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Published in:
Diversity & Distributions

DOI:
[10.1111/j.1472-4642.2012.00937.x](https://doi.org/10.1111/j.1472-4642.2012.00937.x)

Publication date:
2013

Document Version
Publisher's PDF, also known as Version of record

[Link to publication](#)

Citation for pulished version (APA):

Ikin, K., Knight, E., Lindenmayer, D. B., Fischer, J., & Manning, A. D. (2013). The influence of native versus exotic streetscape vegetation on the spatial distribution of birds in suburbs and reserves. *Diversity & Distributions*, 19(3), 294-306. <https://doi.org/10.1111/j.1472-4642.2012.00937.x>

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The influence of native versus exotic streetscape vegetation on the spatial distribution of birds in suburbs and reserves

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ABSTRACT

Aim Management practices in the landscape matrix can have significant effects on the spatial distribution of animals within adjacent protected areas. This has been well established in agricultural and forested areas, but less is known about how management of the suburban matrix affects adjacent reserves. We argue that it is critically important to understand the impact of suburban management on reserves, as flawed planning decisions can have negative conservation outcomes and waste limited resources.

Location Canberra, Australian Capital Territory, Australia.

Methods We examined bird distribution patterns in suburbs and adjacent reserves to the effects of two suburban management practices: (1) street tree planting and (2) boundary design. We focused on three groups of birds with known responses to urbanization: native urban-intolerant species (native avoiders), native urban-tolerant species (native adapters) and exotic urban-tolerant species (exotic adapters).

Results We found that suburbs with $\geq 30\%$ native (*Eucalyptus*) street trees and reserves adjacent to these suburbs had significantly higher bird species richness, native adapter species richness and probability of reporting exotic adapters than those with exotic trees. The type of street trees, however, did not affect the probability of reporting native avoiders. These species were more likely to be reported when habitat complexity was high. Only native adapters responded to boundary design, with higher species richness when the boundary type was a local or unsealed road as opposed to an arterial road.

Main conclusions Native street trees provide foraging resources for birds that would be reduced or absent in exotic streetscapes, enabling native streetscapes to support a rich community of birds. Furthermore, native streetscapes increase bird richness and diversity in adjacent reserves. This result has important conservation implications for suburb and reserve management practices. Our study provides evidence that the establishment and retention of native suburban streetscapes is an important management strategy for improved bird conservation.

Keywords

Landscape planning, matrix management, protected areas, South-eastern Australia, urban forest, urbanization.

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INTRODUCTION

Management practices in the landscape matrix can have significant effects on the spatial distribution of animals within adjacent protected areas (Noss & Harris, 1986; Sisk *et al.*, 1997; Renjifo, 2001; Kristan *et al.*, 2003). The matrix can affect habitat quality, species' dispersal abilities and population persistence in protected areas, as well as be a source of resources, predators and competitors (Bentley & Catterall, 1997; Sisk & Battin, 2002; Tubelis *et al.*, 2004, 2007; Dunford & Freemark, 2005). This has been well established in agricultural and forested areas (Franklin & Lindenmayer, 2009), but less is known about how management practices in the suburban matrix can influence the number and composition of fauna in adjacent reserves (Fernandez-Juricic, 2001; Brady *et al.*, 2009).

It is well documented that urbanization negatively affects many species (Soule, 1991; Luck *et al.*, 2004; Chace & Walsh, 2006), but also that it is possible to design or manage urban areas to better support wildlife (Marzluff & Ewing, 2001; Turner, 2003; Donnelly & Marzluff, 2006; Mason *et al.*, 2007). Suburban street trees can provide habitat for some species of birds (Fernandez-Juricic, 2000; Murgui, 2007), and their geographic provenance (i.e. native to the country or exotic) can affect suburban bird species richness and composition (White *et al.*, 2005). It is not known, however, how the dominance of native or exotic street trees within the suburban matrix will affect the distribution of birds in adjacent reserves. By providing similar structures and resources to wooded reserves, native streetscapes may mitigate the sharp habitat transition between the reserve and adjacent suburb and reduce the physical or behavioural barriers that prevent many birds using the suburban matrix (Yahner, 1988; Kristan *et al.*, 2003; Reino *et al.*, 2009). For example, Hodgson *et al.* (2007) speculated that insectivorous birds may be more likely to cross from a reserve into a suburb if they perceive a soft, low-contrast edge, and native street trees may encourage this behaviour. Alternatively, positive associations between exotic street trees and some suburban birds (e.g. exotic species) may increase the probability of these birds crossing into reserves when exotic trees are planted in the matrix, to the detriment of birds in reserves (Sewell & Catterall, 1998; White *et al.*, 2005).

The design of boundaries adjacent to reserves also can be important to species distributions, and low-vehicle traffic boundaries between suburbs and reserves may reduce edge contrast (Wiens *et al.*, 1985). For example, Brearley *et al.* (2010) found that the squirrel glider (*Petaurus norfolcensis*) was more likely to cross into the urban matrix when the edge consisted of a minor road, compared with a residential edge or major road. Again however, threats associated with urbanization, for example the increased abundance of avian nest predators (Yahner, 1988; Gardner, 1998), may be more likely with reduced edge contrast.

