LEGISLATIVE ASSEMBLY
FOR THE AUSTRALIAN CAPITAL TERRITORY

SELECT COMMITTEE ON ESTIMATES 2023-2024
Mr Mark Parton MLA (Chair), Ms Jo Clay MLA (Deputy Chair),
Mr Michael Pettersson MLA

ANSWER TO QUESTION TAKEN ON NOTICE DURING PUBLIC HEARINGS

Asked by Dr Marisa Paterson on 20 July 2023: Dr Rosie Cooney took on notice the following question(s):

[Ref: Uncorrected Proof Hansard Transcript 20 July 2023 p 56]

In relation to: Myna Birds in the ACT

DR PATERSON: Do you feel that there needs to be education if the research is so suggestive that these birds do not actually have an impact of native species—and that is a big turnaround from what the minister said two years ago to now. Do you think there needs to be some community education about the impact of these birds?

Mr Burkevics: Quite possibly, and I think we have already engaged with CMAG to explain the evidence that we have available to us and encourage a number of actions. I think we do note that there is a petition before the Assembly at the moment on that issue and in time I am sure the minister will have some remarks to say. But potentially I think that at the moment we are certainly not seeing any need for specific community action on this issue beyond what is already occurring.

DR PATERSON: Is it possible to get references to the research that is provided?

Dr Cooney: Yes, sure.

DR PATERSON: Yes, thank you.

Ms Vassarotti: Yes, so that is something that we can take on notice and provide you with some further information about the research base.

Rebecca Vassarotti MLA: The answer to the Member's question is as follows: –

Key references on Common Myna interactions with native bird species are briefly summarised below and included as attachments (Appendix A-E).

- Between 1998 and 2019, long term monitoring by Canberra Birds (previously Canberra Ornithologists Group) found that Common Mynas have been declining in ACT woodlands over recent decades (Appendix A).
- Studies have found that Common Mynas have little impact on resource use by native bird species in urban environments (Lowe et al 2011, Appendix B; Haythorpe 2013, Appendix C).
- Common Mynas are not considered a major competitor with two threatened species of parrot in the ACT (Superb Parrots and Gang Gang Cockatoos). ACT Government monitoring

has found that Superb Parrots and Gang Gangs compete for hollows more with native species than with Common Mynas (ACT Government, unpublished data).

- Only one study has found a negative association between Common Mynas and native parrot species. Grarock et al. (2013) found a negative relationship between Common Myna nest box occupancy and the abundance of the Crimson and Eastern Rosellas at some sites (Appendix D).
- An ACT-based study that examined the impacts of community-led culling on Common Myna abundance found that while effective at fine scales of <1 km^2, too few animals were being removed to achieve broadscale control (Grarock et al. 2013; Appendix E).

Approved for circulation to the Select Committee on Estimates 2023-2024

Signature: Date: 28/7/2023

By the Minister for the Environment, Rebecca Vassarotti MLA

LONG-TERM TRENDS IN ACT WOODLAND BIRDS 1998–2019

Prepared for Canberra Ornithologists Group (COG) by:

Jenny Bounds, Chris Davey and Nicki Taws
COG Woodland Bird Analysis Project Team

Dr Laura Rayner
Environment, Planning and Sustainable Development Directorate, ACT Government

Dr Maldwyn John Evans
Fenner School of Environment and Society, Australian National University

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Acknowledgments

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The authors thank those who reviewed the report, in particular Thea O'Loughlin (Ecologist, ACT Parks & Conservation Service).

It is acknowledged that bird survey data from the COG Woodland Bird Monitoring Project were collected on lands of traditional custodians: the Ngunnawal people.

Cover photo: Rufous Whistler (male)
Bird photos: Geoffrey Dabb
Habitat photos: Jenny Bounds

CONTENTS

Abstract

A major analysis of twenty-one years of systematic bird monitoring data was undertaken by the Canberra Ornithologists Group (COG) in collaboration with the ACT Government and the Australian National University (ANU). Guided by the Project Team and Dr Rayner this analysis looked at long-term trends in: (i) species richness, (ii) individual land-based birds, and (iii) 12 bird groups of interest across the ACT. This report outlines significant trends and key results for the ACT woodland bird community; a list of Conservation Priority Species has been compiled and presented.

From 1998 to 2019, survey data were collected at 142 woodland sites in 15 woodland locations across the ACT. Data for 168 individual bird species were interrogated for analysis, and long-term modelling was possible for 164 species. This included land-based birds (129 species) and water-based birds (39 species), however, this report focusses on land-based birds only.

The analysis revealed no significant change in species richness through time, or across woodland bird monitoring locations. Of the 129 land-based species analysed, 56 species had statistically significant trends in reporting rate; 32 species showed long-term decline and 24 species showed long-term increase. There are some unexpected results, such as declining trends for some species that are regarded as common.

Of the 12 bird groups analysed, 7 groups showed statistically significant changes through time. There was a significant increase in the reporting rate of large-bodied birds, and birds associated with degraded woodland communities. There was a significant decrease in the reporting rate of small-bodied birds, canopy feeders, ACT woodland-dependent birds, and ACT species of conservation concern.

Although no statistical analysis was performed on the relationship between various population indices and weather, an inspection of annual weather records suggests a role of climate and weather in shaping ACT woodland bird population reduction and growth. The woodland analysis includes survey data from periods of drought (1998 to 2009) and significant rainfall (2010 to 2012), which may have been the drivers of short-term change in reporting rates for some species. Weather-related responses of individual species were not consistent.

This report validates widespread concern for ACT woodland-dependent birds as a vulnerable community group. It raises conservation concern for small-bodied birds, particularly insectivores and canopy feeders. It is argued that declining common birds (both migratory and resident) require ongoing assessment in terms of monitoring their population status and their habitat extent and condition. Increasing numbers of large-bodied birds, and species associated with degraded woodland communities, have implications for urban planning into greenfield sites and buffer areas.

Lastly, this report serves as a support tool for land managers through the identification of Conservation Priority Species that arguably require immediate protection and conservation attention. In some cases, strategic restoration planning to improve woodland condition and arrest current rates of decline may be necessary to avoid future local extirpation events.

1. Background

1.1. The Woodland Bird Monitoring Project

The primary aim of the Woodland Bird Monitoring Project was to develop a statistically robust monitoring program to measure the abundance of birds through time at grassy woodland locations in the ACT (see Cunningham 2003 and Bounds *et al.* 2010). A priority feature of the sampling design was to establish survey sites on-reserve and off-reserve to assess the effectiveness of the ACT reserve network for conserving woodland birds (see Rayner *et al.* 2014).

COG has implemented the Woodland Bird Monitoring Project since 1995, commencing at Mulligans Flat Nature Reserve (then leasehold land). Locations have been added to the Project since its inception, including 5 locations in 1998, and 9 locations between 2000 and 2005. In total, there are 142 monitoring sites, nested within 15 broad ACT locations (Appendix A), all occurring in Yellow Box-Blakely's Red Gum grassy woodland — a critically endangered ecological community. All survey locations are in peri-urban Canberra and include ACT Nature Reserves, rural leasehold properties, and quasi-conservation areas not in the ACT reserve system (Commonwealth Department of Defence land). Four locations have changed tenure during the survey period, from leasehold to Nature Reserve.

A core purpose of the Woodland Bird Monitoring Project is to provide recommendations for land management based on monitoring results. Survey data from the Woodland Bird Monitoring Project have been analysed on several occasions previously, examining trends in reporting rate for a range of bird species. Previous reports published on the COG website, including Bounds *et al.* 2010 and Taws *et al.* 2012, provide additional background information and site details. A Ten-year Analysis of bird survey data (1998 to 2008) was undertaken in 2009 (Bounds *et al.* 2010) and multiple project reports have been published in COG's journal *Canberra Bird Notes*, available on the COG website.

This report presents major results from the analysis of survey data collected over the period March 1998 to September 2019; a period of twenty-one years and six months. The intention was to undertake an analysis once the majority of survey locations had at least 20 years of monitoring data.

Goorooyarroo Nature Reserve - view to Gungahlin suburbs

1.2. Scope of the Analysis

Questions for the analysis were developed by the Project Team and focussed on long-term trends in woodland bird detections. This investigated:

- **changes in species richness** to broadly assess community condition
- **individual species trends** to assess conservation status of woodland birds, and
- **composite trends for bird groups of special interest** to inform targeted conservation work.

It should be noted that the overall focus of the analysis, as determined by the Project Team, was on questions of conservation management relevance. It is not based on pure ecological criteria, which an academic study might focus on. This is particularly relevant to the analysis of bird groups outlined later in this report. The Project Team also considered it important to look at the analysis results in the context of weather patterns experienced during the analysis period, therefore supporting weather data are provided in Appendix B, and discussed where relevant.

The analysis was constrained by available time and funding for a professional analyst. Suggested questions for future projects using this dataset are presented in the Conclusion/Recommendation section of this report. An exhaustive report of statistical outputs is not presented here, and further interpretation of results may be undertaken.

Newline Quarry woodland

2. Methods

2.1. Bird Surveys

Bird survey data are collected on sites of 50-metre radius (*c* 0.8 hectares) within grassy woodlands. Ten of the 15 survey locations have 9 sites each, one has 8 sites, and 3 locations have 6 to 7 sites (Table 1). All survey locations are categorised into low, medium, and high habitat structure (*sensu* Bounds *et al.* 2010) to allow for an overall representation of the location. Mulligans Flat Nature Reserve surveys commenced in 1995 with 24 sites; these sites were selected to represent three habitat types: grassy woodland on flats, dry forest on low ridges, and areas at the ecotone of these communities.

Bird surveys are undertaken seasonally, with 4 surveys per year at each site. Each quarterly survey period varied slightly prior to 2003, when it was standardised to a 9-day survey window capturing two weekends at the end of March, June and September, and the end of November/beginning of December. While not encouraged, surveys can be undertaken within a week either side of the quarterly survey window. This gives the survey coordinators and teams some flexibility, many of whom have undertaken the surveys at their sites since establishment.

Surveys are carried out in the early part of the morning, with a 10-minute observation period at each site. All surveys are undertaken by experienced observers in fair weather. The species and number of birds (seen or heard) is recorded. Birds seen or heard up to 100 metres outside the site are also recorded (i.e. 50 to 100 metres from the centre of the site), however, only bird data collected inside the 50-metre radius sites were statistically analysed for this report.

Restless Flycatcher

2.2. Survey Sites

Survey locations and their management histories in this report (Table 1) are described in detail in the Ten-year Analysis Report (Bounds *et al.* 2010).

Table 1. Locations and date range of woodland bird survey data included in analysis, 1998-2019. "NR" is *Nature Reserve*.
* indicates sites managed primarily by Department of Defence.

Location	Number of sites	Survey period
Hall Common/Gold Creek (grazed leasehold) (HAL)	9	2000 to 2019
Mulligans Flat NR (MUL)	24	1998 to 2019
Goorooyarroo NR – North (GOO)	9	1998 to 2019
Goorooyarroo NR – South (GOS)	9	2004 to 2019
Mt Majura NR (MAJ)	9	1998 to 2019
Mt Ainslie NR/Campbell Park (CMP)	9	2003 to 2019
Symonston (Callum Brae NR/Isaacs Ridge NR) (SYM)	9	6 sites: 1998 to 2019 3 sites: 2004 to 2019
Jerrabomberra West NR (JER)	7	2006 to 2019
Red Hill NR (RED)	9	1998 to 2019
Kama NR (KAM)	9	2005 to 2019
Tuggeranong Hill NR (TUG)	7	2000 to 2019
Newline Quarry woodland* (NLQ)	9	2000 to 2019
Majura Training Area (MJF)*	8	1998 to 2000 2004 to 2019
Castle Hill (grazed leasehold) (CAS)	9	1998 to 2019
Naas Valley (grazed leasehold) (NAS)	6	2004 to 2019

2.3. Bird Groups

The Project Team identified 12 bird groups (below) of conservation interest to be analysed for long-term change. Collectively, groups included 125 species, and their allocation is shown in Appendix C. The method used for allocation is defined below. All bird species are allocated to at least one bird group, and bird species can be allocated to more than one bird group.

1. ACT woodland-dependent species: 48 species

There was no standard list of ACT bird species regarded as 'dependent on woodlands' to use as the basis for analysis. The Project Team considered various criteria for determining the species for this group, and individuals in the Project Team developed their own lists, some with criteria stricter than others. This included definitions of 'woodland conservation dependency', 'substantial dependency on woodlands', and 'not living outside woodland structure'. After the Project Team considered various approaches, it was agreed to base this group on a strict premise, 'if woodlands were removed these species would not occur'.

2. South-east woodland-dependent species: 49 species

This group was adopted from Fraser *et al.* (2019) and includes species considered part of the woodland bird community for the temperate south-east mainland eco-region of Australia by 70% of ornithological experts. Of six eco-regional variants, the 'south-east woodland-dependent' group was the most relevant to the ACT region, but differs from the 'ACT woodland-dependent' group determined by the Project Team (above). For example, the 'south-east woodland-dependent' group includes a number of species that occur in ACT forest (dry and wet) structures as well as ACT grassy woodlands (e.g. Brown Thornbill, Spotted Pardalote, White-browed Scrub-wren, White-eared Honeyeater and White-throated Treecreeper), which did not meet strict criteria for 'ACT woodland-dependent'.

3. Degraded woodland community: 10 species

This group was also adopted from Fraser *et al.* (2019). All but one species in this group (Pied Butcherbird) are common in the ACT. The 9 species common in the ACT are not habitat specialists and are generally tolerant of the ACT's urban related environments, e.g. Australian Magpie, Galah, Noisy Miner. The Pied Butcherbird is uncommon in the ACT but is increasing, and considered at the margin of its distribution, the core range being further inland in more open, agricultural, and drier habitats.

4. ACT threatened species: 11 species

This group comprised species listed as threatened under ACT legislation (*Nature Conservation Act 2014*) that were recorded in the Woodland Bird Monitoring Project. Of these, five species are also listed under Commonwealth legislation (*Environment Protection & Biodiversity Conservation Act 1999*).

5. Species of conservation concern: 14 species

This group is based on the local bird knowledge of the Project Team. The group includes some bird species on the ACT threatened list, particularly resident/sedentary species, and others considered to be in very low abundance, thought to be declining, and/or possibly at risk of moving into a threatened category.

6. Canopy feeding species: 24 species

The Project Team considered various criteria for the canopy feeding group. Reference was made to habitat assessment criteria based on structural elements developed for assessing habitat at monitoring sites by Alison Rowell (Rowell 2004; Bounds *et al.* 2010). It was decided to use vegetation structure over two metres as 'canopy'. This could include tall shrubs such as acacias, tall eucalypt re-growth, small and large trees. Birds were allocated based on their primary (but not exclusive) use of this structure for foraging.

7. Mid-storey feeding species: 3 species

This category is defined as species which use up to two metres of the shrub and eucalypt layer, based on Alison Rowell's habitat assessment criteria (Rowell 2004). Few species fit this category with only 3 species identified to primarily use the mid-storey for foraging: Buff-rumped Thornbill, Dusky Woodswallow, and Yellow Thornbill.

8. Ground feeding species: 17 species

The Project Team determined the criteria for the ground feeding group based on ground-layer substrates, such as grasses (short and tall), leaf litter, low-lying debris, forbs and very low shrubs. The ground feeding group includes some birds in low abundance or of conservation concern, such as three robins and two native finches. Only one large-bodied bird was included in this group: White-winged Chough.

9. Woody weed frequent species: 16 species

This group is known to occupy woody weed structures for food, shelter, or nesting as determined by the local knowledge of the Project Team. Common woody weeds found in ACT woodlands include Sweet Briar/Briar Rose, Blackberry, Hawthorn, African Boxthorn, Cotoneaster, and Firethorns. Birds in this group include a mix of 10 small-bodied species (e.g. finches and wrens) and 6 large-bodied species.

10. Small woody weed frequent species: 10 species

This group comprises the 10 small-bodied species (< 70 g) of birds from the 'woody weed frequent' group. Woody weeds may have an important function for small birds in the absence of native shrubs. At some woodland sites, woody weeds are the only or predominant "shrubs", and their removal has implications for the protection and conservation of cover-dependent birds.

11. Small-bodied species: 74 species, and (12) Large-bodied species: 52 species

Birds were analysed as either small-bodied (< 70 g) or large-bodied (> 70 g) based on body mass measured in grams. The 70 g cut-off reflects the body size of Noisy Miners (71 g). Additional cut-offs were tested (50 g and 100 g), but effects of body size were robust to these small variations in classification. The 70 g class showed the clearest difference in group response.

2.4. Data Analysis

For the analysis, trends in *Species Richness* are derived from the number of species detected at each bird survey, whilst trends in *Individual Species Reporting Rate* (hereafter *Individual Species*) are derived from the detection (presence/absence) of an individual species at each bird survey thereby adjusting for any differences in survey effort between years. The reporting rate is the number of surveys in which a species was recorded out of the total number of surveys for that season. The effect on *Species Richness* and *Individual Species* of the following factors was modelled:

- **Time** – the effect of each year spanning 1998 to 2019 (21 years)
- **Location** – representing the 15 locations at which bird surveys took place, and
- **Site** – representing the 142 survey sites, nested within *Location*.

All survey data were checked to ensure that 'true-zero' observations were included; that is, those surveys in which no birds were observed. All water-based bird species (39 species) were omitted from analysis because site locations do not adequately represent their habitat. Survey effort was consistent across all seasons through time, such that trend profiles for migratory birds are robust to detection bias and presented in this report.

Below, the methods for calculating Individual Species Trends, Rates of Change, and Composite Trends for Bird Groups are outlined. Readers seeking further details of analysis may contact COG, see the COG website for email contact.

Weebill

2.4.1. Individual Species Trends

Detections for individual species were used to model the trends of birds through time. Both generalised additive models (GAM), and generalised additive mixed models (GAMM), were fitted to individual species detections data, providing a smoothed (non-linear) trend line. To test whether there was an overall increase or decrease in detections of individual species from 1998 to 2019, generalised linear models (GLM), and generalised linear mixed models (GLMM), were fitted to individual species detections, providing a straight (linear) trend line.

Interpreting Trend Plots

Adopting the method of Bounds *et al.* (2010), plots for individual species show both non-linear and linear trends with associated confidence intervals (indicating significance of the trend pattern), as well as the raw data. Plot elements should be interpreted as follows (see Figure 1):

1. mean reporting rate in each quarter-yearly survey (thin black line)
2. smoothed *GAM/M* trend line showing short-term change in reporting rate (purple line)
3. 95% confidence intervals of the smoothed trend (purple shading)
4. sections of significant rate of change in the smoothed fit (positive is blue, negative is red)
5. predicted straight *GLM/M* trend line fit (green dashed line)
6. 95% confidence intervals of the linear trend (green shade)
7. P-values of the *GAMM* and *GLMM* indicating trend significance ($P < 0.05$ is significant), and
8. a representation of survey effort in each quarter-yearly survey (red bars below main plot); the maximum number of surveys conducted in a quarter was 236.

For all plots, predicted trends for individual species detections were transformed from their original scale of proportions (0 – 1) to percentages (0 – 100). The percent scale (y-axis) should be interpreted as the mean percent of surveys in which a species is predicted to be present. When reporting significant trends in Individual Species only the straight GLM/M trend line fit is referred to.

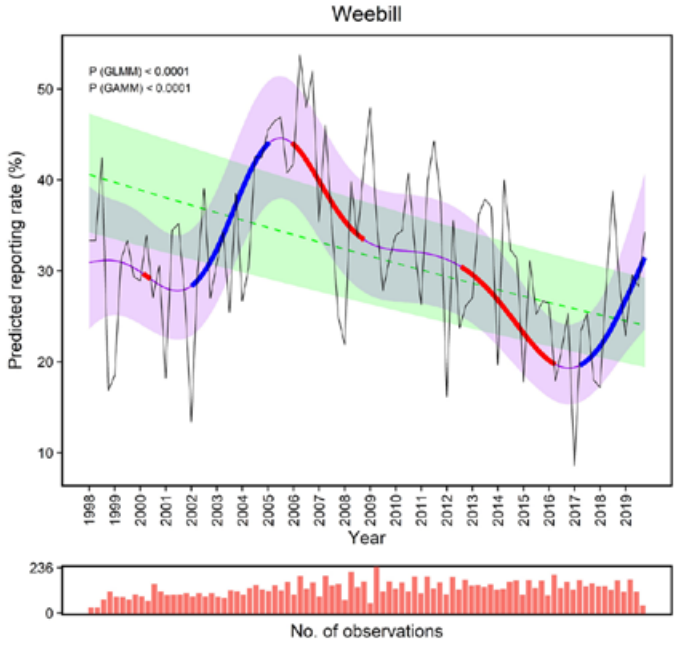

Figure 1. Example *Trend Plot* for interpreting individual species results.

