

Do Common Mynas significantly compete with native birds in urban environments?

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Abstract In Australia, the introduced Common Myna (*Acridotheres tristis*) is commonly believed to aggressively displace native birds and outcompete them for food and nest resources. However, the current paucity of scientific evidence makes it difficult to devise appropriate management strategies for protection of urban bird populations. This study investigates the way in which the Common Myna uses the urban environment and interacts with other species while foraging and nesting in Sydney, Australia. The bird community varied between habitat types along an urbanisation gradient, and the abundance of the Common Myna increased significantly with the degree of habitat modification. Surveys of the frequency of interspecific interactions revealed that the Common Myna did not initiate a significantly greater number of aggressive encounters than did other species. Focal observations of two potential native competitors showed that despite foraging in close proximity, the Common Myna rarely interfered with feeding activity. Assessment of natural tree hollow occupancy found that Common Mynas used significantly fewer tree hollows than did native species. Analysis of nest site selection indicated that Common Mynas chose to nest in more highly modified habitats, and in artificial structures rather than in vegetation. These findings suggest that, in this study area, Common Mynas have little competitive

impact on resource use by native bird species in the urban matrix. The logical conclusion of these results is that the substantial efforts currently directed towards culling of Common Mynas in heavily urbanised environments is misdirected, and resources would be better directed to improvement of natural habitat quality in these areas if the purpose of control is to enhance urban bird diversity.

Keywords Common Myna · *Acridotheres tristis* · Competition · Aggression · Urban

Zusammenfassung Es wird allgemein angenommen, dass der nach Australien eingeführte Hirtenmaina die dort einheimischen Vögel aggressiv verdrängt und im Konkurrenzkampf um Nahrung und Nistmöglichkeiten schlägt. Aber der Mangel an wissenschaftlichen Nachweisen hierfür macht es schwierig, angemessene Strategien zum Schutz der in den Städten heimischen Vogelpopulationen auszuarbeiten und einzusetzen. In dieser Studie untersuchen wir die Art und Weise, auf die der Hirtenmaina in Sydney, Australien, das urbane Ökosystem nutzt und mit anderen Spezies bei der Futteraufnahme und den Nistaktivitäten interagiert. Die Vogelgesellschaft variierte zwischen einzelnen Habitat-Typen entlang eines Verstärkerungs-Gradienten, und das Auftreten des Hirtenmainas nahm signifikant mit dem Ausmaß der Habitat-Veränderungen zu. Untersuchungen zur Häufigkeit zwischenartlicher Interaktionen ergaben, dass der Hirtenmaina nicht signifikant mehr aggressive Aktionen als andere Arten unternahm. Spezielle Beobachtungen zweier einheimischer, potentieller Konkurrenten zeigten, dass sich trotz Nahrungssuche in unmittelbarer Nähe zueinander der Hirtenmaina nur selten störend in die Nahrungssuche einmischte. Die Überprüfung der Belegung natürlicher Baumhöhlen ergab, dass der Hirtenmaina signifikant weniger

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Baumhöhlen nutzte als die einheimischen Arten. Eine Analyse der Auswahl von Nistplätzen ließ vermuten, dass der Hirtenmaina bevorzugt in stärker modifizierten Habitaten und eher in künstlichen Strukturen als in Vegetation nistet. All' diese Ergebnisse legen nahe, dass der Hirtenmaina in dem hier untersuchten Gebiet, einem Stadtbiotop, für die einheimischen Vögel eine nur geringe Konkurrenz um Ressourcen darstellt. Die Schlussfolgerung hieraus ist, dass die derzeitigen substantiellen Bemühungen zur Ausrottung des Hirtenmainas in stark verstädterten Biotopen vergeudet sind und die Ressourcen besser in die Qualitäts-Verbesserung der natürlichen Habitate investiert werden sollten, wenn es darum geht, die Vielfalt der einheimischen urbanen Vögel zu erhöhen.

Introduction

Some of the most intense impacts of human activities result from urbanisation, a process characterised by increased densities of people, buildings, roads and other artificial infrastructure (Hahs and McDonnell 2006). Urbanisation has important consequences for biodiversity as it modifies components of the natural environment, including vegetation cover, atmospheric composition, local climate, hydrological systems, energy flow and nutrient cycles (Bridgman et al. 1995; Alberti et al. 2001). Australia is highly urbanised by world standards, with over 85% of its population living in urban areas (UN Department of Economic and Social Affairs 2008). The built environment covers about 2.4 million ha, or 0.3% of the continent's land surface, at the expense of native vegetation (Stewart et al. 2001).

As urbanisation is expected to increase in the coming decades (UN Department of Economic and Social Affairs 2008), the preservation of species within urban areas will play an important role in regional conservation. A wide range of fauna persists within urban habitats despite their high degree of modification, and populations of rare, threatened or geographically restricted species may rely on urban remnant vegetation for survival (Thompson and Fotso 2000; Hodgkison and Hero 2007; Nunes and Galetti 2007; Saunders and Heinsohn 2008). Cities are typically built along coastlines, which have supported highly productive habitats, and thus may contain remnant patches of highly diverse ecosystems (Melles et al. 2003). The promotion of biodiversity in cities, where people regularly encounter wildlife, also contributes to the education and the development of conservation ethics in the community, and leads to increased support for conservation efforts in general (Miller and Hobbs 2002; Lunney and Burgin 2004; Dunn et al. 2006).

In urbanised areas, birds may suffer negative impacts on fecundity and survivorship through habitat loss and

fragmentation, increased nest predation, brood parasitism, visitation disturbance, collisions, changes in food supply abundance, changes in predator assemblage and introduced competitors (Case 1996). Urbanisation often results in a decline in avian species richness and evenness, and an increase in bird density, although the patterns are inconsistent between study sites and regions (Marzluff 2001). Birds are highly sensitive to changes in vegetation cover, structure and composition, and individual species respond differently to the changes in resource distribution brought about by urbanisation (Chace and Walsh 2006). While many species are disadvantaged by urbanisation, others can successfully exploit the new habitat, leading to significant changes in avian community composition (Clergeau et al. 1998; Sewell and Catterall 1998; Blair 2001; Crooks et al. 2004; Van Heezik et al. 2008). In particular, a suite of exotic birds is able to thrive in human-modified landscapes as they are pre-adapted to live in open and disturbed habitats, and to take advantage of artificial structures (Case 1996), so that urban bird communities worldwide tend to become dominated by a small number of exotic species.

