

Direct impacts of logging and forest management on the brush-tailed phascogale *Phascogale tapoatafa* and other arboreal marsupials in a jarrah forest of Western Australia

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ABSTRACT

The direct impacts of logging on the arboreal, solitary brush-tailed phascogale were investigated in jarrah forest in south-western Australia. Individuals were monitored before, during and after logging in the Kingston Forest in 1995. The study was complicated by the effects of a drought, but sufficient radiotelemetry-based data were gathered to reveal a high level (31%) of foraging activity in logged areas, although negligible (1%) nesting in trees in logged areas. The most parsimonious explanation for the latter was that logging removed most hollow-bearing trees. Data concurrently collected on the number of possums found in the hollows of felled trees supported that view. Phascogale abundance declined throughout the logging study, but this was initially due to drought. When the research ended in 1997 a post-drought population recovery had occurred in the neighbouring unlogged area of Perup, but not in the logged area. No phascogales have been trapped by CALM since 1995 and only five have been seen during CALM spotlight monitoring (1995-2001) in Kingston and surrounding forests. Logging as conducted, coupled with management plans allowing logging of unlogged corridors/buffers, and a recutting rotation shorter than 120 years, would appear unlikely to provide for the long term hollow needs of arboreal marsupials in this area.

Key words: brush-tailed phascogale, logging, jarrah forest, possum, hollow-bearing trees.

Introduction

Logging of Australia's forests has been a major focus of political and scientific inquiry since the 1970s (reviews: Wilson 1991, Norton and May 1994, Recher 1996). In south-eastern Australia, many studies have examined the effects of logging on forest fauna, and several have incorporated or focused on arboreal marsupials (e.g. Tyndale-Biscoe and Smith 1969, Recher *et al.* 1980, Braithwaite *et al.* 1983, Meredith 1984, Lunney 1987, Smith and Lindenmayer 1988, Braithwaite 1991, Traill 1991). In contrast, until recently little such research has been undertaken in south-western Australia, which has comparatively few arboreal marsupials (Tyndale-Biscoe and Calaby 1975). Earlier south-western studies of the impacts of logging on animals were confined to birds (e.g. Abbott and Van Heurck 1985, Norwood *et al.* 1995).

It is generally accepted that reserve systems alone will not retain the original biodiversity of the forests (see Resource Assessment Commission [RAC] 1992; Margules and Pressey 2000, Lindenmayer and Franklin 2002) and in all States a range of fauna management practices is now incorporated in logging policies. Some are intended to mitigate the effects of logging on populations and while the practices vary between regions and forest types, most have certain features in common (Recher 1996). A major feature is the retention of hollow-bearing trees within logged areas for use by hollow-nesting animals (summarised in table 1 of Gibbons and Lindenmayer 1996). The size

of individual logging areas are now generally smaller than they have been in the past, and unlogged forest or corridors are commonly retained between cut areas. Logging in, or near stream zones and gullies are subject to restrictions. Although this is principally undertaken for hydrology reasons, it is considered to be beneficial to wildlife. However, there is little information on which to determine whether these practices reduce the negative effects of logging on wildlife.

Most Australian studies that have explored this issue have used a space for time approach; comparing animal densities and species richness between logged and unlogged areas (as cited in Wilson 1991, Norton and May 1994, Recher 1996, Alexander *et al.* 2002). Few studies have examined the response of species and individuals during the actual process of logging, with notable exceptions being the work of Tyndale-Biscoe and Smith (1969), How (1972), and Friend (1979). Those studies yielded information that cannot be gained in retrospective research, including the role of site fidelity in survival, the impact of predation, and the effect of timing on a species' responses. To refine logging prescriptions further there is need to increase understanding of how logging actually compromises animals.

This study investigated how the process of logging impacted on brush-tailed phascogales *Phascogale tapoatafa* (hereafter phascogale) during and after logging in jarrah *Eucalyptus marginata* forest of Western Australia

(WA). Knowledge of phascogale biology and behaviour at the outset of this study (Cuttle 1982, Soderquist 1993a, Traill and Coates 1993, Soderquist 1994, 1995a, 1995b, also see Rhind 1996, 1998, 2002, 2003, Scarff *et al.* 1998, Rhind and Bradley 2002) allowed predictive (alternative) hypotheses to be proposed. These and their rationale were that:

1. Phascogales would continue to nest in trees in logged areas, but largely confine their nesting to those trees retained as 'habitat' trees. Rationale: (i) phascogales nest in isolated trees; (ii) 'habitat' trees are selected for their value as hollow-bearing trees; and (iii) logging would decrease the availability of other suitable nesting trees.
2. Phascogales would travel through logged areas, but spend negligible time foraging in them. Rationale: (i) phascogales do move through relatively open areas; (ii) the majority of foraging occurs in trees, and (iii) logging reduces the number of trees on which to forage.
3. Territories of phascogales impacted by logging would be larger than those not impacted. Rationale: loss of foraging substrate would necessitate a compensatory increase in range.
4. The physical condition of phascogales in logged areas would decrease in comparison to the condition of those in unlogged areas. Rationale: (i) logging causes the loss of foraging substrate (ii) extra foraging effort and/or lower food availability would cause a decline in condition.

In addition, (5.) the fate of phascogales (and possums; see below) present in hollows at tree felling and (6.) the impacts of logging on phascogale abundance were addressed.

The ability to explore these hypotheses and issues was confounded by the worst drought on record coinciding with the study (detailed in Rhind 2002, Rhind and Bradley 2002). The drought's negative impacts on phascogales were considerable and resulted in poor growth, high mortality, delayed and atypical dispersal behaviour, unusual nesting behaviour, alterations in reproduction, and ultimately a population decline (Rhind 2002, 2003, Rhind and Bradley 2002). These made the original research design unrealistic and limited the ability to interpret results, but sufficient information was gathered to draw a number of reasonable conclusions on the impacts of logging on phascogales.

