CSIRO PUBLISHING

Wildlife Research, 2015, **42**, 324–333
http://dx.doi.org/10.1071/WR14182

The influence of urban encroachment on squirrel gliders (*Petaurus norfolcensis*): effects of road density, light and noise pollution

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Abstract

Context. Loss and degradation of habitat from urban development is a key threat to the squirrel glider (*Petaurus norfolcensis*), because its distribution coincides where most people live in Australia. Squirrel gliders are known to occur in or around urban fringes where native vegetation is retained; however, little is known about specific anthropogenic factors that may affect their persistence.

Aims. We investigated the relative influence of biophysical and urban factors on the use of large hollow-bearing *Eucalyptus* trees, which are a key resource for squirrel gliders. The study was located in a typical urban growth area located in southern New South Wales.

Methods. A stratified random sampling approach was used to survey squirrel gliders in urban and rural areas. Infrared, motion-sensor cameras were placed in 34 selected trees for 17 nights to record squirrel glider occupancy and activity. Data on urban (e.g. light and noise pollution levels, road and housing density) and biophysical (e.g. tree height, hollow-bearing tree density, vegetation cover) variables were recorded at each survey tree. Data were statistically analysed using general linear modelling approaches.

Key results. Squirrel gliders were detected more frequently in the rural matrix (23.4% of camera trap-nights) than in urban areas (9.5%). Model results showed that tree height, and the distance to neighbouring trees, had a significant influence on the occurrence and activity of squirrel gliders. Road density and light pollution were included in 'best' models to explain glider activity (a negative influence), and noise pollution negatively influenced glider occurrence. Although gliders used large trees in both urban and rural areas, activity generally decreased as levels of urbanisation increased.

Conclusions and implications. Access to and availability of key resources such as tall, hollow-bearing trees is critical for gliders to persist in urban environments. Squirrel gliders will tolerate human stressors such as roads, noise and light pollution to a certain extent, but impacts on population viability remain largely unknown. Novel solutions need to be developed to lessen the effects of anthropogenic factors (such as light and noise) on patches of native vegetation retained in urban areas for conservation purposes.

Additional keywords: aboreal fauna, camera traps, gliding, road ecology, tree age, tree height, urbanisation.

Received 11 September 2014, accepted 8 July 2015, published online 19 August 2015

Introduction

Human population growth and concurrent levels of urbanisation are arguably the most damaging anthropogenic threat to biodiversity (Vitousek *et al*. 1997; Garden *et al*. 2006). Urban encroachment is a key ecological process occurring around the world, the impacts of which are accelerating as urban growth expands (United Nations 2012). The urban population in Australia is approaching 20 million (ABS 2013), presenting several critical issues in relation to biodiversity conservation management in and around urban growth centres (Miller and Hobbs 2002).

Urbanisation results in the clearance of native vegetation, and replacement with human land uses and constructions such as houses, roads and gardens. As a result, urban encroachment directly threatens biodiversity (Marzluff and Ewing 2001), leads to simplification and homogenisation of remaining habitat (Blair 2001; McKinney 2002) and is a cause of local species extinctions (Taylor and Goldingay 2012*a*). Urban development can also lead to fragmentation, where remaining habitat patches become increasingly isolated or degraded (Garden *et al*. 2006). In many urbanised landscapes, small remnants or isolated trees are a legacy to once wide-spread vegetation (van der Ree *et al*. 2004), and are critical for the movements and survival of many native species (Potter 1990). However, whether remnant habitats can be utilised in an urban landscape largely depends on the permeability of the landscape, its resources, and resilience of individual species to change (Caryl *et al*. 2013).

In eastern Australia, much of the preferred habitat for squirrel gliders (*Petaurus norfolcensis*) is, coincidently, preferred by humans for urban development. The squirrel glider is a small, predominantly arboreal marsupial that occurs in dry sclerophyll woodlands and forests, and in coastal and wet forests in Queensland as far north as Cape York Peninsula. Remnant populations also occur in river red gum forests in far-western Victoria and South Australia (van der Ree and Suckling 2008). The species relies on large, hollow-bearing eucalypt trees for gliding, nesting and foraging (Beyer *et al*. 2008; van der Ree and Suckling 2008; Sharpe 2009; Crane *et al*. 2012). Squirrel gliders are listed as vulnerable in New South Wales (NSW), threatened in Victoria and endangered in South Australia; however, they are common in Queensland. In certain regions of NSW, squirrel glider populations are listed as endangered, e.g. Wagga Wagga (NSW Government Environment and Heritage 2011). Loss and degradation of habitat as a result of urban development is recognised as a key threatening process for this species (NSW Scientific Committee 2008).