The Common Myna *Acridotheres tristis* (Passeriformes: Sturnidae) is native to southern Asia and was released multiple times in Australia at various locations along the eastern seaboard between 1862 and 1972 (summarised in Long 1981). The Common Myna is now common in Melbourne, the Sydney–Wollongong region, the Australian Capital Territory, in Queensland from Cairns to Townsville and from Toowoomba to Brisbane, and in many towns along coastal New South Wales (Martin 1996; Barrett 2003). Modelling of the distribution of the species using BIOCLIM indicates that the bird has the potential to further extend its range along the east coast of Australia (Martin 1996).

In the public arena, the Common Myna is widely considered to be a major pest because it is introduced, abundant, conspicuous in the places where people live, nests in the rooves of houses, forms noisy roosting aggregations and harbours mites (Thomas 2004). The species has also gained considerable attention from the media for its perceived detrimental impact on native wildlife (e.g. Perry 2008). Respondents to a nation-wide survey voted the Common Myna as 'Australia's Most Significant Pest/Problem' with 82% of the overall vote, beating contenders including the cane toad *Bufo marinus*, European rabbit *Oryctolagus cuniculus* and feral cat (Thompson et al. 2005). A number of local councils and interest groups have devoted considerable resources into the deployment of traps to reduce Common Myna populations (Thomas 2004). The Local Government Association of New South Wales has twice passed resolutions to petition the State Government to implement and fund a Common Myna control programme (Local Government Association of

NSW 2005, 2006). However, unlike for most major introduced pest species, a distinct lack of scientific research exists to quantify or confirm the Myna's actual impacts on native wildlife, and the bird has not been listed as a Key Threatening Process in State or Commonwealth legislation. The perceived pest status of many birds is based on unreliable information, and for some of these species, proper assessment of damage has eventually revealed that their actual impacts are slight (Bomford and Sinclair 2002). Thus, it is important to thoroughly evaluate the extent of the Common Myna problem before implementing control operations.

The potential for the Common Myna to have an impact on native birds arises via the mechanisms associated with competition: the negative effects that one organism has upon another by consuming, or controlling access to, a resource that is limited in availability (Keddy 2001). It is widely postulated that the Common Myna has an impact on the breeding success of other cavity-nesting birds through exploitation of natural tree hollows and aggressive interference during nest site acquisition (Thompson 2002). The Common Myna is also thought to use aggression to out-compete other birds at food resources and drive other birds out of habitats. These claims are built largely from a body of anecdotal evidence, and from only a few scientific sources in Australia. In Canberra, Pell and Tidemann (1997a, b) highlighted the potential for the Common Myna to reduce the breeding success of native parrots as it was the dominant user of available nest boxes and natural hollows in bushland reserves, and was the victor in most aggressive encounters at nest sites. Common Mynas also occupied a large proportion of artificial nest boxes monitored in Melbourne urban remnants (Harper et al. 2005). In contrast, Crisp and Lill (2006) documented foraging behaviour in Melbourne and concluded that the relatively low level of aggression initiated by the Common Myna meant it was not a major competitor for food resources. A Sydney study into the distribution of garden birds found that there were no negative correlations between the presence of Common Mynas and native birds (Parsons et al. 2006), but no other research into competition has been published from Sydney. The results of all these studies, even when combined, do not provide sufficient information to conclude that competition from the Common Myna has a significant impact on native fauna, and it certainly does not provide sufficient basis for implementing active management of the species, particularly given the different histories of urbanisation and different avian communities occurring in different regions of Australia.

The aim of this study was to investigate the importance of competition from the Common Myna to native avifauna and the corresponding role of urbanisation in these interactions. The study was conducted in a cluster of suburbs

within Sydney; the oldest, largest and most populous urban centre in Australia. The specific aims of the study were to: (1) describe the relationships between avian assemblage composition, including the Common Myna, and the degree of urbanisation; (2) compare the amount of background aggression initiated by the Common Myna to that of other species; (3) investigate the frequency with which native foraging competitors encounter the Common Myna and are victims of aggression; (4) document the availability and occupancy of natural tree hollows; and (5) determine nest site preferences of Common Mynas along a gradient of urbanisation.

Methods

Study sites

The study was conducted in an urban area of Sydney approximately 10 km southwest of Sydney airport (Fig. 1). The region has a cool temperate climate, with a mean annual rainfall of 1,214 mm, mean minimum temperature of 14°C and mean maximum temperature of 22°C. The native vegetation of the Sydney region is predominately comprised of eucalypt woodlands and open forest, much of which has been cleared since European settlement (Natural Heritage Trust 2001). The original vegetation is now highly fragmented within the urban matrix, and is confined to isolated pockets particularly along the borders of rivers and drainage systems (Benson and Howell 1990). Native trees and shrubs persist in residential streets and domestic gardens, but are scattered amongst lawn, exotic species, native hybrids and sealed surfaces. In the most highly developed parts of the city, native vegetation is almost entirely replaced by artificial structures.

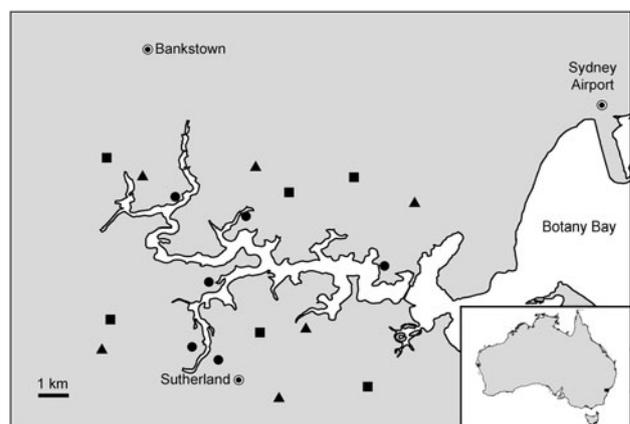


Fig. 1 The study sites in southern Sydney, Australia. *Filled circle* bushland, *filled triangle* residential, *filled square* commercial

