Influence of woodland remnant edges on small skinks (Richmond, New South Wales)

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Abstract   Habitat fragmentation due to clearing often results in an increase in edge areas compared with overall remnant size, but there are limited data on the influence of increased edge areas on generalist species. Therefore, the abundance of small (<55 mm) generalist scinid lizards in small woodland remnants of the Cumberland Plain (western Sydney, Australia) was compared. Using pitfall trapping and points of transect survey techniques, three species were encountered: *Lampropholis guichenoti*, *Lampropholis delicata* and *Cryptoblepharus virgatus*. The abundance and diversity of these skinks differed significantly between edge and core areas; fewer individuals were encountered on the edge of the remnants than in core sites. *Cryptoblepharus virgatus* was only observed in edge sites where *L. guichenoti* made up the greatest skink abundance. In contrast, *L. delicata* abundance was greatest at core sites. There was also a significant difference between the edge and core with respect to the size structure of the *L. guichenoti* population. Larger numbers of subadults were observed in the edge sites, whereas there were significantly fewer adults and juveniles in these areas. Habitat characteristics did not account for the skink distribution observed.

Key words: *Cryptoblepharus virgatus*, Cumberland Plain, edge effect, *Lampropholis*, remnant vegetation, skink, woodland.

INTRODUCTION

Vegetation clearance may radically change the structure of the landscape and isolate natural areas into increasingly smaller remnants (Mader 1984; Lunney 1987; Fox 1990; Saunders *et al.* 1991). In Australia, this phenomenon has resulted in extensive habitat fragmentation (Kitchener *et al.* 1980; Mader 1984; Lunney 1987; Reed 1991; Saunders *et al.* 1991; Bromham *et al.* 1999) with a general expansion of perimeters relative to overall remnant areas. These induced edges may have negative effects on resident populations (e.g. localized extinction) and encourage the expansion of exotic species (Yahner 1988; Saunders *et al.* 1991; Murcia 1995).

To overcome fragmentation, extensive remedial work has been undertaken in Australia. Often the focus has been on corridor formation, via mass tree planting (Tendy 1987). This approach to restoration has further expanded edges in the landscape, especially in areas where these newly formed corridors are narrow and mimic edge ecosystems in their entirety (Sisk & Margules 1993).

The negative impact of edges on the abundance and diversity of endemic species is relatively well documented (Kitchener & How 1982; Mader 1984; Soulé *et al.* 1991; Haila *et al.* 1993; Fleischner 1994; Chiarello 1999; Matthews *et al.* 1999; Berry 2001; Campi & MacNally 2001; Davies *et al.* 2001). Past studies have addressed the decline or loss of specific species due to habitat loss (Soulé *et al.* 1991; Sumner *et al.* 1999; Berry 2001; Campi & MacNally 2001), microclimatic changes (Davies *et al.* 2001), increased competition (Abensperg-Traun *et al.* 1996; Bromham *et al.* 1999) and predation (Matthews *et al.* 1999). However, the focus has generally been on the impact of edges on specialized species (e.g. *Oedura reticulata*, Sarre *et al.* 1995; *Hemibelideus lemurides*, Laurance & Laurance 1999; *Gnypetoscincus queensladiae*, Sumner *et al.* 1999) and there is a paucity of data on generalist taxa (Brown & Nelson 1993; Sarre *et al.* 1995). It has been generally implied that edges are more conducive to habitation by generalists, than specialist species (Leopold 1933; Shaffer 1981; Mader 1984).

Studies have been undertaken on edge usage by reptiles (e.g. Gregory 1978; Gates 1991; Kremaster & Bunnell 1999) but there are limited data on the influence of edges on small endemic generalist skinks. Despite the substantial diversity in skink species (Cogger 2000), most data on habitat usage by such species have been restricted to the impacts of logging (Lunney *et al.* 1991; Webb 1995; Goldingay *et al.* 1996), fire (Taylor & Fox 2001), mine rehabilitation (Twigg & Fox 1991; Taylor & Fox 2001) and grazing (Abensperg-Traun *et al.* 1996; Smith *et al.* 1996). In general, these data indicate that small skinks are the first to invade disturbed areas.

A number of these small skinks are widespread in Eastern Australia (Cogger 2000). Some, such as members of the genus *Lampropholis,* thrive in a broad range of highly modified habitats, including urban gardens (Burgin 1993). Despite them being relatively common, there are limited data on *Cryptoblepharus virgatus*, *Lampropholis delicata* and *Lampropholis guichenoti*. Such skinks have relatively small home ranges (e.g. *L. delicata* and *L. guichenoti*; S. Burgin, unpubl. data) and are less able to disperse or use multiple habitats than are many larger species that have previously been used as the basis for habitat assessment (e.g. mammals, Chiarello 1999; birds, Hagan *et al.* 1996). Small skinks may therefore provide information on the impact of changes to local ecosystems, such as those caused by reduced remnant size. In the present paper we investigate the vulnerability of these endemic generalist skinks to edge effects and the impact habitat alteration has on their abundance and diversity.

Site description

The present study was conducted in remnant woodland of the Cumberland Plain, on the Hawkesbury campus of the University of Western Sydney near Richmond (150°75′E, 33°62′S), New South Wales, Australia, during the autumn and spring of 1999.