As squirrel gliders are known to occupy remnant woodlands or forests in, or at the fringes of urban areas (Rowston *et al*. 2002; Goldingay *et al*. 2006), they an ideal species to examine specific effects of urbanisation on native species. Several studies have been conducted that have compared and contrasted the use of urban versus forested areas by squirrel gliders. Brearley *et al*. (2010, 2011) recorded greater trapping success and observed larger home ranges for squirrel gliders in the interior of forest remnants, as opposed to urban edges. This finding was attributed to more large trees (and associated tree hollows), and greater floristic species richness in forest interiors than in road and urban-edge habitats. Elevated stress levels were recorded for squirrel gliders located near major roads as compared with interior habitats (Brearley *et al*. 2012). However, as Sharpe (2009) described, because squirrel gliders can be found in hollow-bearing trees located in backyards or along busy roads, they may tolerate urban conditions to a certain extent.

In areas experiencing urban growth, anthropogenic factors such as artificial light, noise and road-traffic movements constitute a major threat to native species (e.g. Reijnen *et al*. 1995; Forman and Alexander 1998; Boldogh *et al*. 2007). It is well established that roads pose a significant threat to wildlife, either directly via mortality risks associated with vehicle movements, or indirectly via the impacts on their behaviour associated with noise, light and other forms of pollution (Forman *et al*. 2003; Taylor and Goldingay 2010). However, the specific impact of noise and light pollution on wildlife has received little attention until recently. Noise pollution refers to increased levels of background sound commonly produced by anthropogenic sources (Barber *et al*. 2011). For example, such noise is known to change the vocal behaviours of frogs (Lengagne 2008) and birds (Brumm 2004; Mockford and Marshall 2009). Anthropogenic noise can also block the sounds of animal vocalisations, inhibiting important communications such as those for predator avoidance (Barber *et al*. 2011). Human noise can cause stress in animals (Owen *et al*. 2004); however, some animals can become accustomed and even habituate to increased noise levels from humans (Laurian *et al*. 2008).

Light pollution from street lights, houses, cars, sporting stadiums and other infrastructure is commonplace in urban areas (Longcore and Rich 2004). However, the ecological consequences of light pollution have been poorly studied (Gaston *et al*. 2012). Light pollution is a particular concern to nocturnal animals, where it is known to cause behavioural changes in a range of species (Beier 2006). For example, as prey species naturally change their foraging behaviour in response to changes in moon phase (Daly *et al*. 1992), artificial lighting from anthropogenic sources can change or disrupt foraging movements (Boldogh *et al*. 2007; Stone *et al*. 2009). Similarly, for animals that normally use the moon as a navigation cue, light pollution can cause disorientation and total disruptions to movements (Miles *et al*. 2010).

In Australia, research on the influence of the effects of urban areas on native species is gaining new attention (e.g. Tremblay and St. Clair 2009; Caryl *et al*. 2013; Threlfall *et al*. 2013), where studies have described how responses to urban encroachment are often species specific, and, therefore, difficult to generalise. Despite the documented negative consequences of urban encroachment, few studies have identified the relative importance of specific anthropogenic factors on activity and occurrence of squirrel gliders. Urban areas often possess a complex mosaic of habitats, which can provide critical habitiat for various biota (Garden *et al*. 2006). Hence, gaining a better understanding of these influences is critical to conserving and managing native species in many parts of Australia. In the present study, we examined the influence of urban encroachment on squirrel gliders in a fragmented woodland landscape presently experiencing rapid urban growth. Specifically, our aim was to determine the relative importance of light and noise pollution, in relation to other factors (e.g. tree variables, road density) that are likely to influence the occupancy and activity of squirrel gliders.

Materials and methods

Study area and focus species

The study was conducted in the residential precinct of Thurgoona (–36°2′40″, 146°59′36″), which is located north-east of the city of Albury, southern NSW, Australia (Fig. 1). Thurgoona is situated in a former agricultural landscape experiencing transitional change, where patches of remnant temperate woodlands remain. Because of past clearing, native vegetation in the area is primarily restricted to small roadside remnants, a travelling stock reserve (Bells TSR), several creekline corridors and isolated paddock trees. Dominant tree species include white box (*Eucalyptus albens*), Blakely's red gum (*E. blakelyi*), apple box (*E. bridgesiana*), river red gum (*E. camaldulensis*), red box (*E. polyanthemos*) and yellow box (*E. melliodora*) (Benson 2008). The area also contains extensive revegetation plots (>25 ha) of primarily native species that have been planted since the 1970s (Davidson *et al*. 2004).