Three habitat types along a gradient of urbanisation were selected: bushland remnants, residential streetscapes, and commercial areas, with six replicate 2-ha study sites of each habitat type interspersed throughout the study area (Fig. 1). The six ‘bushland’ sites, ranging in size from 10 to 40 ha, were selected first because they were the most constrained habitat type in the urban environment. In each bushland site, the 2-Ha study areas extended for 100 m along an edge of the remnant and extended 200 m toward the interior. ‘Commercial’ sites were selected next because they were the next most limited habitat in terms of availability. Each commercial site was selected so that it was within 5 km of the nearest bushland site and greater than 1 km from any other site. ‘Residential’ sites were the easiest to find and so were chosen last. Like commercial sites, they were selected so that they were within 5 km of the nearest bushland and commercial site, and greater than 1 km from any other site. Main roads or streets bordering less modified areas (e.g. parklands, reserves, riversides) were avoided, but otherwise they were selected randomly with regard to these constraints. All sites were selected remotely (using Google Earth) such that site selection was not influenced by the bird community, and sampling is therefore considered random. As access to private property was problematic, study areas in residential and commercial sites were restricted to roads, roadsides and front gardens with the width of a site measured across the road from rooftop to rooftop (using Google Earth). The length of each residential and commercial site was varied, depending on the width, to make up a total area of 2 ha.

Distribution of bird species within the urban matrix

To describe the bird community of southern Sydney and to identify variation along the urbanisation gradient, three 40-min point counts and two 20-min line transects were used to quantify the relative number of individual birds of each species present at each site. Point counts of such duration are a standard technique, suitable for conspicuous birds in large areas (Bibby 1992), but do not allow for the entire study area to be searched thoroughly, and so we complimented them with transect surveys. Transects are considered the most efficient and accurate of all general methods, and increase the chances of detecting less conspicuous species (Bibby 1992). Using both methods, all birds seen and heard within the 2-ha study sites were recorded, excluding those flying above the canopy (or rooftops). All data were collected during between October 2008 and February 2009, covering the main breeding season of most birds occurring in the Sydney region (Simpson and Day 2004).

Non-parametric multivariate analyses were conducted with PRIMER v.6 to determine whether the overall

assemblages of sites varied with habitat type. Species abundance data from the five surveys were pooled for each site and fourth-root transformed to reduce the weighting of rare and superabundant species. Bray–Curtis similarity measures were calculated to determine the similarity between sites based on species abundance (Clarke and Warwick 1994). The resulting similarity matrix was used in an Analysis of Similarities (ANOSIM) to test the null hypothesis that there were no differences in the bird communities of the three different habitats. An ordination plot was produced using non-metric Multi-Dimensional Scaling (nMDS) to visually represent the relative similarities of all sites.

Single factor analyses of variance (ANOVA) were used on parametric descriptors of the assemblage data with habitat as an independent factor with three levels. The dependent variables in these analyses were: total species richness, total abundance, native species richness, native species abundance, exotic species richness, exotic species abundance and Common Myna abundance.

Interspecific interference competition

Two surveys of 1 h duration each were undertaken at each 2-ha site during December 2008 and January 2009, one between 0700 and 1000 hours and the other between 1400 and 1700 hours, on separate days. In each survey, from a stationary point, all observed aggressive interactions were recorded, and the species of ‘initiator’ and ‘victim’ were identified. Three different behaviours were classed as acts of aggression: (1) chase: the initiator pursued the victim in flight; (2) supplant: the initiator flew or ran directly at the stationary victim, attempting to replace it at the resource it was using; and (3) harass: the initiator would flit, posture, and/or call in a manner that was clearly directed at the victim. The response of the victim to an act of aggression was categorised as one of three types: (1) retreat: the victim moved away from the aggressor; (2) hold ground: the victim did not move from its original position; and (3) fight back: the victim moved towards the initiator with a counterattack. There were three possible outcomes to aggression: (1) aggressor won: the initiator succeeded in driving the victim away from the resource; (2) indeterminate: the victim did not concede its position, and both birds remained at the resource; and (3) victim won: the victim succeeded in holding its place at the resource and the initiator gave up or was driven away. In addition, the number of birds of all species present was counted during these aggression surveys to determine the relative abundance of potential aggressors and victims.

To obtain a larger sample size of aggressive interactions than could be obtained by a single observer, a volunteer survey ‘Backyard Biffo’ was established through the

Birds in Backyards program (www.birdsinbackyards.net/surveys/backyard-biffo.cfm). Volunteer observers collected data in the same format as above, and data were analysed for all observations collected within the suburbs of Sydney. While quality control could not be guaranteed, the larger sample size provided useful comparative data.

For each data set, Chi-square analysis was used to determine whether there were differences between species in terms of the number of interspecific aggressive interactions they initiated. If all birds are equal competitors, it would be expected that the number of interactions initiated by a species would be proportional to its relative abundance. Thus, the expected values for the Chi-square analysis were determined by dividing the species abundance by the total abundance of birds multiplied by the total number of interspecific interactions. The expected and observed values of aggression for individual species were tested in a separate 2×2 goodness-of-fit analysis against the pooled values for all other species. All species present for any part of the aggression surveys were included in the analysis, even if they were not involved in interactions.

For the main data set (i.e. not including volunteer data), one-way ANOVAs were used to determine whether there was an effect of habitat type on the total number of aggressive interactions and on the number of interspecific aggressive interactions that took place.

Aggression between ground-foragers

Common Mynas are omnivorous ground-foragers (Sengupta 1976; Crisp and Lill 2006; Newey 2007) and the two native bird species observed in this study that would be most likely to compete for foraging resources are the insectivorous, ground-feeding Willie Wagtail *Rhipidura leucophrys* and Magpie-lark *Grallina cyanoleuca* (Cameron 1985; Pizzey and Knight 2007). To investigate interspecific competition amongst these three species, focal Magpie-larks and Willie Wagtails were observed while they foraged on the ground to document how often they encountered Common Mynas and the frequency of competitive interactions. Surveys were not restricted to the formal study sites and included 22 sites, within the broad study area, where the focal species were observed foraging. Fourteen sites were parks or sporting grounds where the ground cover was predominately grass, and seven were along roadsides in residential streets. Thirty-four focal Magpie-larks were observed at 20 sites (12 parks, 8 streets) and 10 focal Willie Wagtails were observed at five sites (all parks). Magpie-larks and Willie Wagtails foraged together at three of these sites. If conspecifics were foraging in a group, one focal bird was selected at random for observation until that bird stopped foraging, at which point observations continued on a different individual. Birds were included in the analysis only if they could be followed for a

minimum of 1 min, with a mean observation period of 8.5 min per bird. No more than three birds of each species were surveyed at any one site. Surveys were conducted on clear mornings during February 2009.

