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

**Author**

le Mar, K

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## Appendix A

### Radio-telemetry: Data set resolution

#### A.1 Introduction

Within radio-tracking studies, sampling frequency or data set resolution can effect movement estimates (Laundré *et al.* 1987). Increased sampling frequency, or higher resolution, appears to result in animals travelling further. The radio-tracking study described in Chapter 7 involved relocating animals at 45 minute intervals. Hence, the aim of this study was to investigate how sampling at a 45 minute frequency compared with results obtained from a more continuous data set.

#### A.2 Methods

##### A.2.1 Data Collection

Four males (two wallabies [# 71 & 73], one pademelon [# 35] and one possum [# 06]) were radio-tracked at 10 minute intervals over a 24 hour period. These data are hereafter referred to as the “continuous” data sets. Two animals were followed over a given 24 hour period. Two data collection periods were carried out over three days in March 1997.

Animals were radio-tracked using fixed towers described in Chapter 6. Five minutes were spent following an individual’s transmitter signal. A control transmitter at a known location was relocated every hour, to check for variation due to equipment problems, weather conditions or human error (fatigue).

Magnetic bearings were processed into mapped locations as described in Chapter 6. Each animal’s continuous data set of mapped locations, was sub-sampled at 45 minute intervals to create three new data sets. These subsets were assigned a letter (a, b or c), and were used to examine potential variation within a single data set.

##### A.1.2 Data Analysis

Distance travelled was calculated as the sum of all straight-line distances between sequential locations. Rate of travel was calculated as the sum of distances travelled divided by the time period involved. Calculations were made using the Animal Movement® (Hooge and Eichenlaub 1997) extension within ArcView® GIS (ESRI 1996). Area covered was calculated using the Minimum Convex Polygon (MCP) estimator in Animal Movement® (Hooge and Eichenlaub 1997).

#### A.3 Results

Results from the 45 minute interval data represented 30-47 % of the distance estimated to be travelled using the continuous data (Table A.1). Results also showed that the 45 minute interval data estimated 31-49 % of the travel rate calculated for the continuous data (Table A.2).

Table A.1 Estimates of distance travelled during the 24 hour period, calculated from the continuous and 45-min interval data sets. Distance, as percentage of the continuous data, is shown in brackets.

Data set	Wallaby #71 (km)	Wallaby #73 (km)	Pademelon #35 (km)	Possum #06 (km)
Continuous data	8.1	14.8	7.1	7.2
45 min sub-set (a)	2.8 (35)	6.0 (41)	2.9 (41)	2.8 (39)
45 min sub-set (b)	2.6 (32)	4.5 (30)	2.5 (35)	3.4 (47)
45 min sub-set (c)	2.9 (36)	5.6 (38)	3.0 (42)	2.4 (33)

Table A.2 Average rates of travel ( $\pm$  s.d.) for the three species, calculated from the continuous and 45-min interval data sets. Rate of travel, as percentage of the continuous data, is shown in brackets.

Data set	Wallaby #71 (m.hr <sup>-1</sup> )	Wallaby #73 (m.hr <sup>-1</sup> )	Pademelon #35 (m.hr <sup>-1</sup> )	Possum #06 (m.hr <sup>-1</sup> )
Continuous data	345 $\pm$ 257	617 $\pm$ 673	306 $\pm$ 266	304 $\pm$ 344
45 min sub-set (a)	119 $\pm$ 99 (34)	257 $\pm$ 310 (42)	122 $\pm$ 124 (40)	123 $\pm$ 123 (40)
45 min sub-set (b)	112 $\pm$ 98 (32)	193 $\pm$ 169 (31)	109 $\pm$ 142 (36)	149 $\pm$ 112 (49)
45 min sub-set (c)	130 $\pm$ 97 (38)	248 $\pm$ 264 (40)	128 $\pm$ 135 (42)	106 $\pm$ 101 (35)

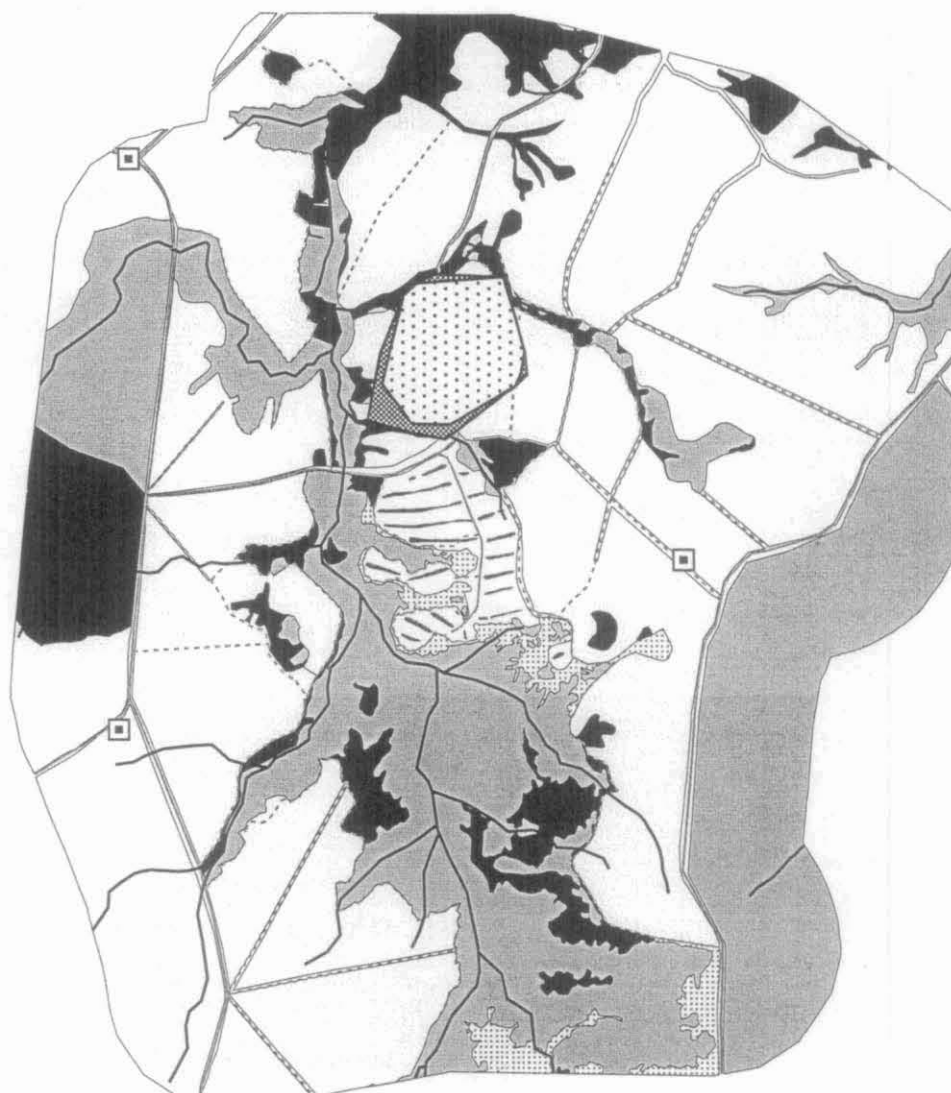
Estimates of area covered by the continuous and 45 minute interval data sets are shown in Table A.3. Values varied considerably amongst species' sub-sampled data sets, estimating between 44-92 % of the area covered by the continuous data. Diagrams showing the area covered, calculated using continuous and subset "a" data sets, are shown in Figures A.1-3.