2.4.2. Rate of Change (ROC)

For each species reporting rate fit, the time in which there was significant increase or decrease in the rate of change (ROC) in the smoothed trend was determined by calculating the first derivative and its associated point-wise confidence intervals. Values were calculated for each quarter year, and only significant values were plotted.

Interpreting ROC Plots

To visualise increases and decreases in the rate of change for multiple species, including bird groups, "ROC plots" were created. As per *Individual Species Trend Plots*, sections of significant rate of change in the smoothed fit are shown in blue for positive change and in red for negative change (Figure 2).

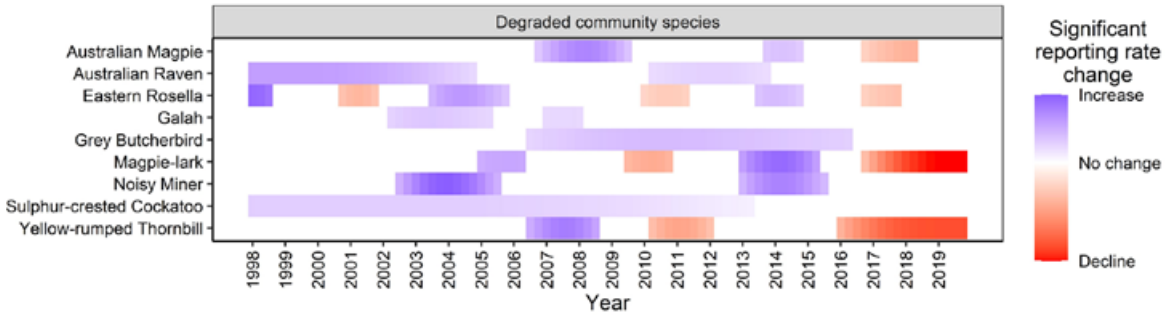

Figure 2. Example *ROC Plot* for interpreting significant periods of change across multiple species

2.4.3. Composite Trends for Bird Groups

To test whether birds of a particular grouping (n = 12, described above) showed common linear trends from 1998 to 2019, a guilds analysis was conducted on linear trend results from the individual species analysis described above. To do this, a new variable representing the time-effect (i.e. trend) for each species was fitted against the two-level factor (included or not) for each bird group.

Interpreting Effects Plots

Summary effect size plots, showing the magnitude of change for individual bird species, are provided. For individual species and bird groups, only species that have shown significant change through time are plotted. Plot elements should be interpreted as follows (Figure 3):

1. a decreasing trend is displayed left of the neutral (dotted) line
2. an increasing trend is displayed right of the neutral (dotted) line
3. the size of the dots represents the relative reporting rate of each species over all surveys, i.e. species with larger points are reported more often than species with smaller points, and
4. lines through the points are confidence intervals, which indicate how reliable the mean effect size is (the longer the line, the more variable or sparse the data are).

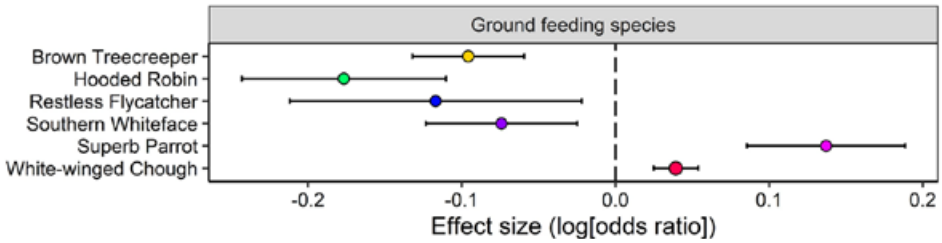

Figure 3. Example *Effects Plot* for interpreting bird group results. Note: colours are for contrast only.

3. Results and Discussion

The analysis incorporated 10,728 surveys undertaken at 142 sites between 1998 and 2019. The number of bird surveys completed per site ranges from 52 to 92 over the observation period (Appendix D). Bird data for 129 land-based woodland bird species (water-based birds excluded) were analysed.

3.1. Species Richness

The analysis showed no significant change in species richness from 1998 to 2019 across woodland bird monitoring sites (Figure 4). There were significant (but small: average 1–2 species) decreases in species richness from 1999 to 2001 and from 2015 to 2019, possibly reflecting significantly drier than average years (Appendix B).

When additional years of data are analysed, it will be interesting to see if the downward trend from 2015 to 2019 continues or not, noting that 2020 had significantly higher than average rainfall (Appendix B) and may signal a positive change in patterns.

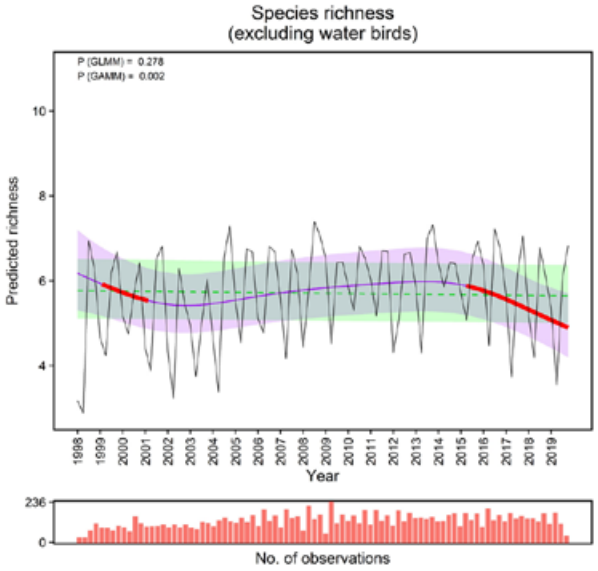

Figure 4. Trend in species richness at 15 woodland locations from 1998 to 2019

3.2. Individual Species

Long-term trends in reporting rate were calculated for 129 land-based woodland species, of which 73 species showed no significant trend pattern, and are not individually discussed. In some cases, non-significant trend results may be due to limitations of the project sampling design. For example, no raptors showed statistically significant trends, likely because the within-site (50 m) survey methodology is not effective for measuring the abundance of raptors (a more targeted sampling design is needed).

Significant trends – either an increase or decrease – were observed for 56 species (see Table 2 below and Appendix E). Table 2 provides an overview of significant trend patterns, and linear effects were plotted on one large effects plot in order of magnitude from greatest decrease to greatest increase (Figure 5). To visualise whether decreases or increases in the rate of change of the non-linear (smoothed) trends were correlated across several species, the rates of change were plotted on a ROC Plot, provided in Appendix F.

3.2.1. Decreasers

Thirty-two species showed a statistically significant decrease in reporting rate over the analysis period (Table 2). Twenty-eight are native species, and include seasonal migrants[1], residents, and a range of sizes of birds (21 small-bodied: < 70 g, and 7 large-bodied: > 70 g). Some of these birds are thought by observers to have declined at some woodland sites or occur in very low numbers at sites (e.g. White-plumed Honeyeater, Southern Whiteface, Crested Shrike-tit). However, the long-term decline of some species was unexpected to the Project Team, including the following 16 species generally regarded as common:

- Tree Martin
- Fuscous Honeyeater
- Pallid Cuckoo
- Dollarbird
- Superb Fairy-wren
- Mistletoebird
- Striated Thornbill
- Weebill
- Yellow-rumped Thornbill
- Rufous Whistler
- Willie Wagtail
- Western Gerygone
- Striated Pardalote
- Leaden Flycatcher
- Buff-rumped Thornbill
- Grey Fantail.

3.2.2. Increasers

Of the 24 species that showed increasing trends (Table 2) most are common, large-bodied birds, adapted to urban environments (i.e. habitat generalists). However, the list of increasers does include 5 small-bodied birds that occur in ACT lowland and mountain forests, as well as woodland habitats (i.e. not strictly woodland-dependent); for example, Golden Whistler, White-eared Honeyeater, and Brown Thornbill.

3.2.3. Consistent linear changes

Species that showed steady, consistent decreases over every year in the analysis period were:

- Hooded Robin
- Tree Martin
- Restless Flycatcher
- Pallid Cuckoo
- Dollarbird
- Rufous Whistler.
- Grey Currawong
- Western Gerygone

Species that showed steady, consistent increases over every year in the analysis period were:

- Rainbow Lorikeet
- Satin Bowerbird
- Little Corella
- Gang-gang Cockatoo
- Australian King-Parrot
- Australian Raven.
- White-winged Chough
- Crimson Rosella

1 *Seasonal migrant* refers to a species that has constant and regular movements, moves into the ACT over the spring and summer months (for breeding), and moves out of the ACT for the cooler months to other locations, e.g. north, coastal. This is distinct from an *altitudinal migrant*, which moves within the local area, breeds in upland forests, and moves to lower and milder altitudes in autumn/winter (including to lowland woodlands).

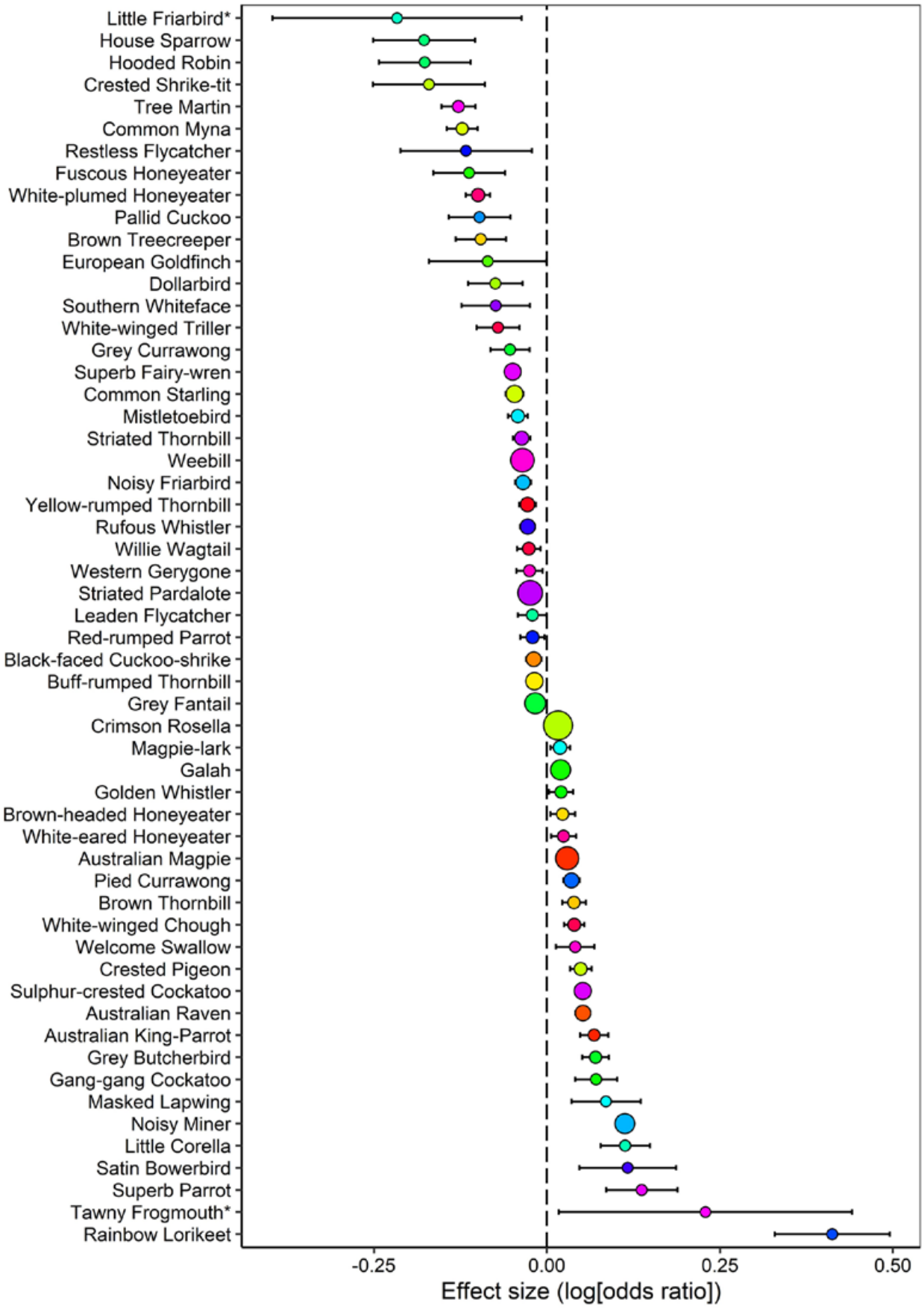

Figure 5. Summary *Effects Plot* showing the magnitude of change in the occurrence of individual bird species that have shown a significant trend pattern. Species marked with an asterisk (*) were rare with fewer than 10 detections over the analysis period.

Table 2. Trend summary for individual species that showed significant change over the analysis period (1998–2019). Species marked with an asterisk (*) show consistent significant inter-annual trend effects, i.e. significant annual decrease or increase in every year of the analysis period. Mean site-level reporting rate (RR) is presented for the first year (1998) and last year (2019) of the analysis to provide indication of linear change through time.

Species name	No. Records	1998 RR (%)	2019 RR (%)	P value	Trend description
DECREASERS ***Greatest to least***					
Little Friarbird	7	0.25	0.00	0.008	Seasonal. Very low RR, most years with no records. Significant decline over the analysis period.
House Sparrow	89	0.23	0.01	< 0.001	Low RR. Significant decline over the analysis period, few records in the last 10 years.
Hooded Robin*	74	0.62	0.02	< 0.001	Low RR declining to very low. Significant steady decline 1998–2014 after which there were no records.
Crested Shrike-tit	44	0.66	0.02	< 0.001	Low RR. Slight peak in 2006, then significant decline 2007–2013 after which there were no records.
Tree Martin*	1,439	2.8	0.20	< 0.001	Seasonal. Medium RR declining to low. Significant decline 1998–2011. Still declining after this but at a less significant rate.
Common Myna	1,171	3.10	0.24	< 0.001	Medium RR declining to low. Significant decline 1999–2002, then increase to a slight peak in 2006, followed by significant decline to 2012. Small increase 2014–2016, then another decline from 2018. Overall downward trend.
Restless Flycatcher*	17	0.36	0.03	0.016	Low RR, more than half the years with no records. Significant decline over the analysis period.
Fuscous Honeyeater	265	1.12	0.11	< 0.001	Seasonal. Low RR. Significant decline 1999–2002. Small increase 2004–2006 followed by significant decline 2009–2010 and ongoing slow decline since.
White-plumed Honeyeater	1,723	5.48	0.71	< 0.001	Medium RR declining to low. Significant decline 1998–2002 then slight increase 2003–2005. Further significant declines 2007–2009 and 2012–2015. Still declining after this but at a less significant rate.
Pallid Cuckoo*	77	1.44	0.19	< 0.001	Seasonal. Low RR. Significant steady decline over the analysis period.
Brown Treecreeper	222	0.36	0.05	< 0.001	Low RR declining to very low. Significant decline from 2011 onwards.
European Goldfinch	59	0.29	0.05	0.048	Sparse observations. Slow marginally significant decrease over the analysis period.
Dollarbird*	127	1.20	0.25	< 0.001	Seasonal. Low RR declining to very low. Significant steady decline over the analysis period.

Table 2 — continued

Species name	No. Records	1998 RR (%)	2019 RR (%)	P value	Trend description
Southern Whiteface	178	0.46	0.10	0.003	Low RR. Significant decline 2009–2014.
White-winged Triller	241	2.44	0.56	< 0.001	Seasonal. Medium-low RR. Significant decline 1998–2000. Small but significant increase in 2010–2011 followed by decline 2013–2014.
Grey Currawong*	201	1.12	0.37	< 0.001	Low RR declining to very low. Significant steady decline over the analysis period.
Superb Fairy-wren	6,535	14.28	5.57	< 0.001	Medium RR. Significant decline 2000–2006, significant increase 2008–2011 followed by significant decline 2013–2019 to give an overall declining trend.
Common Starling	9,547	12.15	4.91	< 0.001	Medium RR. Two periods of significant declines; 1999–2002 and 2010–2012, each followed by 2 years of significant increase; 2003–2005 and 2013–2015. Overall downward trend.
Mistletoebird	900	6.53	2.81	< 0.001	Seasonal. Medium RR. Significant declines in 2000–2001 and 2014–2019 give an overall declining trend.
Striated Thornbill	3,519	6.24	3.00	< 0.001	Medium RR declining to low. Significant decline in 2005–2008 followed by a small but significant increase 2010–2012 then another decline 2014–2017.
Weebill	8,949	40.58	24.49	< 0.001	High RR with large fluctuations. A significant increase in 2002–2005 followed by two periods of significant decline 2006–2009 and 2012–2016, then a significant increase 2017–2019. Overall declining trend.
Noisy Friarbird	1,786	11.70	6.03	< 0.001	Seasonal. Medium RR. Significant decline 1998–2002, followed by significant increase 2004–2006, then ongoing declines, significant in 2009–2011 and 2016–2019.
Yellow-rumped Thornbill	3,941	10.23	5.94	< 0.001	Medium RR. Significant increase 2006–2008 followed by significant declines 2010–2012 and 2016–2019, with an overall declining trend.
Rufous Whistler*	1,631	10.14	5.91	< 0.001	Seasonal. Medium RR. Significant steady decline across the analysis period.
Willie Wagtail	730	3.66	2.15	0.003	Medium to low RR. Significant decline 1998–2002. Non-significant downward trend since.
Western Gerygone*	440	2.33	1.39	0.009	Seasonal. Low RR. Significant steady decline over analysis period.
Striated Pardalote	8,866	42.15	30.45	< 0.001	High RR. A significant increase 1998–2000, followed by a short but significant decline 2001–2002. Another increase 2003–2005 then a significant decline 2010–2013. Overall declining trend.

Table 2 — continued

Species name	No. Records	1998 RR (%)	2019 RR (%)	P value	Trend description
Leaden Flycatcher	386	2.34	1.52	0.048	Seasonal. Low RR. Significant decline 1998–2002.
Red-rumped Parrot	2,143	1.74	1.13	0.019	Low RR. Significant decline 1998–2006.
Black-faced Cuckoo-shrike	1,657	12.42	8.66	< 0.001	Seasonal. Medium RR. Two periods of significant decline; 1998–2002 and 2015–2019.
Buff-rumped Thornbill	7,199	12.14	8.64	< 0.001	Medium RR. Two periods of significant decline; 2000–2002 and 2014–2019.
Grey Fantail	5,088	26.35	20.11	< 0.001	Seasonal. High RR. Significant decline 1998–2000, significant increase 2009–2012, followed by another decline 2013–2016.
INCREASERS ***Greatest to least***					
Rainbow Lorikeet*	285	0.00	0.64	< 0.001	Low RR. First records not until 2008 but significant increase since then.
Tawny Frogmouth	7	0.00	0.24	0.034	Very low RR. First records in 2013. Significant increase since then.
Superb Parrot	168	0.02	0.30	< 0.001	Seasonal. Low RR. First records in 2005 and a significant increase from then until 2012.
Satin Bowerbird*	44	0.01	0.14	0.001	Low RR. First records in 2000 then sporadic until 2011. Significant increase over the period 2000–2019.
Little Corella*	376	0.12	1.24	< 0.001	Low RR. Significant steady increase from first records in 1999 to 2019.
Noisy Miner	7,985	2.97	24.49	< 0.001	Low RR increasing to high. Two periods of significant increase 2002–2008 and 2012–2016 giving a strong upward trend.
Masked Lapwing	250	0.00	0.02	0.001	Very low RR. Significant increase 2007–2010.
Gang-gang Cockatoo*	342	0.07	0.30	< 0.001	Low RR. Slight but significant increase in RR over the analysis period.
Grey Butcherbird	443	1.07	4.51	< 0.001	Low RR increasing to medium. Significant steady increase 2004–2019.
Australian King-Parrot*	1,036	0.39	1.60	< 0.001	Low RR. Significant steady increase over the analysis period.
Australian Raven*	2,528	6.12	16.26	< 0.001	Medium RR. Significant steep increase over the analysis period.
Sulphur-crested Cockatoo	4,576	7.08	18.39	< 0.001	Medium RR. Significant steady increase 1998–2013. Still increasing to 2019 but non-significant.
Crested Pigeon	1,259	1.93	5.20	< 0.001	Low RR increasing to medium. Significant increases 2002–2005 and 2006–2009 to a peak, then significant decrease 2010–2013 before another significant increase 2014–2018.

