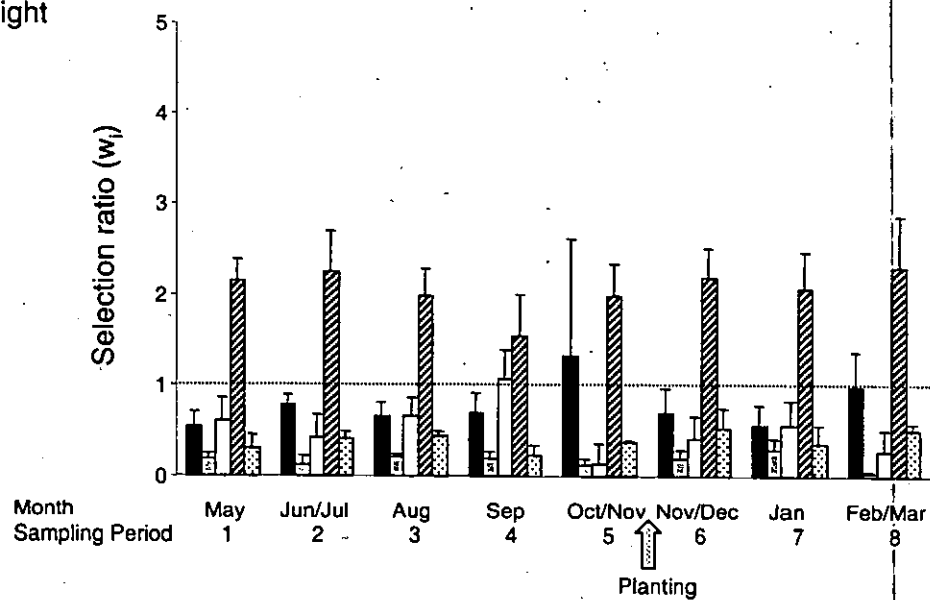
Figure 8.3

Patterns of habitat use (mean proportion of use + 95% C.I.) within pademelons' home-ranges during night and day, over time.

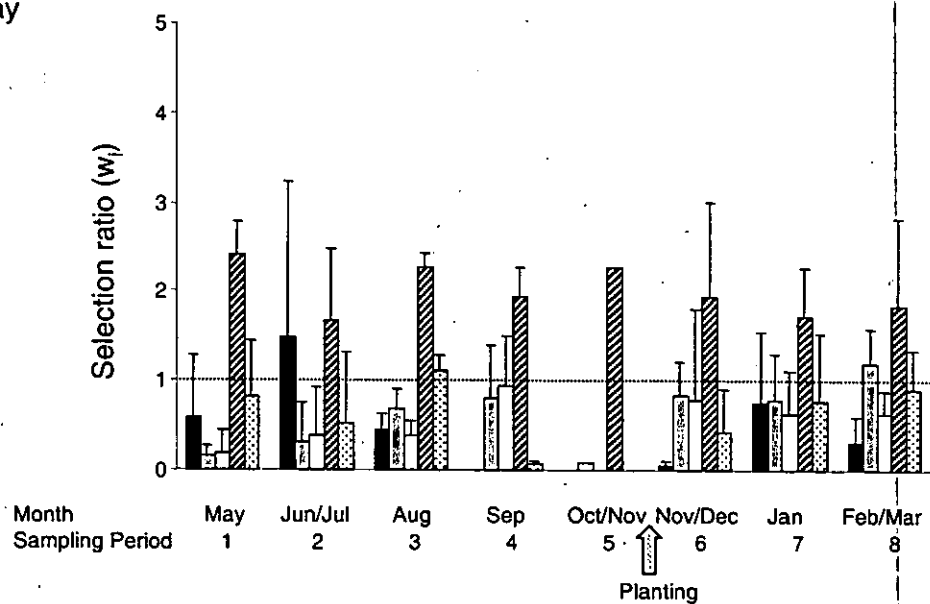
..... indicates no selection

above the no selection line = selected for; below = selected against

(a) Night



(b) Day



Time

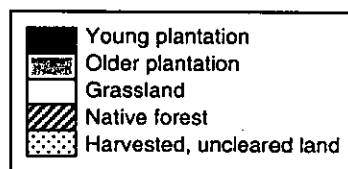


Figure 8.4

Patterns of habitat use (mean proportion of use + 95% C.I.) within possums' home-ranges during night and day, over time.

..... indicates no selection

above the no selection line = selected for; below = selected against

flexible (or less constrained) in their daily scheduling of foraging activities, than these other macropod species on mainland Australia.

Possums primarily used native forest throughout the light cycle. This suggests that animals were both feeding and resting within this habitat. This result is consistent with possums feeding niche of foliovore, and their daytime use of hollow-bearing trees.

It is interesting to note that wallabies appeared to become more active at dawn and dusk in open habitats over summer, when night length was at its shortest. This suggests that animals may require a certain amount of time to conduct their foraging, and in summer this extends beyond the period of darkness. This result is inconsistent with other studies where macropods fed over a shorter period in summer in response to heat stress during the day, for example, eastern grey kangaroos *M. giganteus* (Clarke *et al.* 1989), red kangaroos *M. rufus* (Priddel 1986) and western grey kangaroos *M. fuliginosus* (Priddel 1986). Perhaps with Tasmania's cooler summer climate wallabies within the present study were able to be more flexible (or less constrained) in their daily scheduling of foraging activities, than these other macropod species on mainland Australia.

8.4.2 Habitat selection

Macropods

Wallabies and pademelons displayed highly selective behaviour at both spatio-temporal scales examined. At the larger scale of home-range location within the study area, both species showed very strong selection for young plantation, followed by harvested uncleared land. It is important to note, however, that trapping bias may have confounded animals' patterns of habitat selection. Animals caught and collared for the radio-tracking study were trapped on, or near, the young plantation. Hence, it is probably not surprising that this habitat appears selected for within these animal's home-range.

While young plantation offered a highly productive foraging ground (see Chapter 10), it is unclear why the harvested land would have been a preferred habitat as it contained little forage, although abundant shelter. It is possible that the geographic distribution of this habitat and the use of buffered points, may have resulted in its selection within the analysis, simply due to its proximity to the young plantation.

At the smaller spatial scale of night-range, the two macropod species again selected for young plantation. This suggests that the young plantation was an extremely important habitat for these species. At night, wallabies also selected against older plantation, while pademelons selected against all closed habitats. Consequently, the macropod species based their nocturnal, fine scale, patterns of habitat selection on foraging decisions.

During the day, macropods selected closed habitats, presumably for shelter. It is interesting to note, however, that wallabies and pademelons selected different daytime habitats. Wallabies selected for older plantation, while pademelons selected for native forest. I suggest that this reflects their different predator avoidance strategies, or escape response (Lima 1992). Jarman (1991) suggested that larger macropods rely upon the early detection of predators and their ability to out-run them, while smaller macropods rely on crypsis, to avoid being detected. Consequently, for the wallaby, older plantation would provide shelter through the screening effect of trees' side branches and the presence of windrows, but the relatively open vegetation would enable predators to be detected at a distance, so animals could flee in response. In contrast, native forest would provide pademelons with abundant shelter within dense shrubs and hollow logs, and with their brown and buff pelage, animals would be well camouflaged from predators. A similar pattern of spatial segregation between sympatric naitailed and black-striped wallabies for daytime shelter, has been reported by Evans (1996).

Overall, patterns of habitat selection by macropods at this site were based on foraging decisions at both spatial scales, and predator avoidance decisions only at the smaller scale. Selection of habitats based upon foraging decisions reflects results from some other studies, for example, capercaillie *Tetrao urogallus* (Storch 1993) and red deer *Cervus elaphus* (Langvatn and Hanley 1993). Other species, however, select habitats based upon the risk of predation, whereby animals compromise fitness by foraging in sub-optimal feeding habitats, but benefit because these areas are relatively safe. Examples include, baboons *Papio cynocephalus* (Cowlshaw 1997) and caribou *Rangifer tarandus* (Ferguson *et al.* 1988).

Sexual segregation by macropods at Surrey Hills was suggested by males apparent random use of their home-range. Results should be interpreted with caution, however, due to small sample size. This apparent random use of home-range by males may reflect their larger range size. Alternatively, males may show larger individual variation in their patterns of habitat selection. This may be related to an individual's social status within the population as larger, more aggressive males may hold better quality and/or larger home-ranges than smaller subordinates (Jarman 1991). Further studies with larger sample sizes of radio-collared animals, preferably with males and females ranging in age, are required to clarify this situation.

Possums

Possums' patterns of habitat selection at the larger scale were based upon decisions that are difficult to interpret. Possums showed a strong selection for the young plantation, followed by harvested uncleared land and grassland. Older plantation was also avoided, while native forest was used as available. While selected habitats offered some food and shelter to possums, the habitat with the greatest perceived abundance of food and highest density of hollow-bearing trees was native forest, and this habitat was not selected at this higher level. Consequently, I suggest that selection for the young plantation (and by default harvested uncleared land) reflects trapping bias, as discussed above. This is consistent with animals core home-range areas shown in Figure 7.2, which included little young plantation. Selection of grassland habitat at this higher level, is thought to reflect animals use of individual hollow-bearing trees within these areas, rather than the grassland *per se* (see Chapter 4).

At the smaller scale of habitat selection, possums selected for native forest and avoided all other habitats both night and day. This pattern is consistent with habitat selection decisions based upon the abundance of food and shelter, as discussed above.

8.4.3 Response to planting

Only the macropod species appeared to change their behaviour in response to planting. Pademelons appeared to use this habitat less after planting, but their patterns of selection were unchanged. This perceived behavioural response to planting may reflect a learnt association of seedlings with control operations, such as shooting and/or 1080 poisoning. The act of planting seedlings was thought unlikely to affect pademelons as the process only took one day. In addition, animals were used to people on foot at this site, as I regularly surveyed vegetation and seedlings on the young plantation. Herbiciding was also considered unlikely to negatively impact upon animals use of this site, as only small areas surrounding seedling were sprayed, and it had rained between herbiciding and data collection, which theoretically, would wash the chemical off vegetation.

In contrast to pademelons, wallabies' patterns of habitat use and selection for the young plantation peaked immediately after planting. The decline of pademelon use and increase in wallaby use of the young plantation at this time may have been linked. For example, if

pademelons were dominant to wallabies as discussed in Chapter 4, their reduced use of the young plantation may have enabled wallabies to spend more time feeding within this habitat.

Chapter 9

Radio-Telemetry: 1080-Poisoning Operation

Radio-collared animal data only, published as:

le Mar, K. and McArthur, C. (2000). Re-locating radio-collared targeted marsupials after a 1080-poisoning operation. *Tasforests* 12:155-160.

Dr Clare McArthur provided financial and logistical support, contributed to ideas and assisted with editing manuscripts.

9.1 Introduction

Three native marsupial species reduce productivity in commercial Tasmanian plantations by browsing seedlings (Cremer 1969, Coleman *et al.* 1997, Bulinski and McArthur 1999). These species are the red-bellied pademelon, Bennett's wallaby, and the brushtail possum. To reduce browsing damage, forestry companies poison local herbivore populations with 1080 (sodium monofluoroacetate), as it is assumed that fewer animals will result in less damage.

A handful of studies have examined the large-scale effects of poisoning operations at reducing herbivore abundance (Johnson 1978, Statham 1983, Marsh 1998, Bulinski 1999b). The fine-scale effects of 1080 operations however, have never been quantified and documented. Information such as how far animals travel from the bait-line before death, and where carcasses are located, have important implications for the forestry industry in relation to carcass retrieval and potential effects of poisoning near private land. Consequently, the aims of this study were to:

- (1) determine how far animals travelled from the bait-line before death,
- (2) investigate where carcasses were located,
- (3) examine the sex ratio of animals killed, and
- (4) examine whether any non-target species were killed during this operation.

9.2 Materials and Methods

9.2.1 Study site

The study area was located in Gunns' Surrey Hills Tree-farm, northwest Tasmania (41°28' S, 145°48' E). Five habitats dominated this site (refer to Figure 9.1): (1) a young *Eucalyptus nitens* plantation with an area of 17.8 ha and with relatively high weed cover; (2) older plantations of *E. nitens* (5-7 years of age); (3) grassland; (4) native forest; and (5) uncleared harvested land that consisted of scrub and fallen vegetation.

The young plantation was planted with *E. nitens* seedlings (approximately 20 cm in height) in November 1997, five months before this study began. The young plantation had not been treated with 1080 poison or herbicide before planting, but vegetation within a 0.5 m radius of individual seedlings was treated with Roundup® herbicide approximately one week after planting.



LEGEND

- Radio-collared pademelon
- ⊗ Collar from pademelon
- △ Radio-collared wallaby
- ⊙ Radio-collared possum
- ⊕ Collar from possum
- ⋯ Bait-line
- ~ Waterway
- - - Dirt road
- Gravel road
- Young plantation (▨ windrows) - poisoned plantation
- ▨ Older plantation
- ░ Uncleared, harvested land
- Native forest
- Grassland

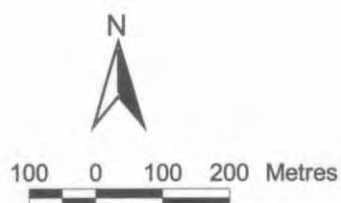


Figure 9.1 Map of the study site, showing the five habitats, location of the bait-line and sites where dead radio-collared animals (or collars only), were found.

9.2.2 Poisoning operation

Staff of Gunns Ltd. carried out the baiting operation under a permit from the Tasmanian Parks and Wildlife Service. Three weeks before poisoning, bait stations were established at 10 m intervals along the young plantation's central access road, the plantation boundary, and nearby firebreaks. Twenty grams of fresh, chopped carrot was placed at each bait station on five occasions (referred to as 'free-feeds'), 3-6 days apart, over a period of 17 days in April 1998. Fifty kilograms of bait was distributed during the first two free-feeds; this was increased to 60 kg during the last three free-feeds. Sixty kilograms of poisoned bait (chopped carrot mixed with 0.014 % 1080 in liquid solution and blue dye) was distributed at the bait stations (hereafter referred to as the 'bait-line', see Figure 9.1) three days after the final free-feed. Bait stations were checked by company staff for remaining poisoned bait 24 hours after distribution. No poisoned bait remained 24 hours after it was distributed.

9.2.3 Radio-collared animals

Animals were caught and trapped as described in Chapter 7. Ten adult pademelons (7 male and 3 female), seven adult wallabies (5 male and 2 female), and nine adult possums (4 male and 5 female) were fitted with a single-stage radio-transmitter (Sirtrack Electronics, New Zealand) attached to a leather collar. Each transmitter emitted a unique frequency between 150.000-151.999 MHz. These animals had been used in a larger study to investigate habitat utilization within the study site. Consequently, some individuals had been collared for up to 15 months before free-feeding. Sample sizes of the radio-collared animals represented approximately 7 % of the pre-free-feeding pademelon population and 21 % of the estimated wallaby population that utilized the young plantation nightly (Chapter 4). More possums were radio-collared than were estimated to use the young plantation at any one time. Results from a larger study estimated a mean nightly abundance (\pm s.e.) of 2 (\pm 1) possums (Chapter 4).

An 'Automated Telemetry Systems' receiver and a three-element yagi hand-held antenna were used to locate animals. Animals were radio-tracked on foot within two days of poisoning. Locations of dead radio-collared animals were mapped and their distances from the closest point on the bait-line were calculated. Females were also checked for pouch young.

9.2.4 Uncollared animals

Carcasses of uncollared animals were found in two separate searches. The first search was conducted 24 hours after poisoning while checking bait stations for residual bait. This search was relatively brief (8 person-hours) and largely restricted to the bait-line. Dead animals were identified to species and sex, and their location was recorded. This search was similar to those typically conducted by forestry operations staff, following a poisoning operation (T. Dick *pers. comm.*).