Table A.3 Estimated area covered by individuals over a 24 hour period, using the continuous and 45 minute interval data sets. Area, as percentage of the continuous data, is shown in brackets.

Data set	Wallaby #71 (ha)	Wallaby #73 (ha)	Pademelon #35 (ha)	Possum #06 (ha)
Continuous data	17.8	64.9	17.9	15.9
45 min sub-set (a)	15.0 (84)	49.9 (77)	12.6 (70)	10.4 (65)
45 min sub-set (b)	13.1 (74)	28.8 (44)	11.9 (66)	11.3 (71)
45 min sub-set (c)	14.0 (79)	60.0 (92)	11.5 (64)	7.5 (47)

#### A.4 Discussion

The underestimation of all results was relatively consistent between species, but somewhat variable between data sub-sets. Results suggest that a sampling frequency of 45 minutes may underestimate animals' linear movements by 50-70 %. This was not surprising, as estimates of linear movement are largely affected by data resolution. Additionally, sampling at 45 minute intervals may also underestimate areal measurements by 10-55 % (average 69 %). Despite this range, mapped results suggest that patterns of habitat utilisation may be



#### LEGEND

- ▣ Radio-tracking tower
- ▤ Wallaby # 71 - 45 min interval data
- ▥ Wallaby # 71 - continuous data
- ~ Waterway
- - - Dirt road
- Gravel road
- ▦ Harvested, uncleared land
- ▧ Young plantation ( ▨ windrow )
- ▩ Grassland
- Native forest
- Older plantation

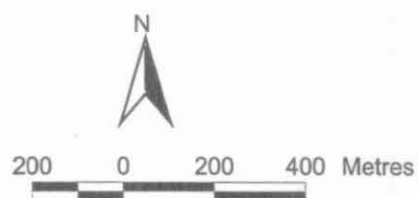













Figure A.1 Map of the area covered by wallaby #71 using the continuous and 45-min interval data sets.



# LEGEND

-  Radio-tracking tower
-  Wallaby # 73 - 45 min interval data
-  Wallaby # 73 - continuous data
-  Waterway
-  Dirt road
-  Gravel road
-  Harvested, uncleared land
-  Young plantation (windrow)
-  Grassland
-  Native forest
-  Older plantation

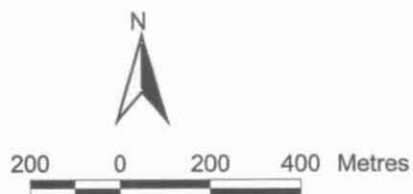
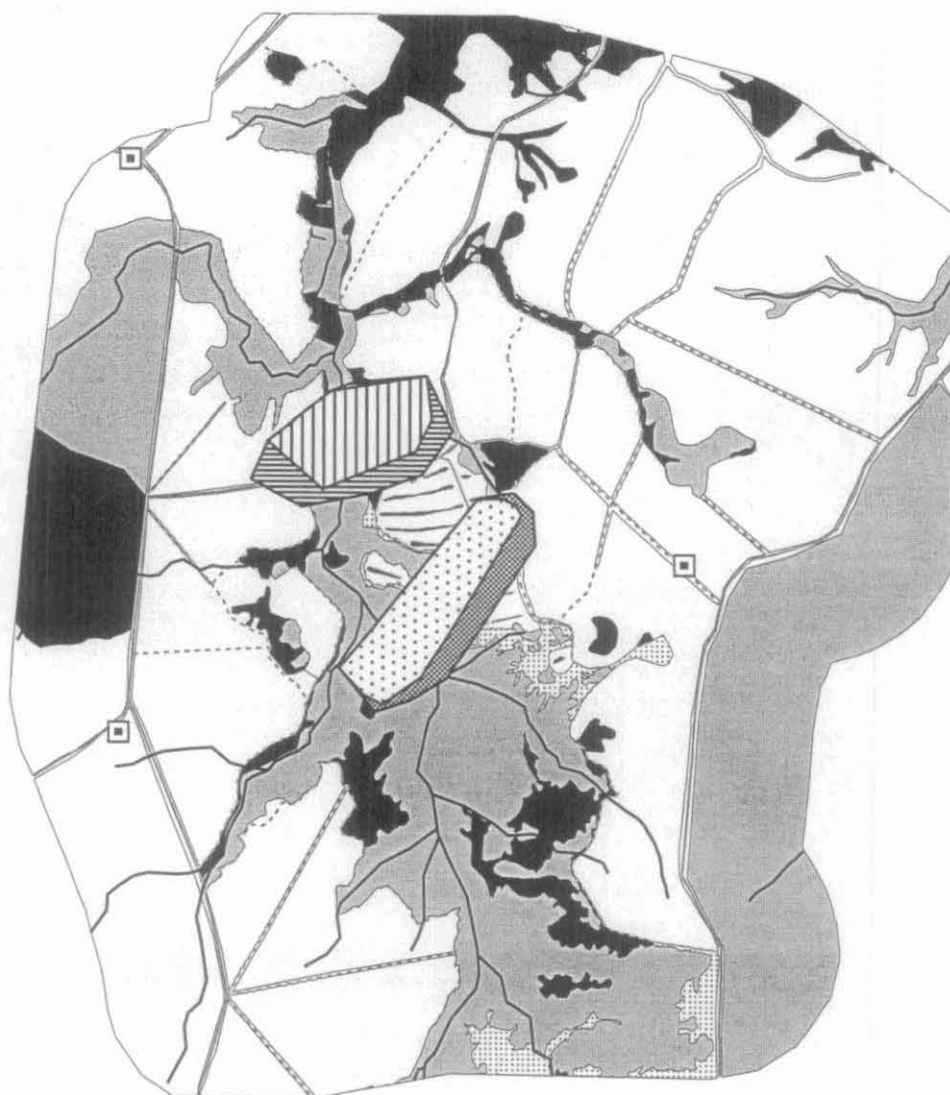















Figure A.2 Map of the area covered by wallaby #73 using the continuous and 45-min interval data sets.