Thurgoona is characterised by a high density of small housing lots, where the population is expected to triple within the next 20 years to ~30 000, with infrastructure and urban development plans having been established until 2045 (RPS Australia East Pty Ltd 2012). Urban encroachment into the rural landscape has led to further fragmentation and isolation of habitat for native species. Native fauna recorded in the area include squirrel gliders, common ringtail possums (*Pseudocheirus peregrinus*), common brushtail possums (*Trichosurus vulpecula*) and over 118 species of native birds (Davidson *et al*. 2004). Because of concerns over urban growth, squirrel gliders have been the focus of several conservation programs in the Thurgoona area (van der Ree 2003). Recent population viability analysis of squirrel gliders in Thurgoona has suggested that the population was viable for at least the next 100 years, provided that large, hollow-bearing trees were retained, nest boxes installed and the understorey of remnants enhanced (Stewart and van der Ree

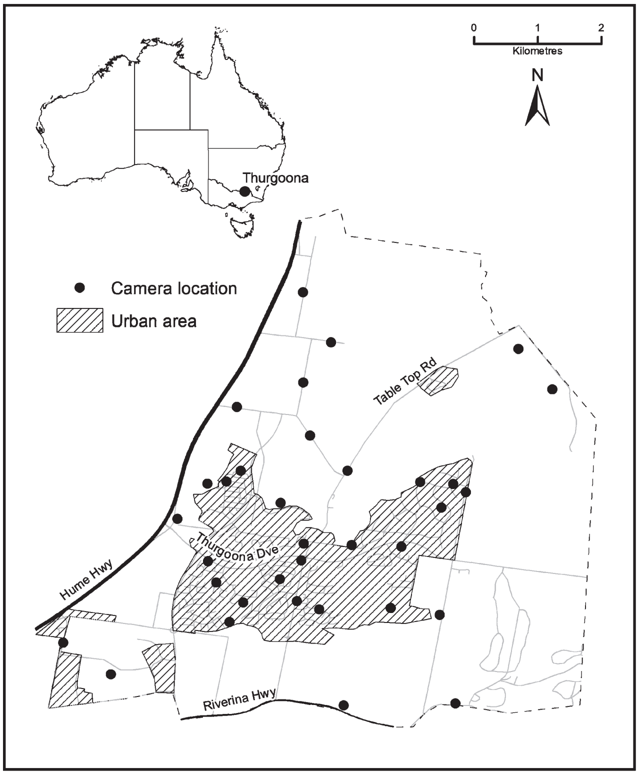

Fig. 1. Location of survey sites in the Thurgoona study area, Australia. Cross-hatched areas indicate urban areas; all other areas are designated as rural matrix (on the basis of housing density).

2009). The survey was carried out in July 2013 at the beginning of the breeding season for squirrel gliders.

Study design

The study area was adapted from the *Thurgoona Threatened Species Conservation Strategy* (Davidson *et al.* 2004; Fig. 1). A stratified random sampling approach was used, where large eucalypt trees (a key habitat, food resource and gliding structure for squirrel gliders: Crane *et al.* 2008, 2012; Sharpe 2009; Goldingay and Taylor 2009) were randomly selected within areas of preferred habitat for gliders in urban and non-urban areas, to assess squirrel glider activity. ArcGIS was used to first stratify the landscape into urban and non-urban areas (hereon known as the 'rural matrix') on the basis of a threshold housing-lot density of 2 lots ha^{-1} (NSW Land and Property Information 2010). High-resolution aerial imagery (dated 2012, supplied by Albury City Council) was geo-referenced in ArcGIS, where an iso-cluster unsupervised classification method was used to group colour pixels into four vegetation classes. A final habitat layer for squirrel gliders was developed using satellite imagery to assign vegetation classes to dominant-eucalypt tree cover. Final mapping was verified by comparing to previous squirrel glider habitat assessments conducted in Thurgoona by Stewart and van der Ree (2009).

To randomly locate large eucalypt trees, ArcGIS was used to provide 30 random points within the constructed squirrel glider habitat layer, in both urban- and rural-matrix areas. At each of the 60 locations, field surveys were then conducted to select the nearest hollow-bearing tree (within 50 m) on the basis of the following criteria: (1) the tree was a rough-barked *Eucalyptus* box species, to maximise detectability of gliders (Crane *et al.* 2008); (2) the tree was at least 70 cm in diameter at breast height (DBH), as large old trees have a high probability of possessing tree hollows (Spooner and Smallbone 2009); (3) that hollows were visible in the tree; (4) the tree possessed at least two main branches below 5 m, for survey equipment to be installed (below); (5) the tree was accessible (either on public land, or permission was obtained for trees on private property); and (6) the tree could be safely accessed (i.e. not infested with bees). To ensure independence among survey sites, trees selected were at least 250 m from each other. Some trees could not be surveyed in urban areas because of the risk of equipment vandalism. A final total of 34 trees in the urban ($n = 17$) and rural matrix ($n = 17$) were selected for sampling.

Glider detection

Infrared (IR), motion-sensor cameras (Reconyx HC500 HyperFire, USA) were used to detect the presence and activity of squirrel gliders at selected trees (survey sites). Cameras were mounted 3–5 m above ground level, aimed at an opposing branch of the tree. To attract squirrel gliders into the field of view of the camera (Fig. 2), Faunatech hair funnels were used as a bait station, and screwed onto the opposing branch. Baits were a mixture of honey, peanut butter and rolled oats, which were placed inside the funnels to attract squirrel gliders, but could not be removed. An additional spray of honey and water was applied to the tree above the funnel, to further attract squirrel gliders to the bait stations and into the field of view of the cameras (see Supplementary Material). Methods followed those described by Paull *et al.* (2011), Meek *et al.* (2012) and Harley *et al.* (2014).