Availability and occupancy of natural tree hollows

The availability and occupancy of tree hollows at each site were surveyed in October 2008, at the peak of the breeding season of most hollow-nesting species (Simpson and Day 2004). Each tree in each study site was examined from the ground to determine the number of hollows that were present. Hollows were counted if they were estimated to be of suitable size for the smallest (Rainbow Lorikeet *Trichoglossus haematodus*) to the largest (Sulphur-crested Cockatoo *Cacatua galerita*) native hollow-nesting bird species known to occupy the region; thus, entrance diameter ranged approximately from 8 to 30 cm. A hollow was deemed to be occupied if a bird was seen to enter or leave the cavity, or was perched above or beside the hollow. When a hollow was detected, it was observed for at least 5 min, and longer if there was bird activity in or around the tree. A second survey was conducted during December 2008 to gain additional occupancy data that may have been missed the first time.

Single-factor ANOVAs were used to determine whether the number of hollow-bearing trees and the total number of hollows present at a site was associated with habitat type. Chi-square analysis was used to determine whether the Common Myna occupied a greater number of tree hollows than expected compared to other hollow-nesting species. All species were expected to use an equal proportion of the occupied hollows. The expected and observed values of occupancy for individual species were tested in a separate 2×2 goodness-of-fit analysis against the pooled values for all other species.

Common Myna nest site selection

To determine habitat preferences for nesting Common Mynas, their nest cavity selection was surveyed in transects along the interface between different habitat types. Twelve 1-km transect lines were surveyed: six traversing the commercial-residential boundary and six traversing the residential-bushland boundary. (Ideally, this would be completed with a third transect type across the bushland-commercial interface, but in the study area these two habitats were not adjacent.) Transects in residential and commercial sites followed the road, and those through bushland ran parallel to the remnant edge. The bushland ($n = 6$) and commercial ($n = 6$) sites were the same as those described above, except for three that were substituted because Common Mynas were absent from one

commercial site and from the residential streets surrounding two bushland sites.

Every Common Myna encountered along a transect line was observed until it flew to a nest site, or until it was not possible to follow it further. Nest sites were categorised as: roof or gutter, hole in built structure, tree hollow, palm tree, hedge, or nest box. Transect lines were walked from one end to the other, and if deviated from in pursuit of a Common Myna, the transect was resumed at the same point once the pursuit was terminated. Transects were each surveyed twice during November and December 2008, once in the morning (0700–1000 hours) and once in the afternoon (1400–1700 hours). Single-factor ANOVA was used to determine whether there were differences in the number of confirmed Common Myna nests between commercial, residential and bushland habitats.

Results

Distribution of bird species within the urban matrix

The bird community differed significantly between habitat types (Global $R = 0.80$, $P = 0.001$), and the low stress level (stress = 0.08) indicates that the data were well represented by the two-dimensional plot (Fig. 2). Residential sites had the highest within-habitat similarity (average 72%), followed by bushland sites (68%) and commercial sites (60%). Bushland and commercial sites had the most different bird communities with an average dissimilarity of 70%. Bushland and residential sites had an average dissimilarity of 46%. Residential and commercial sites had an average dissimilarity of 49%.

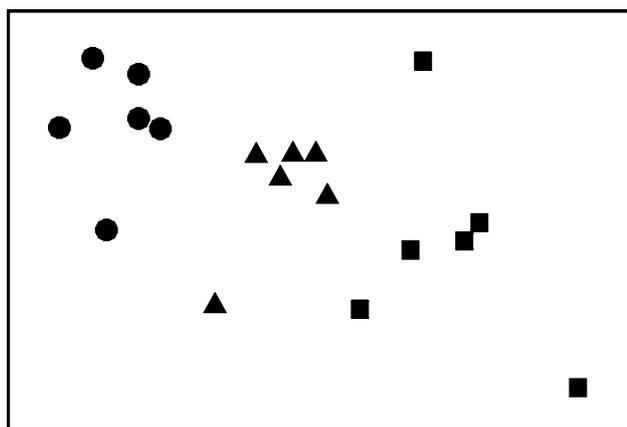


Fig. 2 Ordination of sites based on bird species abundance analysed by non-metric multidimensional scaling using Bray–Curtis similarities between sites. Filled circle bushland, filled triangle residential, filled square commercial

The total abundance of birds was not affected by habitat type ($F_{2,15} = 0.24$, $P > 0.05$), but total species richness decreased significantly ($F_{2,15} = 18.93$, $P < 0.01$) from bushland (mean = 19.7 ± 1.4 SE) to residential (16.5 ± 1.0 SE) to commercial (9.7 ± 1.1 SE). Native species richness declined significantly ($F_{2,15} = 29.13$, $P < 0.01$) with urbanisation, from bushland, through residential, to commercial sites (Fig. 3a). Mean exotic species richness increased significantly from bushland, through residential to commercial sites ($F_{2,15} = 32.00$, $P < 0.01$; Fig. 3a). The abundance of native species also decreased significantly ($F_{2,15} = 13.77$, $P < 0.01$) from bushland, through residential, to commercial sites, while the abundance of exotic species increased significantly from the bushland, through residential to commercial sites ($F_{2,15} = 4.03$, $P < 0.05$; Fig. 3b). The abundance of Common Mynas was significantly associated with habitat type ($F_{2,15} = 11.60$, $P < 0.01$), falling from a mean of $6.7 (\pm 7.9$ SE) in commercial, to $4.9 (\pm 3.9$ SE) in residential, to 0.0 in bushland sites.