Methods

The study species

The brush-tailed phascogale is a medium-sized arboreal dasyurid marsupial found in woodlands and dry forests of mainland Australia (Soderquist 1995a). In most regions there is insufficient information to determine the species conservation status, but they are generally considered rare and of conservation concern. Phascogales are presumed extinct in South Australia, are listed as threatened in New South Wales (*Threatened Species Conservation Act 1995*), as vulnerable in Victoria (*Flora and Fauna Guarantee Act 1988*) and as Near Threatened in Western Australia (Maxwell *et al.* 1996).

Characteristics of the species that contribute to its vulnerability to decline include a dependence on old trees for nesting hollows; an arboreal foraging habit and the dispersed nature of its invertebrate prey (Traill and Coates 1993, Scarff *et al.* 1998); a solitary social structure (Soderquist 1995b); a requirement for large areas to meet its needs (Soderquist 1995b, although see van der Ree *et al.* 2001) and total male die-off each year (Cuttle 1982). Breeding is annual and the toll of the long lactation period (c. five months) is such that few females successfully breed more than once (Soderquist 1993a, Rhind 2002). This life history strategy means that population persistence is dependent on reasonable reproductive success every year.

Study sites and logging procedure

The logging study was conducted during 1995 in Kingston Forest, near Manjimup (34°10'S, 116°35'E) 350 km south of Perth, Western Australia. This work was part of a broader research project that was undertaken on phascogales in both the Kingston forest and neighbouring Perup Nature Reserve between 1992 and 1997 (Rhind 1998). To investigate the impact of logging, three sites were established in Kingston in 1993. Site choice was based on the location of logging activities proposed by the Department of Conservation and Land Management (CALM). One site (Kingston [K] 5) was identified as an impact site and it was logged in 1995; two sites (K4 and K6) were established as control (reference) sites (see fig.1 in Rhind and Bradley 2002). Each site contained a trapping area of approximately 200 ha and was surrounded by a mixture of cleared farmland and continuous forest.

The history of logging in the jarrah forests is described by Abbott and Loneragan (1986) and Stoneman *et al.* (1989). Much of the south-west region is dominated by open, dry sclerophyll forest containing only two major tree species jarrah *E. marginata* and marri *Corymbia calophylla*, and the understorey is typically sparse. In the study area trees are commonly 20-30 m in height, and occasionally exceed 40 m. In both the Kingston and Perup Forests, larger jarrah were removed during selective logging conducted between 1950 and 1970 (CALM staff pers. comm.). Prior to woodchipping in the 1970s, the marri were not taken during logging, so overall these areas had a relatively limited history of logging disturbance. Forestry practices in WA include regular burning and the K5 site was last burnt in 1987/88. These forests were not used for grazing by stock, in part due to the abundance of 1080-producing plant species which are toxic to introduced animals (see Twigg *et al.* 2003).

Relevant silvicultural practices in the jarrah forest are discussed in detail in CALM (1991, 1995) and Stoneman *et al.* (1989). Such practices are revised from time to time, and the following description applies to the period of this study. Two logging styles, 'gaps' and 'shelterwood', were used during the study and are described in Abbott *et al.* (2003). In areas cut to 'gaps', all merchantable timber was felled; in 'shelterwood' areas, typically more trees were retained to provide seed for regeneration. The maximum size of gaps was 10 hectares, and unlogged

corridors of no less than 100 m wide were retained between neighbouring gaps. There was no specified size limit on areas cut to shelterwood specifications, nor was there a requirement for uncut corridors to be left between shelterwood logging and adjoining gaps. Under both prescriptions, three habitat trees per hectare, or 15 trees per 5 hectares, were retained for hollow-nesting fauna (CALM 1991). Gullies were excluded from logging. The K5 logging site was dominated by jarrah and marri, and both species were cut during logging.

Logging of K5 commenced in February 1995 and ceased in May when winter rains began. This area is subject to quarantine regulations to reduce the spread of the jarrah dieback fungus *Phytophthora cinnamomi* and soil

movement is restricted in wet weather. Effectively, this reduces vehicle use to main roads unless wash-down procedures are undertaken. Further logging was undertaken at K5 in early 1996, and Figure 1 shows the site after logging in January 1996. The site was burnt in November 1996 as part of the routine post-logging procedure. Concurrent with this research was a larger study of the impacts of logging undertaken by CALM in the Kingston area and surrounds. As part of that project, intensive fox baiting was implemented from June 1993. A range of published papers (e.g. Morris *et al.* 2000, Abbott *et al.* 2003) and unpublished CALM reports (e.g. CALM 1996, 2001; Wayne *et al.* 2000, 2001) describe that work.



- Area containing nestboxes and used for trapping,
- - - - Area of radiotracking as shown in Figure 2.

Figure 1. Post logging appearance of the K5 site in January 1996 (aerial photo 950422; WA 3695, Frame 5019; Central Mapping Agency, Department of Land Administration, WA).

Study methods

Phascogales were captured using a combination of trapping and nestbox survey methods (described in Rhind and Bradley 2002) and monitoring was carried out by radiotracking. Initial research on phascogales at Kingston began in May 1993. The logging impact study commenced in December 1994 and ended in early August 1995 with continuous fieldwork throughout that eight-month period. Further surveys of nestboxes were undertaken in 1996 and in early 1997.

The research was designed around a BACI (Before/After/Control/Impact) experimental model (Green 1989) and focused on monitoring individuals. The low density and annual life span of the species, combined with the timing and location of proposed logging activity, governed decisions on the location of the study sites, the timing of the study and the area that could be covered during the research. These logistics constrained research to a limited region, and to two control sites and one impact site. The species' response to drought, however, made the design unworkable. Poor survival of collared animals in early 1995 resulted in the decision to redirect the research effort to catching and radiotracking phascogales on the impact site, particularly those individuals being directly affected by logging. Prior information gathered on this species (1992-1994), plus data gained during monitoring phascogales on control sites (early 1995), and subsequent monitoring of phascogale numbers in Kingston and the neighbouring Perup Nature Reserve (1996-1997), provided the context for interpreting the results obtained.