Cameras were mounted in the standard orientation (with a horizontal detection zone), and held in place using a Reconyx camera mount, Python cable locks (Moorebank, NSW), or manufacturer elastic straps as suitable. Where risk of theft or damage was perceived to be high (e.g. in an urban park), the camera was mounted inside a squirrel glider nest box for camouflage. IR cameras were programmed to capture three images per movement trigger at a rate of one per second, with no delay between triggers (Rovero *et al.* 2010). In all, 34 cameras were set to operate in each of the 34 survey trees, and set to continuously operate for 24 h per day, for up to 17 nights (following Meek *et al.* 2012) after the commencement of the survey period. The survey period commenced near the high point of a lunar cycle (full moon), and ended after the low point (no moon), to maximise detection of gliders under different night conditions. The operation and positioning of the cameras was checked during installation, by inspecting images on the SD card using a digital camera. Where surrounding branches or dense foliage would potentially trigger the IR camera, adjustments were made to the final camera installation (for further details, see Supplementary Material).

Measurement of site and landscape attributes

At each survey site, several tree and site attributes were recorded (Supplementary Material). Noise levels were recorded over

Fig. 2. Images of squirrel gliders captured at bait stations using infrared cameras, showing identification features such as facial characteristics, tail shape and colouration.

a 24-h period with a handheld digital sound-level meter (Nadi C-DSM1, CA, USA). Light levels were recorded at night (at 1800–2000 hours and 2200–2400 hours) with an Lutron LX-1108 (Taiwan) light meter. GPS coordinates were recorded with a hand-held Garmin 60 GPS unit (Eastern Creek, Australia). Landscape attributes such as road and house density were measured using ArcGIS. Other landscape factors such as distance to the urban edge, distance to perennial creeks and distance to Hume Highway were calculated and exported to Excel for further data analysis. These methods are further described in the Supplementary Material.

Data collection: IR camera photos

At the conclusion of the survey period, images from IR cameras were examined for evidence of nocturnal animal activity. Photo data (for an example see Fig. 2) were separated into the following three categories: squirrel gliders, other possums and unknown. Squirrel gliders were identified, and distinguished from the closely related sugar glider, by facial characteristics, tail shape and colour (van der Ree and Suckling 2008). Because of variations in image quality, in some cases, a second opinion was obtained from an independent fauna expert to identify the species of glider. In several photos, some animals were classified as 'unknown' because of the animal being either too close to the camera or only partially visible in the image.

For each survey site, squirrel glider occupancy (presence/absence) was determined where at least one photo of an animal was recorded on any camera trap-night. We derived a cumulative frequency probability of detection curve for captures over 17 camera trap-nights for each tree, to determine the appropriate time required to detect whether squirrel gliders were present at a sampling location. Activity was determined from a count of the number of trap-nights where at least one squirrel glider was present (number of nights recorded/number of trap-nights × 100). Owing to difficulty in identifying individuals from images, multiple triggers on a single night were recorded as a single presence for that night. For the present study, trap-nights, rather than days, were used, as squirrel gliders were active during the night and trap-nights spanned two dates.

Statistical analysis

Because of the slight variation in the number of nights each camera was deployed, occupancy and activity were standardised to 14 nights (from the date of deployment) for subsequent statistical analysis. Descriptive statistics (Spearman's rank correlation and Kruskal–Wallace non-parametric tests) were used to explore relationships among squirrel glider occupancy, activity and the site and landscape variables by using Spotfire S+ statistical software (TIBCO Software Inc. 2010). To test for differences in mean values for occupancy and activity between urban- and rural-matrix areas, non-parametric Wilcoxon rank-sum tests were used (Crawley 2002).

Logistic regression was used to investigate the extent to which tree, site and landscape attributes, which we selected *a priori* on the basis of prior studies, contributed to the presence or absence of squirrel gliders within the study region (Hosmer *et al*. 2013). Relationships between variables (such as housing density and road density) were investigated using Spearman's rank correlation test, to ensure that correlated variables were not entered into the same models (Crawley 2002). Variables with a correlation value $|r| > 0.7$ were excluded. In selecting variables, minimum and average light and noise measurements were excluded because these were highly correlated with maximum light and noise. Other variables excluded were the number of street lights within 50 m of the survey tree and the number of houses within 250 m, because of their high correlation with maximum light levels and road density.

In total, 15 variables were considered in the final model-building process. These were GPS coordinates, diameter at breast height (DBH), tree height, distance to the nearest tall tree, distance to the nearest hollow-bearing tree, density of hollow-bearing trees, vegetation cover, number of shrubs, road density, the type of road, maximum light, maximum noise, distance to the urban edge, distance to perennial creeks, and distance to the Hume highway. Initially, all variables were entered into an equal number of candidate models. However, final models were restricted to a maximum of three variables, and four parameters, to ensure that the strength of the model does not decrease with the small sample size (Guisan and Zimmermann 2000). An Akaike's information criterion (AIC) approach was used to select the 'best' models for predicting squirrel glider occupancy. A variant of AIC (AIC_c) was used in the study, corrected for small sample sizes relative to the number of parameters included in the models.