# LEGEND

-  Radio-tracking tower
-  Pademelon # 35 - 45 min interval data
-  Pademelon # 35 - continuous data
-  Possum # 06 - 45 min interval data
-  Possum # 06 - continuous data
-  Waterway
-  Dirt road
-  Gravel road
-  Harvested, uncleared land
-  Young plantation (windrow)
-  Grassland
-  Native forest
-  Older plantation

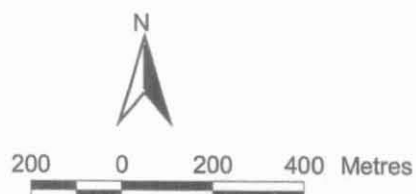


Figure A.3 Map of the areas covered by pademelon #35 and possum #06 using the continuous and 45-min interval data sets.

relatively consistent between the continuous and 45-min interval data sets. This enables us to be confident about the patterns of habitat use and selection determined in Chapter 7.

Data used in Chapter 7 was not adjusted by the errors estimated in the present study because these values were broad and may vary considerably between individuals and across season. In addition, the underestimation of areas used by individuals may have been offset in Chapter 7 by the use of buffered points and buffered home-ranges, within the habitat selection analyses. In conclusion, it is recognised that the 45 minute sampling regime used in Chapter 7 will have an error associated with the resolution of the data set, however, data collection regimes within radio-tracking studies are a trade-off between accuracy, sample size and logistical restraints.

## Appendix B

## Flora on the Young Plantation

Table B.1 A list of most flora species found on the young plantation.

Category	Family	Genus species
Fern	Aspidiaceae	<i>Polystichum proliferum</i>
Forb	Asteraceae	<i>Cassinia aculeata</i>
		<i>Cirisium vulgare</i>
		<i>Cirisium vulgare</i>
		<i>Cotula sp.</i>
		<i>Leontodon taraxaciodes</i>
		<i>Leptinella filicula</i>
		<i>Leptorhynchos squamatus</i>
		<i>Olearia phlogopappa</i>
		<i>Olearia phlogopappa</i>
		<i>Podolepsis sp.</i>
		<i>Senecio jacobaea</i>
		<i>Senecio minimus</i>
		<i>Taraxacum sp.</i>
	Campanulaceae	<i>Wahlenbergia sp.</i>
	Caryophyllaceae	<i>Myosotis sp.</i>
	Fabaceae	<i>Trifolium sp.</i>
	Geraniaceae	<i>Geranium dissectum</i>
		<i>Pelargonium inodorum</i>
	Onagraceae	<i>Epilobium billardierianum</i>
	Oxalidaceae	<i>Oxalis corniculata</i>
	Polygonaceae	<i>Acetosella vulgaris</i>
	Ranunculaceae	<i>Ranunculus lappaceus</i>
	Rosaceae	<i>Acaena novae-zelandiae</i>
	Violaceae	<i>Viola hederacea</i>
Grass	Poaceae	<i>Australopyrum pectinatum</i>
Non-Grass Monocots	Iridaceae	<i>Poa labillardierei</i>
		<i>Diplarrena moraea</i>
		<i>Juncus pallidus</i>
Shrub	Juncaceae	<i>Luzula sp.</i>
		<i>Cyathodes glauca</i>
		<i>Pultenaea juniperina</i>
	Proteaceae	<i>Banksia sp.</i>
		<i>Hakea lissoperma</i>
		<i>Lomatia tinctoria</i>
		<i>Boronia sp.</i>
	Rutaceae	<i>Pimelea sp.</i>
		<i>Pimelea sp.</i>
		<i>Pimelea sp.</i>
Tree	Myrtaceae	<i>Eucalyptus delegatensis</i>
	Fagaceae	<i>Nothofagus cunninghamii</i>
	Mimosaceae	<i>Acacia sp.</i>
Moss/lichen/fungi		<i>Marchantia</i>



# Relocating radio-collared targeted marsupials after a 1080-poisoning operation

K. le Mar\* and C. McArthur

Cooperative Research Centre for Sustainable Production Forestry,  
School of Zoology, University of Tasmania,  
GPO Box 252-05, Hobart 7001

## Abstract

The fate of 26 radio-collared individuals from three targeted marsupial species were followed during a 1080 (sodium monofluoroacetate)-poisoning operation on a eucalypt plantation, to investigate where animals die in relation to the bait-line. Fifteen of the 26 animals died during the poisoning operation: eight of ten Tasmanian pademelons (*Thylogale billardieri*), one of seven Bennett's wallabies (*Macropus rufogriseus rufogriseus*), and six of nine brushtail possums (*Trichosurus vulpecula*). The proportions of radio-collared animals that died within each species did not reflect kill-rates calculated from more reliable absolute density data. Radio-collared carcasses were found between 8 m and 83 m from the bait-line (mean distance 31 m). Seventy-five per cent of carcasses were found inside shelters (i.e. inside windrows, hollow logs, dens or under fallen vegetation). Twelve of the 15 poisoned, radio-collared carcasses were found intact. Three carcasses were not found but recovered collars showed 'carnivores' teeth marks, suggesting that Tasmanian devils (*Sarcophilus harrisii*) or spotted-tailed quolls (*Dasyurus maculatus*) had moved and/or consumed them.

## Introduction

Three native marsupial species reduce productivity in commercial Tasmanian plantations by browsing seedlings (Bulinski

and McArthur 1999; Coleman *et al.* 1997; Cremer 1969). These species are the red-bellied pademelon (*Thylogale billardieri*), Bennett's wallaby (*Macropus rufogriseus rufogriseus*) and the brushtail possum (*Trichosurus vulpecula*). 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 (Bulinski 1999; Marsh 1998; Statham 1983; Johnson 1978). However, the fine-scale effects of 1080 operations 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 aim of this study was to quantify how far targeted marsupials travelled from the bait-line before death, and where carcasses were located following a 1080 operation on one eucalypt plantation.

## Study area and methods

### Study area

The study area was located in North Forest Product's 'Surrey Hills' Tree-farm, north-western Tasmania (41°28'S, 145°48'E). Five habitats dominated this site (refer to Figure 1):

\* Corresponding author:  
e-mail: k.lemar@utas.edu.au



# LEGEND

- ▣ Radio-collared pademelon
- ⊠ Collar from pademelon
- △ Radio-collared Bennett's wallaby
- ⊙ Radio-collared possum
- ⊕ Collar from possum
- ⋯ Bait-line
- - - Dirt road
- ≡ Gravel road
- ~ Waterway



0 100 200 300 400 Metres

- ▬ Coupe (Windrows shown in black) - poisoned plantation
- ░ Older plantation
- ▒ Uncleared harvested land
- Native forest
- Grassland

Figure 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.

(1) a young *Eucalyptus nitens* plantation with an area of 17.8 ha and with relatively high weed cover (hereafter referred to as 'the coupe'); (2) older plantations of *E. nitens* (5–7 years of age, approximately 5 m tall); (3) grassland; (4) native forest (rainforest and wet eucalypt forest); and (5) uncleared harvested land that consisted of scrub and fallen vegetation.