Identification of nest-sites

Phascogales caught at Kingston after January 1995 were radiocollared using an expanding breakaway collar (Soderquist 1993b). They were radiotracked daily to their diurnal nest-sites from the first day of capture until 15 March 1995. After mid-March, tracking on the two reference sites (and the K5 phascogales not directly affected by logging) was undertaken every three to four days. Phascogales moving in areas being logged, or scheduled for logging, were tracked to their nest-sites almost daily until their death or disappearance. Die-off of the males became apparent in late June and radiotracking ceased in July 1995.

Radiotracking was undertaken on foot and nest-sites were located by isolating the radio signal to a specific tree or nestbox. All nest trees were individually measured, marked, and their location was plotted against known reference points on the site. The proximity of nest-sites to the logging activities/boundaries was recorded. Each tracked individual was recorded as using one 'nest-site location' per day as nests are usually occupied exclusively by one phascogale. However, in the autumn/winter of 1995 nest sharing was common (Rhind 2003), hence unique to that year were records of the same site in use by multiple animals on a given day. Repeated use of a nest-site over a number of days by the same animal is normal behaviour for this species (Rhind 1998). Phascogales also rest in hollows at night, but these sites were invariably identified during the day, so night use of nests was excluded from analyses.

Identification of foraging behaviour

Radiotracking to determine foraging locations and behaviour commenced 4 February (all sites) and ended in July 1995 with 11-12 nights per month typically spent tracking (range: 11-20 nights/month). After mid-March, such tracking was largely confined to the K5 site. Night radiotracking fixes were made by tracking directly to a phascogale and briefly observing it if it was active. To maximise spatial independence, a three-hour interval was imposed between fixes on the same individual (see Soderquist 1995b). The phascogale's foraging style was noted (foraging in a logged/unlogged area, in trees or at ground level) and its location was recorded by pacing on a compass bearing to known reference points. Tracking of phascogales spanned the whole night prior to May, after which time it was less successful and more limited. Phascogales spend a greater proportion of the night inactive in winter (Scarff *et al.* 1998), so fewer active foraging records are achieved per hour of observation. The males also become increasingly difficult to find because of ranging behaviour associated with breeding. Finally, rain prohibited vehicle use on many roads so access, hence search ability was restricted in winter.

Interpretation of data in relation to logging

Phascogales on the K5 site were classed as impacted by logging only if they were observed foraging or nesting within a logging area (because not all of the K5 site was logged); the remainder (K4, K6 and some K5 animals) were identified as control animals.

Temporal and spatial distinction between the disturbance phases (before, during and after logging) were defined on an individual phascogale basis. 'During' logging was the period between the first day an animal was observed in a logged area and when logging ceased within its known area of movement (home range or territory). The 'before' phase preceded this, the 'after' phase followed. The amount of time monitored in any one phase varied for each animal depending on the time of capture, logging and death/cessation of monitoring, and not all impacted phascogales were monitored in all three periods or for equivalent times. Accordingly, sample sizes for impacted phascogales range from 7 to 10 depending on the behaviour/phases being addressed.

Interpretation and display of spatial data (location of foraging activity, location of nest-sites, territories) were treated in a range of ways. To describe foraging behaviour, the proportion of foraging time spent in logged and unlogged areas (during and after logging combined) was tabulated for each impacted phascogale. To address nest-site use in logged areas, data were restricted to that gathered after logging had ceased in each individual's area. These post-logging nest-sites were mapped against an aerial photograph taken in January 1996 (Fig. 1; note that additional logging took place after the 1995 study and prior to this photograph being taken; this largely occurred in the southern part of K5 site and does not effect interpretation of the Fig. 2).

Territory and home range sizes

The impacts of logging on the size of territory/home range proved difficult to address. As detailed later, it is unsound to compare phascogale territory sizes without accompanying information on habitat characteristics, so the results presented are accordingly brief. General territory/home range sizes were calculated by combining all night foraging and day nest-site location fixes. These data were analysed using the 100% minimum convex polygon (MCP) method (Mohr 1947) in the Wildtrak non-parametric home range analysis package (Todd 1992).

Determining changes in animal weight/condition in response to logging

The effect of logging on phascogale condition was assessed by comparing changes in weight and tail fat measures between impacted and control phascogales. This required repeated measures on the same animals. Weight was measured to the nearest 0.5g and tail fat was recorded using a five-point scale (Serena and Soderquist 1988). To cater for variable weights between individuals (Rhind and Bradley 2002), the magnitude of differences between weights on recapture was compared, rather than the weights themselves. Phascogales are sexually dimorphic so, with sex as a factor, these data were analysed using a repeated-measures, general linear model for an unbalanced design. Observed power for these data was calculated using power analysis, as was the sample size necessary to detect a small variation in weight (defined as the difference in means/the common standard deviation of 0.556) at a power of 0.85 and a significance level of 0.05 (nQuery Advisor; Elashoff 1997). A power of 0.85 is generally recommended as an appropriate goal in behavioural or ecological studies (Thomas and Juanes 1996).

A similar test comparing tail-fat changes was undertaken using a one-sided t-test for unequal sample size ($P = 0.05$), but with the sexes combined as body condition appeared little influenced by sex at this time. Observed power for these data was also calculated.

Fate of animals at tree felling

Phascogales utilising logging areas were located at dawn prior to each day's logging. Those present in a tree being felled were closely monitored during the operation and over the following nights. Information on the fate of possums present in trees at felling was also gathered. This was undertaken as part of a concurrent investigation on hollows (Rhind 1998). One of the logging crew took considerable interest in the study, inspected trees after felling and maintained records on the number and welfare of the possums found in the felled trees. Those results are included here.

Phascogale numbers

Annual changes in phascogale numbers were determined from nestbox survey results obtained from similarly-timed post-dispersal surveys conducted

between 1994 and 1997. To examine changes in relation to logging, and to include 1997 results (where only one survey was conducted, cf. other years), only findings from those nestboxes available across all years were included. Limiting data in this way reduced bias as nestboxes in certain locations were used more often than others, and logging, road works and burning had destroyed nestboxes on two sites (K4 and K5); elsewhere some had fallen down or were uninhabitable.