Table 2 — continued

Species name	No. Records	1998 RR (%)	2019 RR (%)	P value	Trend description
Welcome Swallow	569	0.56	1.30	0.004	Low RR. Significant increase 2009–2013 then significant decline 2015–2019. Overall subtle, significant increase across the analysis period, most likely due to rainfall response.
White-winged Chough*	3,613	2.26	5.01	< 0.001	Low RR increasing to medium. Significant steady increase over the analysis period.
Brown Thornbill	760	1.58	3.52	< 0.001	Low RR. Steady until small but significant increase 2008–2012.
Pied Currawong	2,319	4.74	9.48	< 0.001	Low to medium RR. Steady until a small but significant increase 2012–2015, then small but significant decrease 2017–2018.
Australian Magpie	6,884	23.96	36.79	< 0.001	Medium RR increasing to high. Steady then significant increases in 2006–2010 and 2014 followed by significant decline 2016–2019. Overall significant linear increase across the analysis period.
White-eared Honeyeater	429	1.81	2.96	0.010	Mostly seasonal. Low to medium RR. Small but significant increase 2006–2009, small but significant decrease 2014–2017. Overall slightly increasing trend.
Brown-headed Honeyeater	1,251	1.49	2.38	0.013	Low RR. Slight increase, (significant only in 2012) through to a peak in 2014, significant decrease 2016–2019. Overall subtle, significant linear increase across the analysis period.
Golden Whistler	484	2.43	3.68	0.024	Mostly seasonal. Low to medium RR with two peaks, 2003 and 2013. Significant changes before and after these peaks, but an overall increasing trend.
Galah	6,453	17.16	23.90	< 0.001	Medium RR increasing to high. An initial significant decline 1998–2000, followed by significant increases between 2002–2008 and an ongoing upward trend.
Magpie-lark	1,411	3.39	4.98	0.008	Low RR increasing to medium. Slightly increasing trend overall with two peaks in 2008 and 2016 with significant changes before and after these peaks.
Crimson Rosella*	14,130	43.16	51.62	< 0.001	High RR. Significant steady increase over the analysis period.

3.2.4. Increasers of interest

Increases in these species are of ecological relevance: a substantial increase in the resident Noisy Miner, and colonisation in recent decades by Rainbow Lorikeet and Little Corella.

Noisy Miner

Noisy Miners showed a significant six-fold increase in reporting rate over the analysis period. This species is an aggressive, medium-sized, colonial honeyeater known to exclude or suppress small birds. There is widespread concern about these impacts, with culling programs implemented across many areas of the Australian temperate zone. Further research is needed to inform Noisy Miner management programs in the ACT.

Rainbow Lorikeet

Rainbow Lorikeets showed a significant increase in the analysis. This is a nectar-feeding parrot typically associated with coastal habitats, and first observed in the ACT around 1986 (Canberra Ornithologists Group). In many parts of Australia, Rainbow Lorikeets are considered an invasive species and, while their reporting rate is low in the ACT, its increase is consistent and statistically reliable.

Little Corella

Little Corellas showed a slight but steady and significant increase in reporting rate over the analysis period. This is a ground feeding, flock species of cockatoo that is very common in open, inland habitats, and has been increasing in the ACT since around 1990 (Canberra Ornithologists Group).

3.2.5. Decreasers of interest

Two small, canopy feeding honeyeaters which have a wide distribution/range outside the ACT are highlighted. The White-plumed Honeyeater is a resident breeder in ACT lowland woodland, and the Fuscous Honeyeater a breeder in mid-altitude woodland.

White-plumed Honeyeater

White-plumed Honeyeater showed an overall decrease of concern. Before 2002, there is a strong decline, and a more gradual decline in the trend since then, to a low level of abundance (Appendix E). This is a widely distributed honeyeater, common along inland river systems. In the ACT, they have been considered a fairly common species, largely sedentary, resident in woodlands as well as parklands with mature eucalypts. The species has been thought by observers to have reduced in numbers or disappeared from some woodland sites, and the analysis result confirms this.

Fuscous Honeyeater

Fuscous Honeyeater showed an overall decrease, with a very low reporting rate since 2011 (Appendix E). The decline is of concern, particularly as the species' core ACT habitat is in the southern half of the ACT where it breeds, at mid-altitude open woodlands in valleys and on footslopes. This area was impacted by bushfires in 2003 and 2020. Some seasonal movement is known in cooler months to ACT peri-urban woodlands where there are nectar sources.

Table 4 has further examples of species with significant decreases.

3.2.6. Reporting rate change and weather

The end of the Millenium Drought (2009) with a period of significantly higher than average rainfall (2010–2012) was a notable event in the analysis period, but most ACT bird species did not show a sustained population-level response (Appendix E).

Some species showed a significant increase in reporting rate from 2010 to 2012 despite decreasing overall over the period 1998 to 2019, such as Grey Fantail, Striated Thornbill, Superb Fairy-wren and White-winged Triller. This could be explained as a positive short-term response to the higher than average rainfall over those years, although seasonal and monthly rainfall was variable (Appendix B). For others, like the Brown Treecreeper and Crested Shrike-tit, their long-term decline was not interrupted by the breaking of the drought.

For the increaser species, there are some species which showed a significant reporting rate increase over the years 2010 to 2012, for example Noisy Miner, Brown Thornbill, Golden Whistler, Pied Currawong and Welcome Swallow, due to a short-term response presumably related to the higher rainfall over that period. The reporting rates of Crested Pigeon and Magpie-lark decreased temporarily at the breaking of the drought, then continued to increase in subsequent years.

Brown-headed Honeyeater, a species showing a small overall increase over the period 1998 to 2019, showed a slight increase in reporting rate during the Millennium Drought, a more significant increase over 2011 to 2012, with a marked decline from 2016, a drier period. This suggests the species is stable overall at ACT woodland sites and appears to tolerate drier conditions; perhaps indicating it is less dependent on nectar sources than other honeyeaters.

3.2.7. Seasonal migrants

The only seasonal (or partial) migrants that showed a significant increase are Welcome Swallow and Superb Parrot. Twelve seasonal migrants (see below) showed decreases; of which 11 are common species that breed in ACT woodlands (excludes Little Friarbird). Trends in seasonal migrants are variable and not clearly explained by local weather.

- Rufous Whistler
- Leaden Flycatcher
- Dollarbird
- Mistletoebird
- Tree Martin
- Pallid Cuckoo
- Noisy Friarbird
- Little Friarbird
- Black-faced Cuckoo-shrike
- White-winged Triller
- Western Gerygone
- Grey Fantail.

Brown-headed Honeyeater

3.2.8. Resident species

Of the 22 resident species that showed a significant increase, 18 are large-bodied species. Of the small-bodied species, two are altitudinal migrants in the ACT (Golden Whistler, White-eared Honeyeater). Weebill and Buff-rumped Thornbill are examples of small, resident species that showed a significant variation, but an overall decrease, in their reporting rates. Significant drops in the reporting rate of Buff-rumped Thornbill appeared to coincide with drier than average periods (e.g. 2000–2003 and 2014–2019), whereas changes in the reporting rate for Weebill do not appear weather-related. The large-bodied Grey Currawong showed a decreasing trend. This species is resident in woodlands. They are at some woodland sites in fairly low numbers, as breeding pairs.

3.2.9. Non-native species

No non-native species were found to have increasing trends. Of the 7 non-native species, 4 showed significant decreases: House Sparrow, Common Myna, European Goldfinch and Common Starling. The remaining 3 are marginal woodland species. Although records are few, the Common Blackbird showed a decline with a positive response with the breaking of the Millennium Drought.

3.3. Bird Groups

Seven bird groups showed significant trends over the analysis period (Figure 6). This included:

- an increase in large-bodied birds, and birds associated with degraded woodland communities, and
- a decrease in small-bodied birds, canopy feeders, south-east Australian woodland birds, ACT woodland-dependent birds, and ACT species of conservation concern.

There was no collective trend in threatened species, ground feeders, or birds associated with woody weeds (all species or small-bodied species). Models did not converge for mid-storey feeders due to sparse data. Additional information showing the magnitude of (significant) change for individual species in a bird group is provided in Appendix G.

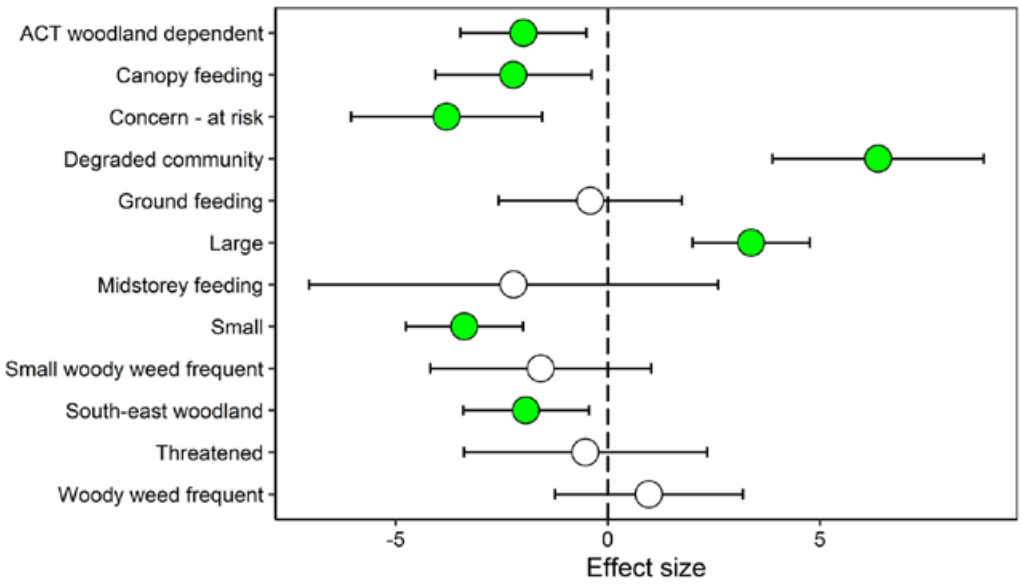

Figure 6. *Effect size plot* for bird groups of interest. Points to the right of the zero line represent that species in that group have collectively increased over time (1998 to 2019), and points to the left of the zero line represent that species in that group have collectively decreased over the time. Bird group trends that are statistically significant are shown in green.

Kama Nature Reserve woodland in autumn

3.3.1. ACT woodland-dependent species

For this group (48 species; Appendix C), a decrease in reporting rate was statistically significant (Figure 6). A similar result was observed for the south-east Australian woodland group (49 species) despite allocation differences for 29 species. Of the 48 species in the ACT woodland-dependent group, 25 species showed a significant trend. Of these 25 species, 21 showed decreases, and 4 showed increases (Brown-headed Honeyeater, Grey Butcherbird, Noisy Miner and Superb Parrot).

3.3.2. Degraded woodland community

This group showed a significant increase across ACT woodland monitoring sites (Figure 6). Of the 10 species in the degraded woodland community group, 2 (Eastern Rosella and Pied Butcherbird) did not show any significant trend. Seven species showed a significant increase, and the Yellow-rumped Thornbill (a small-bodied bird) was the only species that showed a decrease (Appendix G). A general increase in birds associated with a degraded woodland bird community across the analysis period may be indicative of degradation or simplification of important woodland bird habitats. Investigations into woodland condition within the monitoring survey extent could assist land managers in determining if and where restoration action is needed.

3.3.3. Canopy feeding species

The significant decrease in canopy feeders (Figure 6) as a collective was unexpected; there were no obvious indications to observers prior to the analysis. The canopy feeding group (24 species) includes insectivores and nectar feeders, and one specialist feeder (Mistletoebird). Species in this group were various sized honeyeaters, resident and seasonal species, and a range of common species (Figure 7). Despite this mixture, however, rates of change among canopy feeders showed some similarity through time, with multiple group members showing higher reporting rates in 2003–2004 and lower reporting rates since 2015 (Figure 8).

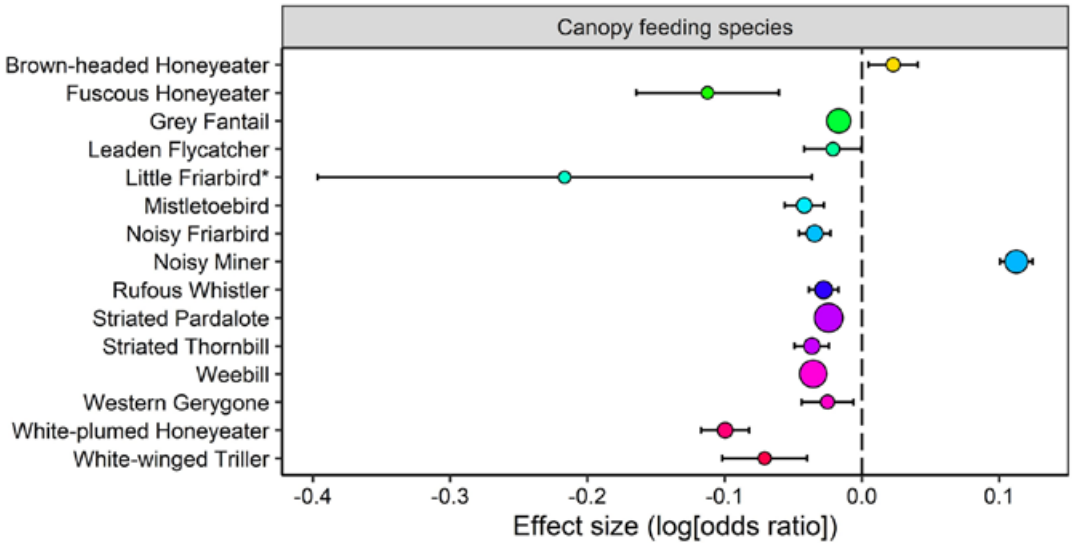

Figure 7. *Effects Plot*: Significant trends in canopy feeding species

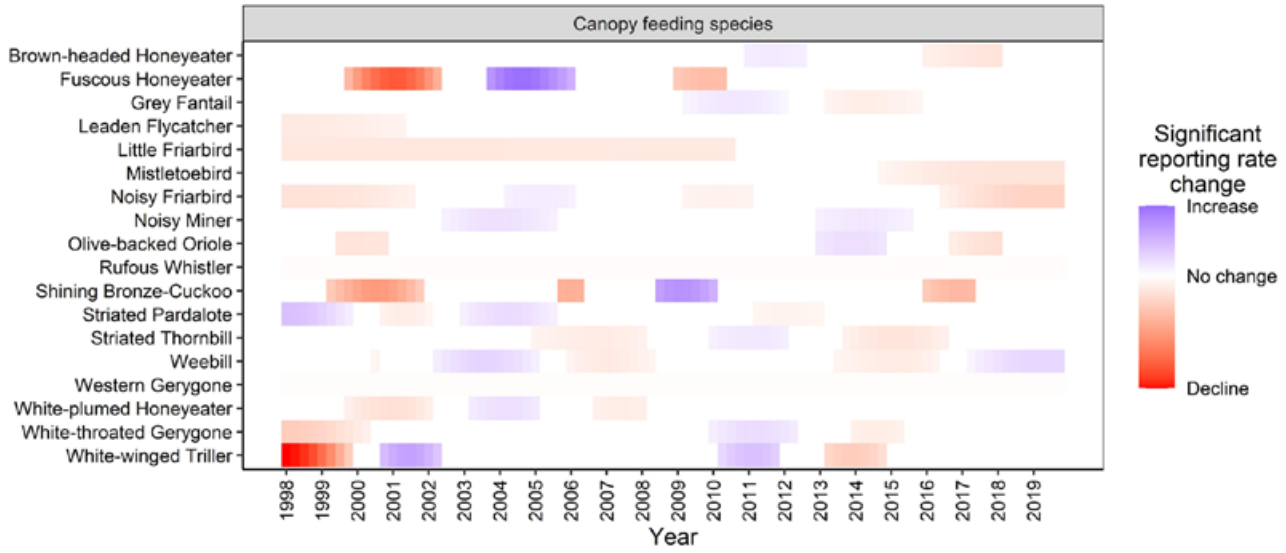

Figure 8. *ROC Plot*: Periods of significant reporting rate change for canopy feeding species

3.3.4. Ground feeding species

Ground feeders (17 species) comprised mostly small-bodied birds and showed no significant collective change in reporting rates (Figures 6, 9). Previous analyses (Bounds *et al.* 2010), and anecdotal information, suggest ground feeders may be of conservation concern. This was based on factors such as increasing weather variability, overgrazing by native herbivores, deterioration of ground layer quality and diversity, and weeds invasion. Furthermore, a number of ground feeding species showed declining trends in previous analyses, or have very low numbers in the ACT region. With an additional 10 years of data, ground feeders as a collective appear relatively stable.

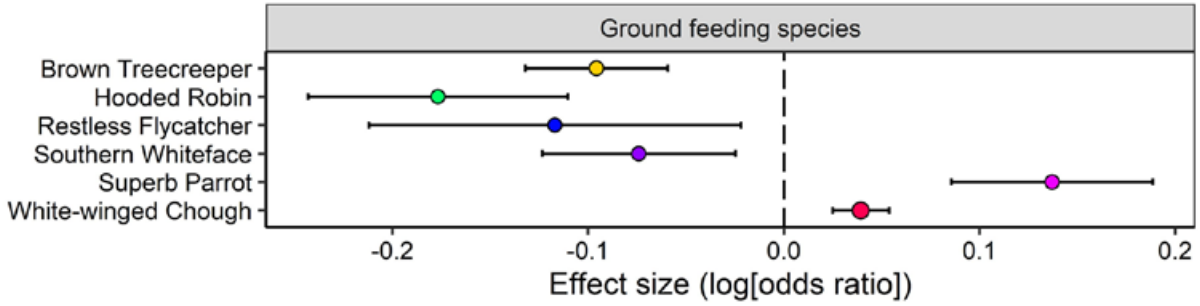

Figure 9. *Effects Plot*: Significant trends in ground feeding species

3.3.5. Woody weed frequent species

Woody weed frequent birds (16 species) showed no significant collective trend (Appendix G). Among the small woody weed frequent birds, 4 were decreasing: European Goldfinch (non-native), Red-rumped Parrot, Southern Whiteface, and Superb Fairy-wren. Where the removal of woody weeds is deemed necessary, land managers should first determine whether the area is occupied by any of the native declining species and, where present, additional measures (e.g. staged or supplementary habitat plantings) should be undertaken to mitigate potential negative impacts of habitat removal.

3.3.6. ACT Threatened species

Birds listed as threatened in the ACT (11 species; Table 3) did not show a consistent collective trend (Figure 10). The analysis showed 4 threatened species with statistically significant trends: 3 with decreasing reporting rates (Brown Treecreeper, Hooded Robin and White-winged Triller) and one with an increasing reporting rate, Superb Parrot. It is worth noting that the Superb Parrot and White-winged Triller are highly mobile species, and seasonal breeding migrants (or partial migrants) in the ACT. Their numbers are likely influenced by seasonal conditions within and outside of the ACT.