The coupe was planted with *E. nitens* seedlings (approximately 20 cm in height) in November 1997, five months before this study began. The coupe 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 (glyphosate) herbicide approximately one week after planting.

#### 1080 poisoning

Staff of North Forest Products 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 coupe's central access road, the plantation boundary, and nearby firebreaks. Twenty grams of fresh, chopped carrots were placed at each bait station on five occasions (referred to as 'free-feeds'), three to six days apart, over a period of 17 days in April 1998. Fifty kilograms of bait were 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 carrots mixed with 0.014% 1080 in liquid solution and blue dye) were distributed at the bait stations (hereafter referred to as the 'bait-line', see Figure 1) three days after the final free-feed. Bait stations were checked by company staff for remaining poisoned bait 24 hours after distribution.

#### Radio-collared animals

Animals to be radio-collared were caught on or near the coupe. Macropods were caught using two methods: (a) dart-gunning animals on the coupe, and (b) trapping on a grassland approximately 75 m north-east of

the coupe. Possums were caught using cage-traps on the coupe and within nearby native forest. Animals were lightly anaesthetised using Zoletil (tiletamine hydrochloride and zolazepam hydrochloride) during radio-collar attachment (7 mg/kg for dart-gunned macropods, 4 mg/kg for trapped macropods and 2 mg/kg for possums). Animals were released at the site of capture within 6 hrs of being caught.

Ten adult pademelons (seven male and three female), seven adult Bennett's wallabies (five male and two female), and nine adult possums (four male and five 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 and 152 MHz. These animals had been used in a larger study to investigate habitat utilisation within the study site (le Mar, unpublished data). 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 Bennett's wallaby population that utilised the coupe nightly (le Mar, unpublished data). 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 ( $\pm$  SE) of 2 ( $\pm$  1) possums (le Mar, unpublished data).

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.

#### Results

No poisoned bait remained 24 hours after it was distributed. Eleven of the 26 radio-

Table 1. Locations of radio-collared animals killed by 1080.

Habitat	Location	Pademelon	Bennett's wallaby	Possum
Coupe	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

collared animals were located alive. The remaining 15 animals were killed during the poisoning operation: eight of ten pademelons, one of seven Bennett's wallabies, and six of nine possums. Both male and female radio-collared pademelons and possums were killed: five of seven male pademelons and three of three female pademelons; four of five male possums and two of six female possums. One Bennett's wallaby was killed, a female with a pouch-young (furless, dead in pouch).

Twelve of the 15 animals that died during the poisoning operation were found

undamaged by carnivores. These animals were found within 83 m of the bait-line (Figure 2); mean distance 31 m (range 8-83 m). Eighty-four per cent of carcasses were located within 40 m of the bait-line (Figure 2). Few animals died out in the open: 75% of carcasses were found inside windrows, under fallen vegetation or inside dens (Table 1). Blood was present in the nostrils and ear canals of two pademelons suggesting internal bleeding as the result of poisoning.

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.

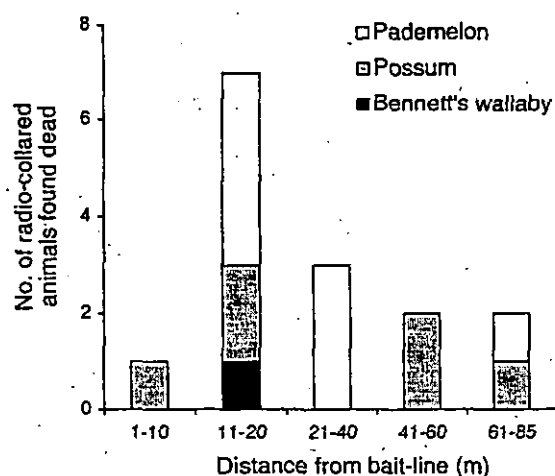


Figure 2. The distances dead radio-collared animals (killed by 1080) were found from the bait-line.

## Discussion

The poisoning operation killed individuals from all three targeted marsupial species. The sample sizes of radio-collared animals were too small to draw conclusions about kill-rates for the three species. However, the effects of the poisoning operation on species densities were collected during another study, and will be reported elsewhere.

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 (Cahalan 1998). Results for Bennett's 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 (McArthur, unpublished data) that Bennett's wallabies can travel substantially further (c. 240 m) from the bait-line than indicated by this study. Hence, Bennett's wallabies may travel further from the bait-line than possums or pademelons.

Results from this study also suggest that 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). 1080 inhibits citrate and succinate

metabolism in the Krebs cycle, blocking energy production within cells and manifesting as cardiac failure in herbivores (Statham 1983). It seems unlikely, therefore, that animals would seek water from dehydration as a response to this.

A major implication of this research is that 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.

### Acknowledgements

We thank North Forest Products Ltd for financial and logistic support and, in particular, David de Little, Ian Blanden, Calton Frame, Trevor Docking, Trevor Dick and Andrew Walker. Christine Mann, James Dick and Jeremy Wilson kindly provided aerial photos and produced maps from the North Forest Product's GIS. The Cooperative Research Centre for Sustainable Production Forestry provided financial support and the use of a vehicle. Roger Martin dart-gunned animals, Miles Lawler and Julianne O'Reilly assisted with catching animals, Stuart Millen assisted with data collection, and Nadia Marsh, James Bulinski, Mick Statham and Chris Fabian provided valuable discussion of management issues. Animals were caught and radio-collared under Parks and Wildlife Permit # FA96071 and FA97006, and University of Tasmania Animal Ethics Permit # 95052.

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## Evaluation of line-transect sampling to estimate nocturnal densities of macropods in open and closed habitats

K. le Mar<sup>A</sup>, C. Southwell<sup>B</sup> and C. McArthur<sup>A</sup>

<sup>A</sup>CRC for Sustainable Production Forestry, School of Zoology, University of Tasmania,  
GPO Box 252-05, Hobart, Tas. 7001, Australia.

<sup>B</sup>Australian Antarctic Division, Channel Highway, Kingston, Tas. 7050, Australia.

**Abstract.** Walked line transects were evaluated for estimating nocturnal densities of red-necked wallaby (*Macropus rufogriseus*) and the red-bellied pademelon (*Thylogale billardierii*) in a range of open and closed habitats. The use of cleared transect lines in densely vegetated habitats reduced noise produced by travelling, while permanent grid markers facilitated collection of perpendicular distance data at night. Results from sighting histograms indicated that animals did not display evasive movement in response to the observer before detection. The probabilities of detecting the macropod species varied significantly between habitats. Significant differences in probabilities of detection were also recorded within species between habitats. Consequently, line-transect sampling is recommended over strip-transect sampling for estimating species abundance when more than one species and/or habitat are of interest. Recommendations are made, however, for forest managers wanting to monitor macropod populations on newly established plantations, when line-transect sampling is not feasible and strip-transect sampling is the only alternative.

### 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, red-necked wallaby (*M. rufogriseus* – sub-species *M. r. rufogriseus*, Bennett's wallaby) and red-bellied pademelon (*Thylogale billardierii*), 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 broad 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.