Models with ΔAIC_c values of ≤ 2 were considered to be the best models, because they have substantial support compared with the best-fitting model in the set (Burnham and Anderson 2001). To determine which variables were best at predicting squirrel glider occupancy, weights were summed across all candidate models (Δ_i) for each variable included the top models. Model goodness-of-fit statistics were calculated by comparing classification results of the 'best' models to the null value model, to obtain model pseudo-r^2 and significance P-values (Hosmer *et al*. 2013).

To predict activity levels for squirrel gliders, a zero-inflated generalised linear model (GLM) was used to account for the high number of zero observations in the data from sites where squirrel gliders were absent (Martin *et al*. 2005). These absences, which were considered true 'zero' values, could greatly influence results, and were, therefore, reduced or effectively filtered from the dataset, through zero-inflated modelling. Zero-inflated poisson regression was undertaken in the statistics program R (R Core Team 2013). As with the logistic occupancy models, zero-inflated models of activity were assessed using the AIC$_c$ model-selection process, with a maximum of three predictors included in any one model. The same approach to determine the top models through AIC$_c$ was used; weights were once again summed across all candidate models for the variables included in the top models. Model goodness-of-fit statistics were calculated by using log-likelihood ratio tests between the 'best' models to the null model, and P-values were computed to determine significance.

Results

Squirrel glider surveys

Squirrel gliders were recorded in 18 survey trees (53%), including 10 in the rural matrix and eight in urban areas. After five trap-nights, squirrel gliders were observed in 16 survey trees (89%) where curve fitting to the frequency distribution of occurrence indicated that cameras were installed for a suitable detection period (minimum 14 trap-nights for each tree). On average, squirrel gliders were detected more frequently at sites located in the rural matrix (23.4% of camera trap-nights; mean = 3.53 ± 1.05 nights) than in the urban areas (9.5% of camera trap-nights; mean = 1.41 ± 0.52 nights), with the difference being significant ($\chi^2 = 5.48$, $P = 0.02$). There was a weak, but non-significant, correlation between distance to the urban edge and squirrel glider activity (Spearman's rank correlation $r_s = 0.20$, $n = 34$, $P > 0.05$).

Mean maximum noise was significantly greater (Fig. 3, $Z = 2.15$, $P = 0.03$) at survey trees where squirrel gliders were absent (present: 51.6 dBA ± 2.1, absent: 55.9 dBA ± 1.5). Similarly, mean maximum light was darker at locations where squirrel gliders were present; however this difference was not significant (Fig. 4, $Z = 1.3445$, $P > 0.05$). The mean height of survey trees where squirrel gliders were present was also greater than those trees where they were absent; however, this difference was not significant ($Z = –1.7053$, $P > 0.05$). Where squirrel gliders were present in trees, the mean distance to the nearest tall tree was significantly greater ($Z = –2.0322$, $P = 0.04$; 13.30 m ± 1.35) than for those trees where they were absent (9.48 m ± 1.29).

Models to predict the occurrence and activity of squirrel gliders

There were seven models within two AIC$_c$ values ($\Delta_i < 2$) that were all significant ($P < 0.05$) in predicting glider occupancy in trees (Table 1). The top model (with the lowest AIC$_c$; pseudo-$r^2 = 0.23$, $P = 0.01$, classification success = 76%) included variables for tree height, distance to the nearest tall tree and latitude. Maximum noise was included only in the seventh-top occupancy model (pseudo-$r^2 = 0.18$); however, the contribution of this variable to squirrel glider occupancy was strongly supported by its summed Akaike weights (0.22), where it was ranked third highest of all predictors (Table 1). The likelihood of squirrel glider occurrence declined as maximum noise increased and the density of neighbouring hollow-bearing trees decreased. Road density also had some support in the fourth-best model, where the probability of gliders occupying a hollow-bearing tree increased as road density decreased, being a variable strongly tied to urbanisation. Road type, and several variables relating to neighbouring hollow-bearing trees, was also included in the top models (Table 1).

For squirrel glider activity, generalised linear (zero-inflated) models were constructed using tree, site and landscape variables

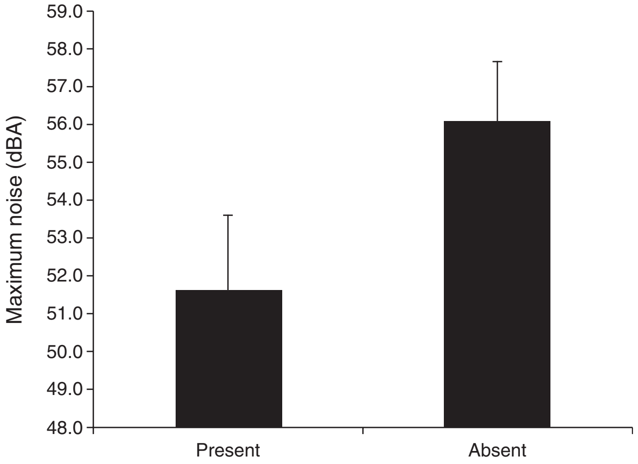

Fig. 3. Average maximum visible light recorded nightly at survey trees monitored for squirrel glider occurrence. Error bar shows one standard error (+s.e.).