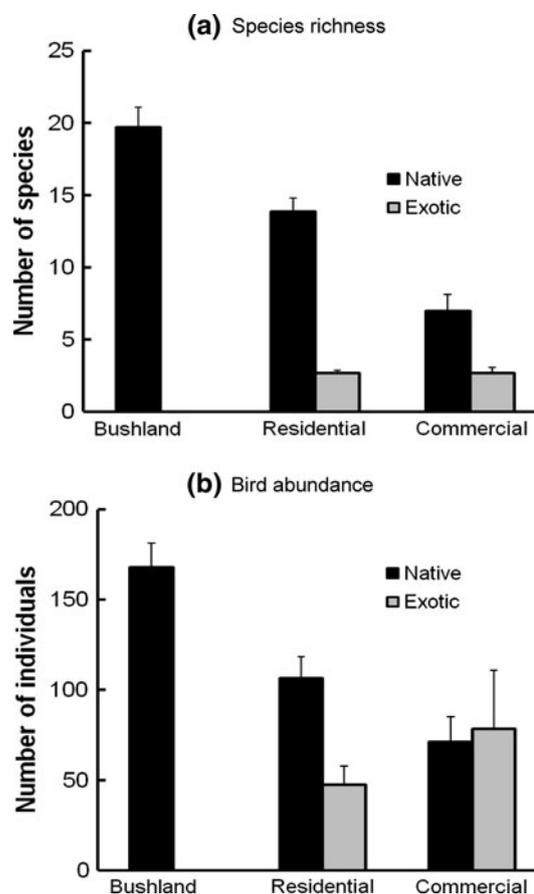


Fig. 3 **a** The mean species richness, and **b** mean abundance, of native and exotic species in each habitat type. Black bars native species, grey bars exotic species. Error bars SEs

Of the 36 native species detected in bushland remnants, 10 were also found in residential areas, a further 11 were found in all habitats, while 15 were present only in bushland sites. Another 12 species were absent from bushland but present in residential and/or commercial sites, of which six species were non-native. In bushland sites the five most abundant species accounted for 63% of the total abundance and all were native species (Rainbow Lorikeet, Noisy Miner *Manorina melanocephala*, Sulphur-crested Cockatoo, Laughing Kookaburra *Dacelo novaeguineae* and Pied Currawong *Strepera graculina*). In residential sites, the five most abundant species also accounted for 63% of the total abundance, but included three native species (Rainbow Lorikeet, Noisy Miner and Red Wattlebird *Anthochaera carnunculata*) and two non-native species (Common Myna and Spotted Dove *Streptopelia chinensis*). The five most abundant species in commercial sites made up 85% of the total abundance with the most abundant species being non-native (Common Myna and Rock Dove *Columba livia*) along with three native species (Welcome Swallow *Hirundo neoxena*, Noisy Miner and Rainbow Lorikeet).

Interspecific interference competition

A total of 39 acts of aggression was observed during 36 surveys and there was no significant effect of habitat type on the total number of interactions ($F_{2,15} = 1.57, P > 0.05$). An average of $0.83 (\pm 0.28 \text{ SE})$ aggressive interactions per hour occurred in bushland remnants, $1.50 (\pm 0.39 \text{ SE})$ interactions per hour in residential streetscapes, and $0.92 (\pm 0.15 \text{ SE})$ interactions per hour in commercial areas. Twenty-seven of the 39 interactions were interspecific with an average number of $0.42 (\pm 0.24 \text{ SE})$ interspecific interactions per hour in bushland remnants, $1.17 (\pm 0.31 \text{ SE})$ in residential streetscapes and $0.67 (\pm 0.17 \text{ SE})$ in commercial areas. The effect of habitat type on the number of interspecific interactions was also non-significant ($F_{2,15} = 2.44, P > 0.05$).

The most frequent attack type used by all aggressors was chase ($n = 20$) followed by supplant ($n = 16$) and harass ($n = 3$). The most frequent outcome of aggression was that the victim retreated and the initiator won control of the resource, which occurred on 29 occasions. The victim held its ground and the outcome was indeterminate on seven occasions; the victim fought back on three occasions, winning one, with the other two indeterminate.

Nine species were responsible for initiating interspecific aggression, the most frequent being the Noisy Miner and Red Wattlebird, followed by the Sulphur-crested Cockatoo and Common Myna (Fig. 4). The remaining five species that initiated acts of aggression (Black-faced Cuckoo-shrike *Coracina novaehollandiae*, Australian Magpie *Gymnorhina tibicen*, Rainbow Lorikeet, Rock Dove and Willie Wagtail) initiated just one interaction each.

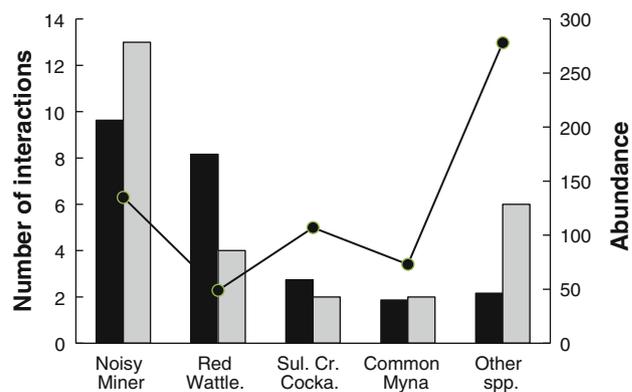


Fig. 4 The number of interspecific interactions initiated by each species of aggressor (grey bars) and numerical abundance of each species (black line) observed during focal site surveys. The black bars show relative levels of interspecific aggression after adjusting to account for abundance

The Common Myna initiated two interspecific acts of aggression, both of which resulted in an indeterminate outcome. In one case, a Common Myna lunged at a Sulphur-crested Cockatoo that incidentally flew close by its nest site. The other was an unsuccessful attempt to supplant a Rock Dove from a ground-based food resource. Two intraspecific interactions between Mynas also occurred over food on the ground.

The observed level of aggression from the Common Myna was not significantly different from the average level of aggression of all other species combined ($\chi^2 = 0.33, df = 1, P > 0.05$). The Rainbow Lorikeet ($\chi^2 = 3.58, df = 1, P > 0.05$) and the Sulphur-crested Cockatoo ($\chi^2 = 0.00, df = 1, P > 0.05$) also showed non-significant amounts of aggression. The observed number of aggressive acts by the Noisy Miner ($\chi^2 = 27.08, df = 1, P < 0.01$) and the Red Wattlebird ($\chi^2 = 5.53, df = 1, P < 0.05$) were significantly greater than expected.