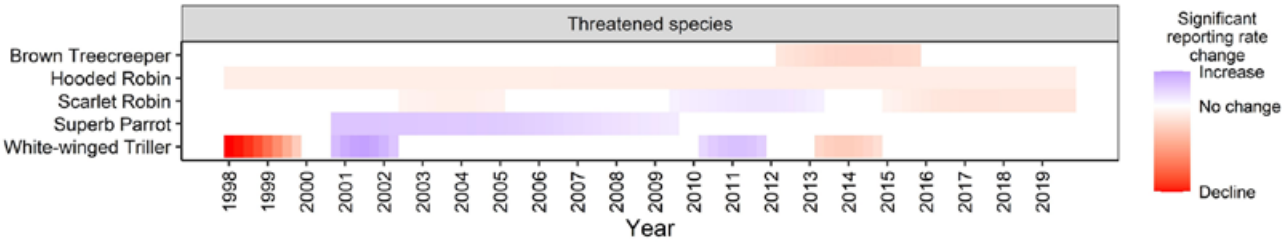

Figure 10. *ROC Plot*: Periods of significant reporting rate change for threatened species

Table 3. ACT threatened species, their listing source, and trend result. Species with significant trends are shown in bold. Asterisk (*) indicate rare species that are not resident in ACT woodlands.

Species	Listing	Trend
Brown Treecreeper	**ACT**	**Decrease**
Hooded Robin	**ACT**	**Decrease**
Little Eagle	ACT	No trend detected
Painted Honeyeater*	Commonwealth and ACT	n/a
Rainbow Bee-eater	Commonwealth and ACT	No trend detected
Regent Honeyeater*	Commonwealth and ACT	No trend detected
Scarlet Robin	ACT	No trend detected
Superb Parrot	**Commonwealth and ACT**	**Increase**
Swift Parrot*	Commonwealth and ACT	No trend detected
Varied Sittella	ACT	No trend detected
White-winged Triller	**ACT**	**Decrease**

Brown Treecreeper & Hooded Robin

Brown Treecreeper and Hooded Robin are flagship woodland-dependent species and ACT listed threatened species, in very low abundance in ACT woodlands, and have disappeared over time from peri-urban woodland sites. These species have been in steady decline as shown in previous analyses (Bounds *et al.* 2010, Bounds 2007). This analysis confirms significant, steady, ongoing declines. These species appear unable to take advantage of and recover with what would typically be considered favourable weather conditions (e.g. higher rainfall periods), possibly due to their now very small sub-populations and factors relating to habitat loss and fragmentation. The ROC Plot result (Figure 10) for Hooded Robin is of note; a consistent, small decline over every year in the analysis.

Superb Parrot

Superb Parrot

Superb Parrot showed a significant increasing trend (Figure 10, Table 3) driven by a jump in reporting rates from 2005 to 2012 (Figure 11). While their reporting rate remains below 1% on woodland sites, this trend corresponds to increased sightings of the species across Canberra, from small numbers known historically to breed around northern ACT. At the present time, there are at least two breeding colonies established in ACT peri-urban woodland. Local research on Superb Parrots suggests their increasing trend may be climate related.

Scarlet Robin

Scarlet Robin did not show a statistically significant trend. However, for Scarlet Robin short-term changes in reporting rate appear to reflect wetter and drier periods; i.e. decrease over the Millennium Drought to 2008, and some recovery to 2014 coinciding with higher rainfall over 2010 to 2012 (Figure 11). As such, the species appears to have some ability to recover their abundance over a breeding season following high rainfall.

Of concern is the marked decrease in reporting rate of Scarlet Robin since 2014 (mostly drier than normal years), to the lowest reporting rate in 21 years of woodland monitoring. The Scarlet Robin was listed as an ACT threatened species in 2015, based on COG's Ten-year analysis (Bounds *et al.* 2010) and other supporting data, which showed a gradual, steady decline.

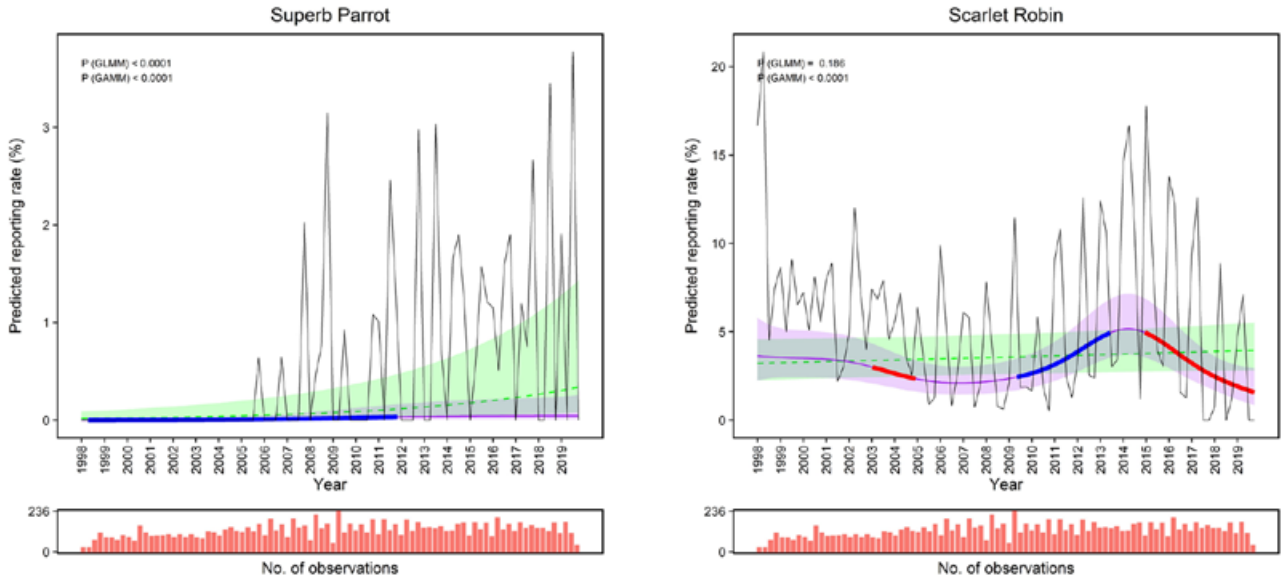

Figure 11. Species trend plots for the Superb Parrot and Scarlet Robin

3.3.7. Species of conservation concern

The result for this group (14 species) is a significant collective decrease, and the effect size is the largest of the declining bird groups (Figure 6). The 8 species with significant trends showed decreases (Table 4). A further 5 species showed large confidence intervals, reflecting low abundance/sparse data. Eleven species (including the 8 decliners) showed variable periods of change (Figure 12). However, Hooded Robin, Restless Flycatcher and Tree Martin all showed consistent declines in every year of the analysis. Five species in this group that are of interest to bird observers due either to their patchy distribution or infrequent detection are Southern Whiteface, Diamond Firetail, Double-barred Finch, Jacky Winter and Flame Robin.

Table 4. Species of concern (at risk) and their trend results. Species with significant trends are shown in bold. Asterisk (*) indicate rare species and/or sparse data.

Species	Trend
Brown Treecreeper	**Decrease**
Crested Shrike-tit*	**Decrease**
Diamond Firetail	No trend detected
Double-barred Finch	No trend detected
Flame Robin*	No trend detected
Hooded Robin	**Decrease**
Jacky Winter *	No trend detected
Mistletoebird	**Decrease**
Painted Button-quail *	No trend detected
Restless Flycatcher	**Decrease**
Scarlet Robin	No trend detected
Southern Whiteface	**Decrease**
Tree Martin	**Decrease**
White-plumed Honeyeater	**Decrease**

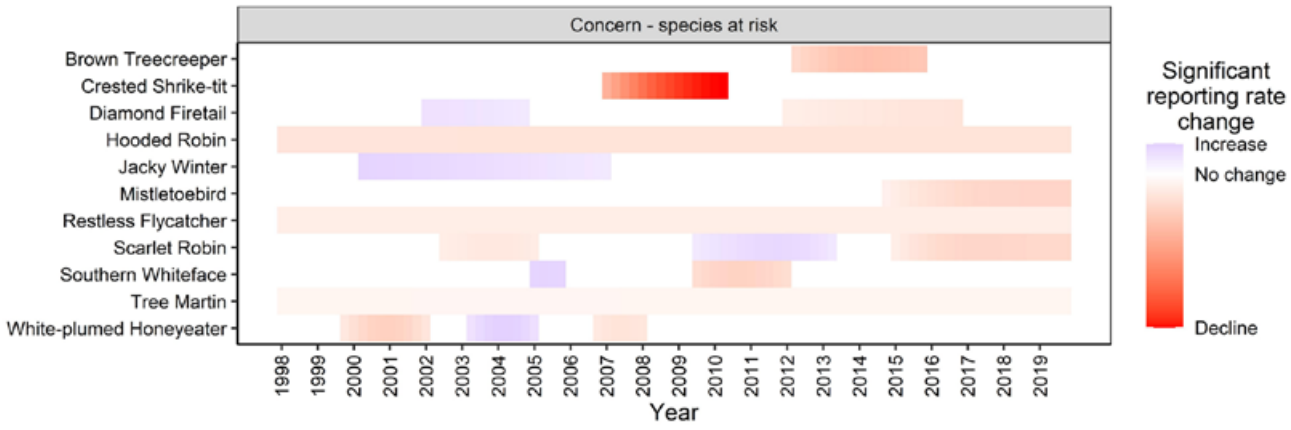

Figure 12. *ROC Plot*: Periods of significant reporting rate change for species of concern

Southern Whiteface

Southern Whiteface showed very low reporting rates, a small but significant decrease in trend overall, and a wide range in the confidence interval, reflecting the very low numbers at woodland sites. Southern Whiteface is a ground feeder, usually recorded in open sites, grassy rocky sites and woodland/grassland margins, where grasses are short or sparse. The species can be associated with woody weeds.

Diamond Firetail

Diamond Firetail showed a significant decrease from 2012 to 2017, corresponding with the end of a wetter period and start of a drier period. The species is resident and typically recorded in small groups, infrequently at particular locations, and probably moves around to take advantage of seed sources. Weed cover (e.g. in high rainfall periods), loss of favoured native grasses, or overgrazing by native herbivores may be influencing factors in their decline.

Double-barred Finch

Double-barred Finch are recorded infrequently and at only a few woodland sites. They occur in small, localised groups in some reserves and urban/rural edge sites. They have some association with woody weeds. Overgrazing and extended dry periods may be limiting their numbers and distribution.

Jacky Winter

Jacky Winter showed no detectable long-term trend. This is a seasonal species in low abundance at woodland sites. The ROC Plot (Figure 12) showed a significant rate of change (increase) during the Millennium Drought from 2000 to 2007; possibly indicating birds moving further east from drier areas inland.

Flame Robin

Flame Robin showed no detectable long-term trend. There is very low abundance of this altitudinal migrant species, occurring at a few woodland sites, generally in cooler months. Flame Robins have been identified as a species potentially affected by climate change.

3.4. Conservation Priority Species

The Project Team has compiled a list of Conservation Priority Species in ACT woodlands and assigned a management **priority ranking** (high or low) to each based on an assessment of significant statistical results collectively. The purpose of this list is to support land managers in conservation and restoration planning for woodland birds. To be considered a Conservation Priority Species, a species had to meet ALL THREE (3) of the following criteria:

1. a significant decliner (Table 2), and
2. an ACT woodland-dependent bird species (Page 4, Figure 6), and
3. (i) a consistent, every-year decliner (Page 10, Table 2)
   (ii) a canopy feeder (Page 5, Figure 6), and/or
   (iii) a small-bodied bird (Page 6, Figure 6).

The **priority ranking** takes account of these criteria and the local knowledge and expertise of the Project Team, and relates to the ACT region only.

Fuscous Honeyeater

Table 5. Species prioritised for conservation attention in the ACT due to concerning long-term status and trait-based vulnerability (i.e. small-bodied and/or canopy feeder). *Reminder: Canopy feeders feed primarily, but not exclusively, in the canopy.*

Species	RR trend	Consistent decline	Canopy feeder	Small-bodied	Priority ranking	Notes on ecology and trend pattern
Brown Treecreeper	Low-very low			✓	High	Mainly ground feeder of insects, also taken from crevices/bark in trees. Reliant on dead & fallen timber. Hollow nester. Requires well connected habitats/patches. No longer resident breeding groups at woodland sites. Rare/uncommon in rural areas.
Buff-rumped Thornbill	Medium			✓	Low	Abundant flock species in good quality woodland structures/habitats. Insect feeder, utilises lower/mid-storey and ground, including eucalypt re-growth.
Dollarbird	Low-very low	✓			High	Seasonal, long distance migrant from areas in Asia and PNG. Canopy and aerial feeder/hawker of insects. Nests in tree hollows, often dead trees.
Fuscous Honeyeater	Low		✓	✓	High	Moves seasonally between mid-altitude and lowland woodlands in ACT. Widely distributed in SE Aust. woodlands. Nectar feeder in eucalypts, shrubs. Takes insects in absence of nectar. Core ACT habitat impacted by bushfires 2003 and 2020.
Grey Currawong	Low-very low	✓			High	Widespread species. In ACT occurs in low numbers/pairs, in woodlands. Takes insects, fruits/berries, probes bark/litter. Records in COG Area of Interest (AOI) show slow decline from 1991 (Canberra Ornithologists Group).
Grey Fantail	High		✓	✓	Low	Mostly migratory/seasonal. Occurs in habitats of varying structures/quality. Insect feeder, taken in flight.
Hooded Robin	Low-very low	✓		✓	High	Mainly ground feeder of insects. Requires large, undisturbed, well connected patches/habitats. No longer resident breeding groups at woodland sites. Rare/uncommon in rural areas.
Leaden Flycatcher	Low		✓	✓	Low	Seasonal species. Occurs in good quality woodland habitats with large eucalypts, open forest. Insect feeder, in trees and in flight amongst treetops.

Table 5 — continued

Species	RR trend	Consistent decline	Canopy feeder	Small-bodied	Priority ranking	Notes on ecology and trend pattern
Little Friarbird	Very low		✓		Low	Very sparse data, marginal ACT species, core range outside ACT.
Mistletoebird	Medium		✓	✓	Low	Seasonal species, some overwinter. Feeds largely on mistletoe fruits, and insects. Tree health decline is possible factor (loss of/reduced mistletoes, herbivores/possums).
Noisy Friarbird	Medium		✓		Low	Seasonal, migratory species. Insects and nectar feeder.
Restless Flycatcher	Low	✓		✓	High	Widely distributed species. Mainly hawks insects on or from the ground. Tolerates open treed habitats, farmlands and riverine.
Rufous Whistler	Medium	✓	✓	✓	Low	Seasonal migratory species. Occurs in good quality woodland habitats with large eucalypts. Insect feeder in trees and shrubs layer.
Southern Whiteface	Low			✓	Low	Ground feeder, on insects and seeds. Some disturbance tolerance, occurs in modified habitats and farmlands. Prefers areas of short grass, patchy ground layer.
Striated Pardalote	High		✓	✓	Low	Widely distributed species. Feeds on tree frequenting insects, including lerps. Disturbance tolerant, occurs in degraded, treed habitats, and fragmented patches.
Tree Martin	Medium-low	✓		✓	High	Seasonal migrant, some overwinter. Hawks insects from or between trees and aerial. Tolerates open treed habitats. Nests in small hollows in eucalypts.
Weebill	High		✓	✓	Low	Lerp and insect feeder, mostly taken from outer foliage of eucalypts. Abundant at woodland sites of varying quality/structure. Utilises eucalypt re-growth.
Western Gerygone	Low	✓	✓	✓	Low	Seasonal migratory species. Small insect feeder, usually in outer foliage. Occurs in better quality woodland habitats. Widespread species, more common further inland.

Species	RR trend	Consistent decline	Canopy feeder	Small-bodied	Priority ranking	Notes on ecology and trend pattern
White-plumed Honeyeater	Medium-low		✓	✓	Low	Widely distributed species in woodland structures and riverine habitats. In ACT, occurs in woodlands, in treed urban/reserve edge green strips, and parklands with mature eucalypts. Insects and nectar feeder.
White-winged Triller	Medium-low		✓	✓	Low	Widespread, mobile, irruptive species, numbers vary seasonally. Insect feeder, including on or close to the ground.

Notes on some declining species in Table 2 list, not included as Conservation Priority Species in Table 5:

- Crested Shrike-tit (RR low, never a common woodland species, uncommon/rare in ACT forest habitats).
- Striated Thornbill (RR medium declining to low, flock and canopy feeding species, not woodland-dependent, also forest species).
- Pallid Cuckoo (RR low, is a consistent decliner, not woodland-dependent, is a widespread species more common inland in open habitats).

Hooded Robin

4. Conclusions & Recommendations

This report provides a timely update on the conservation status of ACT woodland birds. Here, long-term woodland bird population trends have been elicited from the Woodland Bird Monitoring Project database, and interpreted with expert knowledge, to provide an overview of desirable and undesirable changes to woodland avifauna. Further, this report serves as a support tool for conservation land managers through the identification of Conservation Priority Species that arguably require immediate protection and conservation attention. In some cases, strategic restoration planning to improve woodland condition and arrest current rates of decline may be necessary to avoid future local extirpation events. Below, the authors call attention to some key conclusions from the analysis.

The increase of large-bodied bird species, and the strong, unabated increase in aggressive species like Noisy Miner and Rainbow Lorikeet, are important considerations for the management of woodland-dependent birds. This is particularly relevant in the consideration and planning of conservation buffers — including their size and nature — at the interface of new urban and greenfield developments. It is likely that high-functioning conservation buffers will be needed to minimise the intrusion and dominance of large and/or aggressive, urban-tolerant bird species in nature reserves and rural woodland habitats.

There were three important new findings from the analysis:

1. Observed declines were expected for some species (e.g. White-plumed Honeyeater, Southern Whiteface, Crested Shrike-tit), but there were a number of additional species showing declines that were unexpected to the Project Team (e.g. Fuscous Honeyeater, Mistletoebird). Similarly, some species regarded as fairly common or abundant, showed long-term downward trends, (e.g. Rufous Whistler, Weebill, Yellow-rumped Thornbill, Superb Fairy-wren), albeit only small declines in some cases (e.g. Leaden Flycatcher, Red-rumped Parrot). Declining groups were not restricted to native species with 4 of the 7 introduced species showing a decrease (the other 3 showing no trend).
2. A significant collective decline in canopy feeders was unexpected and warrants further investigation. Possible drivers of decline in this bird group include eucalypt dieback and crown condition, climate-related habitat effects (e.g. nectar availability), and overabundant canopy browsers such as possums.
3. Before the analysis, it was thought by the Project Team that ground feeders may be the group of most concern, given previous analyses and factors such as increasing periods of drought, variable rainfall, overgrazing by native herbivores, deterioration of ground layer habitat condition and plant diversity, and weeds invasion. However, the ground feeders group did not show a collective trend.

Drought and rainfall are generally considered important determinants of bird distribution and abundance, especially for small passerines. From this analysis, it appears some species (e.g. Scarlet Robin) responded positively to increased rainfall, while others (e.g. Hooded Robin) showed no response at all. Without a more tailored analysis, it is unclear if (or how) weather affects woodland bird reporting rates. It is plausible, however, that species occurring only at low densities, or in fragmented populations, lack the means to exploit favourable weather to recover their numbers.

The analysis confirms a focus on small birds as a conservation priority is valid. There may be a need to consider the nomination of some species as *Threatened* (Vulnerable status) under the ACT *Nature Conservation Act 2014*, such as those considered 'High' priority in Table 5 (e.g. Tree Martin, Fuscous Honeyeater), or the upgrading of some species already listed as *Threatened* (e.g. Hooded Robin). Additional supporting data, or further analysis, may be needed to underpin a strong case for new threatened species nominations. In the interim, all species named as 'Conservation Priority Species' (Table 5) should form part of a formalised 'Watching Brief' that implies ongoing species assessment in terms of their population status and their habitat extent and condition.

4.1. Future work

Further exploration of results presented here could be undertaken, as well as new projects with questions arising from, or based on, this report. Future work using COG's Woodland Bird Monitoring Project database, dependent on available funding, resources and priorities, could include:

- Targeted analysis of species of interest, including species of concern, which may warrant nomination for listing as *Threatened*. This should be driven by clear questions (e.g. *What do we need to know about species X?*). Action Plans produced by the ACT Government may provide some guidance.
- Analysis of landscape-scale woodland change and associated bird responses. This could include:
  - Assessing whether habitats critical to the maintenance of woodland bird diversity are being maintained.
  - Analysing how woodland birds are responding to ongoing urbanisation and reservation (an update of Rayner *et al.* 2014).
  - Investigating the relationship between dieback and woodland bird abundance, density and distribution.
- Examination of the impacts of dominant avian competitors on vulnerable woodland birds that would include a description of Noisy Miner colonisation dynamics across the ACT.
- An analysis of site-based species occurrence and community composition, which may be influenced by site-related attributes such as patch size, patch connectivity, habitat structure, land tenure, grazing regimes, the urban context, or other geographical factors.
- A case study of Mulligans Flat and Goorooyarroo Nature Reserves where survey data are available from 1995, before/after and within/outside the predator-proof Sanctuary.
- Identification of climate-sensitive species, based on research on how woodland birds are responding to weather variability and broad-scale climate signals.