The responses of birds and other wildlife to suburb-reserve edges and the suburban matrix are highly species specific

(Mortberg, 2001; Trollope *et al.*, 2009; Huste & Boulinier, 2011). Previous studies, however, have effectively grouped birds by their functional responses to urbanization (e.g. Crooks *et al.*, 2004; Donnelly & Marzluff, 2006; Catterall, 2009; Huste & Boulinier, 2011). Following Blair (1996), birds that are sensitive to urban disturbance have been termed 'urban avoiders'. These species are intolerant or unable to use the suburban matrix and are mostly confined to reserve areas, although they may occasionally be recorded in suburban areas. Other species that respond positively to urbanization are termed '(sub)urban adapters' and 'urban exploiters' (Blair, 1996). Urban adapters are tolerant of moderate levels of urbanization, for example, suburbs, and in Australia include many native species that take advantage of flowering garden plants and mown grassy lawns (Green, 1984; French *et al.*, 2005), as well as species that are ubiquitous worldwide. By comparison, urban exploiters reach their highest densities in highly urbanized areas, for example, city centres (Conole & Kirkpatrick, 2011; Huste & Boulinier, 2011). The use of these 'response guilds' in urban studies enables findings to be generalized across geographic regions (Catterall, 2009). However, although the response of each guild to increasing urbanization can be assumed, the effects of suburban management practices on these responses remain unclear. For example, urban avoiders can be expected to decline in numbers from reserves to suburbs, but does the planting of native street trees reduce the strength of this response? Or alternatively, does the construction of a low-contrast edge boundary increase numbers of urban avoiders in suburbs or urban adapters in reserves?

It is important that we improve our understanding of how suburb design influences the impact that adjacent urban development has on reserves (and vice versa) to successfully manage and conserve wildlife populations (Noss & Harris, 1986; Riley, 2006). This is because of the rapid, and continued rise in the world's urban population (United Nations, 2011), and the associated increase in urban land cover (Forman, 2008). Decisions are often made between different suburb management options, with the aim of conserving wildlife, even though the extent that these management options heighten or reduce negative effects on wildlife is poorly understood (Soule, 1991; Gordon *et al.*, 2009). It is critically important that decisions are evidence based, as flawed planning decisions can have negative conservation outcomes and waste limited resources (Sutherland *et al.*, 2004b).

The aim of our study was to investigate the effect that different suburb management practices had on the spatial distribution of birds in suburban areas and adjacent nature reserves. We focused on three groups of birds with known responses to urbanization: native urban avoiders, native urban adapters and exotic urban adapters. Specifically, we asked: Are bird distribution patterns and community composition in suburbs and reserves affected by (1) different kinds of suburb street tree planting and (2) different boundary types? We assumed that the effects of these management

practices on reserves would decline with distance from the suburbs; but to test this, we distinguished between reserve edge and core areas. Similarly, we distinguished between suburb core and edge areas. We expected that the geographic provenance of street trees (native or exotic) would affect responses in the suburb and at the reserve edge, but that the reserve core would be unaffected by the influence of suburban management. We also expected that boundary design would be important, and that low-vehicle traffic boundaries would soften the contrast, and facilitate bird movement, between the suburb and reserve compared to high-vehicle traffic boundaries.

METHODS

Study area and design

Our study was located in Canberra, Australian Capital Territory (ACT), in south-eastern Australia (Fig. 1). Canberra covers approximately 800 km², and current population density is 440 people km⁻² (ABS, 2010). It is a 'planned city' (MacKenzie & Barnett, 2006), and the presence and composition of street trees has been important since Canberra's inception in 1911 (Banks & Brack, 2003). Initially, tree planting was aimed at creating a 'garden city', with formal, exotic-dominated streetscapes (e.g. *Quercus*, *Prunus*, *Ulmus* and *Fraxinus* species). Following World War II, street plantings became more informal in arrangement but exotic tree species were still preferred. It was not until the 1960s that native *Eucalyptus* species began to be favoured in new street and suburb plantings. Since that time, street tree plantings have also become more diverse, and informally arranged multi-species plantings are now common. New developments in Canberra, however, are beginning to reflect a changed social and planning preference towards exotic-dominated street tree plantings (Banks & Brack, 2003).

There are 33 designated nature reserves in Canberra, and many occur in the hills and ridges surrounding urban areas (ACTPLA, 2008). We identified interface locations where reserves directly adjoined suburban residential areas. Each reserve was wooded and larger than 20 ha and each suburb was at least 15 years old with established street trees. All interfaces were located in suburban areas of similar high socio-economic status (which can be interpreted as a proxy for similarity and quality of private gardens: Strohbach *et al.*, 2009) and similar access to public parkland (see Table S1 in Supporting Information). We stratified the interfaces by (1) the provenance of dominant tree cover along the suburban streets and (2) the type of interface boundary separating the reserve and suburb. To classify tree cover, we used a geographic information system spatial layer of street tree cover. We then ground-truthed these data to estimate an average per cent suburban *Eucalyptus* tree cover for each group of sites. We classified the suburb as either native ($\geq 30\%$ mature *Eucalyptus* species tree cover) or exotic ($<30\%$) in provenance. We excluded suburbs with $<30\%$ *Eucalyptus* species but $\geq 30\%$ tree cover of other native species, for