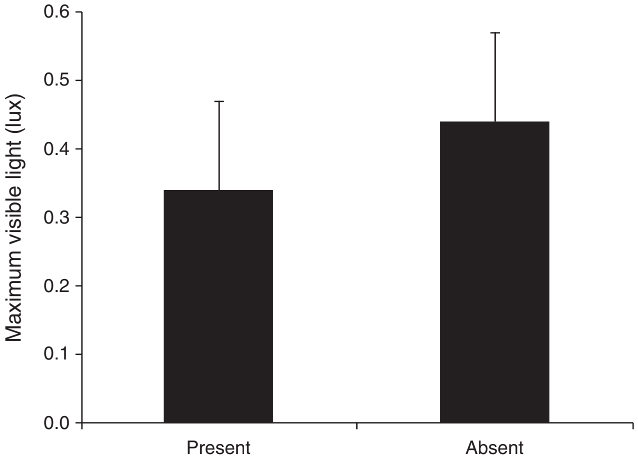

Fig. 4. Average maximum noise recorded at survey trees (24-h period) in relation to squirrel glider occurrence. Error bar shows one standard error (+s.e.).

(Table 2). In the best model ($P = 0.02$), road density and distance to the nearest tall tree (summed Akaike weights = 0.57 and 0.36, respectively) were both negatively related to squirrel glider activity. In the second-best model, distance to the nearest hollow-bearing tree was also a strong predictor variable. Although only included in the third-best model, maximum light was a strong predictor variable (summed Akaike weights = 0.34), and was negatively related to squirrel glider activity (Table 2).

Discussion

Squirrel glider occupancy of large trees was similar in both the urban and rural areas; however, gliders were much more active in trees located in the rural matrix. In one of the first studies of this kind, noise and light pollution were attributed to the absence or reduced activity of the squirrel glider in large trees. Our models showed that tree height, and the distance to neighbouring trees, had a significant influence on the use of trees by squirrel gliders. Road density was a significant negative predictor of glider activity. Other anthropogenic factors such as noise and light pollution also influenced glider occupancy and activity, respectively, although to a lesser extent, as discussed below.

Importance of tall trees for occupancy

In our study, one of the strongest determinants of squirrel glider occupancy (presence) on the urban fringe was the height of hollow-bearing trees. Other studies have identified large, hollow-bearing trees as being important for squirrel gliders (Crane *et al*. 2012), but none has identified tree height as an important factor influencing squirrel glider occupancy. It is more energy efficient to climb a tall tree to glide long distances, than to expend energy on more frequent glides from smaller trees (Scheibe *et al*. 2006; Flaherty *et al*. 2008; Suzuki *et al*. 2012). Goldingay and Taylor (2009) found that squirrel glider glide distances were highly correlated with glide launch height, where tall trees (>25 m) were used to glide long distances up to 43 m. Studies by van der Ree (2003) showed that gaps of up to 75 m are crossed by *Petaurus* spp. These studies and others (van der Ree *et al*. 2010) have shown the importance of tall trees to facilitate movements of gliders through both fragmented and urbanised landscapes. Alternatively, the selection of taller trees by squirrel gliders may be related to predator and barrier avoidance, particularly from introduced cats and foxes (van der Ree 2002).

Squirrel gliders were more likely to be present in more isolated tall trees, which can be attributed to several factors. First, if tall trees are preferred, but occur in a clump, then squirrel gliders are less likely to be present in a particular tree within that clump, so this resource is effectively diffused. Second, more isolated tall trees provide advantages in terms of line of sight and access, as opposed to difficulties presented in gliding through clumps of trees. For example, Selonen and Hanski (2003) found that the flying squirrel (*Pteromys volans*) moved faster and more directly through less dense habitat to avoid high-risk, non-preferred habitat. Because much of the Thurgoona landscape is dominated by patches of regrowth, revegetation areas, and urban housing, isolated tall trees may confer advantages to squirrel gliders in terms of avoiding barriers during movements.