The Cumberland Plain effectively encircles western Sydney. In the north it stretches west from Parramatta to the Blue Mountains and in the south to Campbelltown. Richmond is located on a flat plain in its northwest sector (Benson 1992). Historically the area was covered by tall, open forest, dominated by eucalypts (e.g. *Eucalyptus moluccana, Eucalyptus tereticornois*) and melaleucas (e.g. *Melaleuca decora*). Dominant understorey shrubs included *Bursaria spinosa* and *Acacia parramattenis* (Robinson 1991; Benson 1992).

Since European settlement, more than 94% of the native vegetation has been removed from the Cumberland Plain for agriculture and urban development (Benson & Howell 1990). Remnant forest stands are fragmented and patchy (Robinson 1991; Benson 1992).

METHODS

Remnant description and selection

The present study spanned seven remnant woodlands that ranged in size from 20 to 176 ha (Fig. 1). The largest remnant is divided by a number of fence lines. All remnants contained *Eucalyptus* and *Melaleuca* forests interspersed with exotic weeds, such as Blackberry (*Rubus* spp.), *Lantana* spp. and Kikuyu grass (*Pennisetum clandistinum*). The remnants are bounded by agricultural land, easements and/or roadways. Although the time of initial fragmentation is unknown, current land use practices have been in place for 50–110 years, and fire has been excluded for more than 20 years (S. Pearce, pers. comm., 2001).

Four perimeter sites (1–4) in remnants of 20–56 ha and four core sites (5–8) in remnants of 52–88 (ha) were selected (Fig. 1). Edge sites 1, 2 and 4 were intersected by powerlines and were disturbed by infrequent maintenance checks. Sites 2 and 3 were grazed by cattle during the present study. All sites were affected by traffic and aircraft noise.

The climate in the Richmond area is temperate with warm summers and cool winters. The average long-term maximum temperatures (1940–1991) range between 16.4°C in July to 26.9°C in January (minimum 3.6°C–17.6°C). The average long-term maximum rainfall for the period 1940–1991 was 805 mm, with most falling in the summer months (lowest monthly mean = 34 mm, highest = 99 mm; Ferraro & Burgin 1993). During sampling in March and April 1999, days were cool, sunny and overcast, the average maximum daily temperature was 24.2°C and the total rainfall was 3.7 mm. Sampling ceased over winter then continued in September 1999 when the average maximum daily temperature was 21.3°C and the total rainfall was 1.6 mm.

Skink sampling

At each study site, 15 pitfall traps of PVC pipe (11 cm diameter and 25 cm deep) were sunk into the ground.
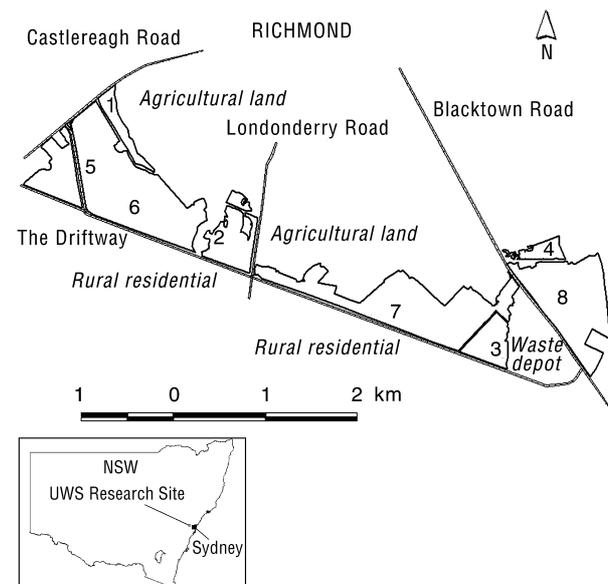

Fig. 1. Map of the study area showing edge sites (1–4) and core sites (5–8). Inset shows the study site in relation to Sydney, New South Wales (NSW).

Table 1. Combinations of the 12 environmental variables yielding the best matches of habitat characteristics and skink assemblage similarity

	P value	Ground cover	Litter depth	Leaf litter cover	Grass cover	Herb cover	Bare ground	Fallen logs	Tree stumps	Trees >10 m	Trees 1–10 m	Shrubs <1 m	Canopy cover
1	0.306									X			
2	0.432*									X*		X*	
3	0.425									X		X	
4	0.325			X					X	X		X	
5	0.292		X		X				X	X		X	
6	0.269		X		X	X		X	X	X		X	
7	0.260		X		X	X		X	X	X		X	
8	0.204		X	X	X	X		X	X	X		X	
9	0.160		X	X	X	X	X	X	X	X		X	
10	0.111	X	X	X	X	X	X	X	X	X		X	
11	0.063	X	X	X	X	X	X	X	X	X	X	X	X
12	0.03	X	X	X	X	X	X	X	X	X		X	X

Measured by Spearman's rank correlation (ρ); *indicates the overall optimum match.