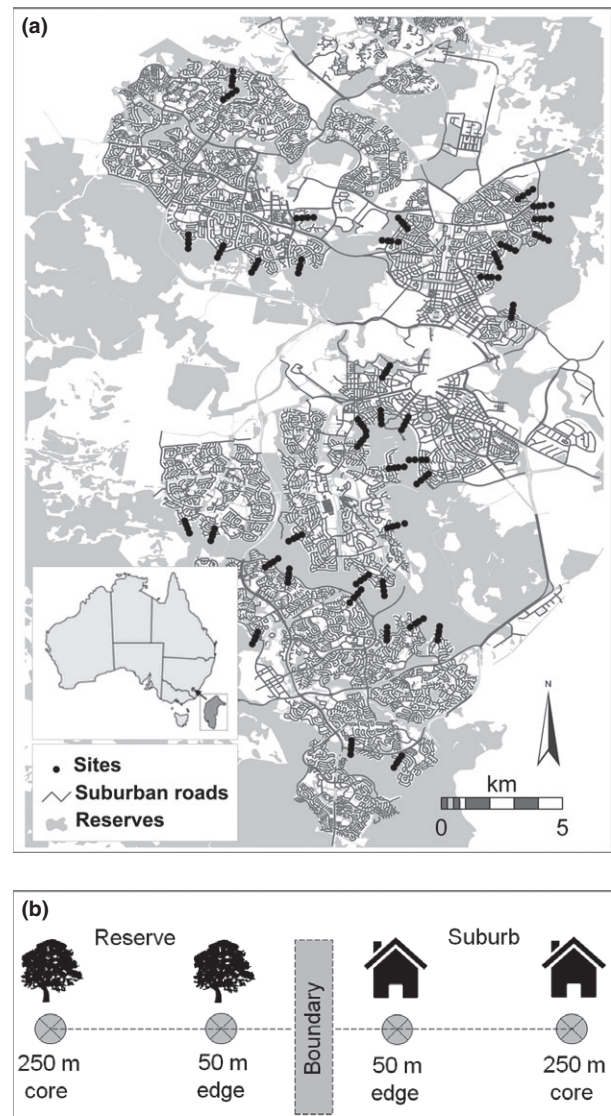


Figure 1 The study was located in Canberra, Australian Capital Territory, Australia. At forty locations, groups of four 50-m-radius sites were placed at the interface of adjoining suburbs and nature reserves (a). Within each group, two sites were located within the suburb and two within the reserve, at 50 m and 250 m distances from the boundary (b).

example, *Casuarina* species. To determine boundary type, we used aerial photography and street maps to classify the interface boundary as unsealed management track or sealed public road. We then further divided sealed roads by road classification into arterial and local roads, corresponding to high- and low-vehicle traffic intensity, respectively. We found that the mean width of each boundary type was related to traffic intensity: arterial roads had a mean width of 72 m, local roads 31 m and unsealed roads 39 m.

We selected 40 interface locations that were separated by at least 0.5 km. Of these, six were 'native/arterial', two were 'native/local', 14 were 'native/unsealed', one was 'exotic/arterial', five were 'exotic/local' and 12 were 'exotic/unsealed'

(Fig. 1). We placed four 50-m-radii sites at each interface: two sites within the reserve at approximately 50 m and 250 m distances from the boundary ('reserve edge' and 'reserve core'), and two sites at the same distances in treed suburban streets ('suburb edge' and 'suburb core'). Hereafter, we refer to this position along the suburb-reserve interface as 'site location'.

Suburb and reserve habitat complexity

At each site, we measured the per cent cover of the canopy, large and small shrubs, grass, leaf litter and impervious surfaces. We scored the per cent cover of each of these components and summed the six scores to achieve a habitat complexity score (HCS) for each site (Table 1). The HCS corresponds to vegetation complexity within a site: higher numbers of vegetation components and/or higher scoring components give a higher HCS. We used analysis of variance to compare HCS (1) between suburbs and reserves, and (2) within suburbs and reserves, between sites grouped by street tree provenance.

Bird surveys and response groups

We surveyed birds during spring 2009. This is the peak breeding season in south-eastern Australia when birds, including summer migrants, exhibit strong site fidelity owing to the establishment of breeding territories (Montague-Drake *et al.*, 2009). We used the 50-m-radius point count method to record all birds seen or heard within a 10-min period (Sutherland *et al.*, 2004a). We surveyed each site twice, once each by two observers (K. Ikin and S. Holliday). All of the sites within a group were surveyed on the same day, although not always by the same observer, and repeat surveys took place on a separate day. We surveyed between dawn and 10 am, and avoided periods of rain, high wind or high temperatures (>30 °C).

We assigned each bird species into one of six response groups: Australian urban 'native adapters' and 'native avoiders', non-native 'exotic adapters' and 'exotic avoiders', and

native and exotic 'neutral' bird species (see Table S2 in Supporting Information for classification). We classified these response groups using independent data from the Canberra Ornithologist Group Garden Bird Survey (GBS). The GBS is a community-based study where volunteers record species observed in their garden site during each week of the year (Fennell, 2009). We classified native/exotic adapters as native/exotic bird species present at ≥ 75% of GBS sites in 2007–08 (n = 72). The exotic adapter response group included common Northern Hemisphere species, such as the house sparrow (*Passer domesticus*) and common starling (*Sturnus vulgaris*). We classified native/exotic avoiders as natives/exotics present at ≤ 25% of GBS sites (COG, 2009). We only recorded one exotic avoider, the spotted dove (*Streptopelia chinensis*), and excluded this response group from further analysis. Finally, we classified other native/exotic bird species present at 26–74% of sites as native/exotic neutral species.

We calculated the reporting rate for each bird species at each site. Reporting rates are the proportion of surveys at each site that a species was recorded (Cunningham *et al.*, 2008; Lindenmayer & Cunningham, 2011), and thus can range from 0 (species was not recorded) to 1 (species was recorded during both surveys for all sites). We then calculated an overall mean reporting rate for each bird species, as well as their mean in suburbs and reserves classified by street tree provenance. This enabled us to rank and compare species by their reporting rates and to assess each species response along the suburb-reserve interface.

Bird responses to native versus exotic street trees and the interface boundary

We wanted to compare how well site location, suburb street tree provenance, boundary type and habitat complexity explained the distribution pattern of all bird species and those with known responses to urbanization, that is, native adapters, native avoiders and exotic adapter bird species. Native and exotic neutral species did not have a predictable response to urbanization, and for this reason we did not

Table 1 Summary of Habitat Complexity Scores (HCS) for suburb and reserve sites grouped by suburb street tree provenance. Every site was given a HCS score for each canopy, shrub and ground layer per cent cover component. Scores ranged from 0 to 3, and higher per cent covers were scored higher, except for grass and impervious surfaces which were scored lower. The stacked bar charts show the proportion of sites that were assigned each component score (the width of the stacked bars for each component shows the relative proportion of sites with each score).