Urban effects on squirrel glider occurrence

Logistic models for squirrel glider occupancy showed that noise had a negative effect on squirrel glider presence; however, this factor was included only in our lowest-ranked model. In general,

Table 1. Top Akaike's information criterion corrected for small sample sizes (AIC_c) models ($\Delta_i < 2$) for squirrel glider occupancy, using logistic regression

K, parameter in the model; D-AIC_c, difference in AIC_c from the top model ($\Delta_i < 2$); Akaike weights are summed (ΣW_i) across all candidate models in the candidate set. Linear relationship is represented by (+) or (–)

Logistic model rank	*K*	AIC_c	D-AIC_c	Tree height	Distance to nearest tall tree	Latitude	Density of hollow-bearing trees	Road density	Road type	Distance to nearest hollow-bearing tree	Maximum noise
1	4	45.81	0	(+)	(+)	(+)					
2	3	46.01	0.20	(+)	(+)						
3	4	46.22	0.41	(+)	(+)		(+)				
4	4	46.59	0.78	(+)	(+)			(–)			
5	4	47.16	1.35			(+)			(–)		
6	4	47.32	1.51	(+)	(+)					(+)	
7	4	47.63	1.82	(+)	(+)						(–)
$\Sigma \boldsymbol{W}_\mathbf{i}$				0.45	0.41	0.16	0.13	0.19	0.08	0.14	0.22

Table 2. Top Akaike's information criterion corrected for small sample sizes (AIC_c) ($\Delta_i < 2$) for squirrel glider activity, using a zero-inflated model

K, parameter in the model; D-AIC_c, difference in AIC_c from the top model ($\Delta_i < 2$); Akaike weights are summed (ΣW_i) are across all candidate models in the candidate set. Linear relationship is represented by (+) or (–)

Zero-inflated model rank	*K*	AICc	D-AICc	Distance to nearest tall tree	Road density	Distance to nearest hollow-bearing tree	Maximum light
1	4	124.88	0	(–)	(–)		
2	4	125.54	0.66		(–)	(+)	
3	4	126.00	1.12	(–)			(–)
$\Sigma \boldsymbol{W}_\mathbf{i}$				0.36	0.57	0.17	0.34

noise emanating from anthropogenic sources (principally human transport) was significantly louder in areas where squirrel gliders were absent. Several studies have inferred that noise pollution is the principal cause for changed spatial movements for many animals, leading to avoidance of noisier areas; however, few studies have measured noise effects directly (Forman *et al*. 2003; Bautista *et al*. 2004).

Variability in our results in relation to noise and the occurrence of squirrel glider was partly explained by local factors. A major highway located on the western boundary of the study area was a strong source of noise, which, depending on wind direction, pervaded the rural matrix to an extent that was unexpected. Noise from the highway was consistent, rather than spontaneous loud noises that often occur in urban environments and are known to scare animals (Maier *et al*. 1998; Wright *et al*. 2010), or even awaken them (Arroyo-Solís *et al*. 2013). As a result, anthropogenic noise was experienced by squirrel gliders in most locations of this study area to varying degrees. Given their persistence in the landscape, these results suggest that noise is tolerated by squirrel gliders in many situations (in terms of occupancy); however, impacts on breeding and movement behaviour are unknown.

Although gliders were generally found in quiet locations, two individual survey trees yielded contradicting results. Both trees were occupied by squirrel gliders, yet both were in the loudest locations of the study area (one within 10 m of a busy 100 km h^{-1} road, and the other next to a high-traffic intersection). These findings suggest that the benefits (i.e. key resources such as nest sites and food) provided by specific hollow-bearing trees in the landscape may outweigh the deleterious effects of human noise levels, at least in the short term. This hypothesis is supported by other studies of squirrel gliders, finding populations of squirrel gliders inhabiting linear roadside vegetation patches (van der Ree 2002), where noise from traffic had not appeared to have an impact on their persistence.

Road density also influenced squirrel glider occupancy, and was included in our third 'best' model to predict glider occupancy. Brearley *et al*. (2010) found similar results, where squirrel gliders were far more abundant in interior habitat sites away from roads, than in habitat patches along major and minor roads. Given that squirrel gliders are found in roadsides despite the adverse effects of roads *per se* (i.e. risks involved with moving vehicles) suggests that they can tolerate anthropogenic influences to a certain extent, providing other key resources are available.

Factors influencing the activity of squirrel gliders in urban trees

Following above, generalised linear modelling showed that road density was a significant negative predictor of squirrel glider activity. This result was not surprising; as urban development intensifies, road and housing density increases, as well as associated noise from cats and dogs. In contrast, the total area of parks or other conservation reserves normally decreases with road density (Brady *et al*. 2009). For squirrel gliders and many other species, roads represent a barrier to movements, either by the gap-crossing distance, or by moving traffic, or both. Forman and Alexander (1998) have previously outlined the impacts of roads and their barrier effects, highlighting the lack of movement across roads for a range of animal species.

Light pollution from urban sources had a negative influence on the frequency of use of trees by squirrel gliders. Generally, trees with higher activity were located in dark areas (both urban and rural areas). These results are supported by an experiment by Barber-Meyer (2007), who found that captive sugar gliders decreased activity and foraging time under two artificial light treatments, designed to be similar to street lighting. Threlfall *et al*. (2013) suggested that large patches (>40 ha) of native vegetation are required to prevent such light pollution from having an impact on bats and other species. Given that the home-range size for squirrel gliders varies between 4 and 15 ha (Sharpe and Goldingay 2007; Goldingay *et al*. 2010; Brearley *et al*. 2011), but only small habitat patches (such as roadsides) remain in urban areas such as Thurgoona, squirrel gliders are likely to experience constant deleterious impacts from urban light pollution as they move about at night.