Readers are encouraged to explore reports from other woodland bird monitoring projects in the region to position the results of this project in the broader context of temperate woodland bird conservation. For example, the Cowra Woodland Birds Program (CWBP) conducts regular bird surveys in woodlands in the Cowra area north of Canberra, and 17 years of data have recently been analysed (Reid & Nicholls 2020).

Finally, this report has been prepared to enable COG to publish key results, and for the consideration of a journal article for peer-reviewed publication.

4.2. Actions Recommended

1. Continue the survey program and consider a future trend analysis.
   - Most suitable at the 30-year milestone (1998 to 2028).
   - With an emerging pattern of drier years since 2015, there is great interest and, in some cases, concern for what trends will emerge in the next decade and whether some unexpected decliners might recover. Monitoring should continue to determine when/if populations will plateau and how this influences the conservation status of species.
2. Consider publishing results presented in this report in a peer-reviewed journal to enhance the dissemination of key findings and reach a wider scientific audience.
3. Seek funding/resources to undertake further analysis or research projects with a focus on Conservation Priority Species and their primary threatening processes.

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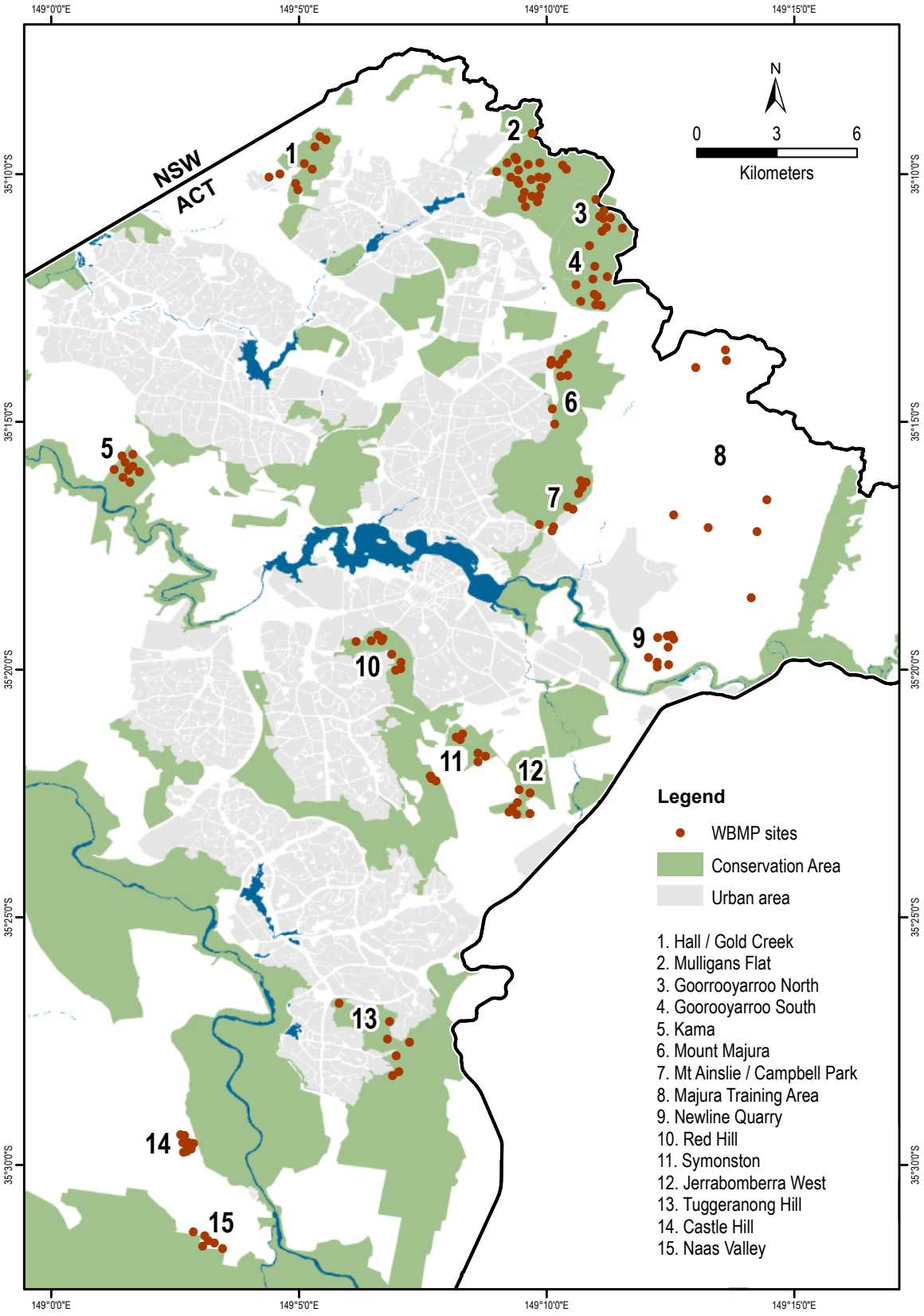
Appendices

Appendix A: Map of survey locations



Appendix B: Weather data from the analysis period

It is important to look at results presented here in the context of weather data. Rainfall data, annual and monthly actual and long-term annual averages, were obtained from the Bureau of Meteorology website, 'Climate Data Online' (www.bom.gov.au/climate), for the Canberra Airport weather station.

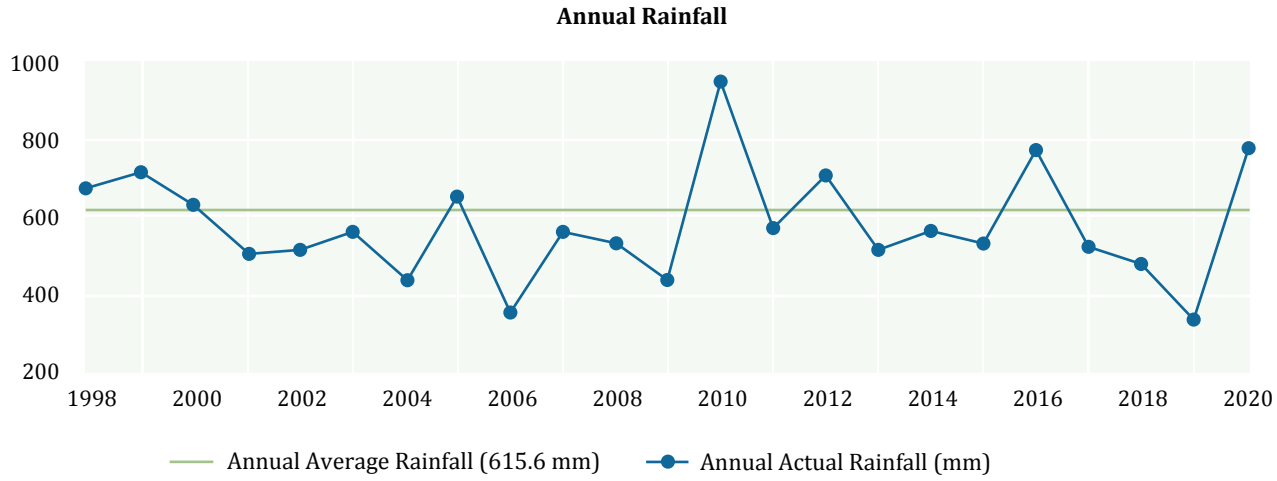


Figure B1. Total annual rainfall and average annual rainfall over the analysis period

The years 1998 to 2008 are within a period generally referred to as the Millennium Drought, notable by below average annual rainfall in most years. Note that four of the years in the data analysed from 1998 to 2008 had annual rainfall well below the average (2001, 2002, 2004, 2006). In that period, there was also more variable rainfall in the spring months (compared with the long-term averages). Spring rainfall is a factor that influences successful bird breeding; indeed the spring or warmer months are critical breeding months for most birds.

The year 2005 had slightly more rainfall than average, after four years of below average rainfall, but notably, 2005 had a third more than the average spring rainfall. COG's Ten-Year Analysis showed some increase in abundance, of both resident and migrant species, as a result of that good spring rainfall event within a generally drier decade (Bounds *et al*. 2010).

Annual rainfall was above average, or close to the average, from 2010 to 2016, but has been below average over the following years, 2017 to 2019. Rainfall on a monthly or seasonal basis has also been quite variable over the years 2010 to 2019.

- 2010 had a third more annual rainfall than the annual average total.
- The equivalent of a year's average rainfall occurred over six months from September 2010 to February 2011, peaking in December 2010 at 198 mm for that month — these are months important for bird breeding.
- The five months from November 2011 to March 2012 had a significant increase in total rainfall, almost equivalent to the average annual rainfall, peaking in the month of March at 197 mm.
- 2012 had above average annual rainfall, 693 mm; spring rainfall was below average and variable.
- 2013, 2014 and 2015 had annual rainfall below average (533 mm, 568 mm and 545 mm respectively — compared with average 616 mm), and rainfall in the warmer months was variable.
- Over two months December 2014 and January 2015, rainfall total was significant at 199 mm, almost a third of a year's average rainfall for these two months.
- 2016 had significantly higher total annual rainfall at 788 mm, with the spring months September to December recording significantly more than the usual averages.
- 2017, 2018 and 2019 had significantly below average rainfall, with 2019 (358 mm) the driest year since 1982, with only 58% of the long-term average rainfall (616 mm).

Although not in the analysis period, 2020 had annual rainfall significantly higher than average at 790 mm, a similar level to 2016. Monthly rainfall was quite variable in 2020, most of the late autumn/early winter months were drier than average, but the four late winter/spring months (August to November) had very good rainfall of just over 50% of the annual total rainfall.

It is noted that in the last few months of 2019 and first month of 2020, following a widespread drought period across eastern Australia, there were catastrophic bushfires, particularly in forests all along the Great Dividing Range and the south coast of NSW. These very significant climate-related events occurred just after the analysis period and would be relevant to future analysis.

- In the SE region, the Tallaganda National Park to the SE of the ACT, rural lands around Braidwood, large areas of national parks to the east and forests as far as the sea were burnt.
- In the ACT, 80% of the Namadgi NP was burnt, much of it severely, but most of the higher, western edge of the park, mainly wet forests and alpine habitat, was not burnt.
- Much of the high country to the south and SW of the ACT was also burnt.
- Most of one woodland location, Newline, was burnt in a local outbreak.

Noisy Friarbird

Appendix C: Allocation of species to bird groups

The table below shows for each species their inclusion (1) or exclusion (0) from the 12 bird groups of interest, as identified by the Project Team.

Species	ACT woodland-dependent	South-east woodland bird	Degraded community bird	Threatened species	Conservation concern	Canopy feeders	Mid-storey feeders	Ground feeders	Woody weed frequent	Woody weed + small	Body mass: small	Body mass: large
Australasian Pipit	0	0	0	0	0	0	0	0	0	0	1	0
Australian Hobby	0	0	0	0	0	0	0	0	0	0	0	1
Australian King-Parrot	0	0	0	0	0	0	0	0	1	0	0	1
Australian Magpie	0	0	1	0	0	0	0	0	0	0	0	1
Australian Owlet-nightjar	1	1	0	0	0	0	0	1	0	0	1	0
Australian Raven	0	0	1	0	0	0	0	0	0	0	0	1
Black-faced Cuckoo-shrike	0	0	0	0	0	0	0	0	0	0	0	1
Black-shouldered Kite	0	0	0	0	0	0	0	0	0	0	0	1
Brown Falcon	0	0	0	0	0	0	0	0	0	0	0	1
Brown Goshawk	0	0	0	0	0	0	0	0	0	0	0	1
Brown Quail	0	0	0	0	0	0	0	0	0	0	0	1
Brown Songlark	0	0	0	0	0	0	0	0	0	0	1	0
Brown Thornbill	0	1	0	0	0	0	0	0	0	0	1	0
Brown Treecreeper	1	1	0	1	1	0	0	1	0	0	1	0
Brown-headed Honeyeater	1	1	0	0	0	1	0	0	0	0	1	0
Brush Cuckoo	0	0	0	0	0	0	0	0	0	0	1	0
Buff-rumped Thornbill	1	1	0	0	0	0	1	0	0	0	1	0
Cicadabird	0	0	0	0	0	0	0	0	0	0	1	0
Collared Sparrowhawk	0	0	0	0	0	0	0	0	0	0	0	1
Common Blackbird	0	0	0	0	0	0	0	0	0	0	0	1
Common Bronzewing	0	1	0	0	0	0	0	0	0	0	0	1
Common Myna	0	0	0	0	0	0	0	0	0	0	0	1
Common Starling	0	0	0	0	0	0	0	0	0	0	0	1
Crested Pigeon	0	0	0	0	0	0	0	0	0	0	0	1
Crested Shrike-tit	0	1	0	0	1	0	0	0	0	0	1	0
Crimson Rosella	0	0	0	0	0	0	0	0	0	0	0	1
Diamond Firetail	1	1	0	0	1	0	0	1	1	1	1	0
Dollarbird	1	0	0	0	0	0	0	0	0	0	0	1
Double-barred Finch	1	0	0	0	1	0	0	1	1	1	1	0

Species	ACT woodland-dependent	South-east woodland bird	Degraded community bird	Threatened species	Conservation concern	Canopy feeders	Mid-storey feeders	Ground feeders	Woody weed frequent	Woody weed + small	Body mass: small	Body mass: large
Dusky Woodswallow	1	1	0	0	0	0	1	0	0	0	1	0
Eastern Rosella	0	0	1	0	0	0	0	0	0	0	0	1
Eastern Spinebill	0	1	0	0	0	0	0	0	0	0	1	0
Eastern Yellow Robin	0	1	0	0	0	0	0	0	0	0	1	0
European Goldfinch	0	0	0	0	0	0	0	0	1	1	1	0
European Greenfinch	0	0	0	0	0	0	0	0	1	1	1	0
Fairy Martin	0	0	0	0	0	0	0	0	0	0	1	0
Fan-tailed Cuckoo	0	0	0	0	0	0	0	0	0	0	1	0
Flame Robin	1	0	0	0	1	0	0	1	0	0	1	0
Fuscous Honeyeater	1	1	0	0	0	1	0	0	0	0	1	0
Galah	0	0	1	0	0	0	0	0	1	0	0	1
Gang-gang Cockatoo	0	0	0	0	0	0	0	0	1	0	0	1
Golden Whistler	0	1	0	0	0	0	0	0	0	0	1	0
Grey Butcherbird	1	0	1	0	0	0	0	0	0	0	0	1
Grey Currawong	1	0	0	0	0	0	0	0	0	0	0	1
Grey Fantail	1	1	0	0	0	1	0	0	0	0	1	0
Grey Shrike-thrush	1	1	0	0	0	0	0	0	0	0	1	0
Hooded Robin	1	1	0	1	1	0	0	1	0	0	1	0
Horsfield's Bronze-Cuckoo	1	1	0	0	0	1	0	0	0	0	1	0
House Sparrow	0	0	0	0	0	0	0	0	0	0	1	0
Jacky Winter	1	1	0	0	1	0	0	1	0	0	1	0
Laughing Kookaburra	0	0	0	0	0	0	0	0	0	0	0	1
Leaden Flycatcher	1	1	0	0	0	1	0	0	0	0	1	0
Little Corella	0	0	0	0	0	0	0	0	1	0	0	1
Little Eagle	0	0	0	1	0	0	0	0	0	0	0	1
Little Friarbird	1	1	0	0	0	1	0	0	0	0	1	0
Little Lorikeet	1	1	0	0	0	1	0	0	0	0	1	0
Little Raven	0	0	0	0	0	0	0	0	0	0	0	1
Long-billed Corella	0	0	0	0	0	0	0	0	0	0	0	1
Magpie-lark	0	0	1	0	0	0	0	0	0	0	0	1
Masked Lapwing	0	0	0	0	0	0	0	0	0	0	0	1
Masked Woodswallow	1	0	0	0	0	1	0	0	0	0	1	0
Mistletoebird	1	1	0	0	1	1	0	0	0	0	1	0

Species	ACT woodland-dependent	South-east woodland bird	Degraded community bird	Threatened species	Conservation concern	Canopy feeders	Mid-storey feeders	Ground feeders	Woody weed frequent	Woody weed + small	Body mass: small	Body mass: large
Musk Lorikeet	0	0	0	0	0	0	0	0	0	0	1	0
Nankeen Kestrel	0	0	0	0	0	0	0	0	0	0	0	1
Noisy Friarbird	1	1	0	0	0	1	0	0	0	0	0	1
Noisy Miner	1	0	1	0	0	1	0	0	0	0	0	1
Olive-backed Oriole	1	1	0	0	0	1	0	0	0	0	0	1
Painted Button-quail	1	1	0	0	1	0	0	1	0	0	0	1
Pallid Cuckoo	0	0	0	0	0	0	0	0	0	0	0	1
Peaceful Dove	0	0	0	0	0	0	0	0	0	0	1	0
Peregrine Falcon	0	0	0	0	0	0	0	0	0	0	0	1
Pied Butcherbird	0	0	1	0	0	0	0	0	0	0	0	1
Pied Currawong	0	0	0	0	0	0	0	0	1	0	0	1
Rainbow Bee-eater	0	0	0	1	0	0	0	0	0	0	1	0
Rainbow Lorikeet	0	0	0	0	0	0	0	0	0	0	0	1
Red Wattlebird	0	0	0	0	0	0	0	0	0	0	0	1
Red-browed Finch	0	1	0	0	0	0	0	0	1	1	1	0
Red-capped Robin	1	1	0	0	0	0	0	1	0	0	1	0
Red-rumped Parrot	0	0	0	0	0	0	0	0	1	1	1	0
Restless Flycatcher	1	1	0	0	1	0	0	1	0	0	1	0
Rock Dove	0	0	0	0	0	0	0	0	0	0	0	1
Rose Robin	0	0	0	0	0	0	0	0	0	0	1	0
Rufous Fantail	0	0	0	0	0	0	0	0	0	0	1	0
Rufous Songlark	1	0	0	0	0	0	0	1	0	0	1	0
Rufous Whistler	1	1	0	0	0	1	0	0	0	0	1	0
Sacred Kingfisher	1	1	0	0	0	0	0	1	0	0	1	0
Satin Bowerbird	0	0	0	0	0	0	0	0	0	0	0	1
Satin Flycatcher	0	0	0	0	0	0	0	0	0	0	1	0
Scarlet Robin	1	1	0	1	1	0	0	1	0	0	1	0
Shining Bronze-Cuckoo	1	1	0	0	0	1	0	0	0	0	1	0
Silvereye	0	0	0	0	0	0	0	0	1	1	1	0
Southern Boobook	0	1	0	0	0	0	0	0	0	0	0	1
Southern Whiteface	1	1	0	0	1	0	0	1	1	1	1	0
Speckled Warbler	1	1	0	0	0	0	0	1	1	1	1	0
Spotted Pardalote	0	1	0	0	0	0	0	0	0	0	1	0

Species	ACT woodland-dependent	South-east woodland bird	Degraded community bird	Threatened species	Conservation concern	Canopy feeders	Mid-storey feeders	Ground feeders	Woody weed frequent	Woody weed + small	Body mass: small	Body mass: large
Striated Pardalote	1	0	0	0	0	1	0	0	0	0	1	0
Striated Thornbill	0	1	0	0	0	1	0	0	0	0	1	0
Stubble Quail	0	0	0	0	0	0	0	0	0	0	0	1
Sulphur-crested Cockatoo	0	0	1	0	0	0	0	0	0	0	0	1
Superb Fairy-wren	0	0	0	0	0	0	0	0	1	1	1	0
Superb Parrot	1	0	0	1	0	0	0	1	0	0	0	1
Swift Parrot	1	1	0	1	0	1	0	0	0	0	1	0
Tawny Frogmouth	0	0	0	0	0	0	0	0	0	0	0	1
Tree Martin	1	0	0	0	1	0	0	0	0	0	1	0
Varied Sittella	1	1	0	1	0	1	0	0	0	0	1	0
Wedge-tailed Eagle	0	0	0	0	0	0	0	0	0	0	0	1
Weebill	1	1	0	0	0	1	0	0	0	0	1	0
Welcome Swallow	0	0	0	0	0	0	0	0	0	0	1	0
Western Gerygone	1	1	0	0	0	1	0	0	0	0	1	0
Whistling Kite	0	0	0	0	0	0	0	0	0	0	0	1
White-browed Scrubwren	0	1	0	0	0	0	0	0	1	1	1	0
White-browed Woodswallow	1	0	0	0	0	1	0	0	0	0	1	0
White-eared Honeyeater	0	1	0	0	0	0	0	0	0	0	1	0
White-naped Honeyeater	0	1	0	0	0	0	0	0	0	0	1	0
White-plumed Honeyeater	1	0	0	0	1	1	0	0	0	0	1	0
White-throated Gerygone	1	1	0	0	0	1	0	0	0	0	1	0
White-throated Needletail	0	0	0	0	0	0	0	0	0	0	0	1
White-throated Treecreeper	0	1	0	0	0	0	0	0	0	0	1	0
White-winged Chough	1	0	0	0	0	0	0	1	0	0	0	1
White-winged Triller	1	1	0	1	0	1	0	0	0	0	1	0
Willie Wagtail	0	0	0	0	0	0	0	0	0	0	1	0
Yellow Thornbill	1	1	0	0	0	0	1	0	0	0	1	0
Yellow-faced Honeyeater	0	1	0	0	0	0	0	0	0	0	1	0
Yellow-rumped Thornbill	0	0	1	0	0	0	0	0	0	0	1	0
Yellow-tailed Black-Cockatoo	0	0	0	0	0	0	0	0	0	0	0	1

Appendix D: Distribution of surveys in the Woodland Bird Monitoring Project

The table below shows the distribution of survey effort during the observation period (1998–2019) of the COG Woodland Bird Monitoring Project. Each cell shows the number of surveys undertaken per year; lighter green indicates fewer surveys, darker green indicates more surveys.