Red-necked wallabies and pademelons have characteristics that make estimation of both relative and absolute abundance very difficult: they are relatively small, they are mostly or entirely nocturnal, they shelter during daylight in heavy cover, and they 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 red-necked wallabies and red-bellied pademelon, as well as other species with similar characteristics. In this paper we describe modifications to the line-transect methods previously used for large macropods and discuss the perceived success of these modifications.

## Study Area and Methods

### Study area

The 428-ha study area was located within the forestry company North Forest Products' 'Surrey Hills' Tree Farm, in north-west Tasmania (41°28'S, 145°48'E). Four habitats dominated the site: established plantation, native forest, grassland, and a prepared site that was planted during the fieldstudy (hereafter referred to as 'the coupe') (see Fig. 1).



**Fig. 1.** Map of the 428-ha study site in north-west Tasmania showing the four main habitat types: native forest, established plantation, grassland and a newly established coupe, plus waterways, cleared land and roads.

Vegetation within established 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. Established 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 to 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 coupe consisted of introduced herbaceous dicots, pasture grasses, native grasses and occasional shrubs (*Hakea* and *E. delegatensis* seedlings). Visibility on the coupe was excellent but interrupted by windrows spaced at 40–80-m intervals. *E. nitens* seedlings were planted on the coupe 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 coupe produced little noise.

### Transect lines

In total, 84 permanent transect lines were established. 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 1). Within each of the four habitat types, transect lines were located at random. The coupe was heavily sampled for its size, as this habitat type was of particular interest for a larger study. Spotlighting surveys were not conducted in the fifth habitat type (cleared land) due to extremely restricted visibility (often less than 1 m).

Transect lines were cleared in native forest to reduce noise produced while travelling. Cleared paths were wide enough to accommodate only 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 established 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 coupe 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 familiarising 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.



**Table 1. Properties of transect lines in different habitats and the number of observations recorded for the macropod species during field-trips used in estimating probabilities of detection (Field-trips 1–11)**  
Species/habitat combinations, where adequate sample sizes allowed line-transect analysis, are indicated with an asterisk

	Coupe	Grassland	Established 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 red-necked wallabies sighted	226*	55*	5	4
No. of red-bellied pademelons sighted	538*	244*	55*	14*

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 high) 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 1). A minimum of six distance categories was used. Within established plantations, tree rows lying parallel to the transect line were used to mark grid lines.

#### Survey

Data were collected over 11 field-trips between May 1997 and July 1998. Each field-trip consisted of six nights spotlighting (2–3 nights apart), taking place over 17 days. Field-trips were approximately four weeks apart with the exception of Field-trips 5 and 6, and 9 and 10, which took place immediately before and after forestry operations (planting [Trips 5, 6] and poisoning the coupe with Compound 1080 (sodium monofluoroacetate) [Trips 9–11]).

A two-person team walking in single file at 1 km h<sup>-1</sup> surveyed the line transects at night. Transect lines were divided into six groups of 14; one group was walked per night. Data collection commenced 2 h after sunset and was completed within 6 h.

In established plantations, native forest and the coupe, only one person searched with a 100-W spotlight. The second person recorded the data 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 (the 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 field-trip.

Data for individual transect lines on the coupe 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 recommenced 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 continued only 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.

#### 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 spotlights 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 coupe, 50 m for grassland, 35 m for native forest and 21 m for established 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 established 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 cm 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.* 1993). Data were analysed for individuals rather than clusters (Buckland *et al.* 1993) as red-necked 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 red-necked wallabies and pademelons within the same habitats, for red-necked wallabies between two habitats and for 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 red-necked wallabies and pademelons on the coupe. 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 (underestimation 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 distance transects (Buckland *et al.* 1993). Values for  $\hat{f}_{(0)}$  were calculated from data pooled over time (Field-trips 1–11) for each

species using the computer program DISTANCE (Laake *et al.* 1992) (le Mar, unpublished data). 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  $\text{cv}$  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

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

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

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

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

## Results

Sample sizes were large enough to produce valid sighting histograms for pademelons in three habitats (coupe, grassland and established plantation) and red-necked wallabies in two habitats (coupe and grassland) (Table 1, Fig. 2). Few pademelons (14) were sighted in native forest but they are included here for indicative purposes.

Few animals were detected on cleared paths: 2 of 57 pademelons in established plantation and 1 of 15 pademelons in native forest. These sightings were removed from the data set before analysis.

The sighting histograms were shouldered (Fig. 2f), relatively flat to Distance Category 4 (Fig. 2a, e) or gradually stepped (Fig. 2b–d). 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 red-necked wallabies on the coupe (30 m).

Detection probabilities for pademelons and red-bellied wallabies varied significantly between habitats (Table 2). On the coupe, the probability of detecting red-necked 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 coupe, followed by grassland, established plantation and native forest (see Table 2).

Results for the percent relative bias (PRB) and percent relative cv (PRCV) are given in Table 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 red-necked wallabies, 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-tran-