Results

General

One hundred and eighteen different phascogales was captured on the Kingston sites between mid December 1994 and July 1995 (K4 = 35, K5 = 46, K6 = 37). Another 67 individuals were caught on the neighbouring Perup sites during this time. Of these, 50 were undispersed or dependent young and were too small to collar. Radio collaring at Kingston commenced 1 January 1995. Excluding phascogales that died/disappeared shortly after collaring, the following numbers were radiotracked in 1995: K4 = 13, K5 = 29, K6 = 16 (total = 58). The number of nest-site locations recorded on each site in 1995 were: K4 = 410, K5 = 1537, K6 = 616 (total = 2563). Ten of the radio-collared phascogales were directly impacted by logging on the K5 site.

Nesting behaviour

A total of 763 nest-site locations was observed in use by the 10 animals impacted by logging (Table 1). Of these, 329 were recorded after logging had ceased in each animal's area of movement. Only 1% of the post-logging nesting occurred in standing trees in the logged areas and none occurred in retained 'habitat' trees (Table 1, Fig. 2). Nesting in nestboxes in cut areas accounted for 8% of post-logging nesting instances. Six other instances of nesting in logged areas (1.8%) involved ground nesting in a log and among the remaining parts of felled trees.

Foraging behaviour

Foraging behaviours were recorded for seven phascogales impacted by logging. They spent an average of $31.5\% \pm \text{s.e. } 3.5$ of time (range 18.1-40.5%, observations = 202) foraging within cut areas (Table 2). Most of those observations included foraging on the ground among the logging debris.

Territory and home range sizes in relation to logging

The home range/territory boundaries of the 15 most intensively monitored phascogales (K5 site; 1995, Fig. 3) illustrate the variation in the area used by individuals. Examination of female territory sizes on this site (1994 and 1995) showed that variability is correlated with habitat type/quality, i.e. females

Table 1. Nest-sites recorded for 10 phascogales impacted by logging. 1a: tree shared once by two individuals; 1b: tree retained due to falling hazard (suspended limb), used twice; 1c: log used three times; 1d and 1e: nests among logging debris, each used once. Those individuals marked * are excluded in Table 2 (foraging observations) due to few night-time foraging records, and individual + is not represented in Fig. 2 (post-logging nest-site locations).

Sex	Tracking start	dates finish	Days tracked to nest-sites (before/ during and after logging)	Number of different nest sites identified (before during and after logging).	Number of different nest trees used in cut areas (after logging)	Number of ground nests in cut areas (during and after logging)
F*+	4/4/95	19/6/95	39	12	0	0
M	1/3/95	17/5/95	68	10	0	0
F	26/2/95	7/5/95	70	8	0	0
F	2/3/95	22/7/95	117	27	1a, 1b	1c
F	25/2/95	10/7/95	111	23	0	0
M*	10/5/05	6/7/95	39	10	0	1d
M*	22/5/95	27/6/95	24	10	1a	0
M	8/2/95	29/6/95	129	12	0	0
M	10/4/95	8/7/95	70	13	0	1c, 1e
M	11/3/95	8/7/95	96	20	0	0
Total			763	145	2	3



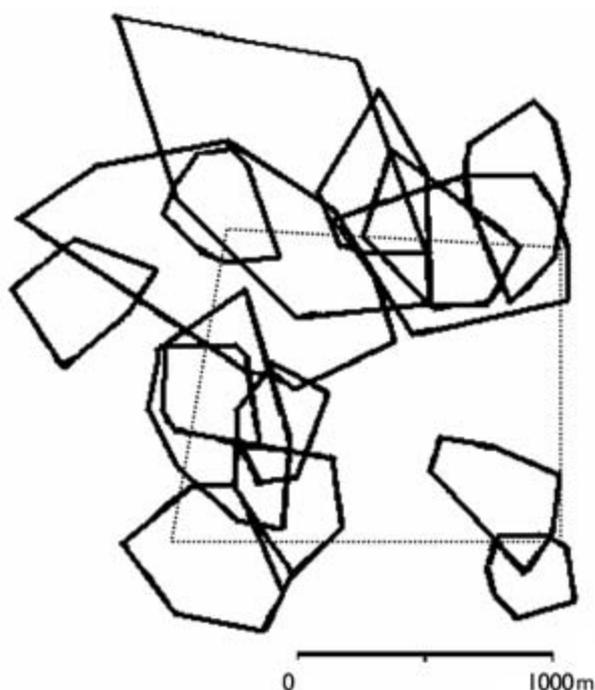
Figure 2. The location of post-logging nest sites ($n = 46$) used by nine phascogales impacted by logging. Two nest sites, and one phascogale (denoted + in Table 2), are not shown as they were outside the illustrated area.

residing in the unlogged gully system had significantly smaller territory sizes ($12 \text{ ha} \pm 3 \text{ s.d.}$, $n = 7$) than those occupying unlogged, adjacent upland areas ($19 \text{ ha} \pm 8 \text{ s.d.}$, $n = 4$; Rhind 1998). Data (all sites/years) also revealed differences in territory/home range sizes between the sexes and the time

of year. It is therefore invalid to compare territory/range sizes of phascogales, in relation to logging, without measurements of habitat characteristics and stratification of samples/results by sex, time of year and habitat type. The impacts of logging alone on territory sizes remain undetermined.

Table 2. The number of night-time foraging observations recorded for phascogales impacted by logging and the percentage of foraging events recorded in logged areas. Data presented are restricted to observations made after foraging was first noted (for each animal) within a logged area. Phascogales are not active all night (Scarff *et al.* 1998) and only those records where the animals were active are included.

Sex	Tracking start	Dates finish	Number of times observed foraging	Percentage of foraging observations recorded in logged areas
M	1/3/95	17/5/95	33	36.3
F	26/2/95	7/5/95	33	18.1
F	2/3/95	22/7/95	37	40.5
F	25/2/95	10/7/95	21	19.0
M	8/2/95	29/6/95	19	36.8
M	10/4/95	8/7/95	21	38.0
M	11/3/95	8/7/95	38	31.5
Total			202	mean = 31.5



Trapping area as shown in Figure 1.

Figure 3. The home range and territory boundaries of 15 phascogales tracked on the K5 site, 1995.