Negative edge effects can penetrate up to several hundred metres from an edge but impacts are influenced by the effects being examined. We followed Kremaster and Bunnell (1999) in assuming that species distribution is most influenced within 50 m of the edge. Edge pitfall traps were located within the 10-m outer wooded perimeter and core traps were ≥150 m from the edge. Because the largest home range identified for the local populations of *Lampropholis* sp. was 20.2 m^2 (*L. guichenoti,* S. Burgin, pers. obs., 1995) and no recaptures were obtained by Jelbart (1998) in traps placed ≥45 m apart, we also spaced traps >45 m apart to ensure independence.

A small amount of leaf litter was placed in the base of each trap to protect captured lizards and a small, inverted plastic bowl was held by bamboo skewers 3–5 cm above each trap.

Pitfall trap sampling was conducted in non-rainy periods over 10 weeks in March, April and September 1999. Up to four sites were sampled daily, alternating inner and edge sites.

Traps were cleared daily between 06.30 and 11.30 hours. Each skink was marked on the abdomen with nail polish (cf. Jardine 1998; Jelbart 1998) and released at the point of capture. This method is a non-intrusive way to identify recaptures. Captured animals were identified (cf. Swan 1990; Cogger 1992) and snout–vent length was used to categorize *Lampropholis* individuals as juveniles, subadults or adults (see Joss & Minard 1985).

Pitfall traps are a relatively simple method of sampling animal populations but their success relies on the target species' trapability (Melbourne 1999; Webb 1999). In the present study, pitfall trapping yielded few captures and was therefore supplemented with points of transect survey (cf. Buckland *et al.* 1993). This technique requires the researcher to count the target animals at predetermined points along a transect. The survey was undertaken over 2 days (8 and 10 October 1999). The researcher walked slowly from one pitfall trap to another (our points of transect), in sequence, taking precautions to minimize noise. Each lizard seen was identified and recorded.

Analyses of skink data

Initially data from pitfall trapping were compared with points of transect survey counts and because trends were similar, the data were combined for analyses.

Similarity, based upon species abundance and diversity, was analysed using multivariate techniques within the PRIMER program. The method was chosen because this analysis does not require species to be present in similar numbers across sites and therefore eliminates many right skewed problems (Clarke & Warwick 1994).

Bray–Curtis similarity cluster matrices (cluster analysis) were used to determine the similarity between and within core and edge sites. This analysis enabled examination of sites to determine similarity, based on species co-occurrence across edge and core sites and the numbers of those species within sites (Clarke & Warwick 1994). Similarity occurs at higher percentages and allows for identification of the species responsible for similarity. A dendrogram was used to show similarity groups. Similarity, within and among sites, was tested by using one-way ANOSIM (Clarke & Warwick 1994; Carr 1997). Chi-squared analysis was used to determine the relationship between distribution and size of *Lampropholis* skinks.

Estimation of habitat variables

Habitat variables were sampled on core and edge sites in September 1999. The 12 habitat characteristics used (see Table 1) were a modified subset of those identified as essential for skink populations (Mather 1989).

To allow for objective evaluation of the habitat, at five pitfall traps on each site, a 20-m $\times$ 20-m plot was delineated. This area was then subdivided into 16 5-m $\times$ 5-m quadrats. A row of four of these, with the pitfall trap adjacent to the mid-line, acted as a transect. One transect (north, east, south or west of the pitfall trap) was selected from the two possible horizontal and two vertical transects.

Habitat characteristics were estimated visually (%) or directly measured, as appropriate. Ground cover height, litter depth, bare ground, leaf litter, grass and herbaceous plant coverage were sampled in a 1-m² quadrat at opposite corners of each of the four 5-m $\times$ 5-m transect quadrats. Other variables were measured across the transect. The basal diameters of trees and shrubs were measured and canopy cover was estimated at ground level.
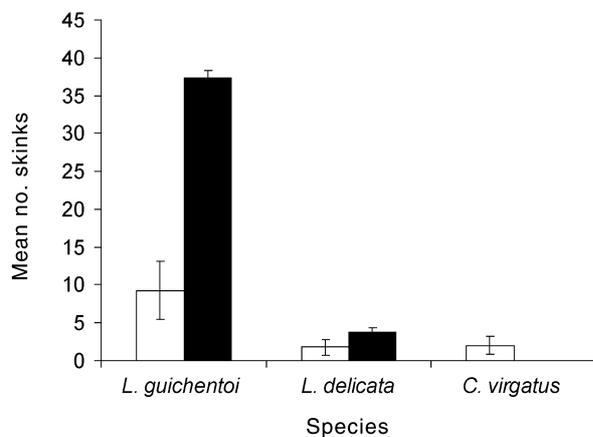

Fig. 2. Mean (and SE) number of *Lampropholis guichenoti*, *Lampropholis delicata* and *Crytoblepharus virgatus* per site encountered at remnants of bushland: (□), edge; (■), core.

Analyses of habitat characteristics

These data were again analysed by PRIMER. Multi-dimensional scaling (MDS; an ordination technique) was used to determine, via graphical representation, site similarity. A two-way nested ANOSIM assessed the habitat similarity within and between sites. Spearman's coefficient of rank correlation was used to calculate the 'best fit' combination of habitat variables with the pattern of lizard abundance and diversity overall (Clarke & Warwick 1994; Carr 1997).