HCS components and scores	Score				Visualisation of site scores			
					Reserve		Suburb	
	0	1	2	3	Native	Exotic	Native	Exotic
Canopy	0–10	11–20	21–50	>51	(n=44)		(n=36)	
Large shrubs	0–10	11–20	21–50	>51	(n=44)		(n=36)	
Small shrubs	0–10	11–20	21–50	>51	(n=44)		(n=36)	
Leaf litter	0–10	11–20	21–50	>51	(n=44)		(n=36)	
Grass	>51	21–50	11–20	0–10	(n=44)		(n=36)	
Imperv. surfaces	>51	21–50	11–20	0–10	(n=44)		(n=36)	

consider them in these analyses. We used generalized linear mixed models, with a Poisson distribution for total species and native adapter species richness, and a binomial distribution for native avoider and exotic adapter probability of reporting (for which we recorded too few species to meet normality assumptions). We assumed that the four sites within a group were not independent. We therefore tested for and found no significant spatial autocorrelation *within* each group of four sites. We thus simplified the random effects model and fitted 'site group' as a random effect to account for possible dependence *between* sites within a group. There were no interactions between the explanatory variables.

Last, we wanted to explore bird community composition in the different site locations and to see if it was related to street tree provenance and boundary type. We used canonical correspondence analysis of species presence/absence data to investigate community structure and to relate this structure to site location, street tree provenance and boundary type (ter Braak, 1986). We removed species from the analyses that we recorded at only one site because uncommon species can obscure community patterns (MacFaden & Capen, 2002). We tested for overall significance, as well as the significance of the marginal effects of the individual variables, using permutation tests, with a maximum of 1000 permutations.

RESULTS

Suburb and reserve habitat complexity

We found that suburbs had a significantly lower mean habitat complexity score (HCS) than reserves (mean \pm SE, suburb: 4.4 ± 0.2 vs. reserve: 8.1 ± 0.3 , $P = 0.011$) but street tree provenance did not significantly affect the complexity of suburb vegetation (native: 4.7 ± 0.3 vs. exotic: 4.0 ± 0.3 , $P = 0.425$) or adjacent reserve vegetation (native:

7.9 ± 0.4 vs. exotic: 8.4 ± 0.6). This was reflected in the contribution of each HCS component to mean scores (Table 1). Native suburbs tended to have higher litter cover than the grassier exotic suburbs, but there was little difference in the relative contribution of each component within reserves.

Bird responses to native versus exotic street trees and the interface boundary

We recorded 66 species of birds, of which 17 were native adapters, 20 were native avoiders, four were exotic adapters, one was an exotic avoider, 23 were native neutral species and one was an exotic neutral species (Fig. 2; Table S2 in Supporting Information). Total species richness ranged from four to 18 species per site (mean 10.25) and the regression analysis showed a significant relationship with street tree provenance (Table 2, Figs 3 & 4). Native adapter species richness ranged from one to 10 species per site (mean 6.42) and was related to street tree provenance and boundary type (Table 2, Figs 3 & 4). We found that the probability of reporting native avoiders was related to site location and habitat complexity, and the probability of reporting exotic bird species was related to site location and street tree provenance (Table 2, Fig. 3, 4 & 5).

Our data showed that species richness and probability of reporting was higher for all response groups at all site locations when street trees were native (Table 2, Fig. 3). This relationship, however, was not significant for native avoiders. Richness and reporting probabilities in reserves adjacent to native suburbs were higher than in reserves adjacent to exotic suburbs.

Native adapter species richness was significantly higher at all site locations where the boundary type was a local or unsealed road (Table 2, Fig. 4). This pattern also was true for total species richness, although this relationship was not

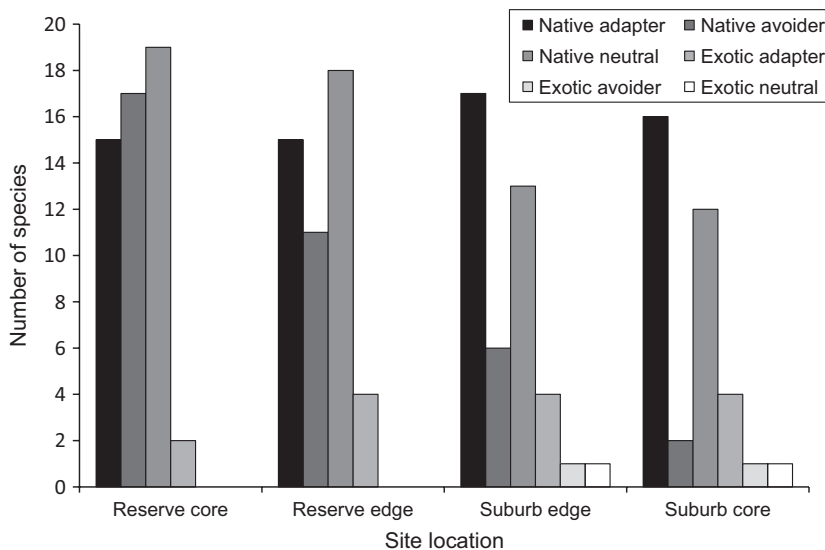


Figure 2 Number of species from each response group recorded at each site location. In total, 17 native adapters, 20 native avoiders, four exotic adapters, one exotic avoider, 23 native neutral and one exotic neutral bird species were recorded (see Table S2 in Supporting Information individual species classification).

Table 2 Generalized linear mixed models for total species and native adapters, native avoiders and exotic adapters response groups. The shape of each relationship is shown in Figs 3, 4 & 5. Total species and native adapter richness were analysed with a Poisson distribution, and native avoider and exotic adapter probability of reporting were analysed using a binomial distribution.

	Species richness		Probability of reporting	
	Total spp.	Native adapters	Native avoiders	Exotic adapters
Site location				
Wald statistic	2.11	7.38	12.40	22.06
d.f.	3	3	3	3
P-value	0.551	0.065	0.008	<0.001
Street trees				
Wald statistic	7.29	6.73	0.36	5.13
d.f.	1	1	1	1
P-value	0.011	0.010	0.553	0.029
Boundary type				
Wald statistic	5.07	8.24	0.05	0.19
d.f.	2	2	2	2
P-value	0.093	0.018	0.975	0.908
Habitat complexity				
Wald statistic	0.61	0.15	8.37	1.99
d.f.	1	1	1	1
P-value	0.436	0.700	0.004	0.160

significant. In contrast, boundary type did not have a discernible effect on the probability of reporting either native avoiders or exotic adapters (Table 2).