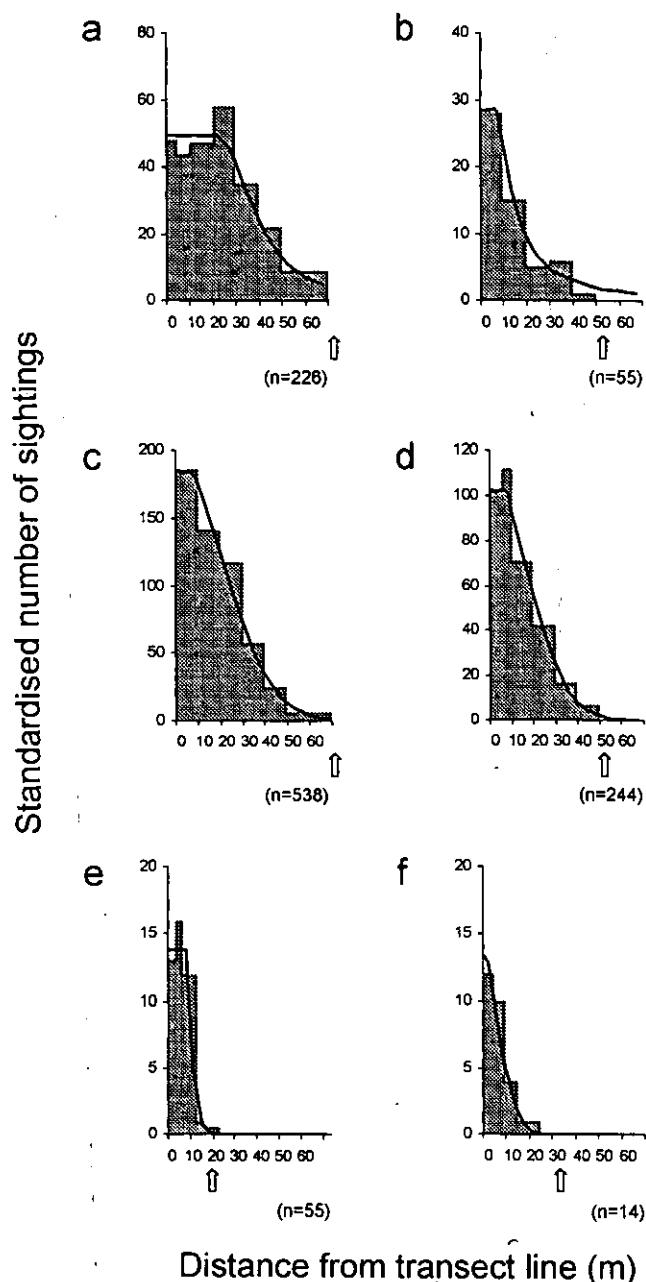


Fig. 2. Sighting histograms and fitted detection functions for red-necked wallabies in two habitats and red-bellied pademelons in four habitats (Field-trips 1–11): red-necked wallabies (a) on the coupe, (b) in grassland, and red-bellied pademelons (c) on the coupe, (d) in grassland, (e) in established plantation and (f) in 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.

sect 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.

### 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).

*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 doesn't move, the observer will literally intercept it.

*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, minimising visual cues when on the coupe and in grassland habitats was important, and reducing noise produced while travelling was critical within native forest and established 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 the density of (untamed) populations exceeded 30 animals  $\text{km}^{-1}$ .

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

**Table 2.** Estimated detection probabilities (% cv) 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

Species	Coupe	Grassland	Established plantation	Native forest
Red-necked wallaby	0.8313 <sup>a</sup> (5.49%)	0.3802 <sup>d</sup> (18.33%)	n.a.	n.a.
Red-bellied pademelon	0.5550 <sup>b</sup> (3.70%)	0.4589 <sup>c</sup> (5.02%)	0.2358 <sup>e</sup> (7.21%)	0.2062 <sup>f</sup> (19.92%)

**Table 3.** The percent relative bias (PRB) and percent relative cv (PRCV) results for data for red-necked wallabies and pademelons for the coupe, using strip-transect analyses and varying strip half-widths compared with line-transect analysis

Strip half-width (m)	Red-necked wallaby PRB	Red-necked wallaby PRCV	Red-bellied pademelon PRB	Red-bellied pademelon 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

closed habitats, noise produced by travelling was minimised by clearing transect lines and access paths. Established plantation transect lines and paths were also raked before the first field-trip and at subsequent four-monthly intervals to remove dry leaves and twigs. To reduce visual cues on the coupe, 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 coupe to reduce the chance of this happening. Firstly, transect lines were positioned perpendicular to windrows so these structures could be utilised 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.

*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 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.

*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 generalised 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 generalised 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 concur with our pre-survey expectations. For example, we expected the sighting probability to be greater for red-necked 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 in distinguishing the pelage and silhouette of red-necked 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 established plantation). In contrast, red-necked 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 red-necked 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 red-necked wallaby.

In addition, results from truncated data sets with varying width distances for red-necked wallabies and pademelons on the coupe 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.

#### 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 red-necked wallabies or pademelons in the closed habitats (native forest and established plantation) or for red-necked 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 red-necked wallabies, pademelons, possums or rabbits in these respective habitat types may require greater effort than used

here. This could be achieved by increasing either the transect length or the number of replicates.

#### 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 new coupes and in established plantations. Within the more densely vegetated habitats (native forest and grassland), 20-m intervals may be too large for accurate data collection.

#### 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 coupes, 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<sup>-1</sup>, 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 minimised 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 red-necked wallabies and pademelons on a newly established plantation. The use of a 30-m strip half-width distance accurately estimated the density of red-necked wallabies on the coupe (PRB = 0.1) and was relatively precise (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 red-necked wallabies on the coupe. 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.

### Acknowledgments

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# Changes in marsupial herbivore densities in relation to a forestry 1080-poisoning operation

K. le Mar\* and C. McArthur

CRC for Sustainable Production Forestry, School of Zoology, University of Tasmania  
GPO Box 252-05, Hobart, Tasmania 7001, Australia

\*Corresponding author email: k.lemar@utas.edu.au

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## Summary

Tasmanian forestry companies commonly manage browsing damage to seedlings on plantations through the lethal control of native mammalian herbivores, using '1080' (sodium monofluoroacetate). Although this practice is controversial, there is a lack of published quantitative data on its success in reducing animal numbers. We present the first published study quantifying the effects of a poisoning operation on the densities of mammalian herbivores both on a poisoned plantation and in the surrounding environment.

Line transect surveys were used to monitor species densities before and after poisoning. The species targetted by the poisoning operation were the red-bellied pademelon (*Thylogale billiardieri*), the red-necked wallaby (*Macropus rufogriseus* subspecies *M. r. rufogriseus*), the common brushtail possum (*Trichosurus vulpecula*) and European rabbit (*Oryctolagus cuniculus*). One non-target species, the common wombat (*Vombatus ursinus*), was also monitored.

Poisoning significantly reduced pademelon density on the poisoned plantation (decline of 98%). A significant decrease in red-necked wallaby density was also detected at this time (decline of 60%) but a general declining trend was present within the data, so we interpret this result with caution. No significant effects were detected for possums, rabbits or wombats. The high kill-rate for pademelons may reflect the dominance of this species over others at bait stations and/or greater sensitivity to 1080. Six weeks after poisoning, the density of pademelons on the treated plantation had increased but was still significantly lower than before poisoning. At the same time, red-necked wallaby density on the treated plantation had increased, with animals moving onto the plantation from surrounding habitats. The decline of pademelons was hypothesized to enable an influx of red-necked wallabies onto the poisoned site, through reduced inter-specific dominance behaviour.

**Keywords:** poisoning of animal pests, pesticides, population control, forest management, population ecology, wild animals, *Thylogale billiardieri*, *Macropus rufogriseus*, *Trichosurus vulpecula*, *Oryctolagus cuniculus*, *Vombatus ursinus*, Tasmania, Australia

## 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' 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 targetted 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 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 red-necked 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).



## Methods

### Study site

The 428 ha study area is in North Forest Product's Surrey Hills Tree-Farm, north-west Tasmania ( $41^{\circ}28'S$ ,  $145^{\circ}48'E$ ). Four habitats dominated this site: (1) a young *Eucalyptus nitens* plantation with relatively high weed cover, 17.8 ha; (2) older plantations of *E. nitens* (5–7 years of age, about 5 m tall), 274.7 ha; (3) grassland, 38.4 ha; and (4) native forest (rainforest and wet eucalypt forest), 89.1 ha (Fig. 1). The young plantation was planted with *E. nitens* seedlings (about 20 cm high) 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® (glyphosate) herbicide about one week after the *E. nitens* seedlings were planted.