RESULTS

A total of 216 individuals from three species (*L. guichenoti, L. delicata* and *C. virgatus*), 52 on edge sites and 164 on core sites, were encountered (Fig. 2). None were recaptured and because a single point of transect survey was undertaken at each site, multi-counting was assumed not to have occurred.

There was no significant similarity in skink populations between edge and core sites ($R = 0.417$, $P < 0.029$); however, the core sites were more similar (group average of 97.8%) than perimeter sites (group average of 69.4%; Fig. 3). *Lampropholis guichenoti* was most commonly encountered and contributed most to the similarity of edge (87%) and core (65.3%) sites, whereas *L. delicata* contributed relatively more to core similarity (34.6%) than did the edge (6.4%).
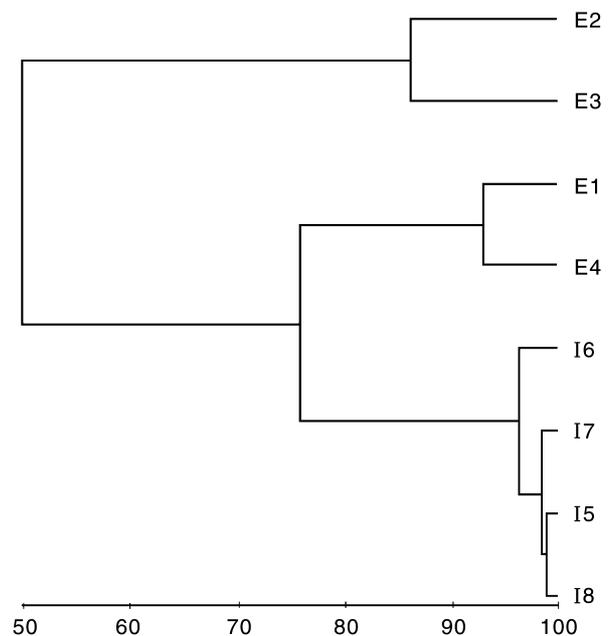

Fig. 3. Similarity of edge (E1–E4) and core sites (I5–I8), based on Bray–Curtis similarity matrices. Core sites display a high level of similarity as a group (average 97.75%). The edge sites group is less similar (average 69.4%). There is no significant similarity between core and edge sites ($R = 0.417$, $P < 0.029$).

Cryptoblepharus virgatus was only encountered at edge sites and contributed 6.4% to similarity.

The size structure of *L. guichenoti* varied significantly between the edge and core ($\chi^2_2 = 28.592$; insufficient *L. delicata* for valid analysis). In general, adults and juveniles populated the core and subadults the edge (Fig. 4).

There was no significant similarity in habitat variables among core or edge sites ($R = 0.343$, $P < 0.00$) and differences between core and edge areas were significantly similar ($R = -0.021$, $P < 0.486$: Fig. 5).

The examination of the effect of all possible combinations of habitat patterns on population dynamics showed no strong correlation between the two.

The lack of significant similarity among habitat characteristics was further demonstrated with the Spearman's coefficient of rank correlation. The weakest correlation ($P = 0.03$) occurred when all 12 habitat variables were correlated with skink abundance and diversity patterns (Table 1). The optimum match (although not strong) was trees >10 m and shrubs <1 m ($P = 0.432$), whereas trees >10 m high consistently appeared in all 'best matched' combinations. Habitat characteristics were, however, unlikely to have caused the variation in skink assemblage observed because of the relatively weak patterns observed.

DISCUSSION

Despite an expectation to the contrary, the abundance of these generalist skinks differed between perimeter and interior bushland habitats. Fewer individuals were encountered on the edge, whereas *C. virgatus* was only observed there. In addition, *L. guichenoti* size differed significantly between edge and core areas. Sub-adult numbers were greatest on the edge but the reverse was observed for juveniles and adults. These differences could not be attributed to habitat characteristics.

The overlap in the distribution of *L. delicata* and its congener, *L. guichenoti* was expected. These species are sympatric throughout much of their range, have similar
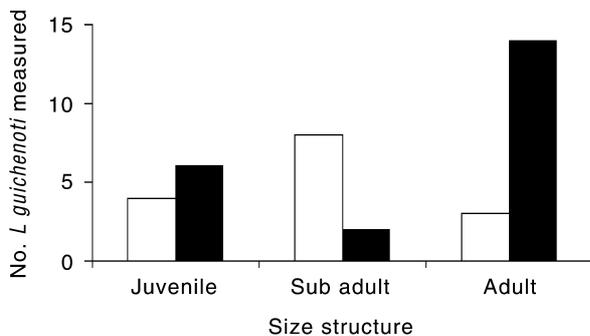
feeding habits (Warner 1995) and reproductive cycles (Burgin 1993) and have been observed to overlap in the study area (e.g. Warner 1995; Giffney 1997; Jelbart 1998). However, their habitat requirements differ (Lunney *et al.* 1991). *Lampropholis delicata* tends to prefer moister microclimates, with lower air temperatures, than *L. guichenoti* (Graham 1987). Although both species were encountered on all sites, *L. guichenoti* were more abundant in edge and core sites than *L. delicata*. However, because conditions were unseasonally moist and cool (conditions apparently better suited to *L. delicata*), a more uniform distribution of *Lampropholis* species was expected.