We performed a canonical correspondence analysis (CCA) on the 57 species recorded at more than one site. We found community composition was significantly related to site characteristics ($P = 0.001$) and that the first two axes explained 73% of variation. We also found that site location ($P = 0.001$) and street tree provenance ($P = 0.023$) were significantly related to species composition, but there was no relationship with boundary type ($P = 0.144$). The first CCA axis represented the suburb-reserve interface and explained 61% of the total variation in the ordination (Fig. 6). Dominance of the response groups changed along this axis, with native and exotic adapters related to suburb sites and native avoiders related to reserve sites. The single exotic avoider recorded was related to suburban sites suggesting that it is not a true urban-intolerant species. The first CCA axis suggests that native avoiders were being replaced by native and exotic adapters with distance from the reserve core. The second CCA axis represented street tree vegetation, and explained 12% of the total variation in the ordination. This suggests that more bird species seem to be related to native street trees than to exotic street trees.

DISCUSSION

We investigated the influence of different suburb management practices on the spatial distribution of birds in suburban areas and adjacent nature reserves. Specifically, we examined the effect of (1) different kinds of suburb street tree planting and (2) different boundary types on bird distribution patterns and community composition. We found that suburbs with $\geq 30\%$ native (*Eucalyptus*) street trees and reserves adjacent to these suburbs had significantly higher richness of all species and of native urban-adapted species, and higher probability of reporting exotic urban-adapted species, than those with exotic trees, even at the reserve core where no effect was expected. Street tree provenance, however, did not affect the probability of reporting native urban avoiders. Only native adapters responded to boundary design, with higher species richness when the boundary type was a local or unsealed road. In the following sections, we discuss potential mechanisms underpinning these responses. We also discuss a range of conservation aims for birds in suburbs and reserves, and how alternative suburban planning options may lead to different bird communities.

Bird responses to native versus exotic street trees and the interface boundary

When street trees were of native ($\geq 30\%$ *Eucalyptus*) provenance, we found more bird species overall and more native adapters. *Eucalypts*, being by far the dominant native tree taxon, provide diverse foraging resources for birds, via foliage, flowers, bark, canopy air spaces, and a leaf litter or coarse woody debris ground layer (reviewed by McElhinny *et al.*, 2006). These components afford a source of invertebrates, nectar and plant exudates that are missing or less abundant in non-eucalypt species (Woinarski & Cullen, 1984). Insectivores and nectarivores comprised 70% of overall bird species, and 65% of native adapters (K Ikin, unpublished data), and the planting of native street trees increases available foraging resources for these species.

We defined a suburb as being planted with native trees when more than 30% of trees were eucalypts. However, this should not be interpreted as a rule of thumb. More research is needed to quantify a threshold for when a suburb becomes 'native' (if one exists) and to determine if it would apply in other geographical regions and if other taxa would respond the same way. Furthermore, we did not identify the species of eucalypts planted in each street, but this may be important to bird richness and composition. For example, eucalypt species of the *Monocalyptus* subgenera, for example, stringybarks and scribbly gums, have lower foliage nutrient levels, poorer invertebrate assemblages and are less likely to produce hollows compared with species of the *Symphomyrtus* subgenera, for example, boxes and red gums (McElhinny *et al.*, 2006). Further, even within the same subgenus, different eucalypt species can differ significantly in the richness and