Site ID	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	Total
CAS.101	3	7	7	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	92
CAS.102	3	7	7	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	92
CAS.103	3	7	7	4	4	3	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	3	90
CAS.104	3	7	7	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	3	91
CAS.105	3	7	7	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	92
CAS.106	3	7	7	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	92
CAS.107	3	7	7	4	4	4	4	4	3	4	4	4	4	4	4	4	4	4	4	4	3	3	90
CAS.108	3	7	7	4	4	4	4	4	3	4	4	4	4	4	4	4	4	4	4	4	3	3	90
CAS.109	3	7	7	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	92
CMP.1201						3	4	4	4	4	4	3	4	4	4	4	4	4	4	4	4	3	65
CMP.1202						3	4	4	4	4	4	3	4	4	4	4	4	4	4	4	4	3	65
CMP.1203						3	4	4	4	4	4	3	4	3	4	4	4	4	4	4	4	3	64
CMP.1204						3	4	4	4	4	4	3	4	4	4	4	4	4	4	4	4	3	65
CMP.1205						3	4	4	4	4	4	3	4	4	4	4	4	4	4	4	4	3	65
CMP.1206						3	3	4	4	4	4	3	4	3	4	4	4	4	4	4	4	3	63
CMP.1207						3	4	4	4	4	4	3	4	3	4	4	4	4	4	4	4	3	64
CMP.1208						3	4	4	4	4	4	3	4	3	4	4	4	4	4	4	4	3	64
CMP.1209						3	4	4	4	4	4	3	4	3	4	4	4	4	4	4	4	3	64
GOO.201	2	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	2	4	4	4	3	84
GOO.202	2	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	2	4	4	4	3	84
GOO.203	2	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	2	4	4	4	3	84
GOO.204	2	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	2	4	4	4	3	84
GOO.205	2	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	2	4	4	4	3	84
GOO.206	2	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	2	4	4	4	3	84
GOO.207	2	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	2	4	4	4	3	84
GOO.208	2	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	2	4	4	4	3	84
GOO.209	2	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	2	4	4	4	3	84
GOS.1301							4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	63
GOS.1302							4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	63
GOS.1303							4	4	4	4	4	3	4	4	4	4	4	4	4	4	4	3	62
GOS.1304							4	4	4	4	4	3	4	4	4	4	4	4	4	4	4	3	62

Site ID	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	Total
GOS.1305							4	4	4	4	4	3	4	4	4	4	4	4	4	4	4	3	62
GOS.1306							4	4	4	4	4	3	4	4	4	4	4	4	4	4	4	3	62
GOS.1307							4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	63
GOS.1308							4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	63
GOS.1309							4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	63
HAL.1101			3	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	78
HAL.1102			3	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	78
HAL.1103			3	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	2	76
HAL.1104			3	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	2	76
HAL.1105			3	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	2	76
HAL.1106			3	4	4	3	3	4	4	4	4	4	4	4	4	4	4	4	4	4	3	2	74
HAL.1107			3	4	4	4	3	4	4	4	4	4	4	4	4	4	4	4	4	4	3	2	75
HAL.1108			3	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	2	76
HAL.1109			3	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	2	76
JER.1601									1	4	4	4	4	4	4	4	4	4	4	4	4	3	52
JER.1602									1	4	4	4	4	4	4	4	4	4	4	4	4	3	52
JER.1603									1	4	4	4	4	4	4	4	4	4	4	4	4	3	52
JER.1604									1	4	4	4	4	4	4	4	4	4	4	4	4	3	52
JER.1605									1	4	4	4	4	4	4	4	4	4	4	4	4	3	52
JER.1606									1	4	4	4	4	4	4	4	4	4	4	4	4	3	52
JER.1607									1	4	4	4	4	4	4	4	4	4	4	4	4	3	52
KAM.1501								1	4	4	4	4	4	4	4	4	4	4	4	4	4	3	56
KAM.1502								1	4	4	4	4	4	4	4	4	4	4	4	4	4	3	56
KAM.1503								1	4	4	4	4	4	4	4	4	4	4	4	4	4	3	56
KAM.1504								1	4	4	4	4	4	4	4	4	4	4	4	4	4	3	56
KAM.1505								1	4	4	4	4	4	4	4	4	4	4	4	4	4	3	56
KAM.1506								1	4	4	4	4	4	4	4	4	4	4	4	4	4	3	56
KAM.1507								1	4	4	4	4	4	4	4	4	4	4	4	4	4	3	56
KAM.1508								1	4	4	4	4	4	4	4	4	4	4	4	4	4	3	56
KAM.1509								1	4	4	4	4	4	4	4	4	4	4	4	4	4	3	56
MAJ.301	2	4	4	4	3	3	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	2	83
MAJ.302	2	4	4	4	3	3	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	84
MAJ.303	2	4	4	4	3	2	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	83
MAJ.304	2	4	4	4	3	3	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	84
MAJ.305	2	4	4	4	3	3	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	83
MAJ.306	2	4	4	4	3	3	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	84
MAJ.307	2	4	4	4	3	3	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	84

Site ID	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	Total
MAJ.308	2	4	4	4	3	3	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	84
MAJ.309	2	4	4	4	3	3	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	84
MJF.401	1	3	4				1	5	4	4	4	4	4	4	4	4	4	4	4	4	4	3	69
MJF.402	1	3	4				1	5	4	4	4	4	4	4	4	4	4	4	4	4	4	3	69
MJF.403	1	3	4				1	5	4	4	4	4	4	3	4	4	4	4	4	4	4	3	68
MJF.404	1	3	4				1	5	4	4	4	4	4	3	4	4	4	4	4	4	4	3	68
MJF.407	1	3	4				1	5	4	4	4	4	4	4	4	4	4	4	4	4	4	3	69
MJF.408	1	3	4				1	5	4	4	4	4	4	4	4	4	4	4	4	4	4	3	69
MJF.409	1	3	4				1	5	4	4	4	4	4	4	4	4	4	4	3	4	4	3	68
MJF.410	1	2	4				1	5	4	4	4	4	4	4	4	4	4	4	4	4	2		63
MUL.501	5	4	4	4	4	4	4	4	4	3	4	4	4	4	4	4	4	4	4	4	4	3	87
MUL.502	5	3	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	87
MUL.503	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	88
MUL.504	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	88
MUL.505	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	88
MUL.506	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	88
MUL.507	5	4	4	4	4	4	4	4	4	3	4	4	4	4	4	4	4	4	4	4	4	3	87
MUL.508	5	2	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	86
MUL.509	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	88
MUL.510	5	4	4	3	4	4	4	4	4	3	4	4	4	4	4	4	4	4	4	4	4	3	86
MUL.511	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	88
MUL.512	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	88
MUL.513	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	88
MUL.514	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	88
MUL.515	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	88
MUL.516	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	88
MUL.517	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	88
MUL.518	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	88
MUL.519	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	88
MUL.520	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	88
MUL.521	5	3	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	87
MUL.522	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	88
MUL.523	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	88
MUL.524	5	4	4	4	4	4	4	3	4	4	4	4	4	4	4	4	4	4	4	4	4	3	87
NAS.1401							3	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	62
NAS.1402							3	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	62
NAS.1403							3	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	62

Site ID	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	Total
NAS.1404							3	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	62
NAS.1405							3	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	62
NAS.1406							3	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	62
NLQ.801			3	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	78
NLQ.802			3	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	78
NLQ.803			3	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	78
NLQ.804			3	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	78
NLQ.805			3	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	78
NLQ.806			3	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	78
NLQ.807			3	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	78
NLQ.808			3	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	78
NLQ.809			3	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	78
RED.601	2	4	4	4	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	86
RED.602	2	4	4	4	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	86
RED.603	2	4	4	4	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	86
RED.604	2	4	4	4	5	3	2	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	83
RED.605	2	4	4	4	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	86
RED.606	2	4	4	4	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	86
RED.607	2	4	4	4	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	86
RED.608	2	4	4	4	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	86
RED.609	2	4	4	4	5	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	86
SYM.701	2	4	4	5	4	4	4	4	5	4	4	4	4	4	4	4	4	4	4	4	4	3	87
SYM.702	2	4	4	5	4	4	4	4	5	4	4	4	4	4	4	4	4	4	4	4	4	3	87
SYM.703	2	4	4	5	4	4	4	4	5	4	4	4	4	4	4	4	4	4	4	4	4	3	87
SYM.704	2	4	4	5	4	4	4	4	5	3	3	3	4	4	2		2	4	4	4	3	3	75
SYM.705	2	4	4	5	4	4	4	4	5	3	3	3	4	4	2		2	4	4	4	3	3	75
SYM.706	2	4	4	5	4	3	4	4	5	1	3	3	4	4	2		2	4	4	4	3	3	72
SYM.713							3	4	5	3	4	4	4	4	4	4	4	4	4	4	4	3	62
SYM.714							3	4	5	3	4	4	4	4	4	4	4	4	4	4	4	3	62
SYM.715							3	4	5	3	4	4	4	4	4	4	4	4	4	4	4	3	62
TUG.1001			3	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	78
TUG.1002			3	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	78
TUG.1003			3	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	78
TUG.1004			3	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	3	77
TUG.1005			3	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	4	4	4	3	77
TUG.1006			3	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	3	78
TUG.1007			3	4	4	4	4	4	4	4	4	4	4	4	4	4	3	4	4	4	4	3	77

Appendix E: Individual species trend plots

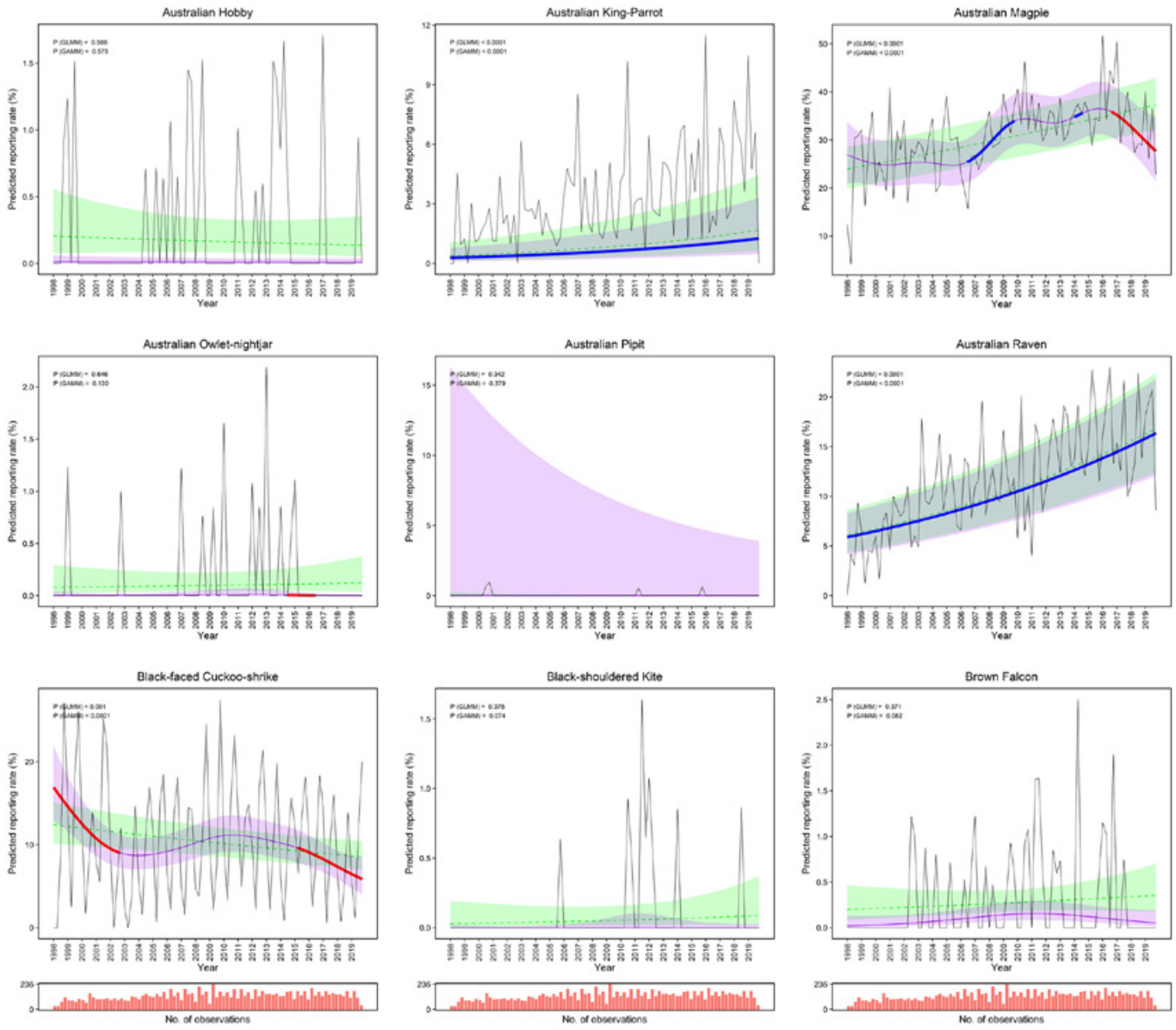
Trends in detections for individual species through time.

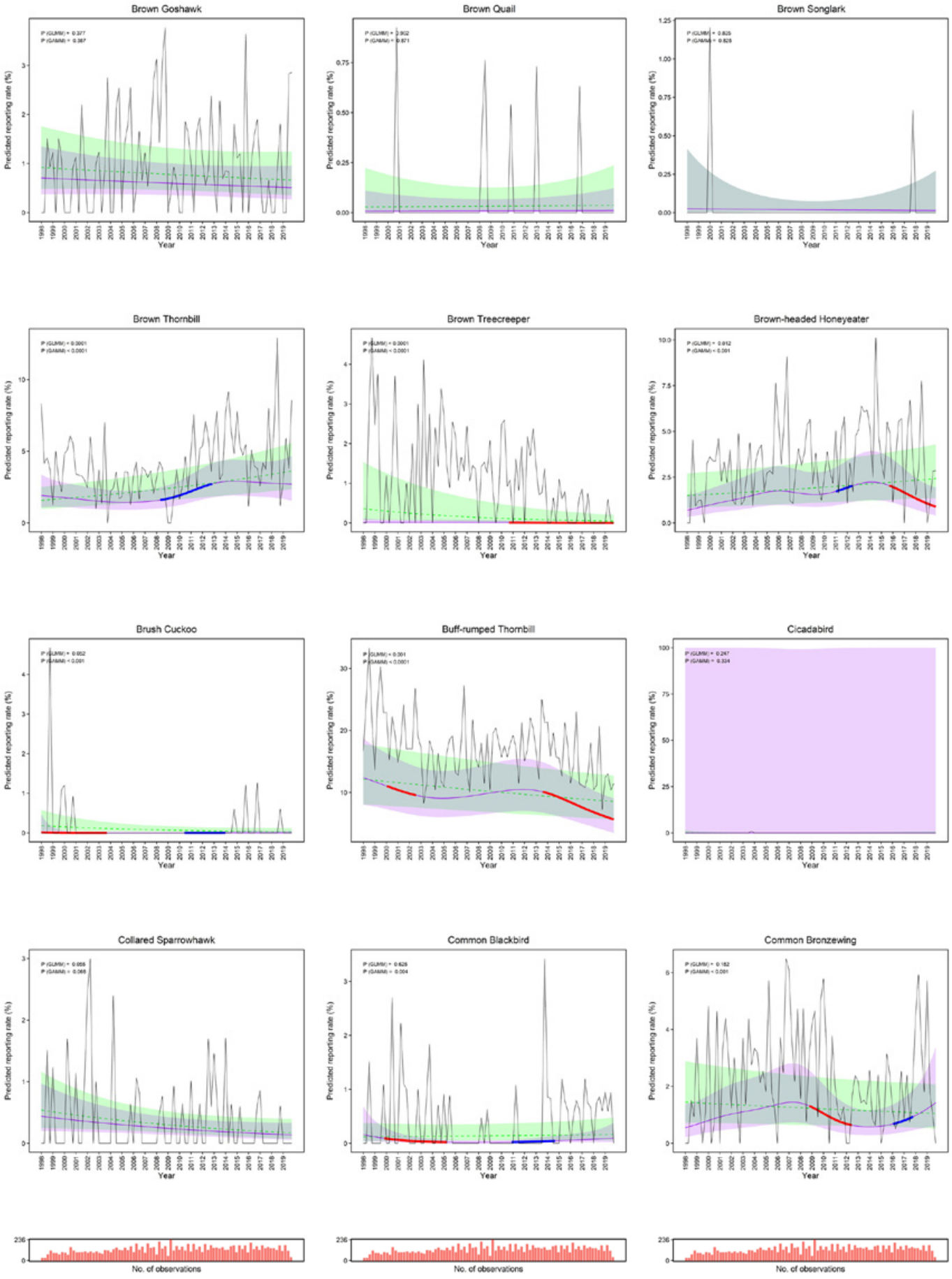
Interpreting Trend Plots

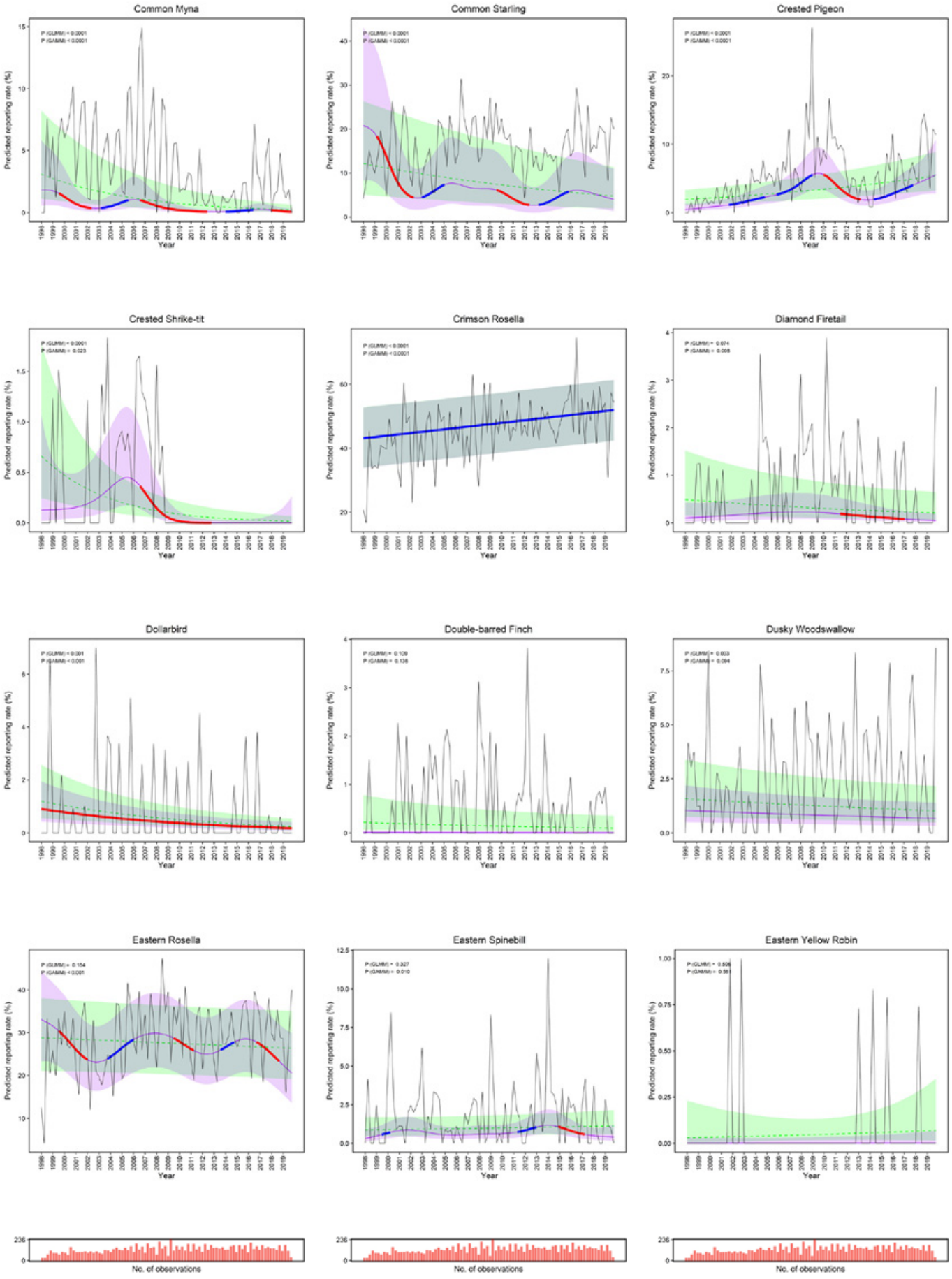
Plots for individual species show both non-linear and linear trends with associated confidence intervals. The percent scale (y-axis) should be interpreted as the mean percent of surveys in which a species is predicted to be present. Plot elements should be interpreted as follows:

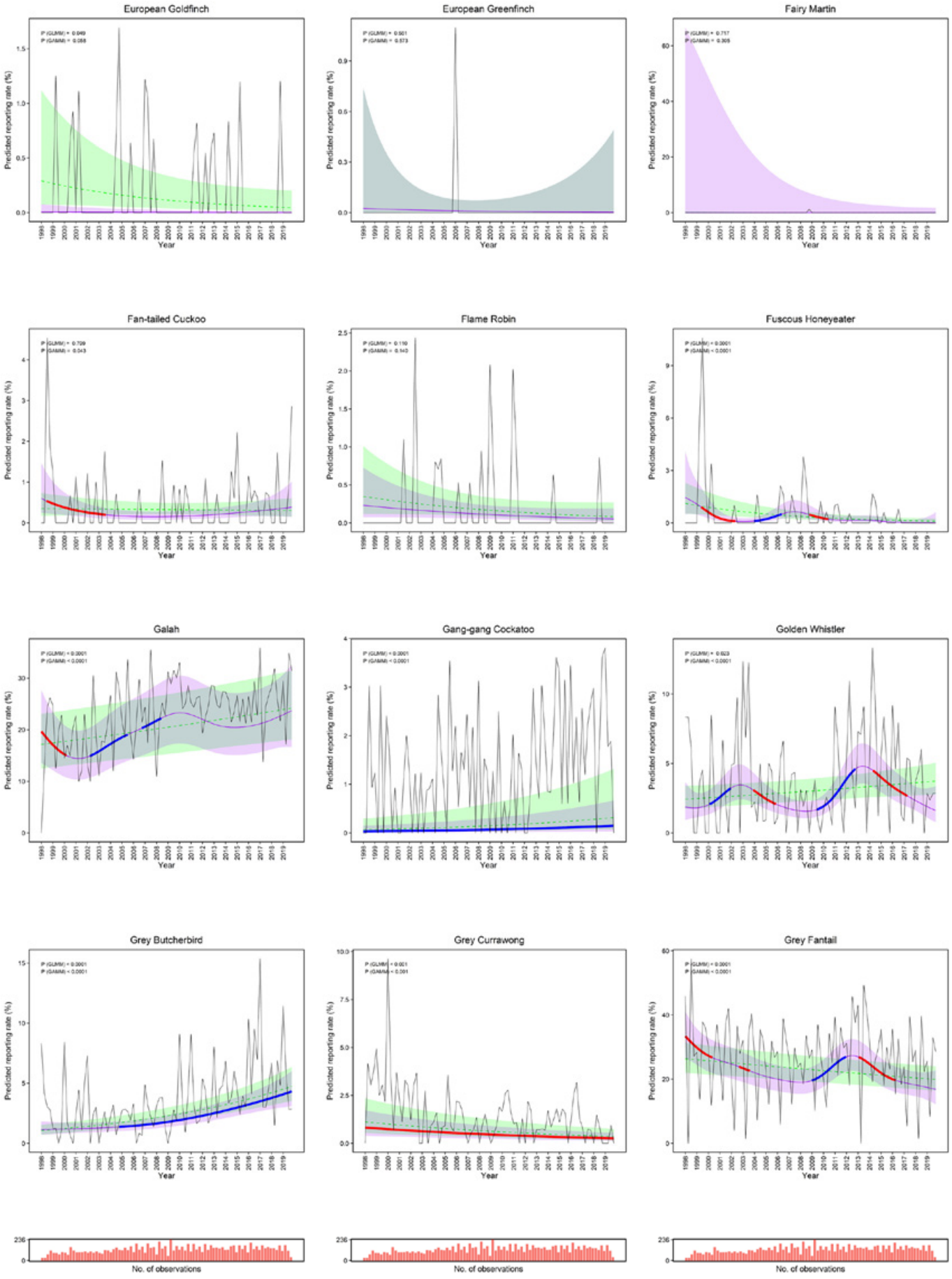
1. mean reporting rate in each quarter-yearly survey (thin black line)
2. smoothed *GAM/M* trend line showing short-term change in reporting rate (purple line)
3. 95% confidence intervals of the smoothed trend (purple shading)
4. sections of significant rate of change in the smoothed fit (positive is blue, negative is red)
5. predicted straight *GLM/M* trend line fit (green dashed line)
6. 95% confidence intervals of the linear trend (green shade)
7. P-values of the *GAMM* and *GLMM* indicating trend significance ($P < 0.05$ is significant), and
8. a representation of survey effort in each quarter-yearly survey (red bars below main plot)

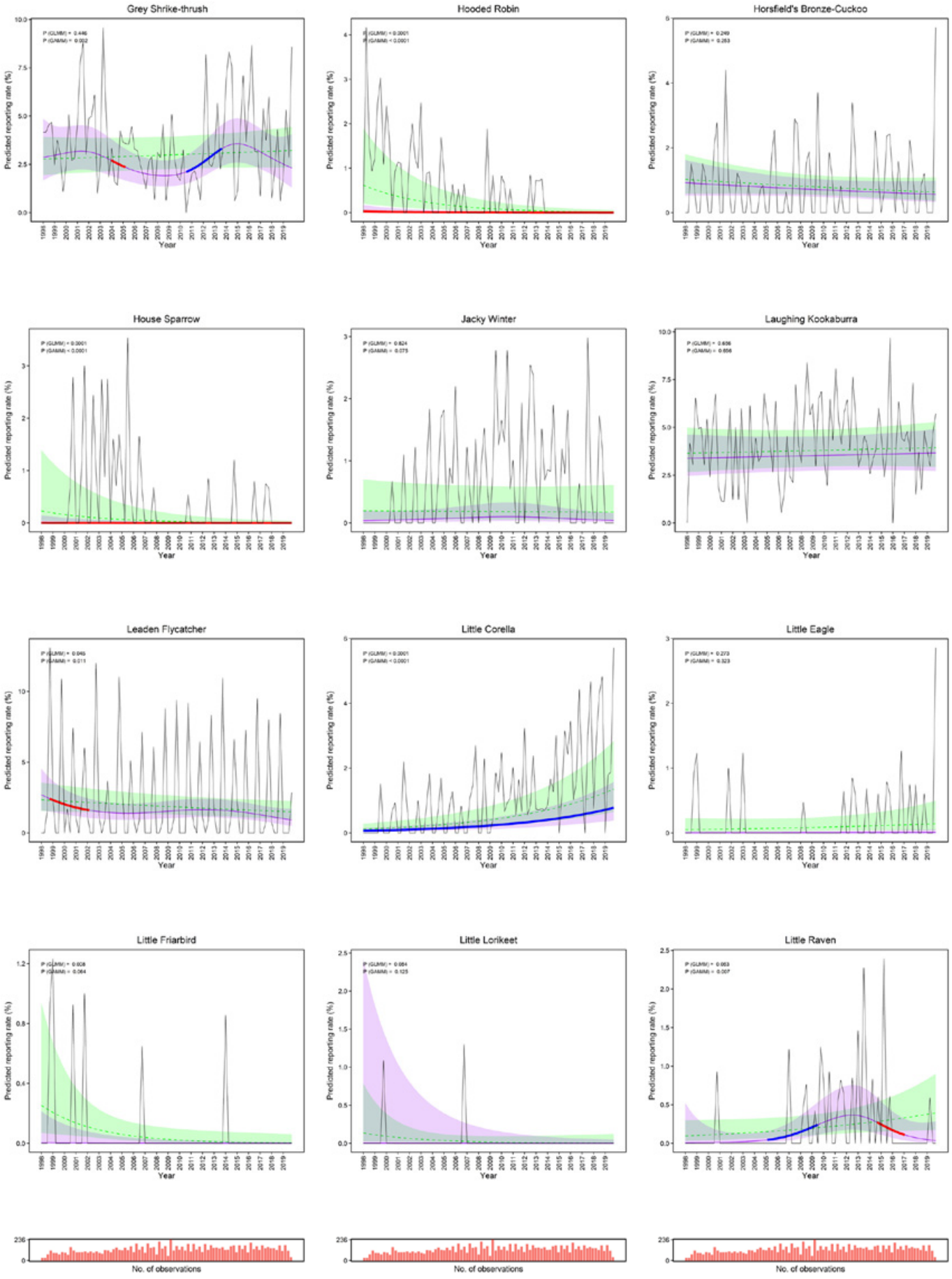
Note: Four rare species shown in Appendix E (Regent Honeyeater, Scarlet Honeyeater, White-bellied Cuckoo-Shrike, Zebra Finch), are not included in Appendix C (groups analysis); because of their rarity it was not possible/appropriate to assign to groups.

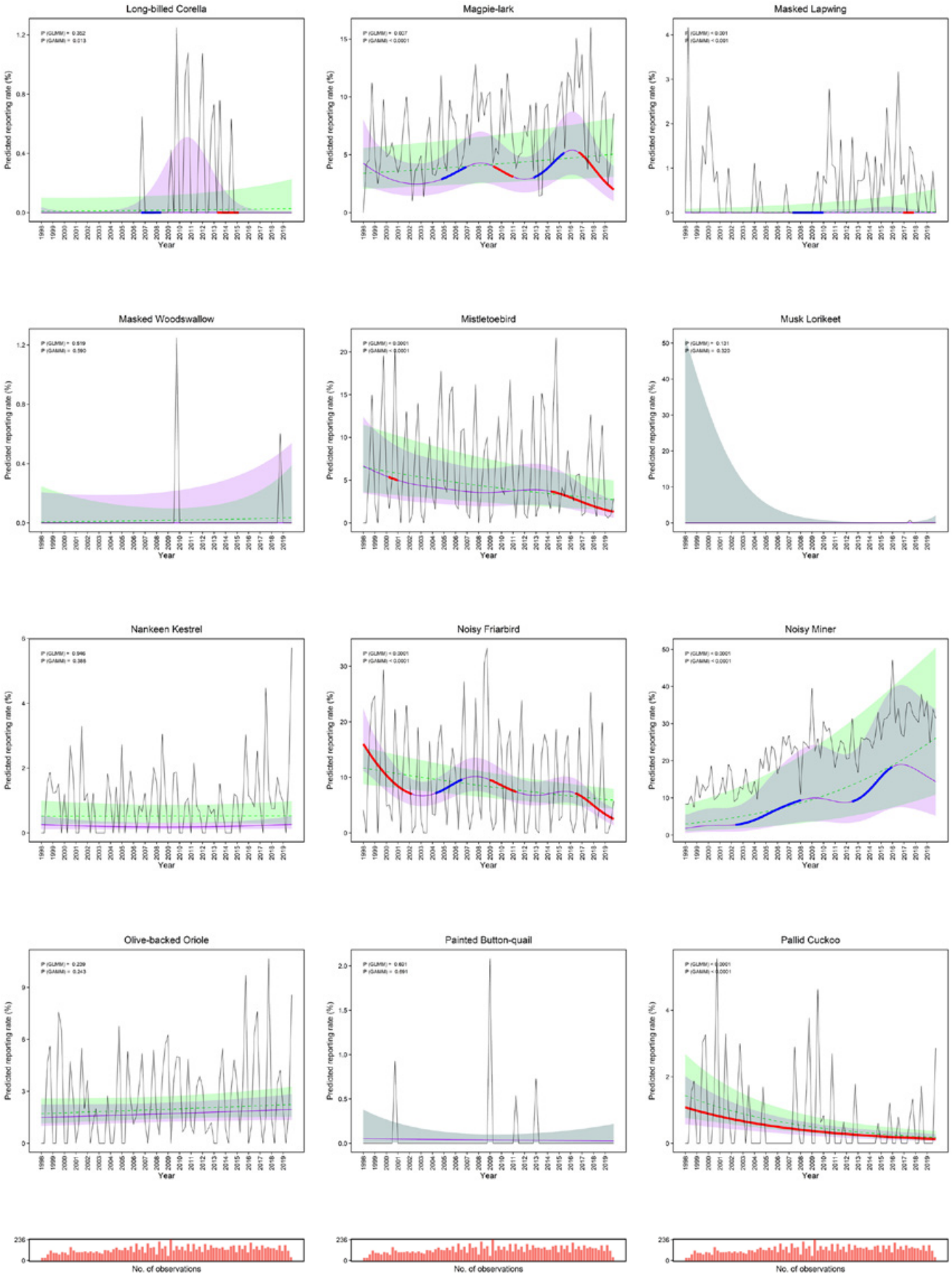


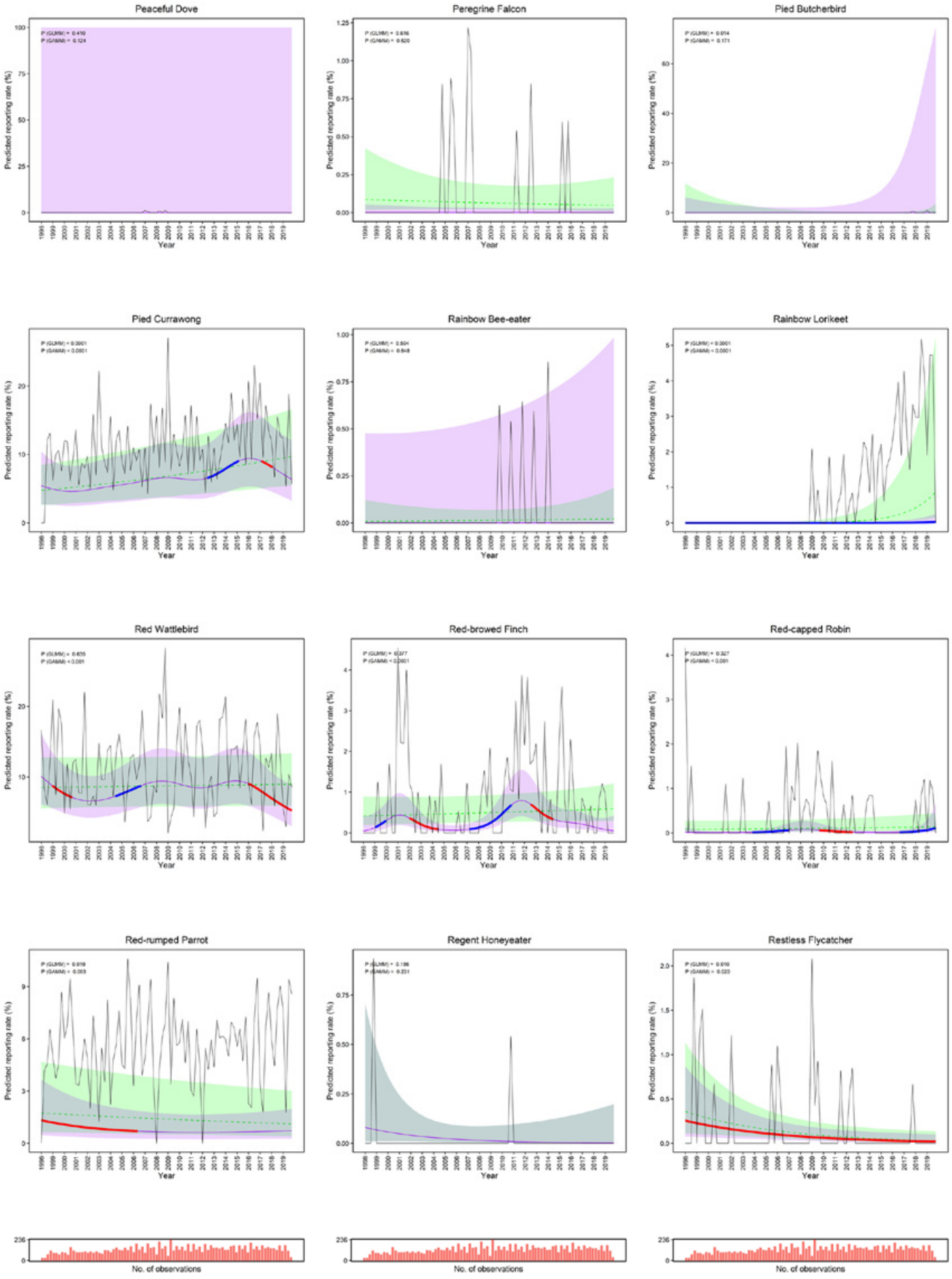

Brown Goshawk
Brown Quail
Brown Songlark
Brown Thornbill
Brown Treecreeper
Brown-headed Honeyeater
Brush Cuckoo
Buff-rumped Thornbill
Cicadabird
Collared Sparrowhawk
Common Blackbird
Common Bronzewing
Predicted reporting rate (%)
Year
No. of observations