The second search for uncollared dead animals occurred while re-locating radio-collared animals. Consequently, these carcasses were found incidentally and usually in association with dead radio-collared individuals. Data collection was as above. It is important to note that most of these animals would not have been detected in the absence of searching for radio-collared animals, due to their location within shelters (for example, windrows and logs). Dead animals found incidentally while conducting spotlighting surveys on the young plantation during Sampling Period 10 (the 17-day period following poisoning), are also reported here.

9.3 Results

9.3.1 Radio-collared animals

Eleven of the 26 radio-collared animals were located alive. The remaining 15 collared animals were killed during the poisoning operation: 8 of 10 pademelons, 1 of 7 wallabies, and 6 of 9 possums. Both male and female radio-collared pademelons and possums were killed: 5 of 7 male pademelons and 3 of 3 female pademelons; 4 of 5 male possums and 2 of 6 female possums. The one wallaby that was killed was a female with a pouch-young (furless, dead in pouch).

Twelve of the 15 radio-collared animals that died during the poisoning operation were found undamaged by carnivores. These animals were found within 83 m of the bait-line (Figure 9.2); mean distance 31 m (range 8-83 m). Eighty-four percent of carcasses were located within 40 m of the bait-line (Figure 9.2). Few animals died out in the open: 75 % of carcasses were found inside windrows, under fallen vegetation or inside dens (Table 9.1). Blood was present in the nostrils and ear canals of two pademelons suggesting internal bleeding as the result of poisoning.

Table 9.1 Locations of radio-collared animals killed by 1080.

Habitat	Location	Pademelon	Wallaby	Possum
Young plantation	Open area	1	-	-
	Close to windrow (< 2 m)	2	-	-
	Inside windrow	1	-	1
Older plantation	Inside windrow	1	-	-
Harvested land	Under fallen vegetation	1	1	1
Native forest	Inside hollow log	-	-	1
	Under fallen vegetation	1	-	-
	Moved by carnivore	1	-	2
Grassland	Underground in soil chamber	-	-	1

Three of the 15 radio-collared animals that died during the poisoning operation were not recovered. Two of the three radio-collars were relocated and showed evidence of carnivore attack (chewed leather collars and teeth marks in the epoxy resin casing). One of these collars was also located near fur and intestines that were assumed to belong to the radio-collared possum. The third collar was mapped by triangulation of radio-bearings but was not found.

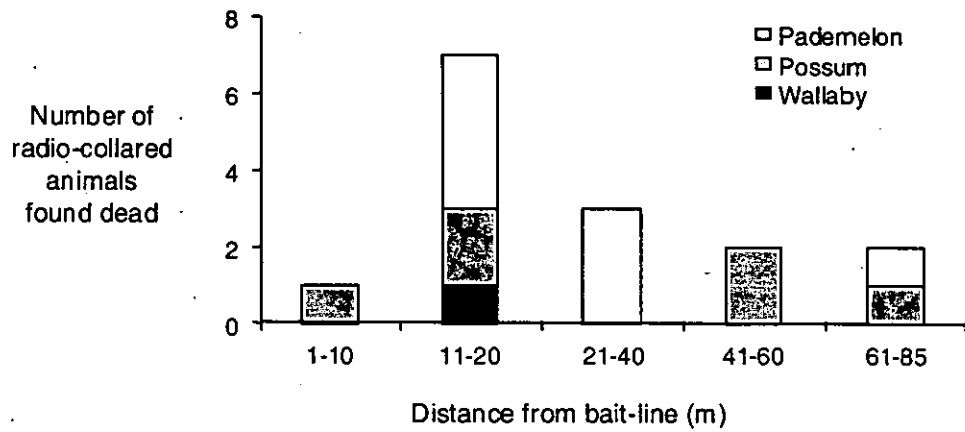


Figure 9.2 The distances dead radio-collared animals (killed by 1080) were found from the bait-line.

9.3.2 Uncollared animals

Fifteen uncollared carcasses (12 pademelon, 1 wallaby, 1 possum and 1 rabbit) were found during the brief search along the bait-line 24 hours after poisoning (Table 9.2). Six carcasses (5 pademelon and 1 wallaby) were found while searching for radio-collared animals (Table 9.3). Additionally, six pademelon (5 males, 1 unknown) and one wallaby (small male) carcass were found while collecting spotlighting data on the young plantation during Sampling Period 10.

Table 9.2 Details of carcasses found around the bait-line and on the open young plantation, 24 hours after poisoning.

Species	Sex	Age	Number
Pademelon	male	adult	7
	male	sub-adult	2
	female	adult	2
	unknown		1
Wallaby	male	sub-adult	1
Possum	female	sub-adult	1
Rabbit	unknown		1

Table 9.3 Details of carcasses found incidentally while searching for radio-collared animals.

Species	Sex	Age	Number	Location
Pademelon	male	adult	2	In windrow
	female	adult	1	In grassland
		sub-adult	1	By windrow
	unknown		1	Open area on young plantation
Wallaby	male	sub-adult	1	Open area on young plantation

9.4 Discussion

The poisoning operation killed individuals from all three targeted marsupial species, as well as rabbits.

9.4.1 Distance from the bait-line

Results showed that all possum and pademelon radio-collared carcasses were found within 85 m of the bait-line. Consequently, data for these species are inconsistent with previous speculation that poisoned animals travel several hundreds of metres, even kilometres, between the time they ingest poisoned bait and death (Johnson 1978, Cahalan 1998). Results for wallabies, however, are inconclusive. Only one radio-collared animal was killed during the operation, and a range of distances travelled from the bait-line is required to assess whether this single result was indicative of the species. Furthermore, there is evidence (C. McArthur unpublished data) that wallabies can travel substantially further (≈ 240 m) from the bait-line than indicated by this study. Hence, wallabies may travel further from the bait-line than possums or pademelons.

9.4.2 Carcass location

Results from this study suggest that most animals sought shelter after consuming poisoned bait. The use of shelter may indicate that animals were aware that they were unwell and were seeking safety from predators. The main predators of these three marsupial species, the Tasmanian devil (*Sarcophilus harrisii*) and the spotted-tailed quoll (*Dasyurus maculatus*) (Jones and Barmuta 1998), were known to inhabit the study site (K. le Mar *pers. obs.*). Results from this study do not, therefore, support previous speculation that animals poisoned with 1080 seek water. Indeed, one possum crossed a river to reach shelter (a known den site). The effect of 1080 is to block energy production within cells, which results in cell death (Atzert 1971, cited in Statham 1983). Consequently, 1080 results in changes to cell structure, not body fluid composition as such, therefore it is unlikely to result in an animal seeking water from dehydration.

9.4.3 Sex ratio of killed animals

The poisoning operation appeared biased towards male pademelons. Although both male and female radio-collared pademelons were killed with 1080, indicating that some females were poisoned during the operation, 80 % of pademelon carcasses found on and around the young plantation were male. This may reflect a male biased sex ratio within this habitat, or male pademelons dominance of the young plantation edge region, where most of the bait was laid. Evidence of the former comes from observations of a highly skewed sex-ratio (90 % male) while dart-gunning animals for radio-collar attachment on the young plantation. Evidence of the latter is based on observations of large male pademelons territorially defending bait stations and displacing subordinates and wallabies (N. Marsh *pers. comm.*), and the majority of bait was laid on the young plantation perimeter.

The poisoning operation failed to kill large male wallabies. These animals may have been bait-shy, through previous experience with poisoned bait. This response has been reported for possums (Morgan *et al.* 1996) but has not been investigated in wallabies. The large body mass of wallabies also makes it susceptible to receiving sub-lethal doses of 1080. It is strongly dimorphic, with adults weighing 20-30 kg. Adult males, in particular, may therefore be prone to sub-lethal poisoning and subsequent bait avoidance. These conclusions are

consistent with anecdotal evidence from operations staff and similar findings by Mooney and Johnson (1979).

The poisoning operation appeared to kill both sexes of possums equally. Similar results were found by Statham (1983), although her sample sizes were relatively low.

9.4.4 Non-target species

No poisoned non-target species were found during the course of this study. Some of the non-target species considered most vulnerable to poisoning with 1080 include the Tasmanian bettong (*Bettongia gaimardi*), long-nosed potoroo (*Potorus tridactylus*) and the common wombat (Statham 1996). A decline in density for brown bandicoots (*Isodon obesulus*) following a poisoning operation has also been reported by Mooney and Johnson (1979). No bettongs, potoroos or bandicoots had been seen within the study area for the 18 months before poisoning. Consequently, the absence of poisoned non-target animals is likely to reflect their absence from the study area, rather than necessarily an escape from poisoning.

Poisoned wombats have been reported by several authors (Mooney and Johnson 1979, McIlroy 1982, Triggs 1996). Wombats were abundant on the young plantation and known to live in windrows (K. le Mar *pers. obs.*), yet no dead wombats were found during the course of this study. However, in the absence of checking burrows for sick and dead animals, these results are inconclusive.

9.4.5 Management implications

Four main implications can be drawn from this work in relation to forestry. First, animal carcasses are extremely difficult to locate following a poisoning operation. Therefore routine carcass collection operations following 1080 poisoning are unlikely to be effective at removing most carcasses. Second, pademelons and possums appear to die close to the bait-line. Results for large male wallabies are not known, but distances are expected to be greater than for the other species. Third, poisoning operations may be biased against female pademelons and male wallabies, and this clearly requires further investigation. Killing only one sex could bring about an overall increase in local herbivore densities by inducing a breeding response to the sudden increase in the availability of local resources. Finally, future investigation into the dynamics between the macropod species is required to fully understand the effects of poisoning. Dominance hierarchies between species may play an important role in the success of present control methods and should be considered during the development of alternative, non-lethal, management methods.

Chapter 10

Vegetation Surveys: Impacts of Foraging Herbivores

10.1 Introduction

This chapter investigates what herbivores ate while foraging on the young plantation. On a hierarchical scale of habitat selection, patterns determined by this study fall between an animal's choice of feeding area within its home-range (Johnson's [1980a] third-order selection), and the plant-parts consumed (Johnson's [1980a] fourth-order selection). Hence, this study determines vegetation use at a relatively small scale in relation to work described in previous chapters.

Consumption of various plant groups by the herbivore species were expected to reflect their different feeding strategies. Based on their dental morphology and gut physiology, the Bennett's wallaby is classed as a grazer (Sanson 1989, Hume 1999), feeding predominantly on grasses (Southwell 1987a, Jarman and Phillips 1989, Statham 1983, Calaby 1991, Sprent 1997). Pademelons are classed as a mixed-feeder, because they are intermediate in form between a grazer and a browser (Sanson 1989), feeding on grasses and forbs and occasionally browsing woody plants (Statham 1983, Johnson and Rose 1991, Sprent 1997). Possums are folivores (Hume 1999), feeding on soft and hard-leaved dicots (Statham 1983, Green 1984, How 1991). Consequently, macropods were hypothesised to forage on grasses and forbs, and possums to forage on forbs and leaves of trees and shrubs.

Results described in this chapter are restricted to ground vegetation data. The proportion of vegetation on the young plantation that comprised commercial seedlings was minute, but important for applied aspects of this thesis. Consequently, animals' consumption of commercial seedlings will be described separately in Chapter 11. Hence, the aims of this chapter were to:

- (1) quantify vegetation biomass removed by herbivores at two temporal scales,
- (2) partition the effects of foraging between macropods and possums,
- (3) investigate the consumption of different plant groups by macropods and possums, and
- (4) monitor herbivores' consumption of ground vegetation before and after the planting of commercial tree seedlings.

This work also has applications for forestry practices, as it examines potential effects of retaining vegetation on animals' use of young plantations. Consequently, the relationship between ground vegetation consumed, herbivore population densities, and seedling growth rate and survival, will be discussed in Chapter 12.

10.2 Materials and Methods

The 18 ha young plantation is described in Chapter 2. Three vegetation plot types (treatments) were used: (1) unfenced [UF]; (2) fenced, no roof [F-R]; and (3) fenced, with roof [F+R]. UF plots were used to examine the effect of all mammalian herbivores (wallabies, pademelons, possums, wombats and rabbits) on vegetation, F-R plots were used to examine the effect of possum browsing on vegetation, and F+R plots were used to examine the absence of mammalian herbivores on vegetation. Unfortunately it was not possible to have a treatment that could target a single macropod species. All three plot types were accessible to insects. A total of 45 plots (15 of each treatment) were erected between February and March 1997; approximately eight months before seedlings were planted on the young plantation.

Plots measured 6 x 8 m and were randomly located on a 50 m grid, with two constraints: (1) at least one plot existed between each set of parallel windrows (or 'bays'), and (2) within a given bay, plot boundaries were at least 50 m apart. UF plots were marked using wire 'pig-tails' and coloured surveyors tape. Fenced plots were constructed using steel star pickets and wire poultry netting (1.8 m high, 1 mm gauge, 50 mm spacing). Fences were approximately 1.3 m high and netting at the bottom of each fence was buried to a depth of 0.2 m to prevent burrowing animals from accessing plots. Although possums can readily climb fence netting, wooden ramps were attached to fences to enhance access to the F-R plots. Two ramps were attached to the outside of a plot and one to the inside. For a given plot, the sides on which the ramps were attached were allocated at random. F+R plots were covered with netting to prevent possum access. Fencing was checked each fieldtrip to ensure neither rabbits nor wombats had dug into plots. There was never any evidence to suggest that herbivores had gained access to F+R plots.

Planting and herbiciding operations, which took place on the young plantation during this study, are described in Chapters 2 and 11. Briefly, 1100 *Eucalyptus nitens* seedlings were planted per hectare, in November 1997. Six seedlings were planted per plot: three trees in two adjacent tree rows. One week after planting, all vegetation within a 0.5 m radius of seedlings were given a spot-application of herbicide.

10.2.1 Data collection

Vegetation data were collected on seven occasions; four before planting and three after planting. The sampling intervals were the same as those used for collecting spotlighting and radio-tracking data, however, it was not possible to collect data at Sampling Period 2 due to heavy snow cover on the young plantation.

Vegetation was classed into seven categories: (1) grasses; (2) forbs (small, herbaceous, non-woody dicotyledons); (3) shrubs (small bushes, often perennials with lignified stems); (4) trees; (5) non-grass monocots (rushes, irises and orchids); (6) moss, lichen or fungi; and (7) ferns. A species list is given in Appendix B. Many specimens could not be identified beyond genus as they were cropped by herbivores and/or without reproductive structures.

Data were collected for plant matter that was considered potential food available to animals. Consequently, data for partially dried-off grasses and dried grasses with seedheads were included (Jarman 1994), but dead vegetation within other categories were not.

Vegetation within plots were sub-sampled, using four haphazardly selected quadrats (1 x 1 m) per plot. Within each quadrat, data collection included percent ground cover and vegetation height. Height was taken as the highest point where vegetation intercepted the ruler at (usually 20) randomly allocated points, within a quadrat. Vegetation that had been clipped in preparation for herbiciding, or had been sprayed with herbicide, were not measured.

Vegetation biomass could not be determined by harvesting due to time constraints. Instead, data were collected on the relative proportions of different vegetation categories using a visual estimate described by Sprent (1997). In brief, species were assessed as either major or minor components of the vegetation, then estimates of relative proportions of biomass were made within each of these categories. An index of absolute biomass was estimated, using the average plot height multiplied by percent cover (hereafter referred to as 'biomass index'). Biomass index was examined at two levels: firstly, at a large scale incorporating all vegetation categories as total biomass; and secondly, at a finer scale, examining the seven plant categories individually. Values for the latter were calculated by multiplying the total biomass index by the proportional biomass estimated visually for each plant category. Although monitoring biomass index is a relatively crude measure for detecting changes in vegetation composition over time, it was considered more meaningful than using changes in plant category cover, as height could increase while cover remained unchanged, and this

change in biomass would be undetected. This observation was confirmed by plotting cover against biomass index data for all quadrats over the entire study period.