Animal condition

No differences were detected in changes of weight or tail fat between logging impacted/control phascogales, but observed power was low (i.e. weight = 0.14, tail fat = 0.06). Two factors influenced this result. Sample size was low for logging impacted animals ($n = 8$, not impacted $n = 24$). Power analysis indicated that repeated weight measures on an implausible 296 phascogales would have been required to detect a small effect-size with a power level of 0.85 and a significance level of 0.05. The primary reason, however, was that the already low weights and poor condition of phascogales in 1995 (Rhind and Bradley 2002) allowed little margin for weight and condition decline (i.e. males in 1995 already weighed 25% less than usual for males in this area). A weight loss of about 10 g in impacted individuals would have been detectable with power of 0.85 and $p = 0.05$, but in 1995 such weight loss would probably have resulted in death.

Fate of animals at tree felling (phascogales and possums)

Three radio-collared phascogales were present in trees during felling. All survived tree felling without apparent injury, although one hollow was opened with a chain saw due to concern that the animal was trapped by collapse of the 2 m long hollow entrance. These phascogales remained in their hollows after tree felling and exited at dusk as usual. Two survived for months afterwards, and one was killed by a predator three weeks after the felling of her nest tree.

On the study site, 65 possums (10 western ringtail possums *Pseudocheirus occidentalis*, 55 brushtail possums *Trichosurus vulpecula*) were found in the hollows of felled trees over an area of about 63 hectare in a 12-week period (pers. comm. E. Vermey, Rose and Bending Forestry Service, Manjimup, Western Australia). Seventeen percent of these possums died at tree fall.

Changes in numbers between years

Nestbox occupancy rates on the three Kingston sites were calculated by averaging occupancy results from two census periods within each year, 1994-1996 (presented in Rhind and Bradley 2002). Those results revealed a population decline that affected all Kingston sites in 1996, i.e. 1994: $23\% \pm 3.5$ s.e., 1995: 21 ± 1.5 , 1996: 7.5 ± 0.7 . A comparable census at Perup Nature Reserve showed a similarly low rate (5%) of nestbox occupancy in 1996. Nestboxes were not used by other species so these results are not confounded by competition.

One census of nestboxes was carried out in 1997. Between-year comparisons based on a single census are not ideal (e.g. January results were not comparable across years due to late dispersal in 1995, Rhind 2002), however, between-site differences were so marked within 1997 that these results are informative. On the Kingston sites, capture rates in 1997 were low (0-11%; Figure 4) and nestboxes showed little sign of activity, e.g. nests, depressions in nesting material, fresh scats (Soderquist *et al.* 1996). In contrast, high occupancy rates (26%), plus considerable activity in boxes at Perup indicated a post-drought population recovery.

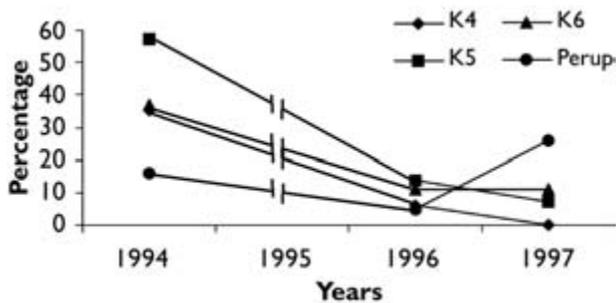


Figure 4. The percentage of nestboxes occupied by phascogales in January/February 1994, 1996, and 1997. Results are confined to nestboxes available across every year (K4 = 17, K5 = 14, K6 = 19, Perup = 19). Results from the 1995 survey were omitted due to delayed dispersal in that year: (An average 21% of Kingston nestboxes were occupied during later surveys in 1995).

Discussion

The continued activity of phascogales within areas after logging highlights their strong site fidelity, as has been found among other marsupials monitored during major habitat disturbances, e.g. greater gliders *Petauroides volans* (Tyndale-Biscoe and Smith 1969), brushtail possums and mountain brushtail possums *T. caninus* (How 1972), and woylies or brush-tailed bettongs *Bettongia penicillata* (Christensen 1980).

Absence of nesting in trees in the logged areas by these same phascogales could have several explanations: that remaining trees did not contain appropriate hollows; that otherwise suitable trees were avoided due to social, spatial or competition reasons; or that the phascogales preferentially chose to nest in the unlogged parts of their range. The latter alternatives are not supported by the broader information gathered on phascogales in this region (Rhind 1996, 1998, 2003). These phascogales readily used isolated trees, were not subject to interspecific hollow competition, concurrently nested in trees occupied by other species and, in 1995 (cf. other years; Rhind 2003) shared hollows with other phascogales. Further, nest-sites were scattered throughout each phascogale's foraging range and individuals often by-passed previously favoured nest-sites and travelled considerable distances at dawn to nest in their current site of preference.

Nest-site choice by phascogales appears to be governed by more than hollow availability or familiarity, or by nest-site proximity to current foraging areas. Social factors, such as maintenance and surveillance of territories/home ranges by visitation or occupation of nest-sites could explain why some phascogales chose to nest in unusual sites within the logged parts of their range, i.e. on the ground, among logging debris, and in logs. Nesting at ground level is rare in this species and accounted for 0.01% of all 4459 nesting records (Rhind 1998). Where observed, it was typically accompanied by other behaviours, which suggested duress, e.g. diurnal activity (Rhind 1998). During the logging study a variety of previously used sites in unlogged areas were known, available and well within range of the individuals that used ground nests within logged areas.

Overall, the most parsimonious explanation for lack of nesting in trees in logged areas is that few remaining trees contained suitable hollows. Data gathered on possums on this site support this view. Trees containing hollows suitable for possums and phascogales have a similar profile (Rhind 1998, Whitford 2002) and the number of brushtail possums found in felled trees on the K5 site (0.87/ha) exceeded the number of possums anticipated to be present in trees (0.42-0.80/ha; derived from pre-logging possum densities and possum refuge sites in CALM 1996, Morris *et al.* 2000).