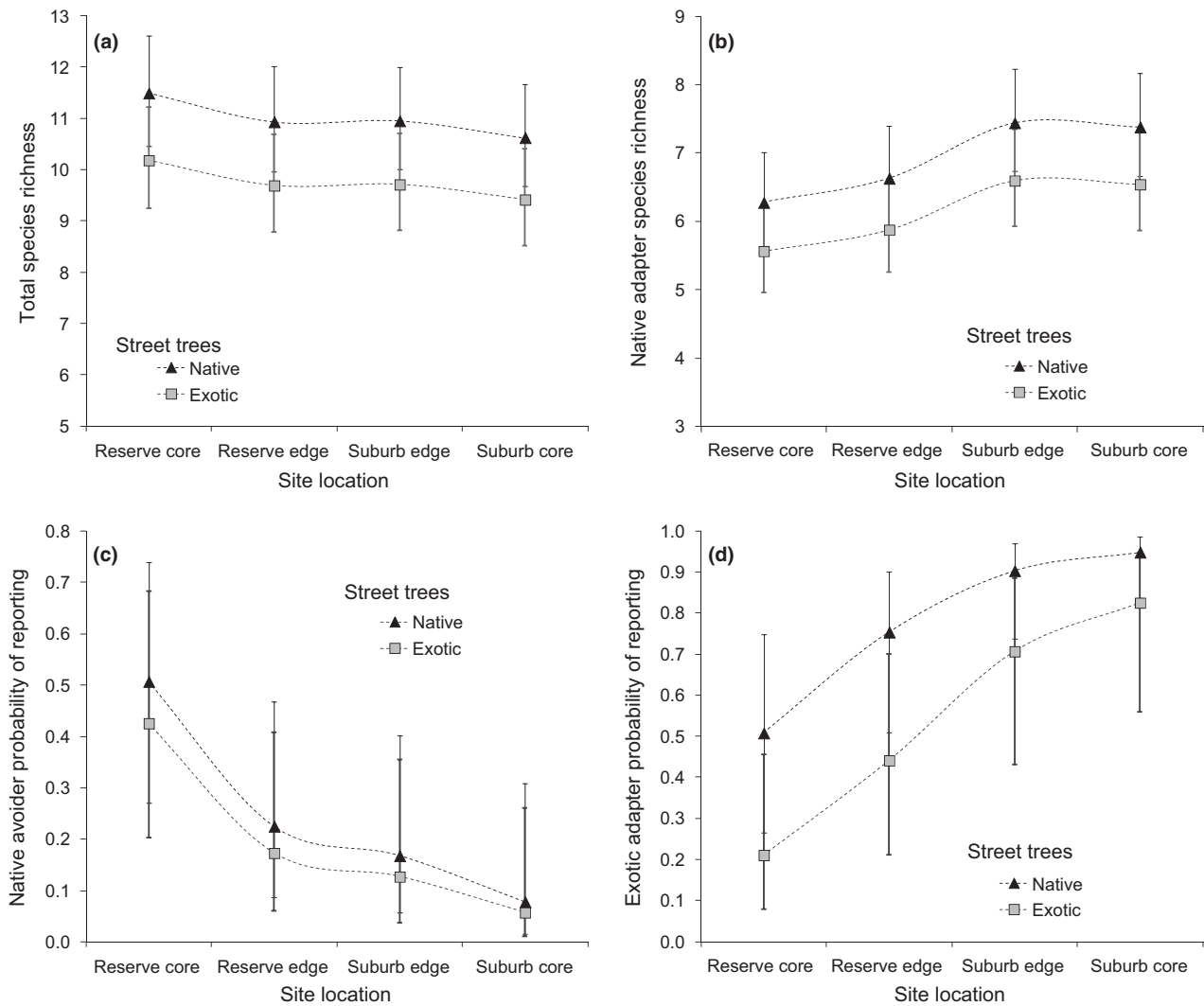


Figure 3 Relationship between site location and street tree provenance for: (a) total species richness, (b) native adapter species richness, (c) native avoider probability of reporting and (d) exotic adapter species probability of reporting, $\pm 95\%$ confidence intervals. Dotted lines visualize general trend.

abundance of insects they support (Barton *et al.*, 2010). The importance of multi-species plantings should also be addressed in future research. Previous studies have found that the range of peeling and non-peeling bark characteristics found in mixed eucalypt communities contributes to high local diversity of birds (McElhinny *et al.*, 2006). This suggests that streets dominated by only a single species, or a few species, of eucalypt would not support the comparatively high number of native birds recorded in the present study. Similarly, further research could explore whether streets with a high diversity of exotic trees support higher bird diversity than streets with single-species plantings, such as those common in our study.

Despite the increased foraging resources provided by eucalypts, the planting of native street trees did not affect the probability of reporting native avoiders, of which 70% were insectivorous or nectarivorous (K. Ikin, unpublished data). In comparison, these species were more likely to be recorded

when site habitat complexity was high. Suburbs with more leaf litter and shrub cover and less grass and impervious surface cover may provide shelter and alternative foraging and nesting resources that assist urban-avoiding species to survive in urban areas. It is also possible that native avoiders experience a behavioural response to suburbs that is independent of street tree vegetation. For example, in Spain, Fernandez-Juricic (2001) showed that species with specialized habitat requirements can be intolerant of human disturbance and occur in lower numbers and breeding densities compared with generalist species habituated to human activities. In addition, more than half of native avoiders in our study weighed less than 30 g and may experience greater predation by avian and domestic predators (Gardner, 1998). The noisy miner (*Manorina melanocephala*), an aggressive native honey-eater, has also been shown to have a strong negative effect on small passerines in Australia (Piper & Catterall, 2003; Clarke & Oldland, 2007; Kath *et al.*, 2009). However, we