10.2.2 Data analysis

Within treatments, plots were used as the unit of replication during analyses. Data were analysed using repeated measures ANOVA's using PROC GLM in SAS (SAS 1989). The dependent variables were height, percent cover and biomass index. The independent variables were treatment (between subject factor) and time (within subject factor). Dependent variables were checked for normal distribution and heteroscedasticity of variance using Proc Univariate in SAS (SAS 1989, Zar 1996). Skewed height and biomass index data were normalised using log transformations. Percent cover data were normalised using a square-root arcsine transformation. Tests for sphericity applied to orthogonal components were used to determine the appropriate analyses for the within-subjects factor (time) and the interaction involving this factor (time*treatment). If the test for sphericity was not significant, an unadjusted ANOVA was used. If the test for sphericity was quite significant ($0.05 < P < 0.0001$), the univariate ANOVA was used with a G-G adjusted P value (SAS 1989). If the test for sphericity was very significant ($P < 0.00001$), a MANOVA with Pillai's Trace was used (Rao 1998, Johnson and Field 1993). Where repeated measures tests showed significant results, multiple comparison tests (with Tukey's adjusted P value) were used to determine differences between plot types, within several individual sampling periods of interest.

To quantify the biomass removed by foraging herbivores, the difference in biomass indices between F-R and UF treatments for grass and forbs over time, and the proportions that these values represented of the available biomass, were graphed.

It should be noted that differences in vegetation biomass and species composition, between fenced and unfenced plots, reflect both mammal feeding as well as plant succession. It was not possible to separate these two factors within the present study.

10.3 Results

10.3.1 Percentage cover of vegetation

Percentage cover of vegetation changed significantly over time but depended upon treatment (Figure 10.1, Table 10.1). In Sampling Period 1 (3-4 months after plot construction), percentage cover was similar between treatments. By Sampling Period 3, percent cover was significantly higher within F-R plots than UF plots ($P=0.002$). Results for F+R plots were similar to the two other treatments. This pattern then remained consistent over time (F-R cf. UF plots: all P values < 0.013).

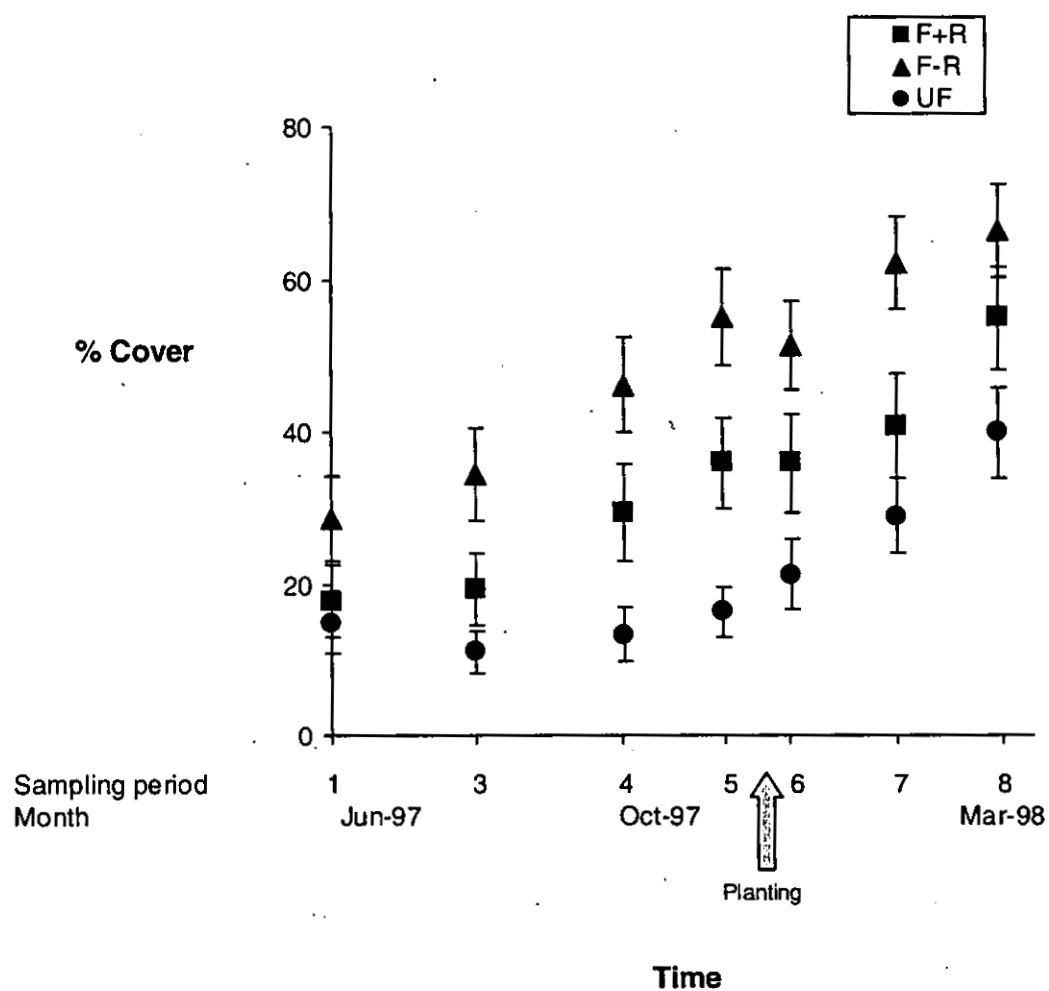


Figure 10.1 Percent cover of vegetation (mean \pm s.e.) within plots on the young plantation over time (F+R=fenced, with roof; F-R=fenced, no roof; and UF=unfenced).

Table 10.1 Results from the repeated measures ANOVA showing effects of time and treatment (plot type) on percent cover of vegetation. Asterisk indicates significant effect.

Source	d.f.	Type III SS	MS	P
Treatment	2	5.8709	2.9354	0.0009*
Time	6	5.5985	0.9331	0.0001*
Time*treatment	12	0.4097	0.0341	0.0056*
Error	252	3.5620	0.0141	

10.3.2 Vegetation height

As with vegetation cover, vegetation height changed significantly over time, depending on treatment (Table 10.2, Figure 10.2). During Sampling Period 1, vegetation was significantly taller within F-R plots than UF and F+R plots ($P=0.0168$ and 0.0386 , respectively). By sampling Period 4 (7-8 months after plot construction), vegetation was significantly higher within both types of fenced plots compared with the unfenced plots ($P<0.013$). This pattern then remained consistent over time (all P values < 0.002). The height of vegetation within fenced treatments increased dramatically between Sampling Periods 6 (December) and 7 (February), associated with warmer weather.

Table 10.2 Results from the repeated measures ANOVA showing effects of time and treatment (plot type) on vegetation height. Asterisk indicates significant effect.

Source	d.f.	Type III SS	MS	P
Treatment	2	14.0452	7.0226	0.0001*
Time	6	41.2086	6.8681	0.0001*
Time*treatment	12	1.8758	0.1563	0.0001*
Error	252	4.0192	0.0159	

10.3.3 Percent cover vs. biomass indices

The scatterplot of percent cover versus biomass index for quadrat data, showed a non-linear relationship (Figure 10.3). For a given percent cover, biomass indices varied substantially, especially at high percent cover values. This reflected the variation in height of vegetation for a given cover. Percent cover underestimates true biomass, while biomass indices will tend to overestimate true biomass because the use of height in the calculation assumes the resulting volume is comprised totally of vegetation, without gaps. Of these two variables, biomass index was considered more likely to represent the range of changes in vegetation occurring over time, and therefore, is used throughout the rest of the study.

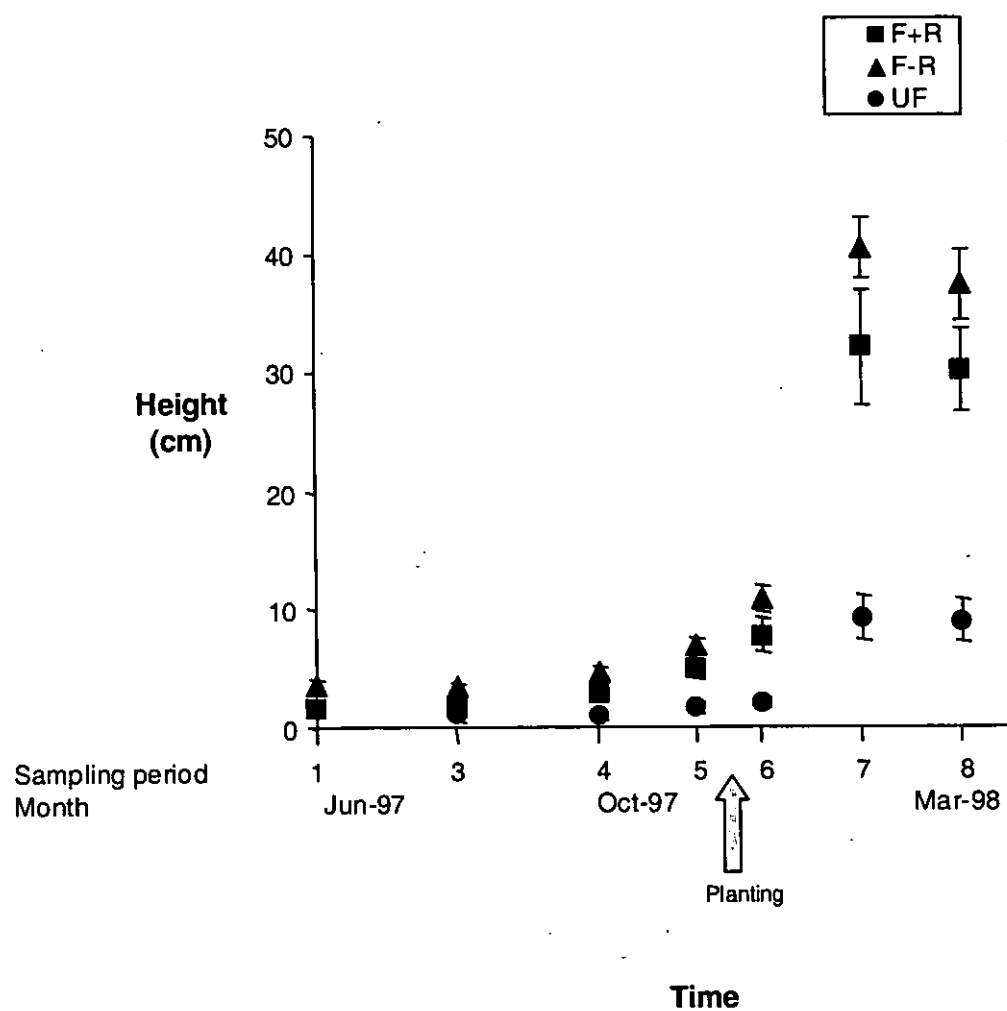


Figure 10.2 Vegetation height (mean \pm s.e.) in plots on the young plantation over time (F+R=fenced, with roof; F-R=fenced, no roof; and UF=unfenced).

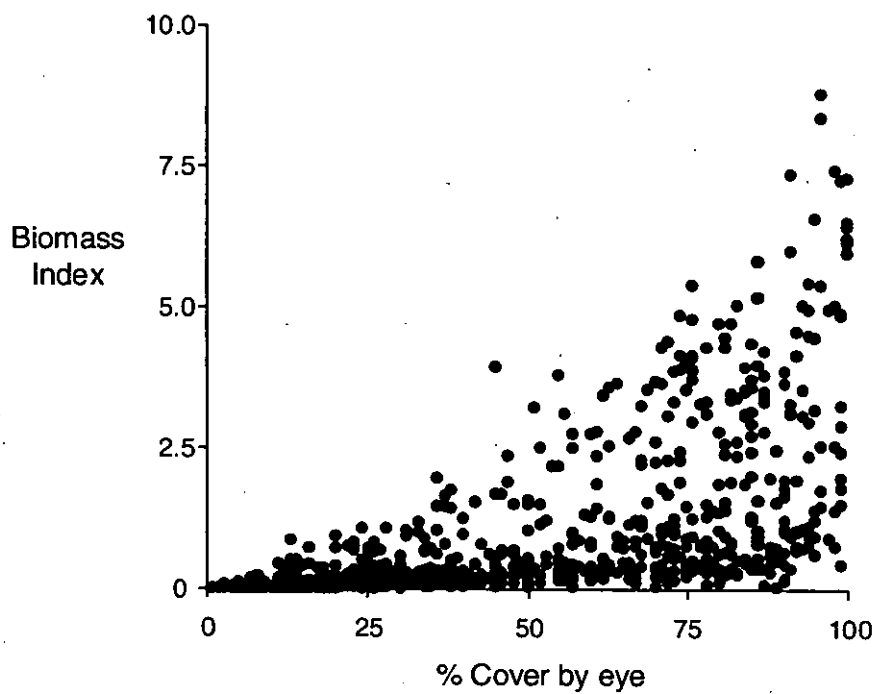


Figure 10.3 Relationship between Biomass Index and % cover for vegetation data, over the entire study period.

10.3.4 Biomass indices for all vegetation

Biomass indices changed significantly over time, depending upon treatment (Tables 10.3 and 10.4, Figure 10.4). During Sampling Period 1, biomass indices were similar between treatments. By Sampling Period 4, the biomass index was significantly higher within both types of fenced plots than unfenced plots (both P values <0.03). As with height, this pattern then remained consistent over time (all P values <0.04), and a dramatic increase in fenced plot biomass took place between Sampling Periods 6 and 7.

Table 10.3 Results from the repeated measures ANOVA for the between subjects effect of treatment (plot type) on biomass index. Asterisk indicates significant effect.

Source	d.f.	Type III SS	MS	F	P
Treatment	2	13.7861	6.8930	17.16	0.0001*
Error	42	16.8689	0.4016		

Table 10.4 Results from the MANOVA for the within subject effects of time and time*treatment (plot type) on the biomass index. Asterisk indicates significant effects.

Source	d.f.	Pillai's Trace value	F	P
Time	6	0.9392	95.2468	0.0001*
Time*treatment	12	0.6401	2.9809	0.0019*

10.3.5 Biomass indices within plant categories

Histograms of changes in biomass indices of the four major vegetation categories over time are shown in Figure 10.5. Plant categories with biomass indices of less than 0.01 are not shown, as they cannot be seen in the figures and represented a minute fraction of the overall vegetation. All seven plant groups appeared within the three treatments, but abundances varied considerably. Plots were dominated by grasses, followed by forbs, then non-grass monocots and shrubs. The remaining three plant categories (moss/lichen/fungi, trees [mostly native *E. delegatensis* and *Nothofagus cunninghamii* seedlings], and ferns) were present in very low quantities.

There was an enormous difference in plant biomass between treatments (Figure 10.5). Fenced treatments contained considerably greater biomass than the unfenced treatment. Tall grass appeared to dominate fenced plots (Plate 10.1). Grasses within unfenced plots were clipped to the ground as a result of herbivore grazing (Plate 10.1). Difference in biomass indices between F-R and UF treatments for grass and forbs over time, and the proportions that these values represented of the available biomass, are shown in Figure 10.6. These results show that herbivores increased their absolute biomass of grasses between Sampling Periods 1-6, reflecting increased availability. Hence, foraging patterns of herbivores did not decline in response to planting (Figure 10.6).