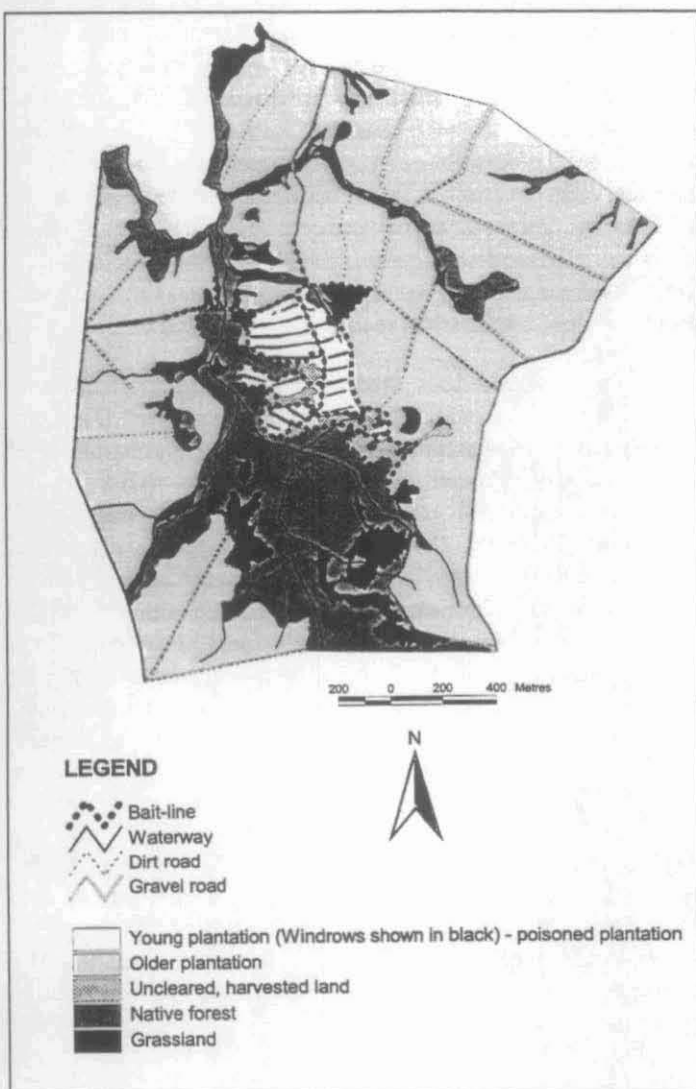


Figure 1. Map of the study site, showing the four dominant habitats and the location of the bait-line

### 1080 poisoning

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 kilograms 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 kilograms 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 Fig. 1) three days after the final free-feed. Bait stations were checked for remaining poisoned bait 24 hours after distribution.

### 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 in le Mar *et al.* (2001). 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.

Only one young plantation was available within this study area, consequently, 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. Second, transect lines were distributed at random throughout the habitat but with a minimal distance of 50 m between transect lines.

Transect lines were surveyed at night by a two-person team walking in single file at  $1 \text{ km h}^{-1}$ , 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.

### 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.* 1998). 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 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).

### Regional density

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

$$\hat{D}_R = \Sigma [\hat{D}_i \cdot (A_i / A)] \text{ and } SE(\hat{D}_R) = \sqrt{[\Sigma \{(A_i / A)^2 \cdot \text{var}(\hat{D}_i)\}]}$$

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

### 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 a  $\alpha=0.012$  after Bonferroni adjustment for multiple comparisons. Statistical analyses were conducted using the univariate procedure in SAS (SAS Institute Inc. 1990).

### Results

Line transect analysis was possible for pademelons in three habitats (young plantation, grassland and older plantations), red-necked 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 observed in only one habitat (young plantation), while wombats were observed in three habitats (no sightings in native forest).

Regional density results showed that pademelons were the dominant species within this environment, followed by red-necked wallabies, wombats, possums and rabbits (Table 1). The five species' populations were relatively stable throughout the study (Fig. 2).

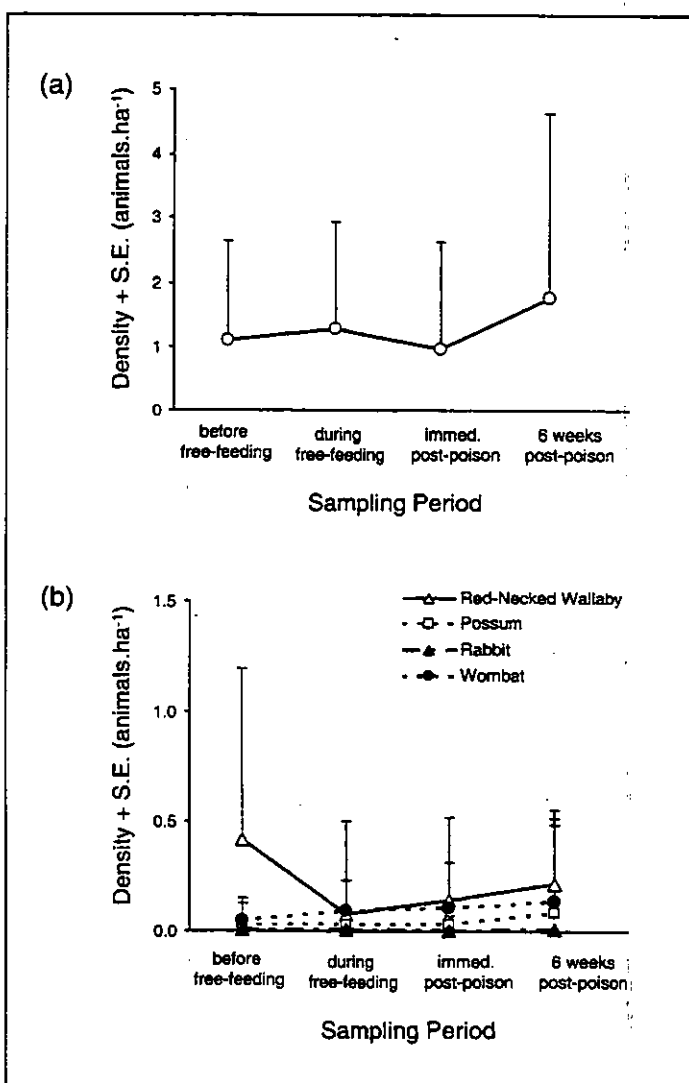
Before free-feeding the ranking of species' densities on the young plantation was consistent with the regional pattern (Table 1). Estimated densities of pademelons and rabbits on the young plantation did not change significantly during, compared with before, free-feeding (Fig. 3) ( $S = 13$ , d.f. = 23,  $P = 0.528$ ;  $S = 0.00$ , d.f. = 23,  $P = 1.00$ , respectively). Estimated densities of red-necked wallabies and wombats on the young plantation tended to be lower during, compared with before, free-feeding but the differences were not significant (Fig. 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.