Assuming correct population estimates, such a rate of encounter at tree felling could only occur if a significant proportion of trees suitable for possum nesting were felled. Mortality rates of CALM-monitored possums at Kingston confirm a high level of impact at the time of logging. Death rates of radio-collared brushtail possums were 34% in gaps and 67% in shelterwood areas within one week of logging (Morris *et al.* 2000), and 70% of ringtail possums died within three weeks of logging (Wayne *et al.* 2000, CALM 2001).

The logged sections of this site will remain hollow-deficient for over a century because jarrah and marri take a minimum of 130 years to develop hollows suitable for phascogales or possums (Abbott and Whitford 2002). In addition, planned logging rotations of 80-120 years will prevent the development of hollows, and the cutting of the unlogged corridors within 15-20 years (CALM 1995) will increase the extent of areas with limited hollows. Collectively these actions and policies appear inconsistent with the requirement that sufficient habitat trees will be retained on logged sites to meet the long term needs of arboreal fauna (see Abbott and Whitford 2002). Fortunately, the need to test the assumption that regrowth will provide adequate habitat when the corridors are logged has been recognised (Abbott *et al.* 2003). This would seem critical as a review of the value of regrowth to fauna (Doherty 1998) failed to identify any Australian experimental studies that have directly examined this question.

Several recent changes have been made to forestry policies in WA, including modification of habitat tree policies (CALM 2001, Conservation Commission of Western Australia: Forest Management Plan 2003). The number of trees to be retained has increased and guidelines on tree selection are more explicit. Nevertheless, predicting the hollow value of single trees in a local area remains highly problematic and leaving a given number of trees does not ensure that they will be useful to fauna (see Gibbons and Lindenmayer 2002). There are, however, now considerable data on the relationships between trees, hollows and fauna requirements in the jarrah forest (e.g. McCombe *et al.* 1994, Whitford 2001, 2002, Whitford and Williams 2002). This provides the opportunity for improving policy by basing it on the probability that trees of certain size/character will contain hollows suitable for fauna. An early example of this approach estimated that retaining 6 trees > 100 cm in diameter, plus 6 > 50 cm, per hectare could provide one possum hollow per 1.7 ha in this habitat type (McCombe *et al.* 1994). More meaningful habitat tree retention policies should be established as modelling and probabilities of correct tree selection are incorporated in calculations.

Increased information on species' requirements gained from the various studies at Kingston will also help guide future policy. Estimates of nest use by possums and phascogales in the study area suggest that 14.5 refuges/ha were collectively used by these species; 3.5 trees/ha (brushtail possums), 7.7 refuges/ha (ringtail possums) (from CALM 2001, Wayne *et al.* 2000) and 3.2 trees/ha (phascogales, Rhind 1998). Not all of the ringtail possums refuge sites cited above were in tree hollows, nor are trees mutually exclusive between species or individuals (cf. Inions *et al.* 1989). For example, at Perup, 16 individuals - four brushtail possums, three ringtail possums and a phascogale with her dependent young - concurrently utilised the one tree at the same time (Rhind 1998). At Kingston, 5 of the 17 (30%) felled trees with possums contained more than one possum, including one tree with three possums sharing the same hollow and another containing a ringtail and brushtail possum sharing the same tree. Identification and retention of trees with the highest probabilities of containing hollows could simultaneously protect refuge sites of several individuals and several species.

At Kingston, the availability of high quality or potentially useable habitat trees was estimated to be around 10/ha by both Rhind (1998) and Whitford (CALM 2001). Strong evidence of occupation (typically possum tracks) was found in 2.13 trees/ha on the K5 site (Rhind 1998). Dead trees were relatively uncommon in the study area (hence may be infrequently scored in surveys of hollow-bearing trees), but they are particularly important and accounted for 28% of the c. 700 different nest trees used by phascogales over the course of the research. Phascogales also showed a preference for making their maternal nests in dead or senescent trees (Rhind 1998). Dead trees are well recognised both internationally and within Australia as an invaluable source of hollows (Gibbons and Lindenmayer 2002). In NSW, their removal is now listed as a key threatening process under the NSW *Threatened Species Conservation Act 1995*. These trees are not protected in WA, nor is this proposed (Conservation Commission of Western Australia Forest Management Plan 2003) and dead trees are often cut down to provide firewood (Driscoll *et al.* 2000). Recommendations relevant to habitat tree retention for phascogales are given below.

Phascogales underwent a broadscale decline during the study and this was primarily due to the effects of drought (Rhind and Bradley 2002). Post-drought recovery was evident at Perup when this study ended in early 1997, but not at Kingston. That finding alone allows the suggestion to be made that logging and accompanying activities at Kingston had a negative impact on phascogales. Results from longer term trapping and spotlighting by CALM on both logged and neighbouring unlogged sites in the Kingston and surrounding areas indicate that this species is now uncommon across a large area of Kingston. CALM staff have not caught any phascogales during the five years of surveys undertaken since June 1995 and only five sightings were made during 189 spotlighting surveys conducted between November 1995 and January 2001

(CALM 2001, Wayne *et al.* 2001). The Perup/Kingston region contains the only known dense populations of phascogales in WA (Rhind 1998) so the failure of this most robust of populations to recover from the disturbance of logging warrants urgent attention.

The reasons for the sustained decline throughout Kingston are speculative, but can be related to the scale of logging operations, the species' life history strategy and the specific timing, number, and nature of the various disturbances. Unlike other mammals in this area, phascogales have a strictly timed life history and males are semelparous (Lee and Cockburn 1985, Rhind 2002). Persistence depends on a reasonable level of reproductive success every year, and a single bad year can have a disproportionately large adverse effect on the population. Three consecutive generations (*i.e.* three years) of phascogales were impacted by negative environmental conditions on the K5 site alone - drought (1994), logging and ongoing drought (1995), further logging and then burning (1996). The burning destroyed the ground debris that phascogales had foraged among after logging and it also removes *Banksia* species. *B. grandis* was the only identified source of nectar/pollen used by phascogales in this forest and it appears to be a seasonally important food (Scarff *et al.* 1998). Wayne *et al.* (2001) estimated that *Banksia* was reduced by 90% in logged areas. In addition, burning was carried out when females were at the energetically vulnerable stage of late lactation (see Soderquist 1993a) and reduction in food resources at this time would be detrimental to maternal survival and the female's ability to successfully wean the 1997 cohort of phascogales. In extent, the K5 study site was only one of several sites logged in this area and the scale of disturbance was extensive. One third of the surrounding 3600 ha was logged, burnt, and thinned from 1995 (figures 1 and text in Wayne *et al.* 2001 and Abbott *et al.* 2003). Logging and burning took place in neighbouring areas in 1997, the K4 phascogale reference site was logged between March 1997 and January 1998, and it was then burnt in November 1998.