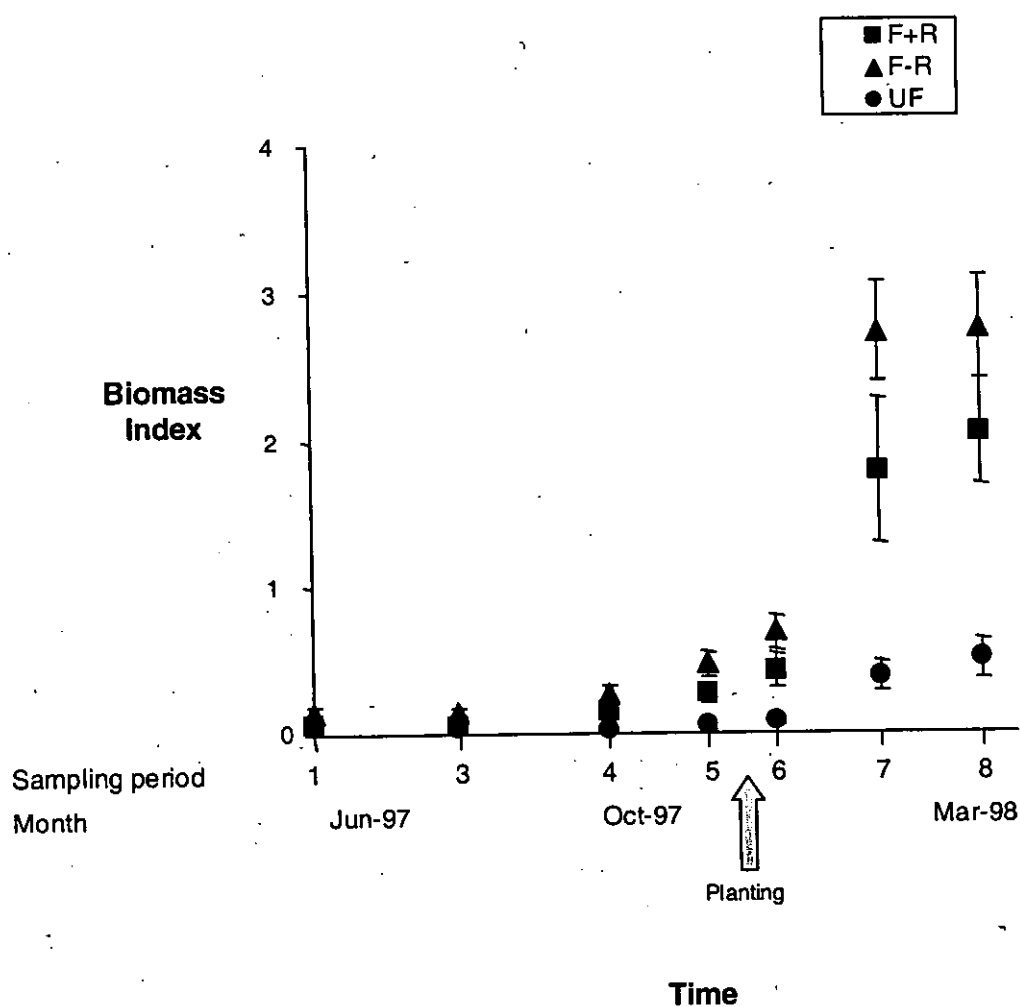


Figure 10.4 Biomass Index (means \pm s.e.) in plots on the young plantation over time (F+R=fenced, with roof; F-R=fenced, no roof; and UF=unfenced plots).

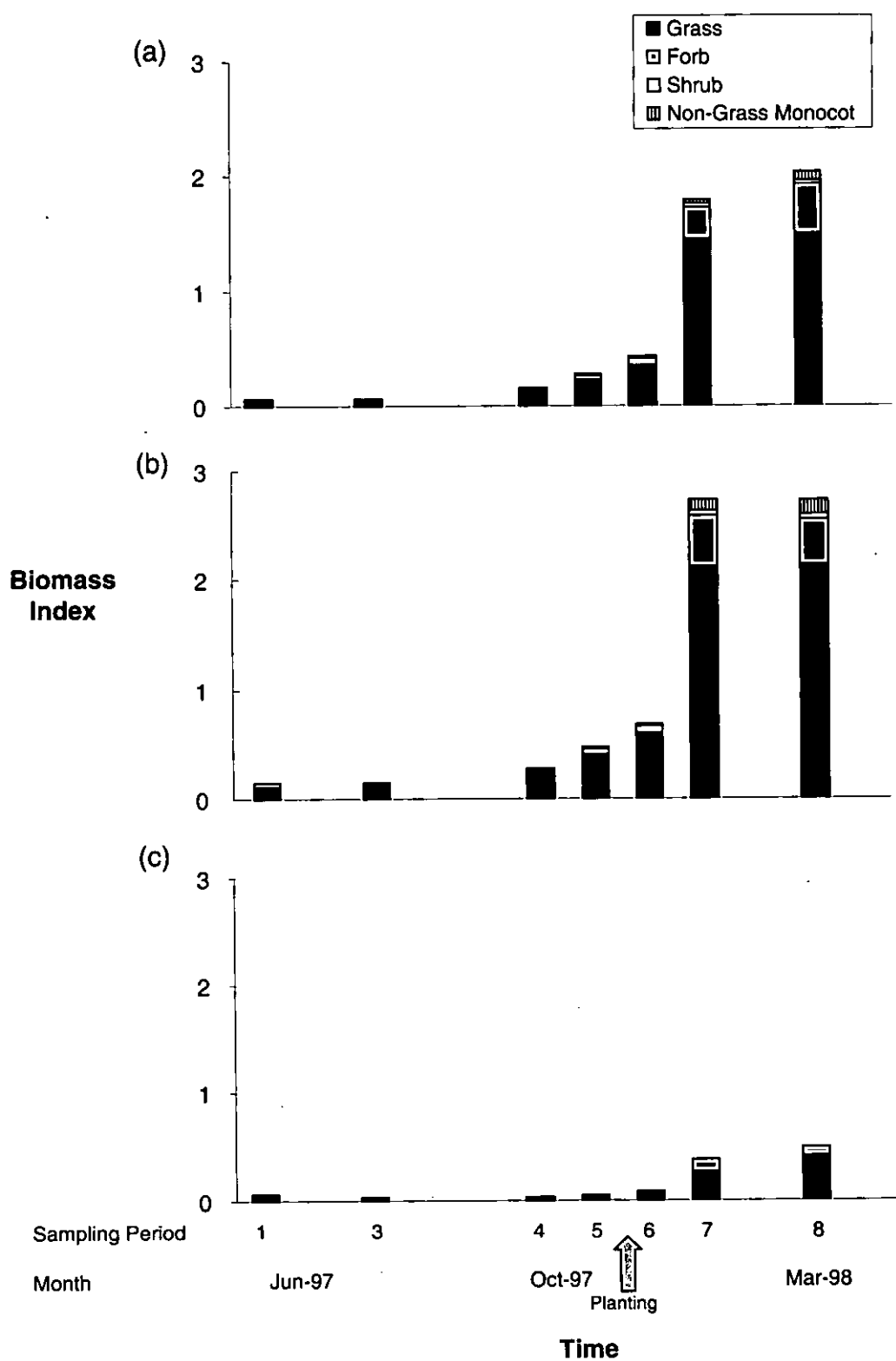


Figure 10.5 The change in biomass indices of the four major vegetation categories within (a) F+R [fenced, with roof]; (b) F-R [fenced, no roof]; and (3) UF [unfenced] plots, over time.

(a) Oct-97



(b) Dec-97



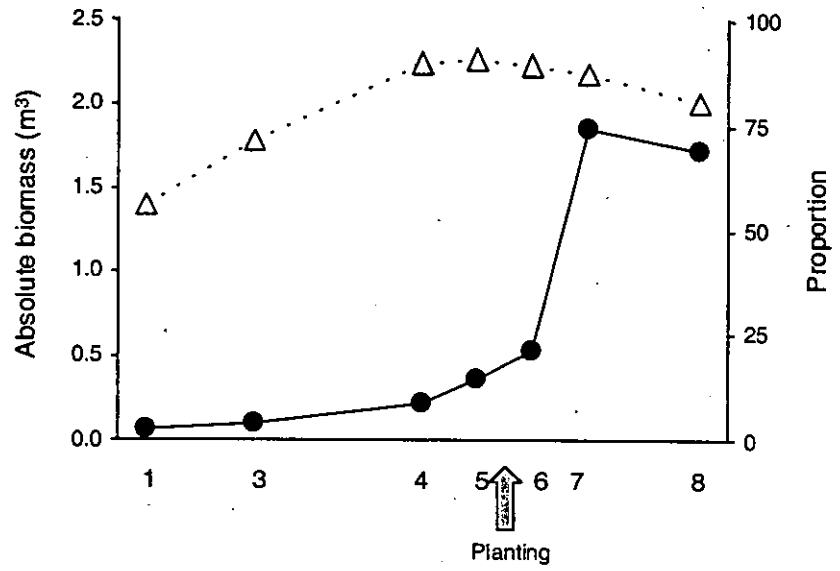
(a) Mar-98



Plate 10.1

High grass cover within fenced plots (left) and the clipped grass within open areas (right) on the young plantation in (a) October 1997, (b) December 1997 [post-planting], and (c) March 1998.

(a) Grasses



(b) Forbs

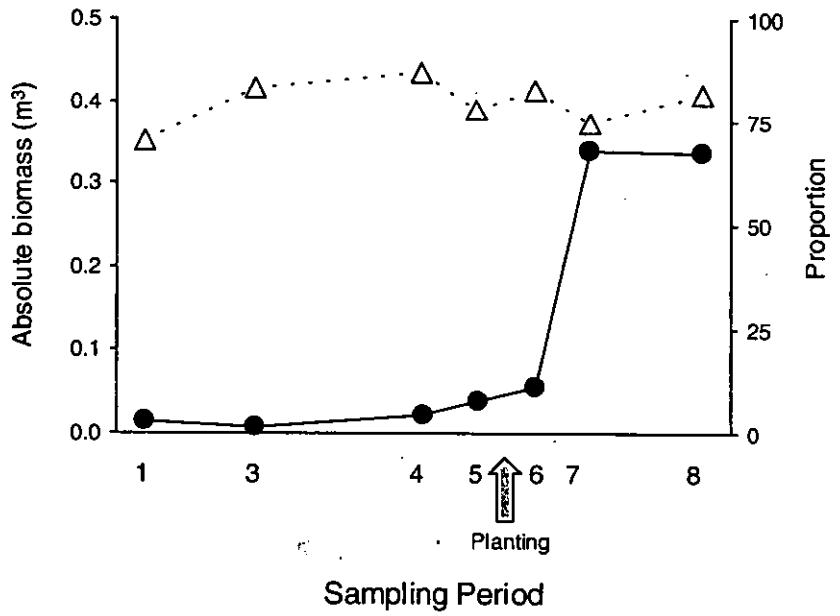


Figure 10.6

Absolute biomass index of (a) grasses and (b) forbs consumed by foraging herbivores on the young plantation (closed circles), and the proportion that this represents of the available biomass index (open triangles).

10.4 Discussion

It was surprising to find plant growth was generally greater in F-R plots than F+R plots, because the only structural difference between the treatments was the fine netting roof on F+R plots. Although the reasons are unclear, the roof may have negatively changed the microclimate within these plots. This could be achieved by netting shading vegetation, and/or lowering ambient air temperature by conducting the cold, particularly over the winter period, and especially overnight. This is supported by a higher extent of frost damage in F+R commercial seedlings, compared with F-R seedlings (Chapter 11).

Both of these scenarios would make conditions less conducive to growth. Henceforth, F-R data were used to represent vegetation available to herbivores, in the absence of browsing.

10.4.1 Effects of herbivores on ground vegetation

Feeding by herbivores had an enormous impact upon vegetation abundance and diversity. Results from Chapter 4 showed that pademelon, wallaby and wombat densities were high on the young plantation, while rabbit density was extremely low. Consequently, the damage reported here is thought to reflect foraging by macropods and wombats. Hence, foraging macropods and wombats appeared to remove most new growth as it became available. The two main plant groups consumed were grasses and forbs, and this result is consistent with other studies (eg. Southwell 1987a, Calaby 1991, Johnson and Rose 1991, McIlroy 1991, Sprent 1997). Additionally, fenced plots contained more non-grass monocots (rushes, orchids and irises), shrub and tree species (native seedlings) than unfenced plots. It is not known whether this reflects animals selective foraging of these plant species, and/or subsequent plant succession. These results, however, support Dickinson and Kirkpatrick's (1986) study where shrubs and graminoids generally grew faster within fenced than unfenced plots in Tasmanian dry eucalypt forest.

Possum browsing appeared to have no impact upon the vegetation. Results are difficult to interpret, however, due to the confounding effect of the netting roof on plant growth within F+R plots. It is also possible that results from F-R plots may have underestimated the impact of possums, if animals avoided these areas. That is, if the effort required to access plots exceeded the benefit of foraging within them, animals may have been disinclined to use them. Results shown within the present study, however, are consistent with low possum densities on the young plantation, reported in Chapter 4.

10.4.2 Ground vegetation before and after planting *E.nitens* seedlings

Patterns of vegetation consumption by herbivores did not change in response to planting. Animals continued to consume almost all of the available grass and forbs. Consequently, these results suggest that grass and forbs were important dietary items both before and after planting.

10.4.3 Management implications

An obvious outcome of this study is that foraging herbivores can substantially reduce weed biomass on plantations: unfenced open areas on the young plantation had the appearance of a mowed lawn, while fenced plots in the same areas, contained thigh-high grass. Herbivore browsing also appeared to delay plant succession and/or consume woody species, which are a major weed problem on many plantations where herbicide has been inadequate (Wilkinson

and Neilsen 1990). The importance of foraging macropods in controlling weeds is presently not recognised. As a result, vertebrate pest control programs, which reduce local macropod populations, are being conducted in isolation of weed management strategies. Hence, I suggest there is scope for a more integrated approach to managing plantations.

Chapter 11

Commercial Seedlings: Impacts of Browsing Herbivores & Other Factors

For herbivores feeding on the young plantation, *Eucalyptus nitens* seedlings represented a minute proportion of the food available within this habitat. Quantifying seedling growth and survival, however, was of great importance to the applied aspects of this project. Consequently, animals' effects on ground vegetation and seedlings have been described separately. Results for ground vegetation were given in Chapter 10, and seedling data are discussed in the present chapter. Due to the applied nature of the seedling data, results will be discussed in relation to productivity of the young plantation.

11.1 Introduction

Tasmanian herbivores are known to damage eucalypt seedlings within commercial plantations during their first few years of growth. The species involved are the native Bennett's wallaby, red-bellied pademelon, brushtail possum and the introduced European rabbit (Gilbert 1961, Cremer 1969, Statham 1983). Browsing damage caused by these species decreases plantation productivity by reducing seedling growth rates, reducing seedling survival and resulting in a change in growth form (e.g. forking), which reduces tree quality (Coleman *et al.* 1997). A key objective of this chapter, therefore, was to quantify the effects local herbivore populations had upon eucalypt seedlings on the young plantation.

A second objective of this chapter was to examine the spatial distribution of browsing damage to seedlings. An edge-effect for pademelon browsing damage has been reported by Statham (1983) and Coleman *et al.* (1997). This edge-effect is the result of animals rarely venturing beyond 100 m of forest edge (Statham 1983, Coleman *et al.* 1997); reflecting their predator avoidance strategy of seeking shelter within dense vegetation when under threat. It was not possible to isolate pademelon browsing damage from that of other herbivores at this site, but damage patterns reported here will be related to results from the radio-tracking study (Chapters 7 and 8) in the final discussion (Chapter 12).

In addition to browsing mammal damage, seedlings at this site were also damaged by other agents, such as insects, frost and shock in response to planting. Consequently, these types of damage were also quantified. The specific aims of this chapter were to:

- (1) describe seedling mortality, extent and severity of browsing, and growth rate over time,
- (2) partition seedling defoliation between browsing mammals and insects,
- (3) partition mammal browsing to seedlings between macropods and possums,
- (4) examine the spatial distribution of mammal browsing to seedlings.

How seedling mortality and growth rate relate to herbivore population densities, and the availability of other vegetation on the young plantation, will be discussed in Chapter 12.

11.2 Materials and Methods

11.2.1 Monitoring of seedlings

Browsing damage to seedlings was thought to be carried out by wallabies, pademelons and possums. Although wombats were considered a major herbivore within this habitat (Chapter 4), they are grazers feeding primarily on native grasses (McIlroy 1991), and hence, were

considered highly unlikely to browse seedlings. Additionally, rabbits were present in such low densities (Chapter 4) that their contribution to seedling damage was considered minute. Consequently, hereafter 'herbivores' refers to macropods and/or possums.

Seedlings were monitored using two methods: plots and transects.