Of the 16 species reported by Bird in Backyards surveys with greater than 2% relative abundance, three showed significantly greater levels of aggression than expected: the Noisy Miner ($\chi^2 = 101.96, df = 1, P < 0.01$), the Australian Magpie ($\chi^2 = 4.25, df = 1, P < 0.05$) and the Pied Currawong *Strepera graculina* ($\chi^2 = 8.44, df = 1, P < 0.01$). The amount of aggression observed from the Common Myna ($\chi^2 = 2.95, df = 1, P > 0.05$) and the Red Wattlebird ($\chi^2 = 0.12, df = 1, P > 0.05$) was not significantly different from that expected on the basis of species occurrence.

Aggression between ground-foragers

Common Mynas were present at 50% of sites where Magpie-larks were observed foraging and at 20% of sites with Willie Wagtails. Focal Magpie-larks spent 8% of their

feeding time with Common Mynas foraging within 10 m, and Willie Wagtails spent 17% of their time with Common Mynas foraging within 10 m. In total, focal birds were involved in 27 aggressive interactions during foraging; in 21 cases, a Magpie-lark was the victim, and six times, a Willie Wagtail was the victim. The Magpie-lark was the subject of aggression from other Magpie-larks, the Australian Magpie, the Noisy Miner and Willie Wagtail, but not from the Common Myna (Fig. 5). The Willie Wagtail was involved in five acts of intraspecific aggression, and was the subject of aggression from the Common Myna on one occasion (Fig. 5).

Availability and occupancy of natural tree hollows

The number of hollow-bearing trees and the number of tree hollows were significantly different between habitat types ($F_{2,15} = 47.95$, $P < 0.01$; $F_{2,15} = 39.22$, $P < 0.01$). Bushland sites contained a mean of 6.2 (± 0.9 SE) hollow-bearing trees per hectare with a mean of 9.6 (± 1.5 SE) hollows per hectare. A mean of 0.2 (± 0.1 SE) hollow-bearing trees per hectare and 0.2 (± 0.1 SE) hollows per hectare were present in residential areas, and zero hollow-bearing trees were identified in commercial sites.

Of the total of 117 hollows identified, 44 were confirmed as occupied, with the Rainbow Lorikeet, Sulphur-crested Cockatoo and feral honeybee (*Apis mellifera*) being the most common occupants and only low rates of occupancy by the Crimson Rosella *Platycercus elegans*, Dollarbird *Eurystomus orientalis* and Common Myna (Fig. 6). The hollow occupied by the Common Myna was one of only two hollows identified in the residential sites; all other species used hollows in bushland sites. Hollow occupancy by the Common Myna was significantly lower than the average occupancy rate of all hollow-nesting species ($\chi^2 = 40.1$, $df = 1$, $P < 0.01$).

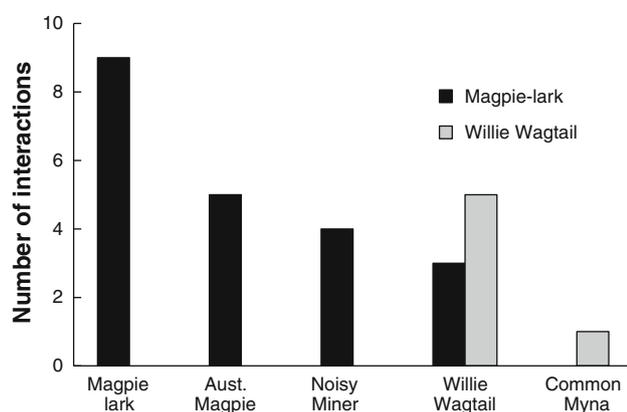


Fig. 5 The total of number of attacks by each species of aggressor on focal Magpie-larks *Grallina cyanoleuca* (black bars) and focal Willie Wagtails *Rhipidura leucophrys* (grey bars)

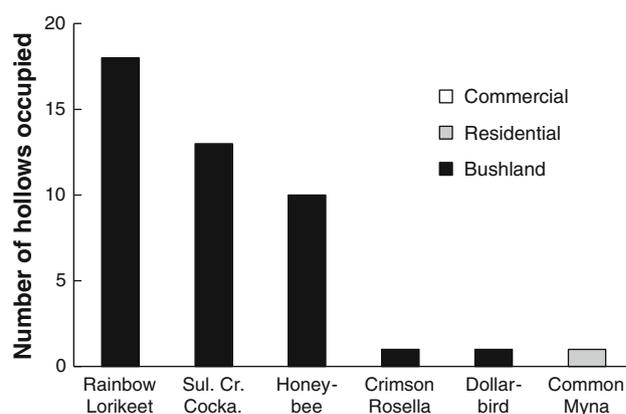


Fig. 6 The total number of tree hollows occupied by each species in bushland (black bars), residential (grey bars) and commercial (white bars) habitats. No occupancy of tree hollows was observed in commercial areas

Common Myna nest site selection

A total of 36 Common Myna nests was found: 20 in commercial sites, 13 in residential sites and three in bushland sites. There was a significantly greater number of Common Myna nests in commercial areas than in residential sites at the commercial–residential interface ($F_{1,10} = 14.55$, $P < 0.01$), and at the residential–bushland interface there were more nests in residential areas than bushland sites although the difference was not significant (Fig. 7). Of the 36 nests located, 29 were constructed in human buildings in rooves, gutters, and in cavities where the built structure had been in some way damaged, such as holes in fibro walls, broken hollow shop signs and air vents. The remaining 7 nests were associated with vegetation including tree hollows, in palm trees, a hedge, and an artificial nest box on a tree (Fig. 8).

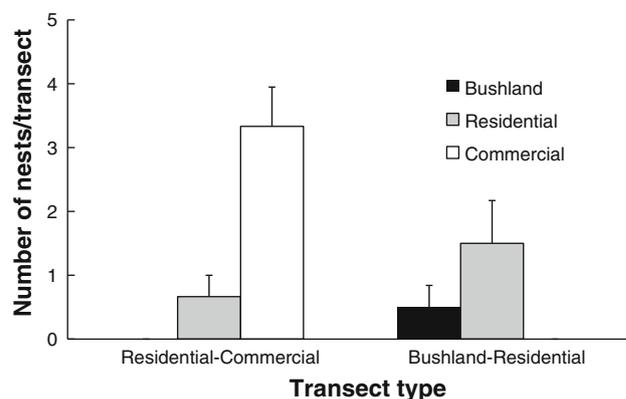


Fig. 7 The mean (+SE) number of Common Myna *Acridotheres tristis* nests detected in bushland (black bars), residential (grey bars) and commercial (white bars) habitats during transect surveys along different habitat interfaces