Ringtail possums also declined (by 85%) in both the logged and unlogged parts of Kingston and surrounds, but not in the Perup Nature Reserve (CALM 2001). Their decline occurred in 1999, well after that of phascogales, and the reasons for the similar geographical trend between species are likely to be different. Ringtail possums share little in common with phascogales other than hollow dependence and a highly arboreal habit, but they do overlap in resource requirements with brushtail possums. Brushtail possums at Kingston increased 10-fold following fox baiting (table 3 in Morris *et al.* 2000). There is no comparable information on ringtail possum numbers in response to baiting. No data were collected on foxes or their effects on potential prey prior to baiting or logging, but it is fair to assume that fox control at Kingston underpinned the increase in brushtail possum numbers. Coupled with logging activities, this increase in the numbers of brushtail possums could have been disadvantageous to ringtail possums. Even if both possum species equally benefited from fox control (reduced predation), the greater competitive ability

of brushtail possums to monopolise shared resources (particularly hollows) could have resulted in ringtail possums being out-competed for limited resources. Both possum species are flexible in the types of refuge sites they use, but successful breeding by ringtail possums, hence recruitment, may be dependent on tree hollows.

Disentangling the effects of fox control from the impacts of logging, burning and drought are problematic given the data available. What is well known is that manipulating predator pressure can alter dynamics among predators, predators and prey, prey species, conspecifics, and it can also change the structure of the environment in which they live (e.g. Risbey *et al.* 2000, Soule *et al.* 2003). In addition to positive responses, neutral or negative responses may also have occurred among the various species at Kingston. For phascogales, there was no compelling evidence to suggest that foxes (or control measures) greatly influenced the changes in phascogale abundance observed during this particular study. Phascogales were readily caught at Kingston in May 1993 prior to baiting (Rhind 2002), they declined despite baiting, and they showed different population trends in two well-baited areas (Kingston and Perup). Fox control remains intensive in Kingston, and is presumably effective in reducing fox numbers, hence the sustained decline by phascogales cannot reasonably be attributed to fox predation.

Recommendations

The following recommendations resulted from this study.

1. *Monitoring of phascogales.* The post-logging status of phascogales at Kingston and surrounds needs to be determined. While there is an ongoing commitment to long term monitoring at Kingston (Conservation Commission of Western Australia, 2003), the trapping and spotlighting methods used are not well suited to detecting phascogales. Re-establishment and monitoring of existing nestboxes at Perup (as a reference site) and at Kingston, with monitoring twice a year between January and June would provide a cost-effective means for obtaining abundance data comparable to that gathered during this study. Collection of measures of animal weight, growth, sex, and condition would aid interpretation of findings if compared to information in Rhind (2002) and Rhind and Bradley (2002). Further radiotracking studies are not justified as extensive and intensive radiotracking, hence resources, would be required to gain any meaningful results.

2. *Fox baiting.* Fox baiting, and intensified baiting is advocated as part of routine logging procedure (Conservation Commission of Western Australia, 2003) especially in relation to ringtail possums (CALM 2001, Wayne *et al.* 2001). Interactions between ringtail and brushtail possums, and other potential effects and interactions caused by altering fox predation need to be examined before it can be assumed that baiting will be effective. Well controlled experiments with adequate control sites, and with emphases on monitoring behaviour rather than abundance, would be needed to avoid

deriving weak inferences. This would require considerable resources. There is no cost-effective way for including phascogales in such a study and currently there is little justification for doing so.

3. *Monitoring of retained habitat trees.* Use of trees retained as 'habitat trees' by hollow dependent species in logged areas of Kingston must be determined to identify their efficacy. Such data may already have been collected, but they are, as yet, unpublished.

4. *Dead trees.* Dead and dying trees must be protected from logging. This single action alone would safeguard a significant proportion of hollows used by phascogales (nearly one third; including maternal nest-sites) and by other species dependent on large hollows e.g. possums, cockatoos and owls. Longevity of dead trees as a standing source of hollows is limited but, on falling, they become valuable as hollow logs. Logging is identified by CALM (CALM 2001) as a practice that will limit log availability for a considerable time so protection of dead trees would additionally and immediately secure a future source of log refuges for endangered species present at Kingston such as numbats *Myrmecobius fasciatus* and chuditch *Dasyurus geoffroii*.

5. *Retention of trees and hollows in logged areas.* Setting targets and devising ideal tree and hollow retention rates in logged areas is difficult, but data gathered on phascogales allow the proposal to be made that retaining 3-4 hollows per hectare is a reasonable target for this species (Rhind 1998). The most meaningful way to devise and report on hollow retention rates is to focus on the probabilities of trees containing hollows, rather than focusing on tree numbers. For example, analyses of simple measures of the major predictors of hollows (e.g. tree diameter and condition) taken during marking trees for retention prior to logging would allow tree retention rates to be adjusted to achieve a desired number of retained hollows.

6. *Unlogged corridors/buffers.* Current policy, allowing logging of corridors/buffers within 15-20 years of the main logging activity, should be revised/deferred until it is established that the regrowth in logged areas will support the habitat requirements of fauna. No studies in Australia have directly examined this issue so such research would be a major contribution to the conservation of forest fauna.