Plots

Three plot types (treatments) were used: (1) unfenced [UF]; (2) fenced, no roof [F-R]; and (3) fenced, with roof [F+R], as described in Chapter 10. Briefly, there were 6 seedlings per plot (3 trees in 2 neighbouring tree-rows), and 15 plots of each treatment. UF plots were used to examine the effect of all mammalian herbivores on seedlings, F-R plots were used to examine the effect of possum browsing on seedlings, and F+R plots were used to examine the absence of mammalian herbivores on seedlings.

Transects

Six transects that ran from one side of the young plantation to the other, were used to investigate the spatial distribution of seedlings damaged by browsing mammals. Transect lines monitored between 60 – 137 seedlings (average 84 seedlings), a total of 502.

The secondary aim for this data set was to validate results collected from the UF plots. Only 45 seedlings, 0.005 % of total seedlings, were monitored within these plots. The seedlings monitored within the six transects, however, represented 2.5 % of all seedlings. The latter sample size reflected those used by Marsh (1998) and Bulinski (1999b) to monitor browsing damage to seedlings over time.

11.2.2 Forestry operations

Eucalyptus nitens seedlings (approximately 20 cm high) were planted on the young plantation on November 27, 1997. Approximately 1100 seedlings were planted per hectare. Seedlings were container-stock, meaning they were grown in the nursery in pots and transferred directly into the field. Due to high weed cover within the fenced plots (constructed eight months before planting), small areas (approximately 8 cm²) usually had to be cleared before the seedlings could be planted.

Seedlings within unfenced plots, and the rest of the young plantation, were planted using a planting tube. This device is essentially a pipe through which seedlings travel from the forester's hip to the ground. In contrast, seedlings within the fenced plots were planted by hand because the roof inhibited foresters from using a planting tube. The planting technique, consistent with using a planting tube, was mimicked as much as possible (for example, the cotyledons were buried). Different planting methods were, therefore, not considered to influence subsequent seedling growth or survival.

Vegetation surrounding seedlings was given a spot application of herbicide one week after planting (see Chapter 2). This operation had been scheduled to occur before planting, but due to logistical constraints (wet conditions and the availability of ground crew), this was not possible. The herbiciding process involved covering a seedling with a plastic container, and spraying Roundup® on all vegetation within a 0.5 m radius. Within fenced plots, tall vegetation, which could potentially be blown onto a seedling, was clipped before spraying.

Seedlings were fertilised six weeks after planting. Approximately 100 g of Pivot® (18N:20P:0K) slow release fertiliser granules were placed 20 cm away from seedling stems.

Seedlings that appeared dead were also fertilised in case they were only suffering from transplant shock. Shock damaged seedlings can occasionally recover (Calton Frame, *pers. comm.*), but appear dead after losing their leaves.

11.2.3 Data collection

Data were collected for plot seedlings on six occasions: (i) on the day of planting; (ii) 2.5 weeks after planting; (iii) 9 weeks; (iv) 15 weeks; (v) 6 months and (vi) 8 months after planting. Seedlings within transects were monitored during periods (ii)-(v). Sampling intervals i-iv were the same as Sampling Periods 5-8 for the spotlighting and radio-tracking studies. Sampling Period 9 followed a free-feeding forestry operation, in preparation of a 1080 poisoning operation on the young plantation (see Chapter 5). Sampling Period 10 immediately followed the 1080 poisoning operation (Chapter 5). Consequently, data presented in this chapter extends beyond the study period for ground vegetation on the young plantation (Chapter 10).

Each sampling period, individual seedlings were classified as live or dead. 'Dead' seedlings included plants that had been pulled out of the ground, or were missing. Seedlings that appeared dead from shock damage were recorded as a separate group, but were included in total mortality. No further information was collected for dead seedlings. Seedlings that had 'died from shock' were checked for recovery each sampling period.

Live seedlings were measured to monitor growth, and damage was assessed to quantify effects of browsers. Seedling height (± 0.5 cm) was measured from the base of the stem to the highest point of stem or foliage in the vertical plane. Damage to seedlings were identified to causal agent: mammal browsing, insect damage, transplant shock damage to planting, and frost damage. A description of each is given below.

- (1) Mammal damage - typically foliage is partially or wholly sheared off, leaving straight, rather than curved edges. Damage can also affect leaf stem, shoots and branches. Usually foliage does not accumulate at the base of the seedling.
- (2) Insect damage - usually confined to the periphery of leaves, characterised by nibbled leaf edges. Damage can also consist of holes in the leaf blade and damage around the leaf skeleton. Stems and branches are not typically damaged. Usually foliage has not accumulated at the base of the seedling.
- (3) Transplant shock damage - hereafter referred to as 'shock damage'. Some or all of the seedling's leaves may be brown, brittle and dead. Dead leaves can accumulate at the base of the seedling.
- (4) Frost damage - sections of seedling's leaves are black and dead.

Distinguishing mammal damage from shock and frost damage was straightforward. It is possible, however, to occasionally mistake insect damage for mammal damage, if seedlings have been heavily browsed to the point where leaves have been damaged down to the leaf stem. During the present study, such damage was generally assumed to be caused by mammals. Conversely, due to the nature of the different bite marks, there would have been few instances where mammal browsing was mistakenly identified as insect damage. Thus, the damage estimates reported here may represent a slight overestimation of mammal damage.

Damage was reported in two ways: extent and severity. Damage extent was defined as the percentage of total seedlings damaged by a given agent. Damage severity was a visual estimate of the percentage of a seedling's biomass that had been removed through mammal browsing, insect browsing or frost. Damage severity was recorded for all live seedlings, and was quantified using a seven-point scale, modified from Marsh (1998) (Table 11.1). Analysis of damage severity was then performed only on seedlings with damage (score >0). To ensure

that this relative measure remained consistent throughout the study, all scoring was conducted by K.le Mar.

The number of leaders was also recorded for each seedling, each sampling period. This enabled changes in seedling growth form to be identified over time.

Table 11.1 The seven-point scale used to quantify severity of damage to seedlings, modified from Marsh (1998).

Score	Estimate of the biomass removed (% of total)	Observation
0	0	no damage
1	1-9	little damage
2	10-29	up to 1/3 of leaves gone
3	30-49	between 1/3 and 1/2 gone
4	50-69	between 1/2 and about 2/3 gone
5	70-89	between 2/3 and most gone
6	90-100	most or all gone (includes no whole leaves)

11.2.4 Data analysis

Plots

Data were analysed as described in Chapter 10. For the UF treatment alone, the proportion of seedlings damaged by browsing mammals versus insects was tested statistically for differences, over time, using Signed Rank tests. Where results were significant, pair-wise comparisons between treatments were made using a Bonferonni adjusted α value of 0.01.

Transects

The distribution of damaged seedlings within transects was mapped to enable a visual assessment of animals' foraging behaviour in relation to the boundary of the young plantation. Maps were constructed using the ArcView® GIS described in Chapter 2.

It should be noted that with each subsequent sampling period, seedlings that had died were no longer monitored in relation to damage or growth. Consequently, sample size of live seedlings decreased over time. This means some care is needed in interpreting results. For example, a particular type of damage may appear to decline over time, when in fact the previously affected seedlings had since died and been removed from the data set. Interpretation of live seedling data, therefore, requires an appreciation of the change in percent mortality over the study period.

11.3 Results

11.3.1 Seedlings within plots

Mortality

The mortality of seedlings within the three treatments, is shown in Figure 11.1. Results for the repeated measures MANOVA showed a significant time and time/treatment effect (Table 11.2). Results from multiple comparison tests are shown in Figure 11.1. Mortality was low within the three treatments 2.5 weeks after planting. By week 9, mortality had increased to around 25 % in UF and F-R plots. Most of this mortality was due to shock (Figure 11.2). Mortality was also significantly higher within UF plots than F+R plots ($P=0.0037$), and this pattern remained for the duration of the study. Eight months after planting, seedling mortality within fenced plots was unchanged, but mortality had increased to 48 % within the UF treatment.

Table 11.2 Results for the repeated-measures MANOVA for seedling mortality.

Source	d.f.	Pillai's Trace Value	<i>P</i>
Time	4	0.7171	0.0001
Time*treatment	8	0.4968	0.0026

Damage

Results of mammal browsing damage to live seedlings, over time, are shown in Figures 11.3a-b. Within UF plots (accessible to all herbivores), damage from browsing mammals increased over time (Figure 11.3a). Damage was moderate 2.5 weeks after planting (14 % of seedlings affected; 11 % biomass removed). This increased to 24 % of seedlings affected at 15 weeks, and peaked six months after planting (51 % affected; 74 % of biomass removed). Eight months after planting, values were relatively consistent with the previous sampling period, although slight declines reflected a proportion of live seedlings dying (Figure 11.1), hence moving out of this data set.

Within F-R plots, possum browsing damage was low (Figure 11.3b). Damage was not recorded until 15 weeks after planting, when a single plot was visited. Possum damage peaked six months after planting when four of the 15 plots were visited, however, damage was neither extensive nor severe (9 % of seedlings affected; 2 % of biomass removed). Eight months after planting, mammal browsing damage largely comprised data recorded during the previous sampling period, because the same individual seedlings showed damage. No mammal damage was observed in F+R plots (Figure 11.3c), as expected.

Results of insect damage to live seedlings within the three treatments, over time, are shown in Figure 11.3. Within all three treatments, 40-50 % of seedlings showed insect damage 2.5 weeks after planting, and damage was relatively severe (Figures 11.3a-c). Insect damage peaked 15 weeks after planting (UF: 60 % seedlings affected; F-R: 67 %; and F+R: 88 %), and damage was severe (UF: 47 % biomass removed; F-R: 25 %; and F+R: 28 %). Six months after planting, extent of damage to seedlings within fenced plots was still high (F-R: 66 % and F+R: 86 %), but had declined within unfenced plots (11 %) as mammal damage began to predominate. Eight months after planting, insect damage values were consistent with the previous sampling period.

Signed Rank tests carried out on UF data showed that patterns of mammal and insect browsing damage to seedlings differed significantly at three time periods. Nine weeks after planting, the extent and severity of insect damage was greater than for mammal browsing

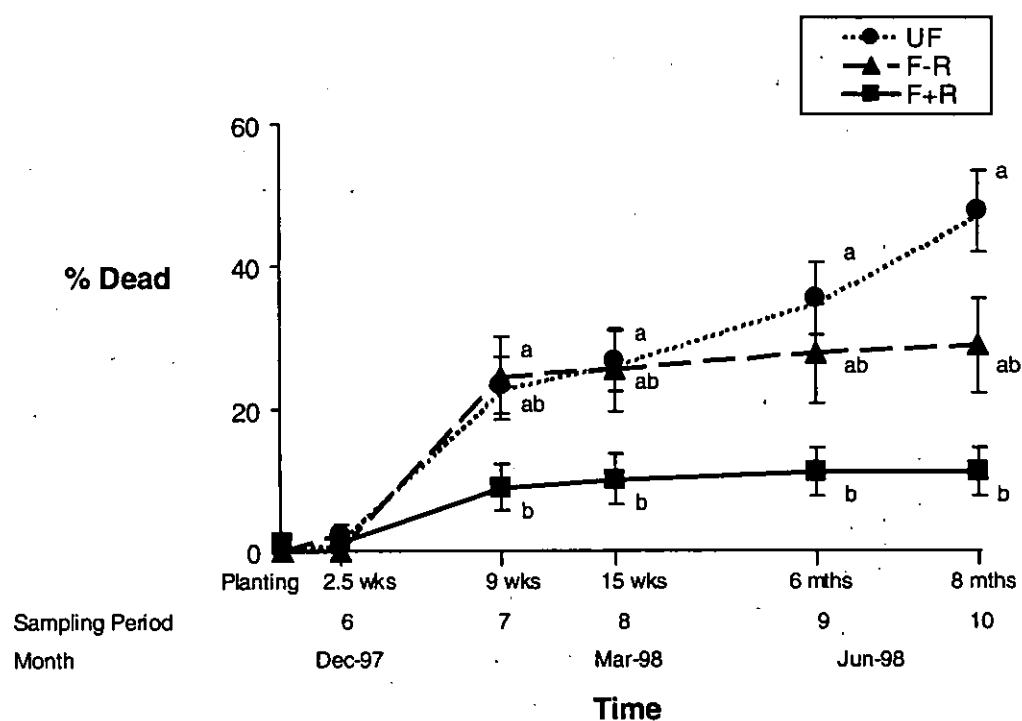


Figure 11.1 The proportion (mean \pm s.e. bars) of dead seedlings (total mortality) within the three treatments (UF=unfenced; F-R=fenced, no roof; F+R=fenced, with roof), over time. Letters show significant differences between treatments ($\alpha=0.01$) within each sampling period.

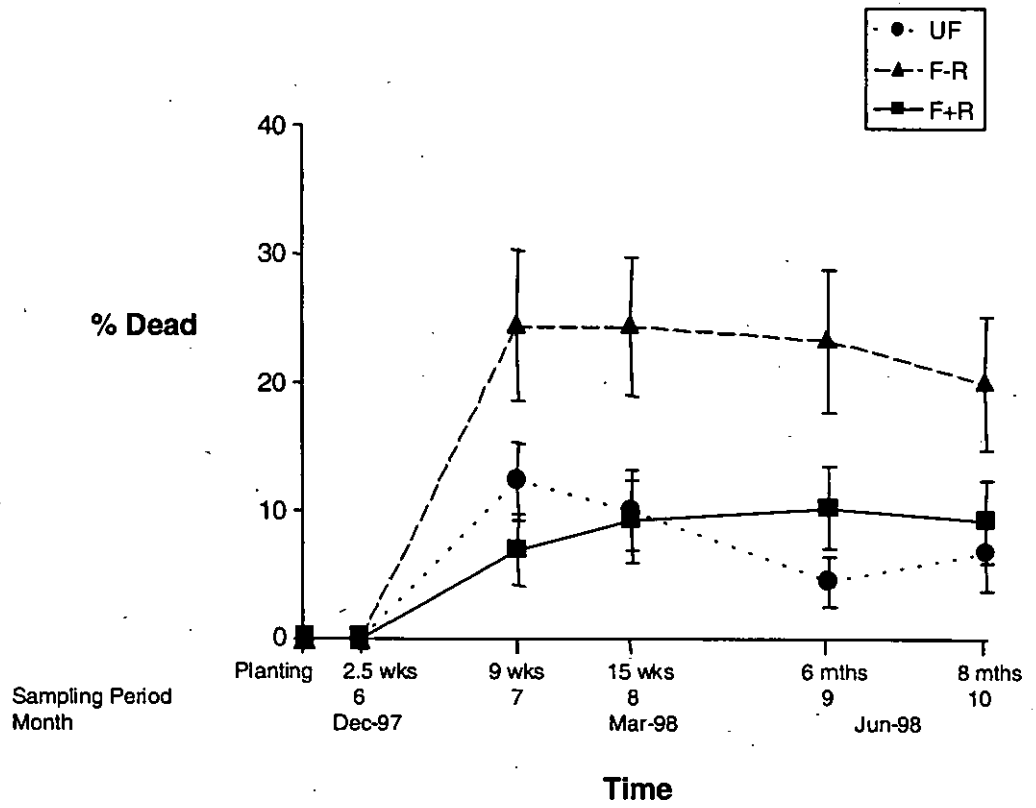


Figure 11.2 The proportion (mean \pm s.e.) of all seedlings, killed by shock damage, over time.