Squirrel glider activity was strongly influenced by distance to the nearest hollow-bearing and other tall trees. However, this result conflicts with model results for occupancy, where the advantages suggested for squirrel glider use of isolated tall trees (as opposed to clumps of tall trees) may not be supported. Further studies are required to ascertain the effects of neighbouring tree densities on squirrel glider tree selection and activity, where (as we hypothesise) certain tall trees may act as 'nodes' in connected aerial pathways for squirrel gliders to forage, whereas other trees located in vegetated patches may provide better conditions for nesting.

Gliding mammals use trees for several reasons, such as for movement, foraging, shelter and breeding (van der Ree and Suckling 2008; Sharpe 2009). Therefore, the factors that influence the activity of gliders in a particular tree may be specific to that tree, and change from season to season, as key resources (e.g. food) also change. In our study, we found that gliders were less active in urban trees. Activity is explicitly linked to detection probability (Manning 2011); therefore, if squirrel gliders behave differently in urban areas, we were less likely to detect them by using the remote camera setup. For example, our camera traps were located lower on the ground than where gliders might typically forage (Crane *et al*. 2012), potentially leading to fewer detections. However, Goldingay and Taylor (2009) showed that the majority of glides land on the trunk of a tree, potentially leading to more detection. These considerations, in conjunction with a small sample size, and complexity of ecological and anthropogenic interactions, may explain some of the variability in our results.

Conclusions

Although squirrel gliders used large trees in both urban and rural areas, the use of such trees generally decreased as levels of urbanisation increased. These results suggest that pursuing conservation-management objectives in the urban 'backyard' is indeed problematic. Urban factors that have an impact on gliders, such as roads, noise and light, are difficult to mitigate. A reduction in road traffic and associated noise is not likely, because urban areas such as Thurgoona will continue to expand. Light reductions do not seem likely either, because the majority of anthropogenic light recorded during the present study was produced by street, house and car lights, all being necessary for human habitation. However, the use of street lighting could be changed between 2400 hours and

0600 hours when human use is limited, or by using alternative lighting that is less intrusive on wildlife (Stone *et al*. 2009; Falchi *et al*. 2011).

The results of the present study underlined the importance of conserving large, hollow-bearing trees for squirrel gliders. However, these trees are often scarce in urban areas, isolated by housing development, and threatened by continued decay (Harper *et al*. 2005; Davis *et al*. 2013). Therefore, the retention of patches of suitable habitat and large eucalypt trees is critical for conserving gliders. With ongoing revegetation and other conservation practices (e.g. installation of artificial nest boxes; Beyer and Goldingay 2006), populations of squirrel gliders may persist in urban areas, where novel resources provided by humans (e.g. gardens) may provide additional benefits (Baker and Harris 2007; Brearley *et al*. 2010). Alternatively, there may be time lags in the eventual demise of local populations undergoing urban encroachment.

The use of infrared, motion-sensor cameras was a highly successful method to survey squirrel gliders, highlighting its potential use for a wide range of other studies, such as, for example, monitoring the use of glider poles (Taylor and Goldingay 2012*b*). Other squirrel glider survey techniques such as spotlighting, physical trapping and stag watching are also proven methods for surveying arboreal fauna (Goldingay and Sharpe 2004; Harley *et al*. 2014); however, these techniques require much time and effort by the researchers involved. In contrast, the use of cameras allows for multiple trees to be analysed simultaneously, without an increase in the number of researchers, and facilitates around-the-clock monitoring. Because individual squirrel gliders were difficult to distinguish from the images obtained, a study of squirrel gliders that combines marking (to identify individuals) with camera monitoring may provide further useful information on the movements of such fauna in urban or rural landscapes elsewhere.

Acknowledgements

Data were collected according to ethical guidelines for treatment of animals, Charles Sturt University (Animal Ethics Approval Protocol No. 13/024), approved 11 April 2013. Funding for this project was provided by the Albury Conservation Co. Staff from the Albury City Council (Dr Rachel Clancy, Steve Onley and Vince Christopherson) provided generous GIS mapping and technical assistance installing cameras, and for arranging necessary work plans and permits. Ross Goldingay, Dan Lunney and three anonymous reviewers provided valuable comments on an earlier draft of this manuscript. Several landholders provided access to their properties who we owe thanks to: Jill and Keith McDonald, Ellen Sampson, Jill Williams and Andrea Zanin. Deanna Duffy and Simon McDonald from CSU provided assistance with GIS and statistical analysis. Grace Eppelstun, Meg Garty and Anna McLean assisted with fieldwork at late hours. Special thanks go to Sam Niedra (Nature Conservation Trust of NSW) for his contributions, and compiling information on the current status of squirrel gliders in Thurgoona.

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