Based on previous observations of *C. virgatus* in the same general area (Giffney 1997), we expected to encounter them across remnants; however, they were only present in small numbers at the edge.

Woodland fragmentation results in structural change and restructuring of habitats at the interface with the disturbance. Many studies (e.g. Kitchener *et al.* 1980; Howe 1984; Mader 1984; Lunney & Ashby 1987; Lunney 1987; Soulé *et al.* 1991; Saunders 1993; How & Dell 1994) have identified that such vegetation change reduces species abundance and diversity. The removal of vegetation and the introduction of agriculture leave 'hard' edges, devoid of appropriate habitat to sustain reptile populations (Demaynadier & Hunter 1998; Voss & Chardon 1998). Ferraro and Burgin (1993) observed that reduction in overstorey around wetlands resulted in loss of frog abundance and diversity. However, in the present study we did not observe
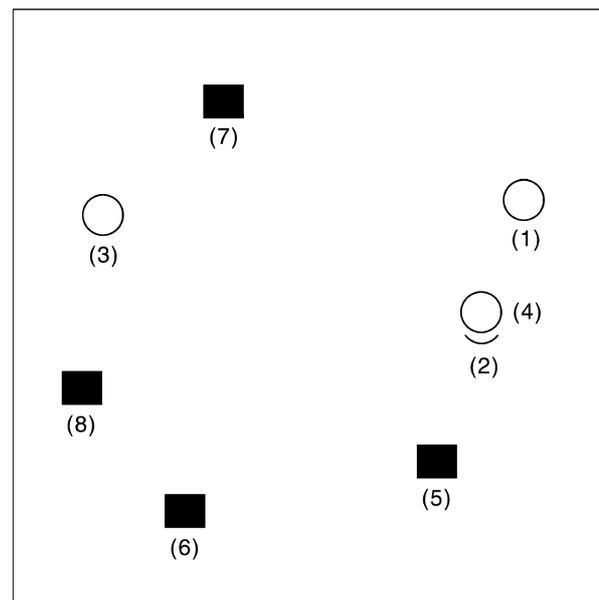

Fig. 4. Size structure of the *Lampropholis guichenoti* captured on the (□) edge and (■) core areas of remnant bushland of the Cumberland Plain.

Fig. 5. Multidimensional scaling (MDS) ordination of habitat variables as they affect abundance and diversity of the skinks found on (■) edge and (○) core areas (stress = 0.03). There is no significant similarity either within ($R = 0.343$, $P < 0.00$) or between ($R = 0.21$, $P < 0.486$) the sites.

that skink habitat characteristics (e.g. Milton 1980; Mather 1989; Lunney *et al.* 1991; Brown & Nelson 1993; Kutt 1993) impacted upon dispersal.

The skinks studied here are opportunistic arthropod predators (Crome 1981; Lunney *et al.* 1989; Brown 1991), although there is some selection based on gape width (Warner 1995). Although initial fragmentation can impact upon the diversity and abundance of arthropods (Abensperg-Traun *et al.* 1996), the remnants used in the present study were created through clearing conducted decades ago, and any reduction in numbers would have been negated.

Repetitive events, such as grazing and vehicle passage, reduce the diversity and abundance of ground-dwelling herptofauna (Bury & Busack 1974; Hadden & Westbrooke 1996). A major impact of such activities is the destruction of understorey vegetation and soil compaction. These impacts may relate to lizard decline (Bury & Busack 1974; Abensperg-Traun *et al.* 1996; Smith *et al.* 1996) and disrupt social structures (Bury & Busack 1974). Although cattle had previously grazed these sites (and incidentally during this study), vehicles were excluded. There was no indication that these activities contributed substantially to skink distribution.

In a wide range of species (e.g. *Anolis aeneus,* Stamps & Eason 1989), subadults disperse into disturbed areas, such as edges. Such dispersal may optimize survival. However, no evidence of such behaviour has been observed in local populations of *Lampropholis* (S. Burgin, unpubl. data) and so differences in population dynamics cannot be related to differential dispersal.

Environmental plasticity can influence population size structure. Qualls and Shine (1998) investigated this in *L. guichenoti* and observed differences due to the environment in populations located 120 km and >1000 m elevation apart, although there were no differences in snout–vent length (i.e. our measure of size structure). In our study, elevation differences were <20 m across our sites, the sites were within a 5-km radius, and habitat structure did not differ significantly. However, the edge (i.e. where there were significantly more subadults) could have been hotter than the core. In the presence of plasticity, substantially more larger animals would be expected on the edge.

Because habitat characteristic differences, disturbance and food limitation did not impact on skink distribution, an alternative hypothesis is that predatory pressure is responsible for variation in skink abundance and diversity. Animals that occupy perimeters are probably more exposed to predation (Soulé & Gilpin 1991). A reduction of canopy cover and lack of adequate shelter can increase predation within disturbed areas (Shaffer 1981). Although not quantified, there may be a link between canopy cover and predation. Casual observation also indicated that predatory

birds (e.g. magpies, *Gymnorhina tibicen*; kookaburras, *Dacelo novaegineae*; Australian ravens, *Corvus coronoides*) were in larger numbers on the edge than in the core and they were observed to predate on skinks.