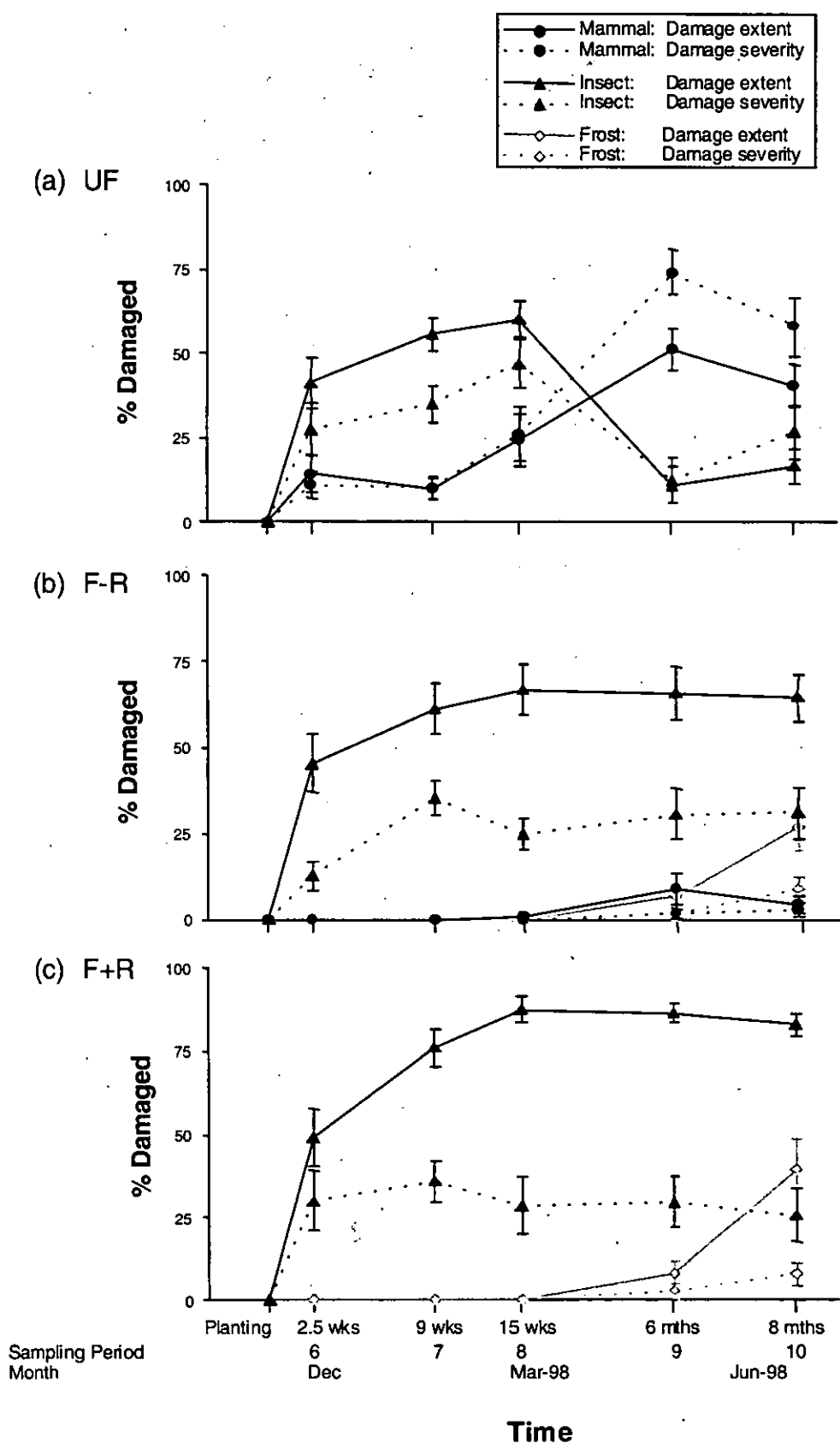


Figure 11.3 Damage extent and severity (mean \pm s.e.) for (live) seedlings within (a) UF unfenced, (b) F-R fenced, no roof, and (c) F+R fenced, with roof plots, over time.

($S=52.5$, $P=0.0001$; $S=49.5$, $P=0.0006$, respectively). Fifteen weeks after planting, the extent of insect browsing was still greater than mammal browsing ($S=37.5$, $P=0.0066$), but severity was similar to that caused by browsing mammals. Six months after planting this pattern was reversed, with extent and severity of mammal browsing significantly greater than that of insect browsing ($S=-44.5$, $P=0.0035$; $S=-49.5$, $P=0.0006$, respectively).

Frost damage to live seedlings within the three treatments is shown in Figure 11.3. Within fenced plots, frost damage was first recorded in May (winter), six months after planting. The extent of damage to seedlings was relatively high (F-R: 27 % and F+R: 39 %) but damage was not severe (F-R: 9 % biomass removed and F+R: 8 %). No frost damage was reported for seedlings within unfenced plots.

Growth

Average seedling height at planting was 15 cm (range: 5-26 cm, $n=90$). Results of height of live seedlings over time, are shown in Figure 11.4. Seedlings within UF plots did not grow. From 2.5 weeks after planting, UF seedlings were significantly shorter than those within fenced plots (all P values <0.0038). Seedlings within the two fenced treatments did grow and growth rate was indiscernible between fence treatments. Seedling height in the fenced treatments doubled within six months.

Change in growth form

The proportion of live seedlings with multiple leaders generally increased over time (Figure 11.5). Within UF plots, the sudden decline in seedlings with multiple leaders, between 15 weeks and 6 months after planting, was caused by the deaths of previously monitored seedlings.

11.3.2 Seedlings within transects

Mortality, damage and growth results for seedlings within the six transects were relatively consistent with the data obtained from the UF plots. Mortality was slightly higher for transect seedlings, but similar proportions had died from shock (Figure 11.6). The proportion of live seedlings with mammal damage was indiscernible between transect and plot groups (Figure 11.7). The extent of insect damage was lower for seedlings in transects compared with UF plots, but data sets followed the same trend over time (Figure 11.7). Seedlings within transects and UF plots also grew at a similar rate (Figure 11.8).

The spatial distribution of damaged seedlings within transects is shown in Figure 11.9. It appears that mammals browsed seedlings irrespective of their location. (Severity of damage to these seedlings is not shown).

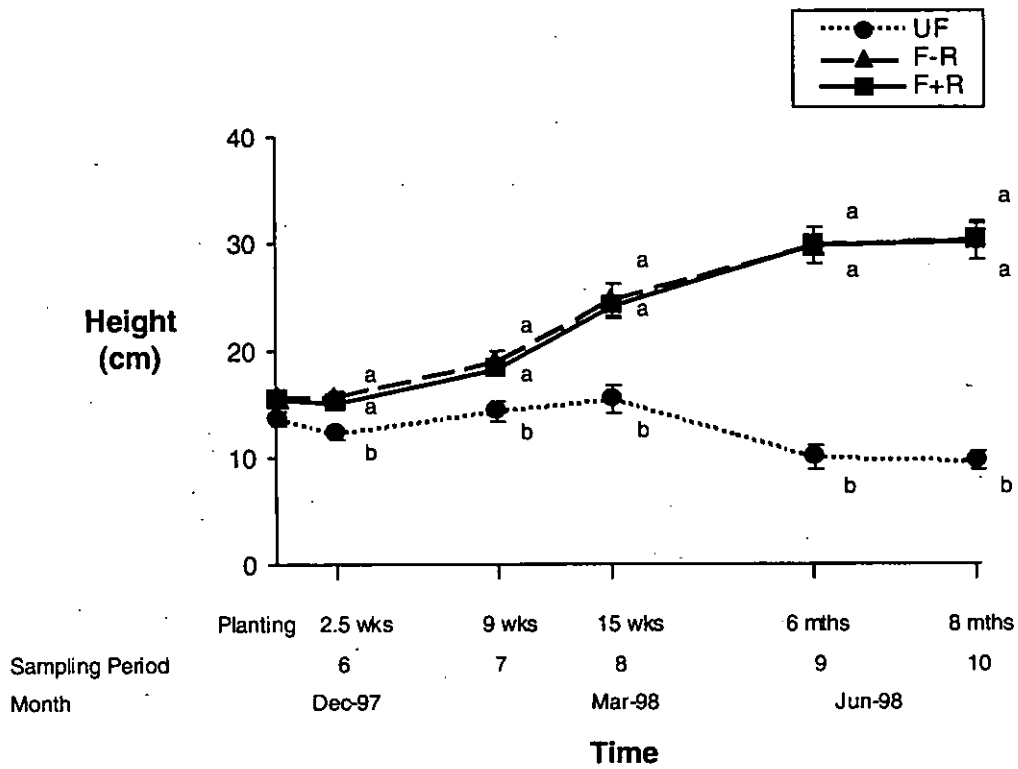


Figure 11.4 Height of live seedlings (means \pm s.e.) within the three treatments (UF=unfenced; F-R=fenced, no roof; and F+R=fenced, with roof, over time). Letters show significant differences between treatments ($\alpha=0.01$) within each sampling period.

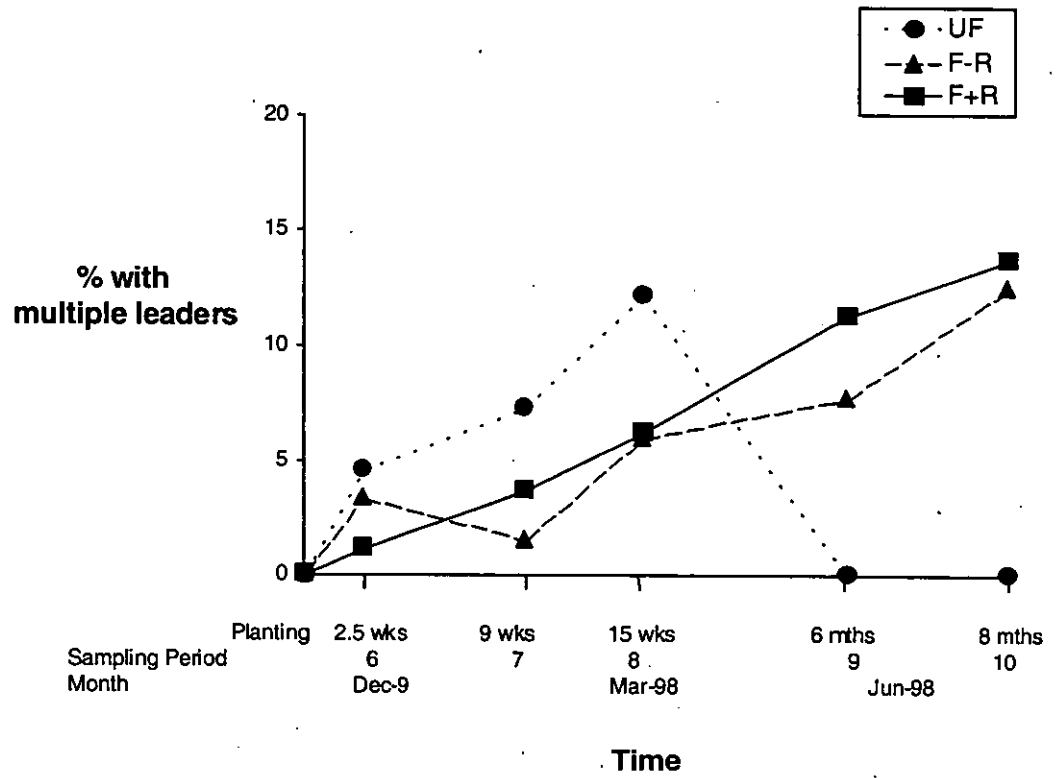


Figure 11.5 The proportion of live seedlings with multiple leaders within UF (unfenced), F-R (fenced, no roof), and F+R (fenced, with roof) treatments, over time.

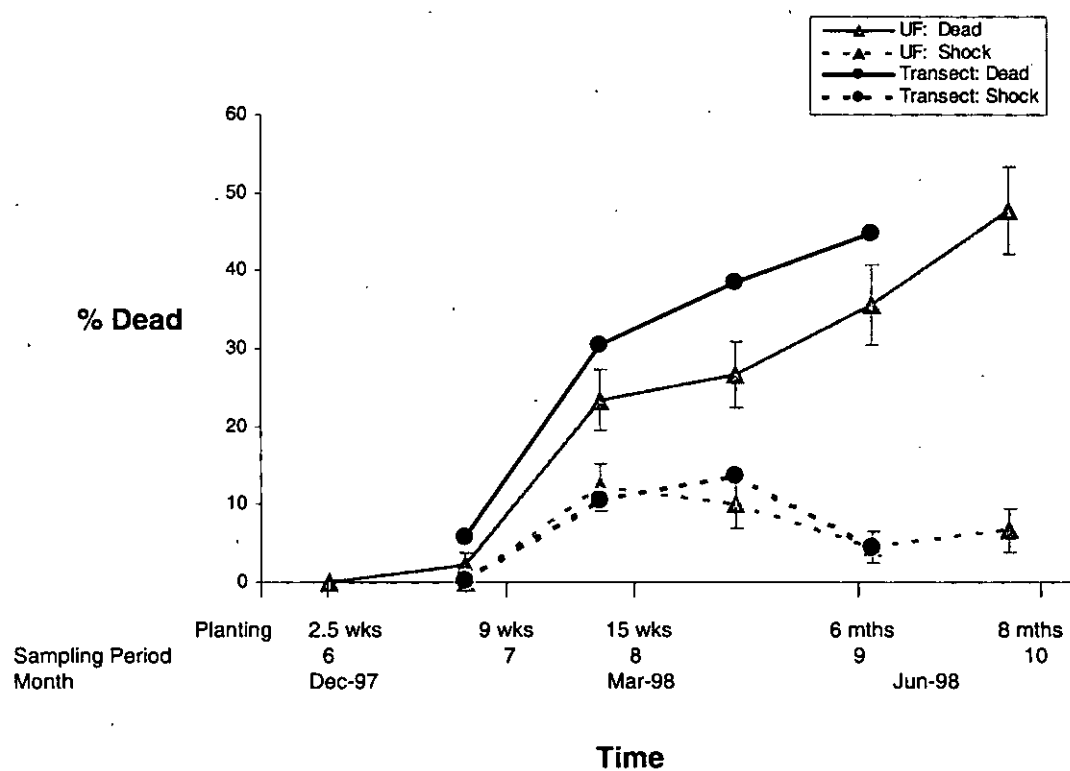


Figure 11.6 A comparison of seedling mortality and death from shock between plants within transects and UF (unfenced) plots (mean \pm s.e.) on the young plantation, over time.

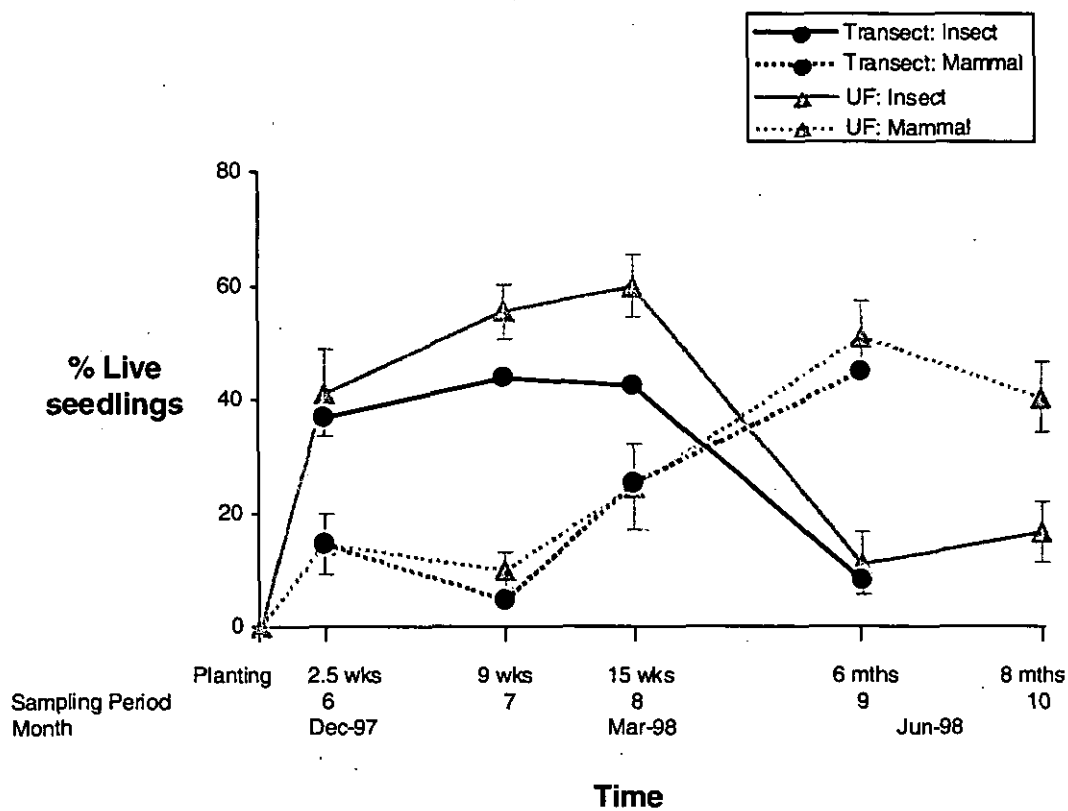


Figure 11.7 A comparison of extent (% of live seedlings) with damage caused by mammals and insects within transects and UF (unfenced) plots (mean \pm s.e.) on the young plantation, over time.