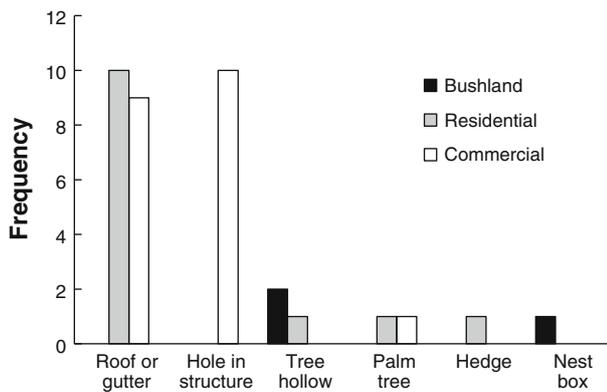


Fig. 8 The frequency of each cavity type used by Common Mynas in bushland (*black bars*), residential (*grey bars*) and commercial (*white bars*) habitats

Discussion

In this study, the abundance of the Common Myna increased with the degree of habitat modification along the urbanisation gradient, and its presence was thus negatively associated with both the diversity and abundance of native bird species. However, there is unlikely to be a causative relationship given that the Common Myna was no more aggressive than other species, rarely interacted at foraging sites, preferred to nest in more highly modified habitats and used significantly fewer tree hollows than native species.

Total species richness and native species richness declined along the urbanisation gradient from bushland through residential to commercial areas, and the proportion of exotic species in the assemblage greatly increased. Native species richness was higher than exotic species richness in all three habitats, but in the commercial areas, the total abundance of exotic birds exceeded that of native birds. Commercial areas, however, were still able to support the same number of individuals as natural bushland vegetation even though total species richness declined. Thus, only a small number of species were able to exploit the most urbanised environment, where they can become highly abundant. This phenomenon of increasing proportions of exotic species richness and abundance in the assemblage has similarly been noted in other studies (Green 1984; Blair 1996; Wood 1996; Sewell and Catterall 1998; Parsons et al. 2003; Crooks et al. 2004; Van Heezik et al. 2008). Although the five most abundant species in commercial areas accounted for a greater proportion of individuals than in residential streetscapes and remnant bushland, the average within-habitat similarity of commercial sites was relatively low. This suggests that the homogenisation effect was not as strong as detected in northern hemisphere studies (Blair 2001; Crooks et al. 2004; Clergeau et al. 2006) perhaps due to the more recent history and lower intensity of urbanisation in Australia, or

differences in native species traits. Importantly, residential sites were used by cavity-nesting parrots (Crimson Rosella, Eastern Rosella *Platycercus eximius*, Sulphur-crested Cockatoo and Rainbow Lorikeet) that would potentially be susceptible to competition for nest sites, as well as by native ground-foragers (Magpie-lark and Willie Wagtail) that use similar food resources to the Common Myna.

The abundance of the Common Myna was positively associated with the increased level of habitat modification. It was common at every residential site and all except one commercial site, but not seen at all in bushland remnants. This is consistent with other studies in Australia where Common Mynas are abundant in highly urbanised landscapes (Wood 1996; Pell and Tidemann 1997a; Parsons et al. 2003; White et al. 2005; Crisp and Lill 2006); however, in these studies, they have usually also been present in remnant vegetation surrounded by an urban matrix (Pell and Tidemann 1997a; White et al. 2005; Crisp and Lill 2006). The reserves where Pell and Tidemann (1997a) conducted their research were mainly savannah type habitat with a very sparse cover of eucalypts compared with Sydney bushland, and more similar to the open woodlands of the Common Myna's natural habitat. The conflicting results between Melbourne studies (White et al. 2005; Crisp and Lill 2006) and those in Wollongong (Wood 1996) and Sydney (Parsons et al. 2003; this study) indicate that regional vegetation characteristics are likely to influence habitation by Common Mynas.

Contrary to popular preconceptions about its behaviour, the Common Myna was found to be no more aggressive than expected given its high abundance, and these findings were corroborated by the similar non-significant results of community backyard surveys. Pell and Tidemann (1997b) observed that the Common Myna won 82% of interspecific aggressive encounters, but did not identify how many of those were initiated by the Common Myna. The rate of aggression initiated by the Common Myna here (0.06/h) was much less than the 0.4/h observed by Crisp and Lill (2006), but again it is unknown against which species the Common Myna initiated its attacks. However, these other studies were concentrated around particular activities, namely nesting and feeding sites, rather than the use of general shared space. This indicates that Common Mynas may only aggressively defend the particular resources they are using, rather than actively excluding other birds from the use of resources within a wider area or territory.

The Red Wattlebird and Noisy Miner initiated more interactions than the Common Myna and showed significantly higher levels of aggression than any other species. The community backyard surveys strongly supported this significant result for the Noisy Miner. The Noisy Miner aggressively defends its territory from intruders and there is now a substantial body of research from different

geographical areas in Australia demonstrating that Noisy Miners act as a 'reverse keystone species' structuring bird communities in the human-modified environments in which they are most common (Grey et al. 1997; Major et al. 2001; MacDonald and Kirkpatrick 2003; Piper and Catterall 2003; Clarke and Oldland 2007; Maron and Kennedy 2007). In Sydney suburban gardens, Parsons et al. (2006) found negative correlations between the presence of the Noisy Miner and that of several small bird species in Sydney gardens, while none were negatively correlated with the presence of the Common Myna. Thus, the perception in the public arena that the Common Myna is a highly aggressive species may be misplaced and attributable to identity confusion with the Noisy Miner.

Native ground-foraging birds seem to experience little competitive interference from Common Mynas over food resources. The Magpie-lark was never the subject of aggression from the Common Myna despite the latter being present at half its feeding grounds in flocks of up to 20 birds. In comparison, Willie Wagtails were responsible for 14% of the attacks on Magpie-larks, though they were only present at three sites and at a maximum abundance of five birds. The opportunity for attack did exist, as on multiple occasions Common Mynas and Magpie-larks foraged within 10 m, and sometimes within 1 m of one another. Willie Wagtails spent a greater proportion of time in the presence of Common Mynas than the Magpie-lark, but were the subject of just one act of aggression from a Common Myna, and in this case both birds remained foraging at the patch. However, sample size for the Willie Wagtail observations was much smaller, and Common Mynas were only present at one out of five sites where the Willie Wagtail was observed foraging. Intraspecific aggression was more important than interspecific interactions for both the Magpie-lark and Willie Wagtail. The simplest explanation for the lack of aggression from Common Mynas is that there was no absolute shortage of food resources for ground-feeding insectivores or that the resource is not economically defensible (Weir and Grant 2004).