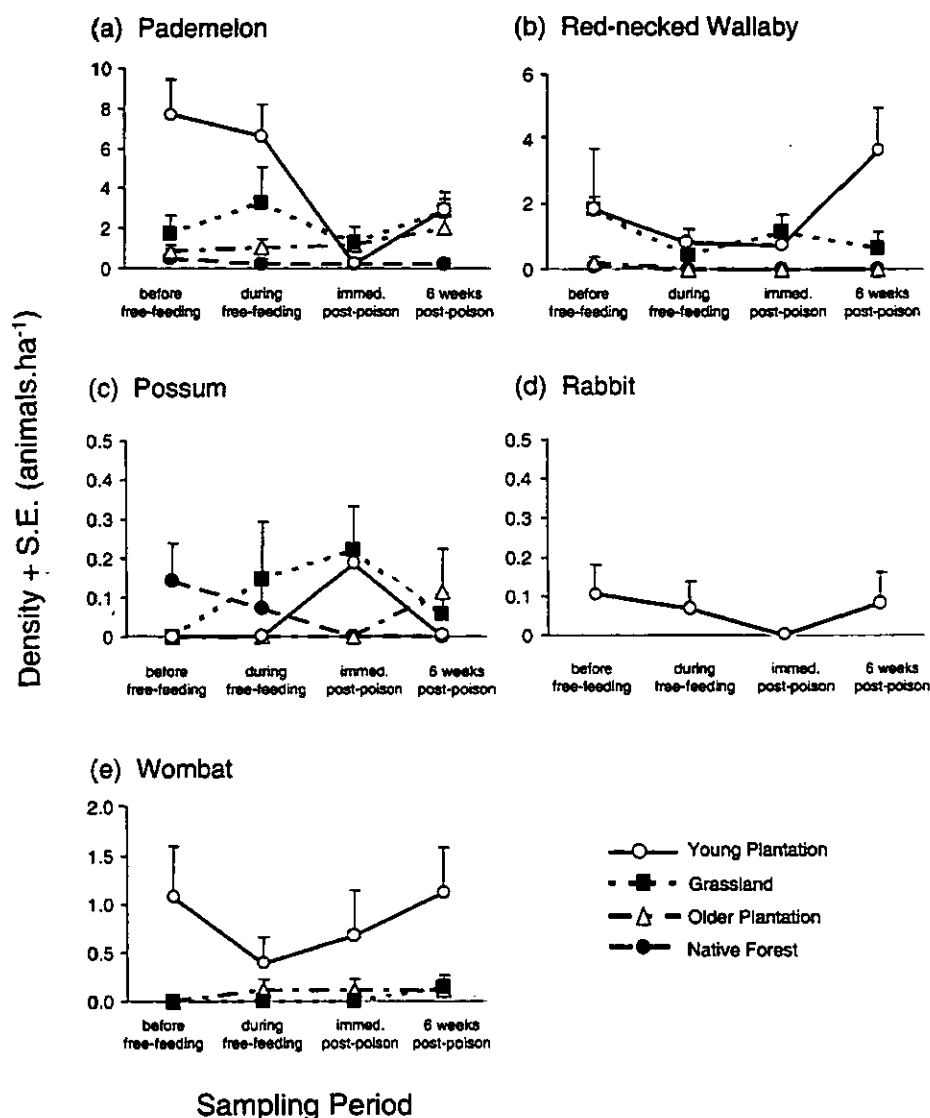
**Table 1.** Species' densities ( $\pm$  S.E.) at a regional scale and on the young plantation before free-feeding

Species	Regional density ( $\pm$ S.E.) (animals ha <sup>-1</sup> )	Density ( $\pm$ S.E.) on the young plantation (animals ha <sup>-1</sup> )
Pademelon	1.12 $\pm$ 1.51	7.73 $\pm$ 1.70
Red-necked 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



**Figure 2.** The estimated regional densities ( $\pm$  S.E.) of (a) pademelons and (b) red-necked wallabies, possums, rabbits and wombats before free-feeding, during free-feeding, immediately post-poisoning and six weeks post-poisoning

Pademelon density declined significantly between pre-free-feeding and immediately post-poisoning on the young plantation ( $S = 65$ , d.f. = 23,  $P = 0.000$ ) (Fig. 3a), but not in other habitats ( $P \geq 0.34$ , in all cases). Red-necked wallaby density on the young plantation also declined significantly during this period ( $S = 32$ , d.f. = 23,  $P = 0.008$ ) (Fig. 3b), while density in other habitats remained unchanged ( $P \geq 0.5$ , in all cases).



**Figure 3.** The estimated densities (mean + S.E.) of (a) pademelons, (b) red-necked wallabies, (c) possums, (d) rabbits and (e) wombats in four habitats (young plantation, grassland, older plantations and native forest) before free-feeding, during free-feeding, post-poisoning and six weeks post-poisoning

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) (Fig. 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 was unchanged throughout this period ( $P > 0.1$ , in all cases) (Fig. 3a). Six weeks after poisoning, red-necked wallaby density on the young plantation was again similar to before free-feeding ( $S = -22$ , d.f. = 23,  $P = 0.268$ ). Red-necked wallaby density in other habitats did not change significantly during this period ( $P \geq 0.5$ ) (Fig. 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) (Fig. 3c-e).

## Discussion

Data presented here provide the first quantitative information on absolute changes in animal densities in relation to a poisoning operation. 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 comparing single site results with results from other studies, if available. Where possible, we do this in the following sections.

### Pademelons

The relative stability of the regional 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 targetted during this operation. Poisoning effectively targetted 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 ( $n=25$ , K. le Mar *pers. obs.*; le Mar and McArthur 2000). This result is similar to findings from Marsh (1998) and Bulinski (2000). Marsh (1998) 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 red-necked wallabies (N. Marsh *pers. comm.*). The possible dominance of pademelons over red-necked 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 red-necked 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).

### Red-necked wallabies

As with pademelons, regional densities results for red-necked wallabies suggested that any effects of the poisoning operation were localised. Free-feeding appeared to cause a slight but not significant decline in red-necked 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. Red-necked 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. Red-necked 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 (K. le Mar *pers. obs.*; le Mar and McArthur 2000). Mixed results have been reported for the effectiveness of 1080-poisoning on reducing red-necked 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, red-necked 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 red-necked 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 red-necked wallabies have not been investigated, but are clearly important before the relative impact of these two species can be determined.

### 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 (K. le Mar *pers. obs.*; le Mar and McArthur 2000).

### Wombats

Density changes for wombats on the young plantation followed a similar pattern to red-necked 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 (K. le Mar *pers. obs.*), although it cannot be inferred that no animals were killed, since carcasses may have been in burrows.

### 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 indicate a large effect of 1080 on the local pademelon population and at least some effect on the red-necked wallabies. The fact that these patterns are similar to those found by Marsh (1998) and Bulinski (2000) 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.

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