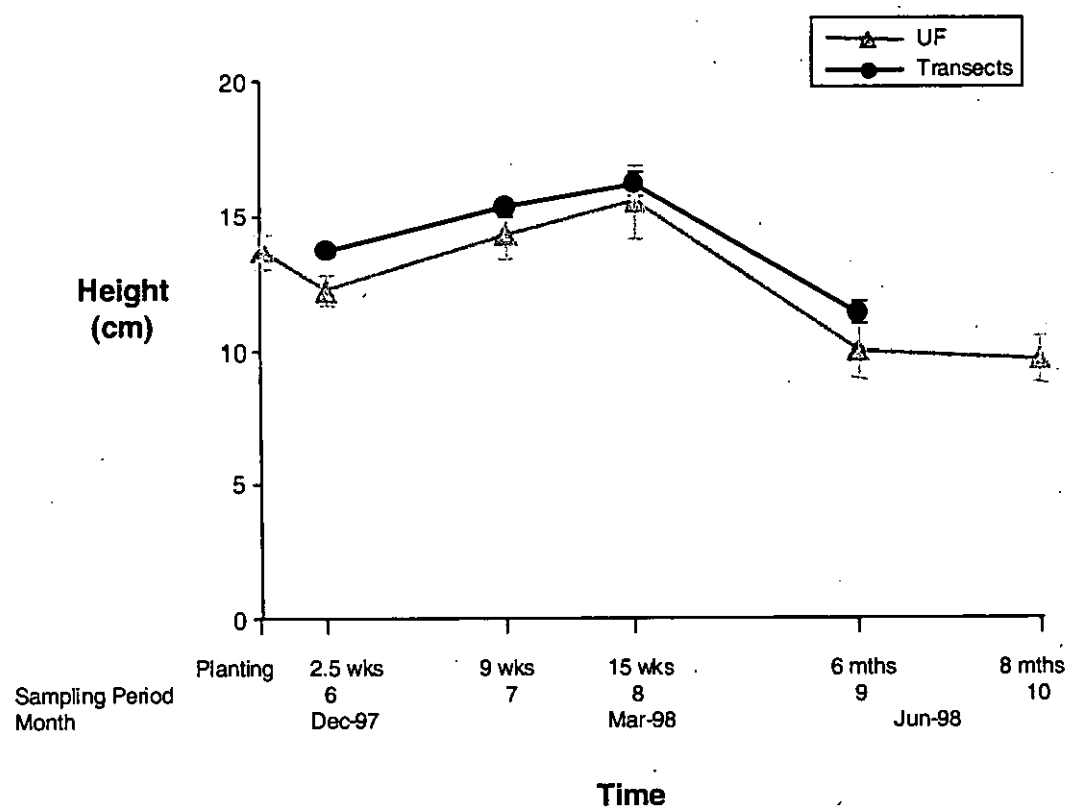
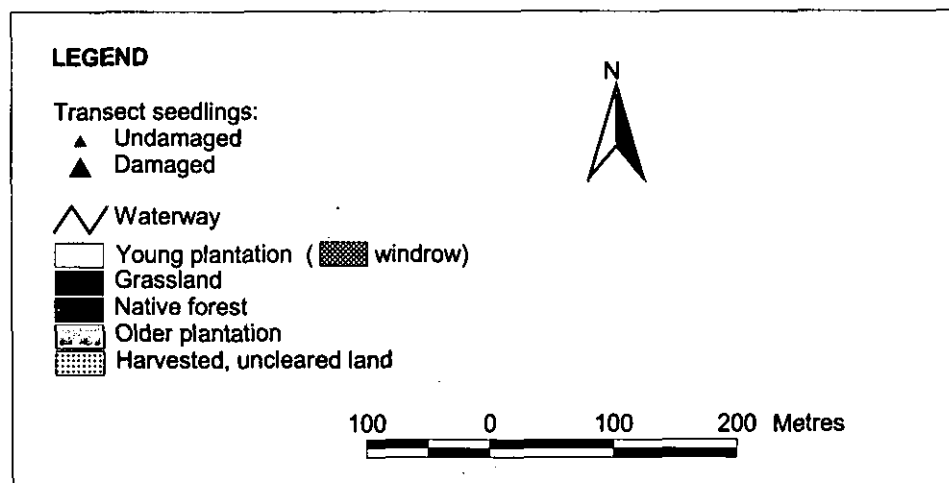


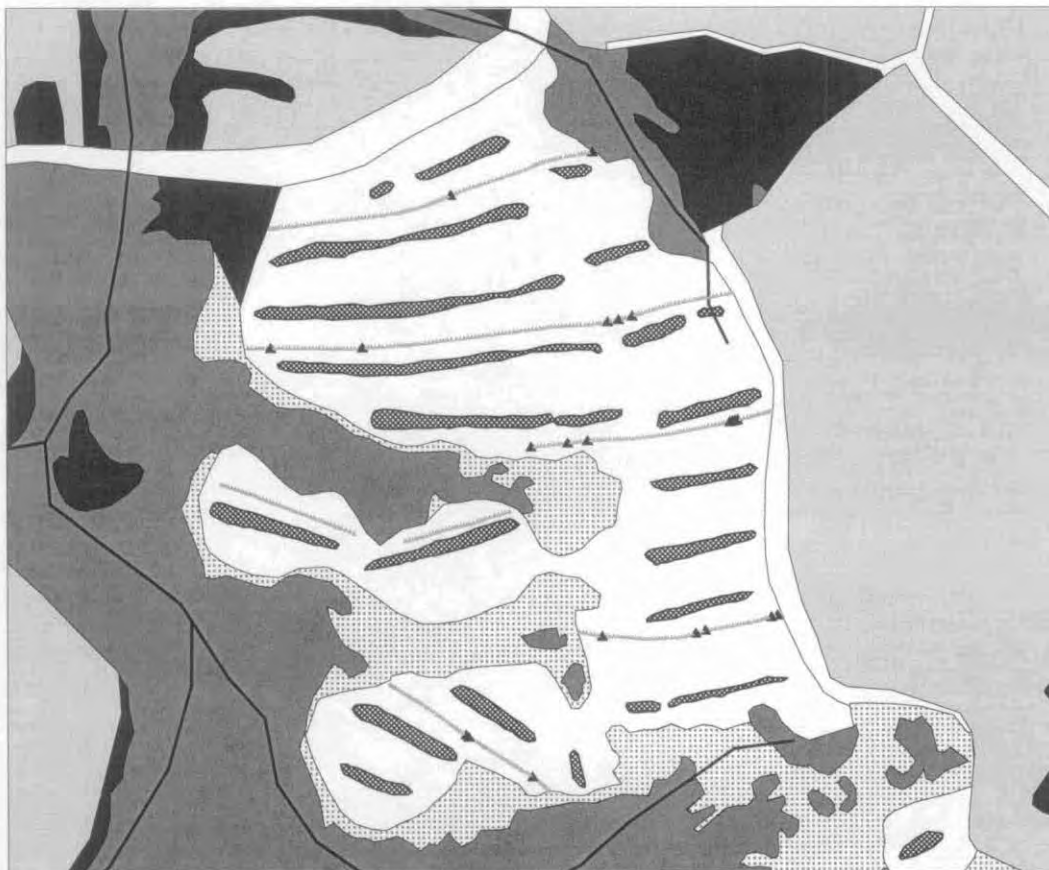
Figure 11.8 A comparison of live seedling height between plants within transects and UF (unfenced) plots (mean \pm s.e.) on the young plantation, over time.

Figure 11.9 The distribution of mammal damaged seedlings along the six transect lines at (a) 2.5 weeks, (b) 9 weeks, (c) 15 weeks, and (d) 6 months after planting. The legend for the maps that follow, is given below.





(a) mammal damaged seedlings 2.5 weeks after planting



(b) mammal damaged seedlings 9 weeks after planting



(c) mammal damaged seedlings 15 weeks after planting



(d) mammal damaged seedlings 6 months after planting

11.4 Discussion

Despite the small sample size of seedlings within UF plots (0.005 % of total), results were representative of seedlings within transects (2.5 % of total). Consequently, we can be confident about results obtained from the plot data.

11.4.1 Mortality

A relatively high proportion of seedlings died from shock damage following planting. It is not known what event triggered seedlings to go into shock, but it probably related to an event that took place in the nursery, during seedling development (Calton Frame, *pers. comm.*). Shock damaged seedlings were not confined to the young plantation; similar damage was observed within other plantations at Surrey Hills, presumably planted with the same stock. It is unclear why mortality from shock damage varied between plot treatments.

Another cause of seedling death was herbicide. A quick survey of dead seedlings on the open plantation suggested that herbiciding after planting killed approximately 0.5 % of plants (Calton Frame *pers. comm.*). These seedlings could be identified by their dead foliage and established root system, which clearly indicated that they had died a couple of weeks after planting. This coincided with herbiciding.

Seedling mortality within unfenced plots, six and eight months after planting, was also attributed to mammal browsing damage (see below).

11.4.2 Mammal browsing

Results from the comparison of fenced and unfenced plots showed that mammal browsing damage reduced seedling survival (40 %) and seedling height (68 %). These results are consistent with other studies (Gilbert 1961, Cremer 1969, Statham 1983, Montague 1996, Bulinski 1999a).

At this site, macropod browsing was largely confined to the winter period, several months after seedlings were planted. Visual observations of plots at this time showed that very little forage was available to herbivores because most of the grasses had died off (K. le Mar *pers. obs.*). Hence, increased mammal browsing to seedlings at the onset of winter may be due to a change in the availability of suitable forage at this time. This result is consistent with findings by Gilbert (1961), Cremer (1969), Statham (1983), Montague (1996) and Coleman *et al.* (1997).

Possum browsing appeared to have little impact upon seedlings at this site, and was much lower than browsing by macropods. A previous study (Gilbert 1961) suggested that the relative importance of possum browsing is site dependent. Results shown in the present chapter are consistent with low possum density at this site (Chapter 4). It is possible, however, that the impact of possums is underestimated if animals avoided F-R plots, as discussed in Chapter 10.

The spatial distribution of mammal browsing to seedlings showed no edge-effect; mammals browsed seedlings throughout the plantation. It is possible, however, that patterns of mammal browsing severity may have existed, although this was not examined within the present study. This aside, I suggest that the presence of shelter, or windrows, within plantations will determine whether or not an edge-effect is detected. Statham (1983) reported an edge-effect of seedling damage within a *Eucalyptus delegatensis* plantation that lacked windrows, while Bulinski and McArthur (2000) found no edge-effect in 15 plantations located around Tasmania, all of which had windrows. In the absence of windrows, pademelons rely on neighbouring habitat for shelter from predators, and 100 m from the

forest edge appears to represent the safety 'cut-off' point (Statham 1983, Coleman *et al.* 1997). In the presence of windrows, however, pademelons can access shelter within the interior of the plantation, which I suggest enables them to utilise the entire site.

Temporal variation in browsing damage can reflect seasonal changes in herbivore densities, and the availability of other forage (Montague 1996). How seedling damage relates to population densities and biomass of ground vegetation on the young plantation, will be discussed in Chapter 12.

11.4.3 Insect damage

Insect browsing to seedlings was immediate (occurred within 2.5 weeks of planting), extensive (45 % of live seedlings) and severe (23 % of foliage removed from damaged seedlings). Scarab beetles (*Heteronyx spp.*) were considered to be largely responsible for this damage, as they were highly abundant at this time, and were actively defoliating seedlings. These beetles are a known pest within northern Tasmanian tree farms (Wardlaw and de Little, 2000), and their impact within southern plantation estates is currently under review (Hurley *et al.* 2000).

Patterns of insect browsing at this site reflected scarab beetles' cycle of seasonal activity. Adult scarab beetles defoliate *E. nitens* seedlings during warm, sunny weather, between October and February (Wardlaw and de Little, 2000). Other times, beetles burrow into the soil surrounding seedlings. High levels of insect damage were reported on the young plantation over summer and this damage was carried over into winter as seedlings slowly recovered and insects were dormant.

11.4.4 Management implications

Slow seedling growth (15 to 30 cm in 8 months), even in the absence of mammal browsing, indicated suboptimal growing conditions at the Surrey Hills study site. As a comparison, a northern Tasmanian *E. nitens* plantation of similar altitude, planted in October, showed seedling height increased from 20-80 cm over 8 months (C. McArthur unpublished data). In addition to the cool climate at Surrey Hills, waterlogged soils and frequent frosts were thought to negatively impact upon seedlings' growth. Competition from weeds within fenced plots may also have resulted in seedlings having to compete for limited resources (nutrients and light), although these still grew better than unfenced seedlings.

Due to the severity of the damage, scarab beetle browsing can easily be misidentified as mammal browsing damage (Wardlaw and de Little, 2000). Consequently, regular monitoring of seedlings (for example, monthly, particularly in warmer months) is important to correctly identify the cause of browsing damage. Insect damage was also responsible for about half of seedlings with multiple leaders having undergone a change in growth form. This effect of damage is usually attributed to mammal browsing. My results suggest that infrequent monitoring (for example, half-yearly), could result in an over-estimation of damage attributed to mammals.

The assumed linear relationship between macropod abundance and mammal browsing damage is unjustified (Bulinski 2000). There were approximately 199 pademelons and 43 wallabies using the young plantation at night, when the seedlings were planted (Chapter 4). These high abundances of herbivores would normally result in forestry staff conducting a 1080-poisoning operation to reduce local herbivore populations before planting. Results, however, showed that mammal browsing damage to seedlings was low, particularly compared with insect damage over the summer months following planting. Additionally, by the time mammal damage to seedlings increased (some time between 4 and 6 months after planting), the effects of any poisoning operation are likely to have ceased (Chapter 5).

Interestingly, at the completion of this study, the young plantation was poisoned for a second time with 1080 (October 1998), herbicided (aerial spray of Roundup® and Gestop® in December 1999), and entirely replanted (January 1999), yet seedling mortality was still high and growth rate low (I. Raven *pers. comm.*). These results are consistent with Bulinski (2000), who found no link between accumulated macropod scat density and browsing damage at 35 eucalypt plantations across Tasmania.

Chapter 12

General Discussion

Previous chapters have discussed results from the spotlighting, radio-tracking, ground vegetation, and seedling data relatively independently of one other. What follows is a synthesis, linking together herbivore population densities, patterns of habitat use and selection, and their consumption of forage within the young plantation. Together, these data provide baseline ecological information on how wallabies, pademelons and possums utilise this patchy forestry environment. They also offer possible alternatives to the present lethal control methods for reducing mammal browsing damage to commercial seedlings within Tasmanian plantations.

12.1 Use of the Forestry Environment

12.1.1 Macropods

Patterns of habitat use and selection by wallabies and pademelons were consistent across spatio-temporal scales. Wallabies and pademelons used all habitats, both night and day (Figure 8.1), but showed strong selection for open habitats at night (Tables 4.2 and 8.3), and closed habitats during the day (Table 8.3). These patterns were supported both by the spotlighting data (population level), and by the radio-tracking data (individual level). Use of open habitats at night by these macropods was consistent with their nocturnal behaviour, and feeding strategies of grazer and mixed-feeder. Use of open habitats for feeding activities was confirmed by the detailed study of vegetation consumed on one of these habitats, the young plantation. Herbivores, most of which were macropods, consumed large quantities of grass and forb biomass from this habitat (Figure 10.5).