Bushland remnants supported a density of hollow-bearing trees (mean = 6.2 hollow-bearing trees per ha) similar to that found in small remnants in urban Melbourne (5.8) (Harper et al. 2005) and reserves in Canberra (4.1) (Pell and Tidemann 1997a). Half the sites had high abundances of hollow-bearing trees, comparable to those found in natural unlogged eucalypt forest and woodland, which can range from 6.2 to 22.8 per hectare (Saunders et al. 1982; Kavanagh and Stanton 1998; Gibbons and Lindenmayer 2002). Very few hollow-bearing trees existed in residential streetscapes, and none in commercial areas, which is to be expected given the lack of suitably aged trees that remain in these habitats. Thus, the potential to

support large populations of cavity-nesting birds is diminished in the most urbanised habitats.

Common Mynas were not seen to use any tree hollows in bushland remnants in the hollow survey, and used only one hollow in a residential site. It seems therefore that, based on these results, exploitation of hollows by the Common Myna is unlikely to affect the breeding success of native cavity-nesters in urban Sydney. This is contrary to the findings of Pell and Tidemann (1997a), who reported that 12% of available hollows in Canberra savannah and woodland reserves were claimed by Common Mynas. Competition from introduced honeybees, which occupied 9% of all available hollows, may potentially pose a greater threat to native birds in Sydney than that presented by the Common Myna (see also Harper et al. 2005).

The Common Myna seems to prefer nest sites in the most urbanised habitats rather than in more natural sites. When birds had a choice between habitat types, they nested in commercial areas rather than residential streetscapes, and in residential streetscapes rather than remnant bushland. Twice as many nests were found in the transects through more urbanised areas (commercial sites and adjacent residential streetscapes) than in less developed areas (bushland remnants and adjacent residential streetscapes). Contrary to this, Pell and Tidemann (1997a) concluded from year-round population estimates that Common Mynas resided in residential areas during winter, but moved into remnant bushland to nest during the breeding season. They did not, however, investigate the use of artificial structures for nesting. The majority of Common Myna nests in this study were created in artificial structures as compared with natural vegetation, with only two nests found in bushland hollows. This finding is supported by that of Councilman (1974) who studied breeding pairs throughout urban habitat in Auckland, New Zealand and determined that Common Mynas nesting in the city avoided native bushland. The pairs nesting in bushland tree hollows in this study were not seen to go very far into the remnants and used the adjacent residential area to perch and forage when not at the nest. Access by Common Mynas to deeper bushland did not appear to be impeded by native birds, as no interactions, displays or calling took place. Thus, the Common Myna seems to actively select urbanised areas and artificial cavities for breeding, possibly because they offer greater nesting success (Dhanda and Dhindsa 1996), and are in close proximity to ample human food resources.

Conclusion

Humans have introduced numerous animal species around the world to environments outside their native ranges (New 2006). The majority of introductions of vertebrate species

to Australia occurred in earlier periods of European settlement, before the potential repercussions for native ecosystems were appreciated, and the subsequent realisation of the negative impacts of some introduced species has led to their widespread persecution in the general community.

Extreme prejudice and vigilante action has been levelled toward the Common Myna in Australia, based on an unconvincing body of evidence. Businesses and dedicated action groups sell Common Myna traps and trap plans to enthusiastic members of the public for use in their backyards. Governments and local councils have jumped on the bandwagon by throwing financial support behind the development of specialist trapping devices to capture large numbers of Common Mynas for extermination (Gallasch 2010).

The results presented in this study suggest that, in southern Sydney at least, the Common Myna does not have a significant competitive impact on native bird species. The high level of interspecific aggression that is often attributed to the Common Myna was not evident during observations, though the reputed behavioural dominance of the Noisy Miner was clearly detected. A case of mistaken identity with the Noisy Miner, with similarities in appearance and name, is likely to have contributed to the negative perceptions of the Common Myna.

Contrary to the findings of other studies, there is no evidence here that Common Mynas exploit tree hollows at the expense of native cavity-nesting birds. The availability of tree hollows was low in urbanised sites, and Common Mynas were generally absent from remnant bushland habitat where the hollow-bearing trees were abundant for native species breeding. Common Mynas primarily nested in buildings and other artificial structures, and only occasionally nested in tree hollows or other vegetation. It seems that habitat and nest-site partitioning alleviates the effects of resource competition, as also suggested by Pell and Tidemann (1997b) where Common Mynas in Canberra mainly used the edge and savannah areas of reserves while native species made more use out of interior and woodland sections. In Sydney, the high abundance and visibility of Common Mynas in the residential and commercial habitats, where people spend the majority of their time and have most of their wildlife experiences, has probably contributed to the view that Common Myna populations have ‘taken over’ the habitat of native species, when in reality they tend to avoid the more natural habitats where a greater number of native species are found.

This study cannot be used to completely discount the existence of competition between the Common Myna and native birds. The lack of corroboration provided by these findings, with respect to previous studies, and derived from an area where the Common Myna has not been studied before, highlights the fact that site-specific variables may

influence the degree to which they negatively interact with other species. The most effective way to reveal the impact of competition interactions is to conduct controlled Common Myna removal experiments and monitor the response of native species.

In New South Wales, current management decisions regarding the Common Myna, with some local councils individually spending money on trapping and reduction programs (Indian Myna Bird Project 2007), are based on a small amount of research from other states. However, as Soule (1990) points out, ‘a policy of blanket opposition to exotics will become more expensive, more irrational, and finally counterproductive as the trickle becomes a flood’. If the policy concern is about a decline in native bird diversity in cities, efforts should focus on native species with low occupancy rates to determine what limits their abundance. If the concern is the spread of an exotic species that may be in a sleeper phase, efforts should focus on the fringe of the species range to prevent establishment beyond its current distribution. Applying resources, in an undirected fashion, to the places where the species is having minimal impact, and where its high abundance may simply reflect its superior ability to exploit urbanised habitats, addresses neither policy concern.

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