Selective predation on larger animals would reduce the numbers of larger adults (the most fecund animals; Burgin 1993) and limit recruitment. Such ongoing predation and subsequent dispersal of subadults into unoccupied habitat may account for the observed lack of adults and juveniles in perimeter areas. Such a 'drain' of individuals and subsequent heavy predation could have the longer-term consequence of reducing abundance throughout the woodland. Landscape management that results in the expansion of edge relative to core habitat may therefore ultimately have a significant impact on even small generalist species, such as the *Lampropholis* and *C. virgatus* that are widespread, even in highly modified habitats.

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REFERENCES

Abensperg-Traun M., Smith G. T., Arnold G. W. & Steven D. E. (1996) The effects of habitat fragmentation and livestock grazing on animal communities in remnants of gimlet *Eucalyptus salubris* woodland in the Western Australian wheatbelt. I. Arthropods. *J. Appl. Ecol.* **33**, 1281–301.

Benson D. H. (1992) The natural vegetation of the Penrith 1:100 000 map sheet. *Cunninghamia* **2**, 503–662.

Benson D. H. & Howell J. (1990) Sydney's vegetation 1788–1988: Utilization, degradation and rehabilitation. *Proc. Ecol. Soc. Aust.* **16**, 115–27.

Berry L. (2001) Edge effects on the distribution and abundance of birds in a southern Victoria forest. *Wildl. Res.* **28**, 239–45.

Bromham L., Cardillo M., Bennett A. F. & Elgar M. A. (1999) Effects of stock grazing on the ground invertebrate fauna of woodland remnants. *Aust. J. Ecol.* **24**, 199–207.

Brown G. W. (1991) Ecological feeding analysis of South-eastern Australian scincids (Reptilia: Lacertilia). *Aust. J. Zool.* **39**, 9–29.

Brown G. W. & Nelson J. L. (1993) Influence of successional stage of *Eucalyptus regnans* (Mountain Ash) on habitat use by reptiles in the Central Highlands, Victoria. *Aust. J. Ecol.* **18**, 405–17.

Buckland S. T., Anderson B. R., Burnham K. P. & Laake J. L. (1993) *Distance Sampling: Estimating Abundance of Biological Populations.* Chapman & Hall, London.

Burgin S. (1993) *Lampropholis*: The new laboratory animals. In: *Herpetology in Australia, a Diverse Discipline: Transactions of the Royal Zoological Society of NSW* (eds D. Lunney & D. Ayers) pp. 279–81. The Royal Zoological Society of NSW, Mosman.

Bury R. B. & Busack S. D. (1974) Some effects of off-road vehicles and sheep grazing on lizard populations in the Mohave Desert. *Biol. Conserv.* **6**, 179–83.

Campi M. J. & MacNally R. (2001) Birds on edge: Avian assemblage along forest-agricultural boundaries of central Victoria, Australia. *Anim. Conserv.* **4**, 121–32.

Carr M. (1997) *PRIMER User Manual.* Plymouth Marine Laboratory, Plymouth.

Chiarello A. G. (1999) Effects of fragmentation of the Atlantic forest on mammal communities in South-eastern Brazil. *Biol. Conserv.* **89**, 71–82.

Clarke K. & Warwick R. (1994) *Changes in Marine Communities: An Approach to Statistical Analysis and Interpretation.* Plymouth Marine Laboratory, Plymouth.

Cogger H. G. (1992) *Reptiles and Amphibians of Australia*, 4th edn. Reed New Holland, Sydney.

Cogger H. G. (2000) *Reptiles and Amphibians of Australia*, 6th edn. Reed Books, Chatswood.

Crome B. (1981) The diet of some ground-layer lizards in three woodlands of the New England Tableland of Australia. *Herpetofauna* **13**, 4–11.

Davies K. F., Melbourne B. A. & Margules C. R. (2001) Effects of within-and between-patch processes on community dynamics in a fragmentation experiment. *Ecology* **82**, 1830–46.

Demaynadier P. G. & Hunter M. L. Jr (1998) Effects of silvicultural edges on the distribution and abundance of amphibians in Maine. *Conserv. Biol.* **12**, 340–52.

Ferraro T. J. & Burgin S. (1993) Amphibian decline: A case study in western Sydney. In: *Herpetology in Australia, a Diverse Discipline: Transactions of the Royal Zoological Society of NSW* (eds D. Lunney & D. Ayers) pp. 197–204. The Royal Zoological Society of NSW, Mosman.

Fleischner T. L. (1994) Ecological costs of livestock grazing in Western North America. *Conserv. Biol.* **8**, 629–44.

Fox B. J. (1990) Two hundred years of disturbance: How has it aided our understanding of succession in Australia. *Proc. Ecol. Soc. Aust.* **16**, 521–9.

Gates J. E. (1991) Powerline corridors, edge effects and wildlife in forested landscapes of the central Appalachians. *Wildlife and Habitats in Managed Landscapes* (eds J. E. Rodiek & E. G. Bolen) pp. 13–32. Island Press, California.

Giffney D. (1997) Weather influence upon the activity of small skinks (*Lacertilia Scincidae*). Unpublished M. App. Sc. Thesis, University of Western Sydney, Hawkesbury.