The fact that wallabies and pademelons selected for open habitats at a range of different scales suggests that food is the driving factor for many of these selection decisions. Rettie and Messier (2000) suggest that the selection of similar resources, across different spatio-temporal scales, has a multiplicative effect on the quality of those resources. This behaviour may, therefore, enable animals to concentrate resources available within the environment. If the resource is food (abundance and/or quality), this consistent pattern of resource selection would increase foraging efficiency, which in turn, promotes survival and reproduction. Similar patterns of selection across spatio-temporal scales have been reported for a range of species, for example, capercaillie *Tetrao urogallus* (Storch 1993), muskoxen *Ovibos moschatus* (Schaefer and Messier 1995), baboons *Papio cynocephalus* (Cowlshaw 1997), caribou *Rangifer tarandus* (Bradshaw *et al.* 1995), and green woodpeckers *Picus viridis* (Rolstad *et al.* 2000).

Increased use of closed habitats by macropods during the day, but only at this temporal scale within the decision-hierarchy, suggests that habitat quality, in terms of shelter, is important but not overriding of food. It also reflects the composition of the landscape within the Surrey Hills forestry environment. Within this particular landscape, closed habitats were more abundant than open habitats (87 % vs. 13 % respectively). Consequently, their lack of selection at the other spatio-temporal scales may simply reflect this high availability (Section 1.2). Additionally, resting animals may require little space which could also result in closed habitats not being selected for at higher scales (Garshelis 2000).

One reason wallabies and pademelons may shift from nocturnal feeding sites of high food quality, to shelter in densely vegetated diurnal habitats, is to avoid extremes of heat and wind

(Priddel 1986, Clarke *et al.* 1989). I consider it unlikely, however, that the macropods at the Surrey Hills site were showing this thermoregulatory behaviour, because the climate was relatively mild and animals showed the same patterns of behaviour throughout the year, irrespective of weather conditions.

A second reason may be that animals were seeking shelter as a predator avoidance strategy (Croft 1989). I consider this situation highly likely because the macropods' two native marsupial predators, the Tasmanian devil and spotted-tailed quoll (Jones and Barmuta 1998), were known to inhabit the Surrey Hills study site (K. le Mar *pers. obs.*). In addition, carnivore damage to radio-transmitters (either before or after the animals had died), was also documented on several occasions (Chapters 7 and 9).

It is interesting to note that this situation differs from mainland studies, where macropods' daytime sheltering behaviour, in the absence of predators, has been attributed to ghosts of predators past. That is, this behaviour has been described as an adaptation to mainland predators from the Pleistocene era, for example, the extinct marsupial lion (*Thylacoleo*), Tasmanian wolf (*Thylacinus*), and Tasmanian devil (Caughly 1964). Hence, the Tasmanian macropods appear to be responding to an actual risk of predation, unlike several mainland species, which appear to be responding to a perceived risk of predation.

12.1.2 Possums

Patterns of habitat use and selection by possums varied across spatio-temporal scales. Spotlighting data showed that at night, at the population level, possums selected for grassland, young plantation and native forest. Radio-tracking data showed selection for grassland, young plantation and harvested uncleared land at the scale of home-range location, but nocturnal selection for native forest only, when animals were foraging. It is unclear whether the differences between the spotlighting and radio-tracking data are an artefact of small sample size for the collared animals. For example, if possums have strong individual variation in their patterns of habitat use and selection, results from the six animals used in the telemetry study may not be representative of the entire population. Alternatively, home-range composition could reflect trapping bias, as collared animals were caught on, or near, the young plantation.

Differences in patterns of selection at the scales of home-range location and nocturnal feeding areas could reflect selection for different resources at these two scales. It is difficult to identify what these resources are at the higher level of home-range location, and further investigation into possums behaviour, including resource use, is required. There are examples of other species that select for different resources at sequential spatio-temporal scales. Female yellow-headed blackbirds *Xanthocephalus xanthocephalus* select for food (insect abundance) at the higher scale of settling area, but select cover habitat at the lower scale of nest site location (Orians and Wittenberger 1991). Woodland caribou *Rangifer tarandus* select for predator avoidance at the higher scale of seasonal range, and select for food at the lower scale of daily area (Rettie and Messier 2000).

Selection for native forest by possums, night and day, is consistent with the arboreal folivore feeding strategy of this species, and their use of tree-hollows for daytime den sites. Selection for grassland is curious, however, as this habitat was thought to offer this species little food or shelter. There are several possible explanations. First, grasslands may have a high food abundance and/or quality for possums that I have failed to recognise. Second, there were occasional, large *Eucalyptus* trees within grasslands, which may provide safety to animals foraging on the ground within this habitat. Therefore, possums may have been associated with these trees within grassland for shelter reasons, but were able to use grasslands for foraging as a consequence of this shelter. Third, the large *Eucalyptus* trees may have offered possums both browse and den sites. In this case, use of grasslands may simply reflect access to food and shelter within these trees. Any, or all, of these scenarios may be true, particularly

as Tasmanian possums are known to spend more time feeding on the ground i.e. be less arboreal than their mainland counter-parts (Statham 1984), despite high predator densities within Tasmania. Further investigation into use of these habitats by possums would be useful.

12.2 Were Herbivores Enhanced or Hindered by Patchiness?

In the absence of comparable data within undisturbed, relatively homogenous Tasmanian environments, I can only extrapolate from my own data to discuss the potential effects of landscape patchiness and quality of habitats on wallabies, pademelons and possums.

12.2.1 *Macropods*

The patchy nature of this forestry environment appeared to benefit macropods. Wallabies and pademelons had high population densities (0.3 and 1.5 animals.ha⁻¹, respectively), and small, round home-ranges (61.6 and 22.3 ha, respectively), which were indicative of highly abundant resources (food and shelter) for these species. Use of the young plantation as a primary feeding ground by macropods was also indicated by the high utilisation of available vegetation within this particular habitat.

Improved conditions for macropods, resulting from increased patchiness within the landscape, reflect the fact that these animals use separate, heterogeneous habitats for different activities, as discussed in Section 12.1.1. Wallabies and pademelons selected to feed mostly within open habitats at night, and to shelter within closed habitats during the day. Consequently, these macropods appear to be advantaged by environments where habitat patches are relatively small (high perimeter to area ratio), and open feeding areas are created near densely vegetated habitats. This is consistent with Bulinski's (1999b) findings, that browsing damage to commercial seedlings within Tasmanian plantations was positively related to a plantation's perimeter to area ratio, and to the proportion of perimeter adjacent to cover habitat.

Animals that benefit from man-modified environments by an increase of habitat edge are described as ecotonal or forest-edge species (Franklin and Forman 1987). Examples of northern hemisphere forest-edge species include a range of deer species *Cervus* spp. in Great Britain (Hannan and Whelan 1989), white-tailed deer *Odocoileus virginianus* in north America (Drolet 1978), and moose in Norway (Hjeljord *et al.* 1990).

Another aspect of the ecology of these forest-edge species is that, because forests are dynamic environments, the quality of habitats may change over time (Franklin and Forman 1987). For example, habitat quality may be low initially after harvesting or clearing, as food and/or shelter has been removed, but food abundance and/or quality may increase over subsequent months and years, before finally declining as trees form a canopy, which then shades, and reduces growth of, ground vegetation (Floyd 1980). Hence, factors, such as rotation length, could be important determinants of the long-term impact of forest patchiness on herbivores. Within south-eastern Australia, rotation length of *Eucalyptus* plantations is relatively short (12-15 years) when producing wood pulp, or up to 35 years for the production of saw logs. The relative proportion of these plantation types could significantly alter the overall impact of the forestry environment on herbivore populations.

12.2.2 Possums

Possums appeared to be disadvantaged by the fragmentation and changed composition of the landscape. Possum population density was very low ($0.04 \text{ animals.ha}^{-1}$), while individual's home-ranges were extremely large (39.1 ha) compared with Australian mainland populations (see review by Green 1984). This may reflect the fact that possums are a forest-interior species, which are usually disadvantaged by reduced native forest patch size and increased edge (Franklin and Forman 1987).

My results suggest that resources were limited for possums at this site. Although food abundance and quality was not quantified within most of the habitats within this study, only native forest and the young plantation were thought to offer potential food to possums. Results from fenced plots on the young plantation did not clarify this situation, because possum densities were so low within this habitat, that their effect on vegetation was indiscernible (Chapter 10). Grassland and harvested uncleared land were thought to offer little food to a folivore, as grassland comprised mostly grasses and sedges, while harvested uncleared land comprised mostly fallen logs. However, as discussed in Section 12.1.1, it is possible that grasslands offered more food than I thought. Despite this, the habitat itself only comprised 9 % of the study area and therefore was not particularly abundant. Additionally, older plantation appeared to represent a biological desert to possums, as animals did not browse juvenile *E. nitens* foliage, and the ground was mostly bare, apart from the occasional rush (*Juncus pallidus*). Sixty-four percent of the study area comprised older plantation, while native forest comprised only 20 %. Hence, within this particular forestry environment, possums appeared to have limited foraging areas. At least 20 % of this native forest was also scheduled for land-clearing to make way for more *E. nitens* plantations. Consequently, food (and shelter) resources were destined to become even less abundant for animals within the local population, over time. Hence, possum conservation within Tasmania's forestry environments may be an issue of concern in coming years.

Loss of high quality diurnal den sites for possums, due to forestry land-clearing practices, has been well documented (McIlroy 1978, Gibbons and Lindenmayer 1996). Use of tree-hollows as diurnal refuge by possums, at the Surrey Hills study site, was confirmed by radio-tracking animals on foot, during which I found that nest sites were generally at least one metre above ground level. This is inconsistent with other studies where tree-hollows have been limited or unavailable and possums have nested in dense shrubs, holes at the base of trees, below ground (e.g. abandoned rabbit warrens), and among rocks and crevices (Troughton 1946, Dunnet 1956, Frith 1973, Green and Coleman 1987). The use of dens at ground level, however, may only be possible when predators are in low densities or absent (Cowan 1989). Apparent high densities of predators (Tasmanian devils and spotted-tailed quolls) at the Surrey Hills site may have prohibited possums from denning at ground level. It is unlikely that climatic factors, such as high year-round rainfall (2200 mm annually), was the cause: possums in New Zealand have used ground dens where annual rainfall exceed 3000 mm (Green and Coleman 1987).

12.3 Conclusions in Relation to Herbivore Ecology

My results suggest that there are four important factors to consider when studying herbivores within patchy environments. First, spatio-temporal scale does affect patterns of habitat use. Second, the nature of those patterns depends on the species and possibly the relative availability and need of high quality habitats. This species effect exists because different animals view the quality of habitats in different ways: what is "good" for one species is not necessarily "good" for another. Third, if habitats are not selected for, it does not necessarily follow that they are not important. It may be that they are abundant already, or that only a small area/amount is required, as discussed earlier in relation to shelter for resting animals. Fourth, in order to fully understand an animal's ecology, it is important to collect both day and night data, irrespective of when animals are active, because resting animals also have

resource requirements. For example, within this study, if the essentially nocturnal macropod species had only been studied at night, their daytime selection of closed habitats, and the different shelter requirements of the two species, would not have been detected (Table 8.3). This is particularly important if data are to be used to develop strategies for species management. Biased findings within data collected during only one light cycle have been reported by Beyer and Haufler (1994) and Arthur and Schwartz (1999).

12.4 Impact of Herbivores on Forestry

The previous sections have described the influence of the forestry environment on herbivores. What follows is a discussion of how the herbivores affect forestry, particularly in relation to the browsing damage to commercial seedlings, and how this could be reduced.

12.4.1 Role of landscape design

If high macropod densities are associated with patchy landscapes, reducing fragmentation of the environment may reduce browsing damage to seedlings within plantations by decreasing regional herbivore densities. This could be achieved through plantation distribution (clustering plantations), plantation design (designing larger plantations), and plantation placement (proximity to other habitats). Results from this study have shown, however, that high herbivore densities are not necessarily associated with severe mammal browsing damage to seedlings. Consistent with this, Bulinski (2000) found no relationship between macropod densities (measured as accumulated scats) and damage to seedlings in 32 Tasmanian plantations. Consequently, landscape design may only provide part of the answer, because the damage to commercial tree seedlings by herbivores is affected by more than just animal numbers (see below). Further investigation into the relationship between herbivore densities and browsing damage to seedlings is required.

12.4.2 Role of plantation vegetation

Results from my study showed that in the absence of broadcast herbiciding, and hence in the presence of ground vegetation, browsing damage to commercial seedlings was low over the summer following planting. A corollary of this is that the current practice of herbiciding before planting may remove a primary food source (grass and forbs) for macropods. Without this primary food source, but with the highly visible and regularly spaced commercial seedlings, animals may be forced to eat these plants in the absence of an alternative. This is especially the case if animals are unable to extend their home-range due to high local population densities (Chapter 8). This suggests that the presence of vegetation, perhaps of particular types, may be important in mitigating damage.

The deliberate retention of ground cover, or provision of alternative food plants for herbivores feeding within plantations, may effectively reduce mammal browsing damage to commercial seedlings (Clunie and Becker 1991). It is interesting to note, however, that my results of high grass biomass and low initial damage to seedlings conflicts with Bulinski's (1990b) findings that grass cover, in a comparison between 32 Tasmanian plantations, was positively correlated with mammal browsing damage to seedlings. Results, from the young plantation that I investigated, suggest that low grass cover may be associated with high macropod densities and that high grass cover can be associated with low browsing damage to seedlings. Again, this implies that the relationship between animal numbers and damage is non-linear, but also that there may be a complex interaction with vegetation that needs to be resolved.

Forestry companies have two concerns over plantation productivity in regard to using ground cover or cover-crops as alternative food sources for herbivores. Firstly, the growth rate of planted trees can be reduced by heavy competition from surrounding vegetation (Waters 1985, Tolhurst and Turvey 1992), and poor height growth exposes planted trees to browsing damage over much longer periods (Clunie and Becker 1991). Competition between commercial trees and surrounding vegetation could be reduced or eliminated by controlling for weeds within the immediate vicinity of plantation trees, such as along tree rows, leaving alternative food sources for browsing herbivores throughout the rest of the plantation. This could be achieved by using strip- or spot-herbicide regimes which create bare patches along tree rows, alternatively, cover-crops could be restricted to regions between tree rows (Clunie and Becker 1991).

The second concern over the presence of food plants within plantations is that this situation may attract animals into the general area, thereby inadvertently increasing herbivore densities on plantations (Clunie and Becker 1991). Within the present study however, as food abundance increased over time on the young plantation (Figure 10.5), local herbivore populations remained stable (Figure 4.1) in the absence of poisoning and shooting operations. Stable populations, even though densities were high, caused little damage to seedlings over the summer period while alternative vegetation was abundant (Figure 11.3). It was only once ground vegetation declined at the onset of winter that browsing damage to seedlings started to increase. Hence, the use of these alternative crops may offer a solution to current lethal control methods, but population stability may be important for this to be effective.

12.5 Conclusions in Relation to Forestry

Results from this study indicate that non-lethal management strategies for reducing herbivore browsing damage to seedlings are a possibility. These strategies can now be based on an understanding of the feeding ecology of these herbivores that has been gained from my research. Techniques could involve improved design of plantation size and shape, reduction of size or removal of windrows, and provision of herbivores with an alternative food source to commercial seedlings. Current practices of broadcast herbicide and 1080-poisoning or shooting before planting, may also need to be reconsidered in light of this study, as population stability may play an important role in future management strategies for reducing damage caused by these herbivore species. Timing of planting may also be important. The common practice of planting seedlings during winter may exacerbate the browsing problem, given the reduced availability of forage for herbivores at this time. Finally, information obtained from my study also suggest that the manipulation of herbivores densities, particularly the abundance of dominating male pademelons, may play a key role in future management methods. In conclusion, this study has succeeded in its original aims of examining herbivore population densities, patterns of habitat use and selection, and specifically the use of a newly established plantation. This information helps fill in the gaps in our understanding of the feeding and population ecology of Tasmania's main native herbivores, which were once abundant in much of SE Australia.

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