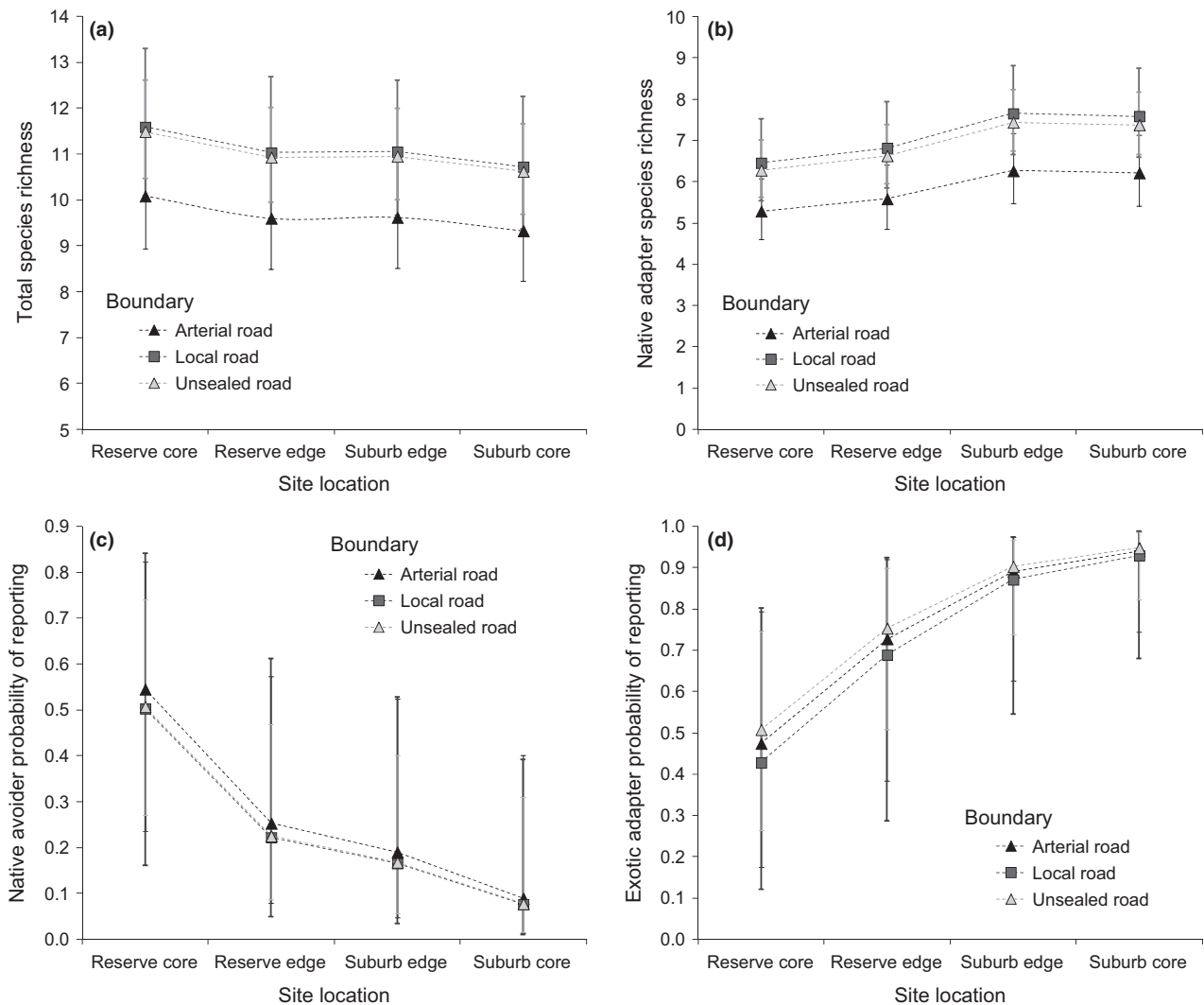


Figure 4 Relationship between site location and boundary type for: (a) total species richness, (b) native adapter species richness, (c) native avoider probability of reporting and (d) exotic adapter species probability of reporting, $\pm 95\%$ confidence intervals. Dotted lines visualize general trend.

reported this species at only 20% of sites, suggesting that it had a limited influence on native avoiders within our study.

The probability of reporting exotic urban-adapted bird species increased at all site locations when street trees were native. This result was unexpected, given that several Australian studies have found either a negative relationship or no relationship between exotic bird species and native street-scapes (Green, 1984; Catterall *et al.*, 1989; White *et al.*, 2005, 2009). However, although these studies recorded a similar suite of exotic bird species to our study, their surveys were conducted outside of the spring breeding season and/or placed a greater emphasis on garden vegetation in the experimental design, and as such, the findings between the different studies may not be strictly comparable. Furthermore, although most exotic street trees and exotic birds were of Northern Hemisphere origin, we did not distinguish between trees and birds native to Europe versus North America. Thus, some species do not naturally occur together, for

example the European blackbird (*Turdus merula*) and American white oak (*Quercus alba*), which means that a positive relationship between exotic trees and exotic birds should not be *a priori* assumed. Urban-adapted exotic species may also be more able to adjust to and utilize novel resources (Clucas & Marzluff, 2011), such as those provided by native eucalypt trees. Cavity-nesting common starlings (*Sturnus vulgaris*), for example, frequently nest in eucalypt tree hollows (Gibbons & Lindenmayer, 2002). More research is needed, therefore, on how exotic urban-adapted bird species respond to suburban street tree provenance.

We found that native adapters were the only guild to respond to the interface boundary, and exhibited greater species richness at all site locations when the boundary type was an unsealed or local road. Positive responses of birds to low-traffic intensity roads have been reported previously (e.g. Forman *et al.*, 2002), and it was surprising that native avoiders and exotic adapters did not show a similar response.

Table 3 Recommended suburb and reserve management practices for different target bird groups

Conservation aim	Target bird group	Recommended suburb management practices	Recommended reserve priorities
Birds present, no matter which species or how many species	'Any and all' Urban-adapted native and exotic species	Native-provenance street trees	Pockets of urban reserves to increase bird numbers in suburbs
High numbers of native bird species, regardless of species diversity or formal conservation status	'Native favourites' Urban-adapted native species	Native-provenance street trees Low-traffic intensity boundaries between suburbs and reserves	Pockets of urban reserves to increase bird numbers in suburbs
High diversity of native birds, possibly but not necessarily including those with formal conservation status	'Native highlights' Urban-avoiding native species	Greater focus on private gardens and habitat-enhanced open space (e.g. parks with a high habitat complexity) – more research needed	Combination of urban reserves to increase bird numbers in suburbs and areas of undisturbed reserves to increase bird numbers in reserves
Increased numbers of birds with formal conservation status, e.g. endangered, threatened or vulnerable	'Conservation concern' Native 'listed' species	Greater focus on private gardens and habitat-enhanced open space (e.g. parks with a high habitat complexity) – more research needed	Large, undisturbed areas of reserves away from suburbs

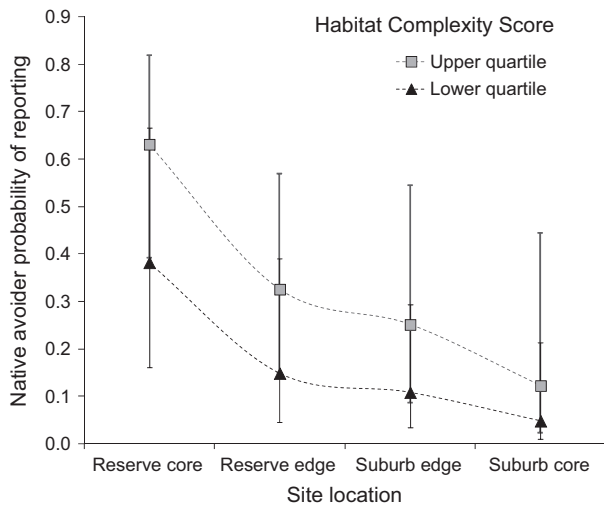


Figure 5 Relationship between site location and habitat complexity score (HCS) for native avoider probability of reporting, $\pm 95\%$ confidence intervals. Dotted lines visualize general trend.