Goldingay R., Daly G. & Lemkert F. (1996) Assessing the impacts of logging on reptiles and frogs in the Montane forests of southern New South Wales. *Wild. Res.* **23**, 495–510.

Graham A. (1987) Habitat selection in *Lampropholis delicata* and. *Lampropholis guichenoti* (*Squamata: Sqincidae*). BSc. (Hons) Thesis, Macquarie University, Sydney.

Gregory P. T. (1978) Feeding habits and diet overlap of three species of garter snakes (*Thamnophis*) on Vancouver Island. *Can. J. Zool.* **56**, 1967–74.

Hadden S. A. & Westbrooke M. E. (1996) Habitat relationships of the herpetofauna of remnant Buloke Woodlands of the Wimmera Plains, Victoria. *Wildl. Res.* **33**, 363–72.

Hagan J. M., Vander Haegen W. M. & McKinley P. S. (1996) The early development of forest fragmentation effects on birds. *Conserv. Biol.* **10**, 188–202.

Haila Y., Saunders D. A. & Hobbs R. J. (1993) What do we presently understand about ecosystem fragmentation. In: *Nature Conservation 3: Reconstruction of Fragmented Ecosystems, Global and Regional.* (eds D. A. Saunders, R. J.

Hobbs and P. R. Ehrlich) pp. 45–55. Surrey Beatty and Sons, Chipping Norton.

How R. A. & Dell J. (1994) The zoogeographic significance of urban bushland remnants to reptiles in the Perth region, Western Australia. *Pac. Conserv. Biol.* **1**, 132–40.

Howe R. W. (1984) Local dynamics of bird assemblages in small forest habitat islands in Australia and North America. *Ecology* **65**, 1585–99.

Jardine A. (1998) An investigation of two sympatric Australian Skinks *Lampropholis guichenoti* and *Lampropholis delicata*: Their home ranges and habitat preference. Unpublished M.App. Sc. Thesis, University of Western Sydney, Hawkesbury.

Jelbart J. E. (1998) An investigation of the influence of edge effects on the distribution and abundance of skinks in remnant Cumberland Plain Woodland in Western Sydney. Unpublished M.App. Sc. Thesis, University of Western Sydney, Hawkesbury.

Joss J. M. P. & Minard J. A. (1985) On the reproduction cycles of *Lampropholis guichenoti* and *L. delicata*. *Aust. J. Zool.* **33**, 699–704.

Kitchener D. J., Chapman A., Dell A., Muir B. G. & Palmer M. (1980) Lizard assemblage and reserve size and structure in the Western Australian wheatbelt: Some implications for conservation. *Biol. Conserv.* **17**, 25–62.

Kitchener D. J. & How R. A. (1982) Lizards species in small mainland habitat isolates and islands off South-western Western Australia. *Aust. Wildl. Res.* **9**, 357–63.

Kremaster L. & Bunnell F. L. (1999) Edge effects: Theory, evidence and implications to management of western North American forests. In: *Forest Fragmentation: Wildlife and Management Implications* (eds J. Wisniewski, J. A. Rochelle & L. Lehmann) pp. 117–53. Leiden, Boston, MA.

Kutt A. (1993) Initial observations on the effect of thinning Eucalypt regrowth on heliothermic skinks in lowland forest, East Gippsland Victoria. In: *Herpetology in Australia, a Diverse Discipline: Transactions of the Royal Zoological Society of NSW* (eds D. Lunney & D. Ayers) pp. 187–96. The Royal Zoological Society of NSW, Mosman.

Laurance S. G. & Laurance W. F. (1999) Tropical wildlife corridors: Use of linear rainforest remnants by arboreal mammals. *Biol. Conserv.* **91**, 231–9.

Leopold A. (1933) *Game Management.* Charles Scribner Sons, New York.

Lunney D. (1987) Effects of logging, fire and drought on Possums and Gliders in the coastal forests near Bega, New South Wales. *Aust. Wildl. Res.* **14**, 263–74.

Lunney D. & Ashby E. (1987) Population changes in *Sminthopsis leucopus* (Gray) (Marsupialia: Dasyuridae), and other small mammal species, in forest regenerating from logging and fire near Bega, New South Wales. *Aust. Wildl. Res.* **14**, 275–84.

Lunney D., Ashby E., Grigg J. & Connell M. (1989) Diets of Scincid Lizards *Lampropholis guichenoti* (Dumeril & Bibron) and *L. delicata* (De Vis) in Mumbulla State Forest on the South Coast of New South Wales. *Aust. Wildl. Res.* **16**, 307–12.

Lunney D., Eby P. & Connell M. (1991) Effects of logging fire and drought on three species of lizards in Mumbulla State Forest on the South Coast of New South Wales. *Aust. J. Ecol.* **16**, 33–46.

Mader H. J. (1984) Animal habitat isolation by roads and agricultural fields. *Biol. Conserv.* **29**, 81–96.

Mather P. B. (1989) A comparison of the normal habitats of Skinks of three electrophoretically distinguishable forms of

Lampropholis delicata (Lacertilla: Scincidae) in South-eastern Queensland. *Aust. Wildl. Res.* **16**, 159–65.