Conclusion

This study, combined with longer-term information gathered by CALM, documented a sustained post-logging decline of phascogales in an area where they were previously abundant. Logging and associated activities are strongly implicated. If phascogales in all areas of the southwest respond to logging as observed in this study, then there is good reason to be concerned for their long-term viability. There is a long history of logging in the jarrah forest, with more recent (>1986) rates averaging 17,000 hectares each year (see Abbott and Whitford 2002). The species is already rare or uncommon throughout much of its remaining southwest range and it is now largely confined to the jarrah forest. Establishing the best forestry management practices will therefore be critical for ensuring their conservation.

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APPENDIX I



Nestboxes. Phascogales were captured from nest boxes and by trapping (photo 2). Nest boxes were placed at a density of about one per nine hectares (every 300 m), and originally at a height of 3-4 m which required two people to check the boxes as shown here at Perup. Nest box height subsequently proved to be unimportant so the boxes at Kingston were typically erected at around 2-3 metres and could be easily checked by one person. Boxes were placed on the largest trees available and the one illustrated is a mature marri. The understorey species in the foreground is a *Banksia grandis*, the only species identified as a source of nectar for phascogales in the study area.

Photo: S. Rhind.



Trapping. Phascogales are generally trap-shy, but can be successfully captured using large Elliott box traps (Elliott Scientific Co., Victoria; size 460 X 150 X 155mm) placed off the ground on large logs and stumps. Door locking devices proved essential to prevent phascogales from escaping, and a vertical bar was fitted over the open end of the trap to stop the abundant woylies (brush-tailed bettongs) entering the traps. Spacing between traps was 100-300 m and they were wrapped in hessian to provide insulation, and baited with oats, honey and sardines. Routine trapping was carried out for 4 nights, but as recapture can cause trap stress all traps within 300m of a trapped individual were closed down for one night after capture. Photo: S. Rhind.



Brush-tailed phascogale. The brush-tailed phascogale is the most widely distributed of extant dasyurids in Australia and one of the few arboreal marsupials present in the jarrah forest. South-western Australia has no gliders, only two large possum species, one *Antechinus* (*A. flavipes*) and one pygmy possum species – and the latter two species are not present in the Perup/Kingston area. Limitation in food availability is one of the most likely explanations for the comparatively depauperate nature of the arboreal fauna in the southwest forests. Featured here is one of the study subjects carrying a 3g radio collar (with whip aerial visible). Ear tags damage phascogale ears so ear tattooing was used to mark individuals; the tattooed '0' is clearly visible on the right ear.

Photo: J. Lochman.



Radiotracking. Recapture rates are typically very low with phascogales so radiotracking was the primary means of gaining information on phascogale behaviour. In this relatively open habitat the radio signal range was typically around 300-400 m when phascogales were in trees. During ground movement, radio-tracking range would deteriorate to 100 m. Despite the species relatively small size they move over large areas with distances of 0.5 km not unusual between nest-sites; dispersal distances - where known - were 6+ kilometers.

Photo: S. Rhind.



Nest boxes. The nest boxes placed up in the Kingston study area were rapidly used by phascogales. The tight entrance (c. 35 mm), and pre-filling the boxes with shredded bark were considered important factors in influencing the speed with which phascogales adopted these boxes. Internal volume of most nest boxes was about nine litres.

Photo: J. Lochman.

APPENDIX I



Checking nest boxes at Perup in 1994-95. The Perup Nature Reserve was chosen as the original study area because of reports that phascogales were using nest boxes that had been put up for birds in the 1980s. It is now clear that phascogale numbers at Perup (and Kingston) are atypical of phascogales in southwestern Australia. Nest box surveys (1993-1996) of 300 nest boxes spread over 10 sites in the northern jarrah forest (representing 460 'nest box years') have failed to catch any phascogales.

Photo: J. Lochman.



Post-logging, Kingston 5 site, mid-1995. This photo shows the typical appearance of one of the logged areas after logging had been completed in 1995, but prior to the post-logging burning that followed in November 1996.

Photo: G. Rhind.



Logging landing. Small parts of each logged area were completely cleared of trees and were the area used to stack and sort logs prior to removal on logging trucks. These logging landings can be clearly seen in Figure 1.

Photo: S. Rhind.



Post logging debris. A considerable amount of fallen timber and branches remained after logging had been completed. One of the unexpected findings of the study was that phascogales – normally reluctant to spend time on the ground - foraged extensively amongst this ground debris. And it may have initially represented profitable foraging as the extensive structural damage to the trees could well have exposed quantities of under-bark invertebrates. Burning in 1996 destroyed most of the ground residue. Photo: S. Rhind.



Phascogale hollows. The presence of collared phascogales in trees being felled provided a means for following their welfare at tree fall, and for definitely identifying phascogale hollows. This dead branch contained a collared male phascogale (camera lens cap provides scale). He remained in the hollow throughout the day, exited at dusk as usual without any sign of injury, and survived for weeks afterwards.

Photo: S. Rhind.



Hollows that contained phascogales at tree fall, such as this one shown intact in photo 9, were dissected to determine the characteristics of hollows used by this species. Searches of felled trees for possums *in situ*, and other felled trees previously used by phascogales also led to the identification of hollows. Where an animal was present, its welfare was checked if possible. After the hollow was vacated (typically the next day), the hollow was opened with a chainsaw, measured, and its contents recorded. The characteristics of 20 phascogale hollows in 18 trees were documented during the study. Phascogale hollows were most commonly found in the main trunk of the tree (70%).

Photo: S. Rhind.

APPENDIX I



Logging also provided the opportunity to measure possum hollows and so identify differences between those used by possums and phascogales. The characteristics of 27 possum hollows (identified in 24 trees) were documented during the study. Major size differences were apparent between the hollows used, but in common with phascogales most possum hollows (73%) were found in the main trunk and first-order branches.

Photo: S. Rhind.



Nesting tree. Phascogales depend on hollow-bearing trees for their nest sites. Dead trees, as shown here, were relatively uncommon in the study area, but accounted for 28% of the c. 700 nesting trees identified during the study (1992-1995), and 88% of trees containing nursery hollows were either dead or senescent. Dead trees are not protected during logging, but changes to logging policy to protect these would safe-guard one of the major sources of nesting sites for phascogales and other hollow-dependent species.

Photo: S. Rhind.