Reduced road intensity should minimize traffic noise and pollution, create less disturbance and decrease edge contrast (Forman *et al.*, 2002; Fernandez-Juricic, 2004). Minor differences in the disturbances exerted by different boundary types, however, may be overridden by greater disturbances exerted by the suburban matrix, especially for species tolerant to this disturbance, i.e. exotic adapters, or intolerant of it, i.e. native avoiders.

Management Implications

There are many reasons to conserve birds in suburban and adjacent reserve areas. In Table 3, we illustrate a range of

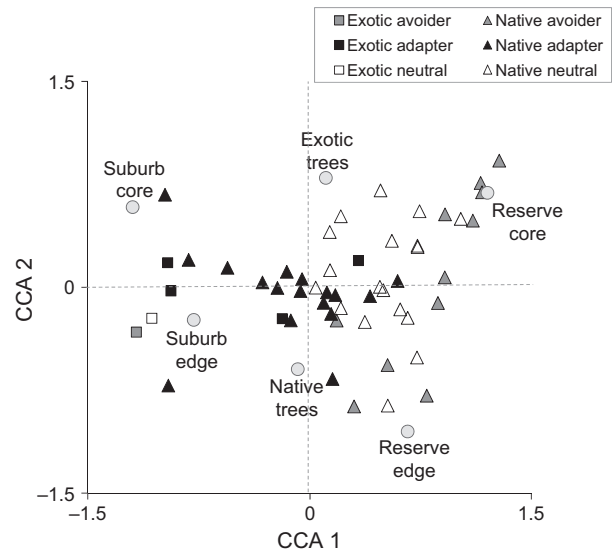


Figure 6 Canonical correspondence analysis (CCA) biplot for common bird species, classified by species response group. The centroids of site location and street tree provenance levels are shown, and proximity of species to a centroid indicates strength of the species-environment relationship. The overall significance of the CCA was $P = 0.001$.

conservation aims and management actions that our study identifies. One key reason for suburban bird conservation is that people enjoy seeing and hearing birds around their home, work and recreational spaces, even if they are not interested in what the individual species are (Recher, 2004). If a management aim is to maintain birds in suburbs and reserves, irrespective of species composition or abundance, then an 'any and all' approach could be taken (Table 3). Birds that are already benefiting from suburban habitat, such

as exotic species and urban-adapted native species, would continue to benefit from this management approach. Alternatively, if managers are interested in sustaining native birds in particular, then they might target practices in favour of 'native favourites'. These species, referred to as 'icons' by Catterall (2004), include urban-adapted native species that are charismatic, colourful, highly visible and identifiable by many 'non-birders'. Retaining and promoting these groups of birds in urban areas will also help people experience and connect with nature more easily (Miller & Hobbs, 2002), leading to important flow-on effects, such as improved health and well-being (Fuller *et al.*, 2007), as well as fostering greater concern for environmental and conservation issues (Miller, 2005). Both the 'any and all' and the 'native favourites' goals would be aided through the planting of native street trees, the retention of suburban reserves, and possibly through the design and implementation of low-traffic boundaries.

Different approaches are needed if managers want to conserve urban-avoiding native species, and increase the possibility of seeing these 'native highlights' occasionally in suburban streets (Table 3). With improved planning and design, the capacity of suburbs to support many of these species might be improved (Marzluff & Ewing, 2001; Mason *et al.*, 2007). The role of habitat-enhancing practices at the suburb-scale, for example, neighbourhood parks (Crocini *et al.*, 2008; Stagoll *et al.*, 2010), and the residential-scale, for example, private gardens (Catterall, 2004; Evans *et al.*, 2009), for this group of species needs to be further researched. Practices such as these may help to sustain these species at the regional scale, improving overall bird diversity. Lastly, there are the 'conservation concern' birds, that is, those that are formally listed in state or national threatened species legislation. In our study, we recorded only two species listed as threatened in the ACT, out of a possible nine (Nature Conservation Act, 1980), indicating that many of these species may have already been lost from suburbs and adjacent reserves. Thus, the conservation of these species may not be attainable in suburban areas, and conservation strategies should perhaps focus on the preservation of large, undisturbed reserve areas not adjacent to suburbs.

CONCLUSIONS

A better understanding of the spatial distribution of birds at the reserve-suburb interface will help guide decision makers in implementing management practices that will improve the ability of suburb and adjacent reserve areas to support bird populations. By demonstrating the importance that native street trees have for many birds within the suburb and in adjacent reserves, our study provides evidence to support the establishment and retention of native suburban streetscapes. This is especially important in contexts where conservation initiatives may contradict socio-economic and cultural preferences for exotic trees (Kirkpatrick *et al.*, 2011).

ACKNOWLEDGEMENTS

We wish to thank S. Holliday for invaluable field assistance; P. Olsen, M. Evans, R. Cunningham, J. Wood, J. Hanspach, J. and J. Stein. C. Davey, J. Bounds, L. Rayner, P. Barton, A. Carter and P. Lentini for useful discussions and provision of data contributing to the experimental design and manuscript preparation; and S. Lane and staff from ACT Parks, Conservation and Lands, R. Thorman and staff from ACT Land Development Agency, and the Canberra Ornithologist Group for their generous assistance. K.I. was the recipient of a Wildlife Preservation Society of Australia University Wildlife Conservation Grants Award, Birds Australia Stuart Leslie Bird Research Award, and the Molonglo PhD Scholarship, jointly funded by the Fenner School of Environment and Society (The Australian National University) and Parks, Conservation and Lands (ACT Government). Ethics approval was given for this work (C.RE.66.09).

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SUPPORTING INFORMATION

Additional Supporting Information may be found in the online version of this article:

Table S1 Comparison of socio-economic status and access to public parkland between interfaces.

Table S2 List of bird species and assigned response groups.

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Editor: Ingolf Kühn