Matthews A., Dickman C. R. & Major R. E. (1999) The influence of fragment size and edge on nest predation in urban bushland. *Ecography* **22**, 349–56.

Melbourne B. A. (1999) Bias in the effect of habitat structure on pitfall traps: An experimental evaluation. *Aust. J. Ecol.* **24**, 228–39.

Milton D. (1980) Some aspects of the population dynamics of *Lampropholis guichenoti* in the Toohey State Forest near Brisbane. *Herpetofauna* **11**, 19–23.

Murcia C. (1995) Edge effects in fragmented forests: Implications for conservation. *TREE* **10**, 56–62.

Qualls F. J. & Shine R. (1998) Geographic variation in lizard phenotypes: Importance of the incubation environment. *Biol. J. Linn. Soc.* **64**, 477–91.

Reed P. (1991) An historical analysis of the changes to the forests and woodlands of New South Wales. In: *Conservation of Australia's Forest Fauna* (ed. D. Lunney) pp. 393–406. The Royal Zoological Society of New South Wales, Mosman.

Robinson L. (1991) *Field Guide to the Native Plants of Sydney*, 2nd edn. Kangaroo Press, Kenthurst.

Sarre S., Smith G. T. & Meyers J. A. (1995) Persistence of two species of gecko (*Oedura reticulata* and *Gehyra variegata*) in remnant habitat. *Biol. Conserv.* **71**, 25–33.

Saunders D. A. (1993) A community-based observer scheme to assess avian responses to habitat reduction and fragmentation in South-Western Australia. *Biol. Conserv.* **64**, 203–18.

Saunders D. A., Hobbs R. J. & Margules C. R. (1991) Biological consequences of ecosystem fragmentation: A review. *Conserv. Biol.* **5**, 1–15.

Shaffer M. (1981) Minimum population sizes for species conservation. *Bioscience* **31**, 131–4.

Sisk T. D. & Margules C. R. (1993) Habitat edges and restoration: Methods for quantifying edge effects and predicting the results of restoration efforts. In: *Nature Conservation 3: Reconstruction of Fragmented Ecosystems* (eds D. A. Saunders, R. J. Hobbs & P. R. Ehrlich) pp. 57–69. Surrey Beatty & Sons, Chipping Norton.

Smith G. T., Arnold G. W., Abensperg-Traun M. & Steven D. E. (1996) The effects of habitat fragmentation and livestock grazing on animal communities in remnants of gimlet

Eucalyptus salubris woodland in the Western Australian wheatbelt. II. Lizards. *J. Appl. Ecol.* **33**, 1302–10.

Soulé M. E., Alberts A. C. & Bolger D. T. (1991) The effects of habitat fragmentation on chaparral plants and vertebrates. *Oikos* **63**, 39–47.

Soulé M. E. & Gilpin M. E. (1991) The theory of wildlife corridors capability. In: *Nature Conservation 2: the Role of Corridors* (eds D. A. Saunders & R. J. Hobbs) pp. 3–8. Surrey Beatty & Sons, Chipping Norton.

Stamps J. A. & Eason P. K. (1989) Relationships between spacing behaviour and growth rates: A field study of lizard feeding territories. *Behav. Ecol. Sociobiol.* **25**, 99–107.

Sumner J., Moritz C. & Shine R. (1999) Shrinking forest shrinks skink: Morphological change to response to rainforest fragmentation in prickly forest skink (*Gnypetoscincus queenslandiae*). *Biol. Conserv.* **91**, 159–67.

Swan G. (1990) *A Field Guide to the Snakes and Lizards of New South Wales*. Three Sisters Publications, Winmalee.

Taylor J. E. & Fox B. J. (2001) Disturbance effects from fire and mining produce different lizard communities in eastern Australian forests. *Aust. Ecol.* **26**, 193–204.

Tendy T. (1987) The value of corridors (and design features of same) and small patches of habitat. In: *Nature Conservation: the Role of Remnants of Native Vegetation* (eds D. A. Saunders, G. W. Arnold, A. A. Burbidge & A. J. Hopkins) pp. 357–9. Beatty & Sons, Chipping Norton.

Twigg L. E. & Fox B. J. (1991) Recolonization of regenerating open forest by terrestrial lizards following sand mining. *Aust. J. Ecol.* **16**, 137–48.

Voss C. C. & Chardon J. P. (1998) Effects of habitat fragmentation and road density on the distribution pattern of the moor frog *Rana arvalis*. *J. Appl. Ecol.* **35**, 44–56.

Warner F. (1995) A Comparison of the dietary contents of two sympatric Scincid lizards, *Lampropholis delicata* and *Lampropholis guichenoti*. BSc (Hons) thesis, University Western Sydney, Hawkesbury.

Webb G. A. (1995) Effects of logging on lizards in Eucalypt forest at Eden, New South Wales. *Aust. For.* **58**, 155–9.

Webb G. A. (1999) Effectiveness of pitfall/drift-fence systems for sampling small ground-dwelling lizards and frogs in southeastern Australian forests. *Aust. Zool.* **31**, 118–26.

Yahner R. H. (1988) Changes in wildlife communities near edges. *Conserv. Biol.* **2**, 333–9.