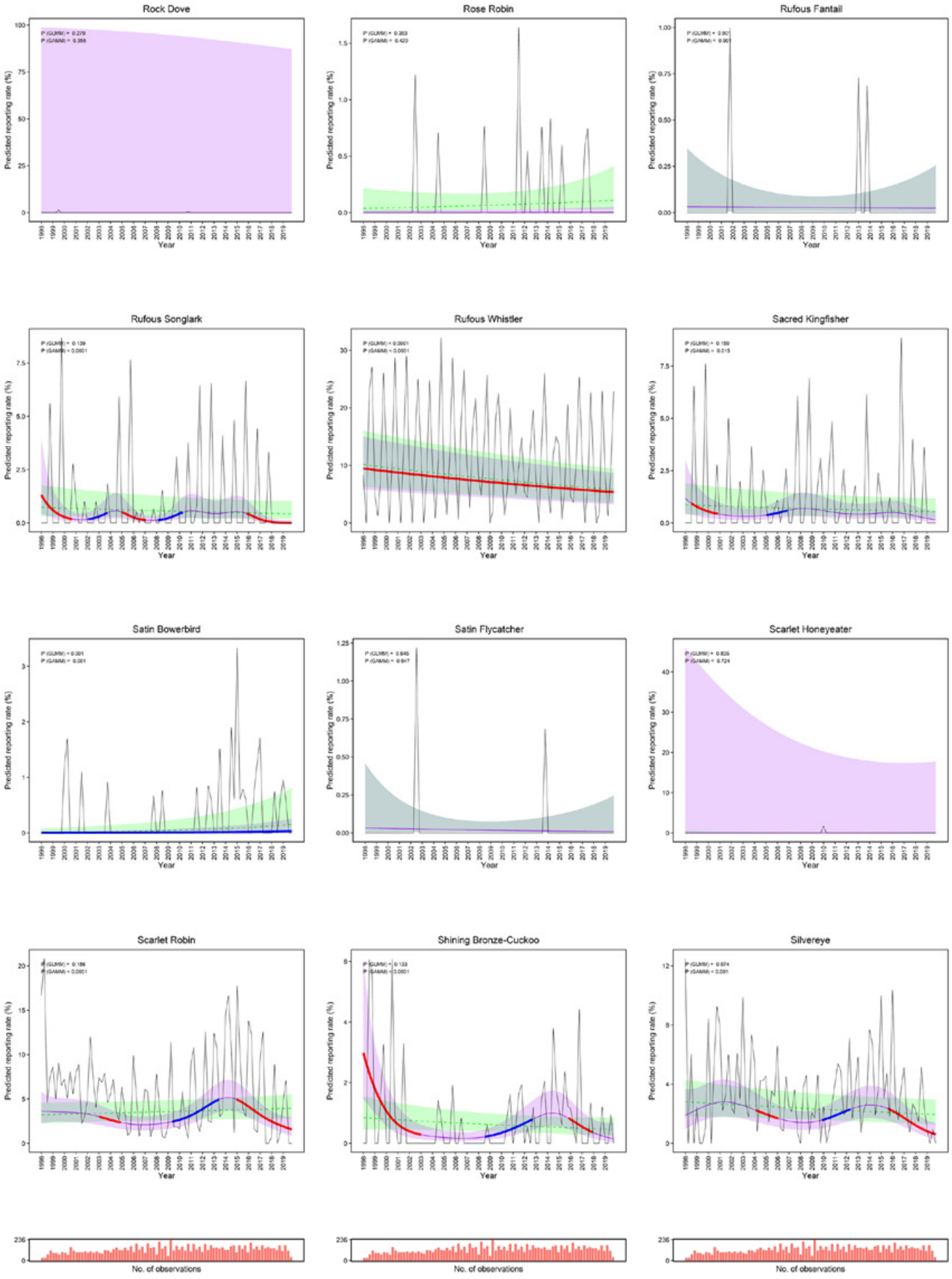

Common Myna
Common Starling
Crested Pigeon
Crested Shrike-tit
Crimson Rosella
Diamond Firetail
Dollarbird
Double-barred Finch
Dusky Woodswallow
Eastern Rosella
Eastern Spinebill
Eastern Yellow Robin
Predicted reporting rate (%)
Year
No. of observations

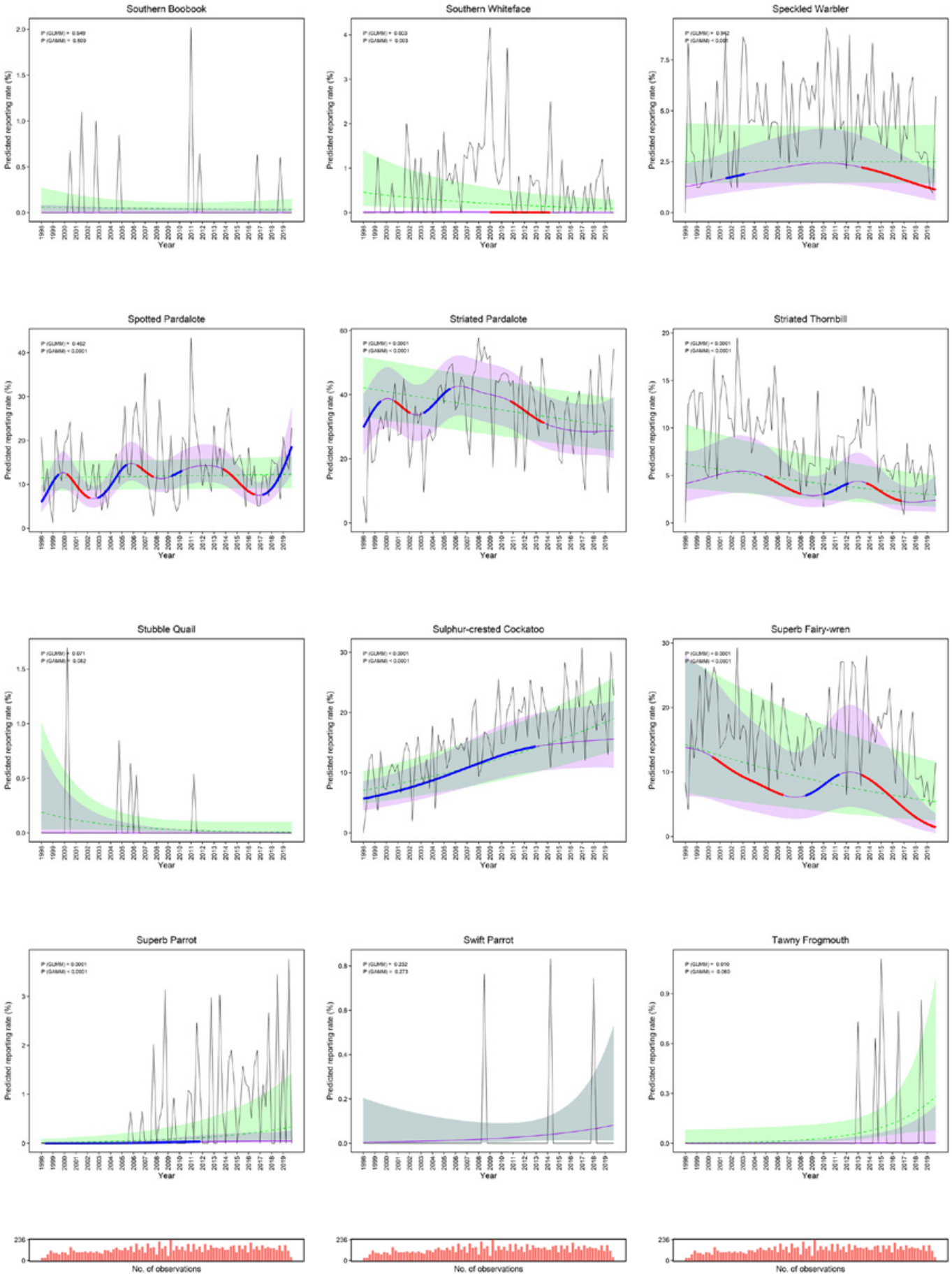

European Goldfinch
European Greenfinch
Fairy Martin
Fan-tailed Cuckoo
Flame Robin
Fuscous Honeyeater
Galah
Gang-gang Cockatoo
Golden Whistler
Grey Butcherbird
Grey Currawong
Grey Fantail
Predicted reporting rate (%)
Year
No. of observations

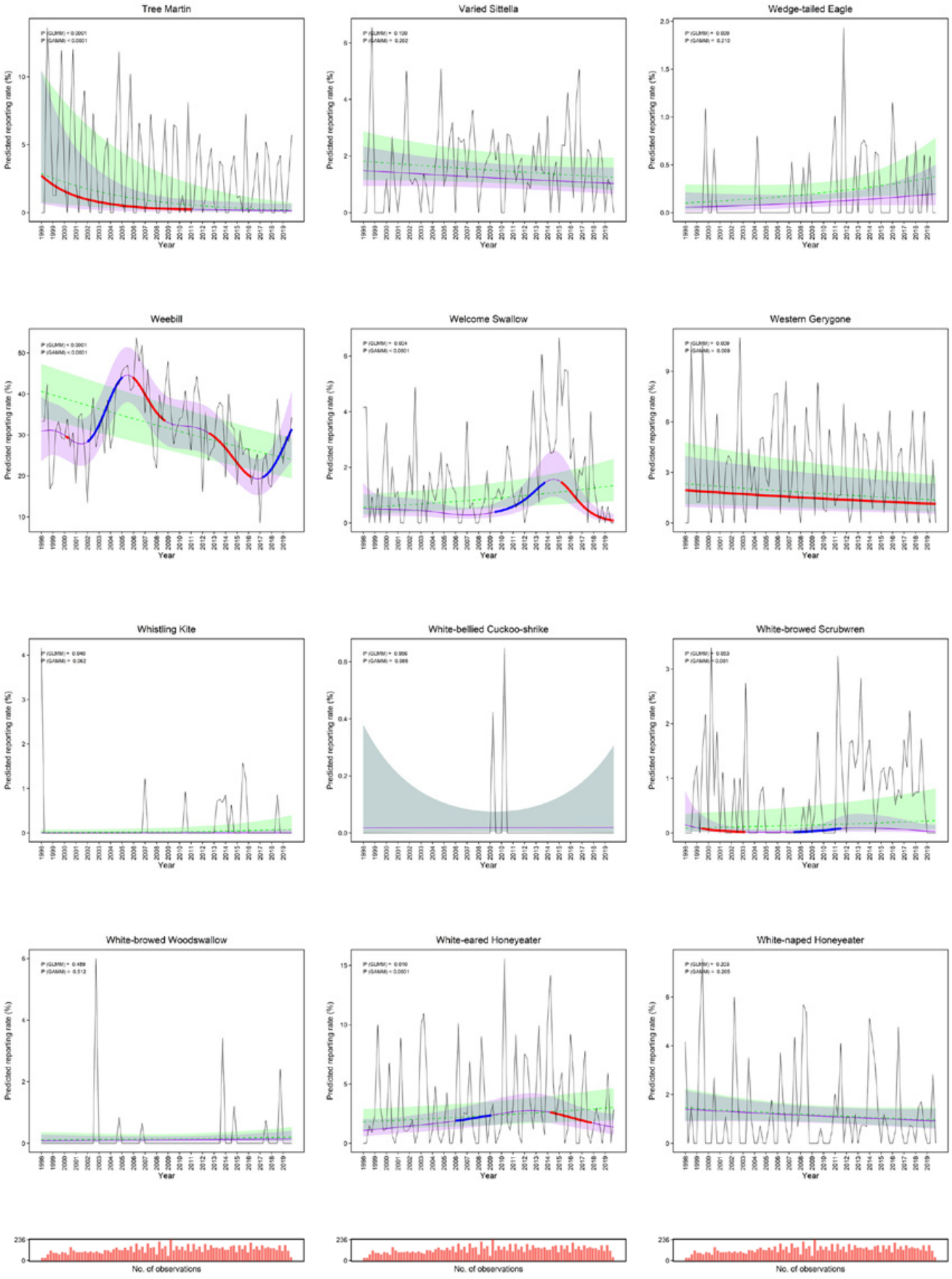

Grey Shrike-thrush
Hooded Robin
Horsfield's Bronze-Cuckoo
House Sparrow
Jacky Winter
Laughing Kookaburra
Leaden Flycatcher
Little Corella
Little Eagle
Little Friarbird
Little Lorikeet
Little Raven
Predicted reporting rate (%)
Year
No. of observations

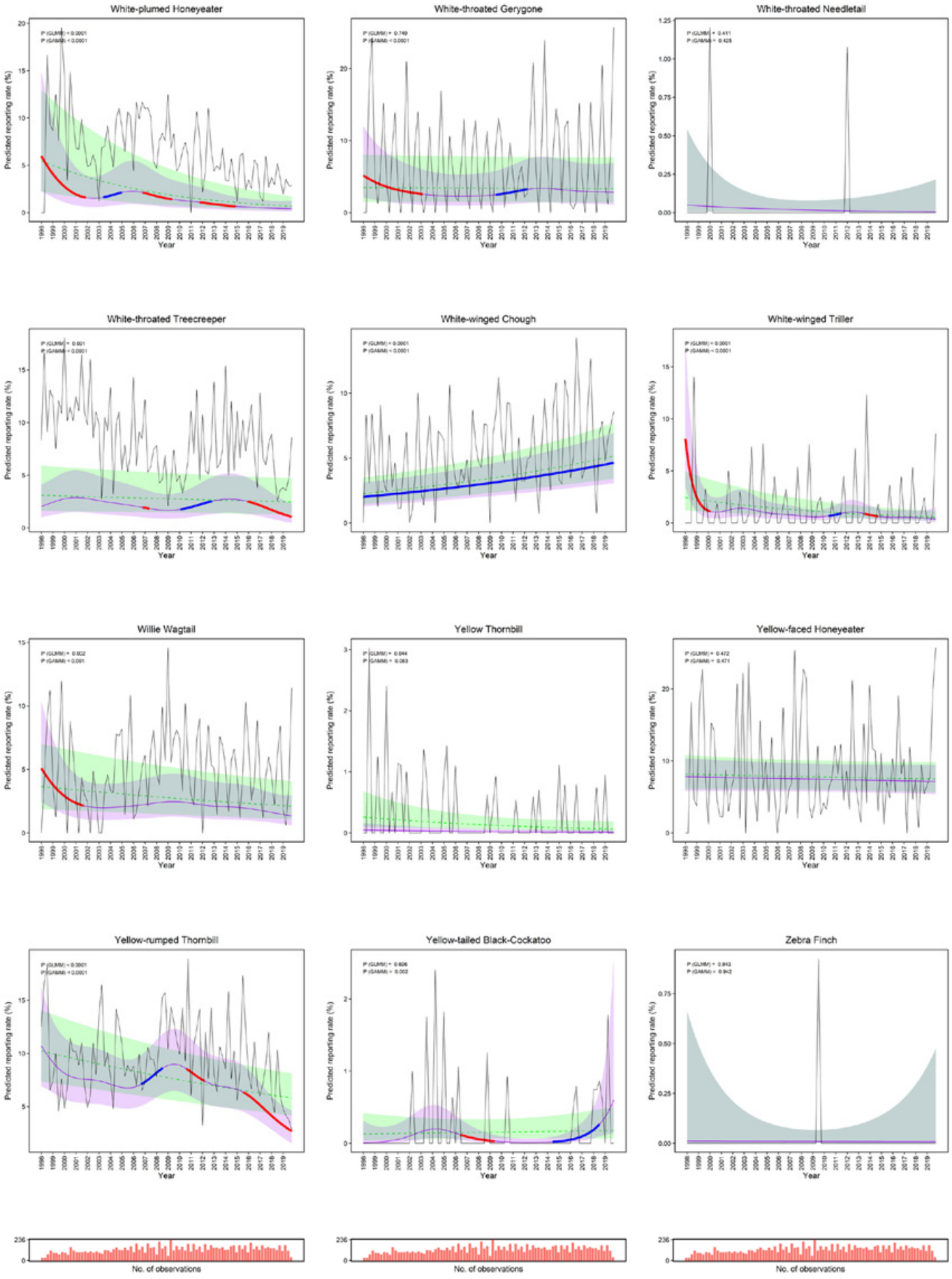

Long-billed Corella
Magpie-lark
Masked Lapwing
Masked Woodswallow
Mistletoebird
Musk Lorikeet
Nankeen Kestrel
Noisy Friarbird
Noisy Miner
Olive-backed Oriole
Painted Button-quail
Pallid Cuckoo
Predicted reporting rate (%)
Year
No. of observations


Peaceful Dove
Peregrine Falcon
Pied Butcherbird
Pied Currawong
Rainbow Bee-eater
Rainbow Lorikeet
Red Wattlebird
Red-browed Finch
Red-capped Robin
Red-rumped Parrot
Regent Honeyeater
Restless Flycatcher
Predicted reporting rate (%)
Year
No. of observations


Rock Dove
Rose Robin
Rufous Fantail
Rufous Songlark
Rufous Whistler
Sacred Kingfisher
Satin Bowerbird
Satin Flycatcher
Scarlet Honeyeater
Scarlet Robin
Shining Bronze-Cuckoo
Silvereye
Predicted reporting rate (%)
Year
No. of observations


Southern Boobook
Southern Whiteface
Speckled Warbler
Spotted Pardalote
Striated Pardalote
Striated Thornbill
Stubble Quail
Sulphur-crested Cockatoo
Superb Fairy-wren
Superb Parrot
Swift Parrot
Tawny Frogmouth
Predicted reporting rate (%)
Year
No. of observations


Tree Martin
Varied Sittella
Wedge-tailed Eagle
Weebill
Welcome Swallow
Western Gerygone
Whistling Kite
White-bellied Cuckoo-shrike
White-browed Scrubwren
White-browed Woodswallow
White-eared Honeyeater
White-naped Honeyeater
Predicted reporting rate (%)
Year
No. of observations

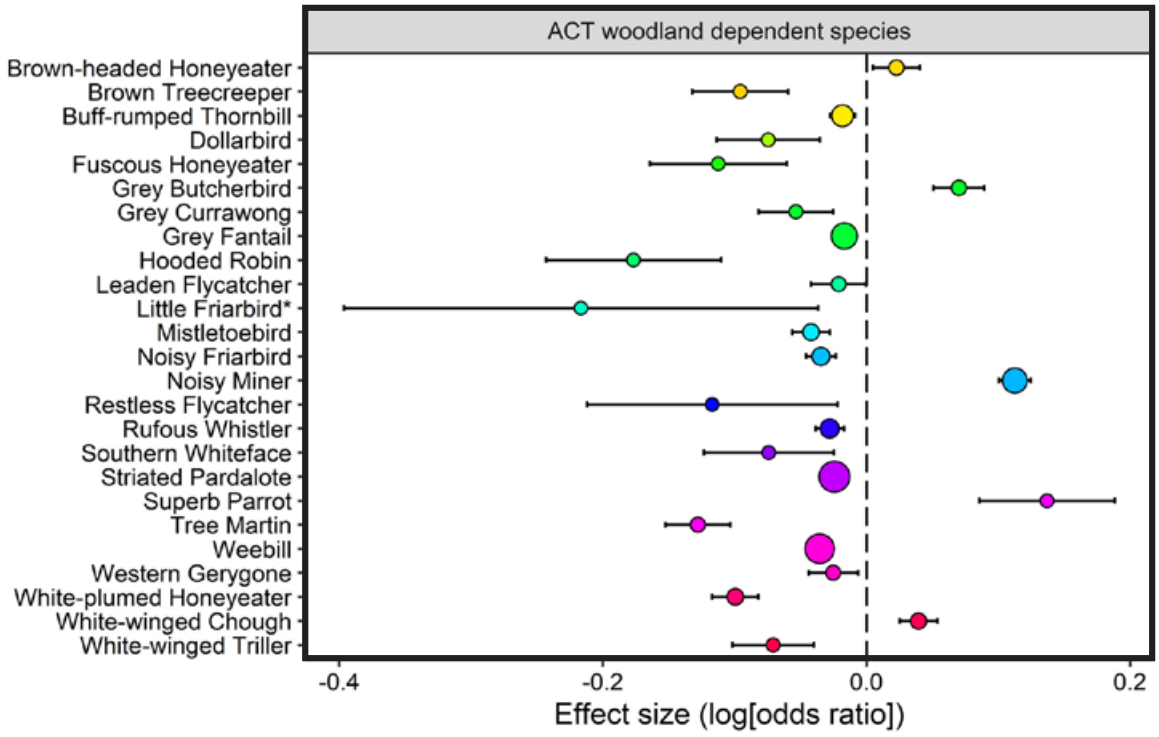

White-plumed Honeyeater
White-throated Gerygone
White-throated Needletail
White-throated Treecreeper
White-winged Chough
White-winged Triller
Willie Wagtail
Yellow Thornbill
Yellow-faced Honeyeater
Yellow-rumped Thornbill
Yellow-tailed Black-Cockatoo
Zebra Finch
Predicted reporting rate (%)
Year
No. of observations

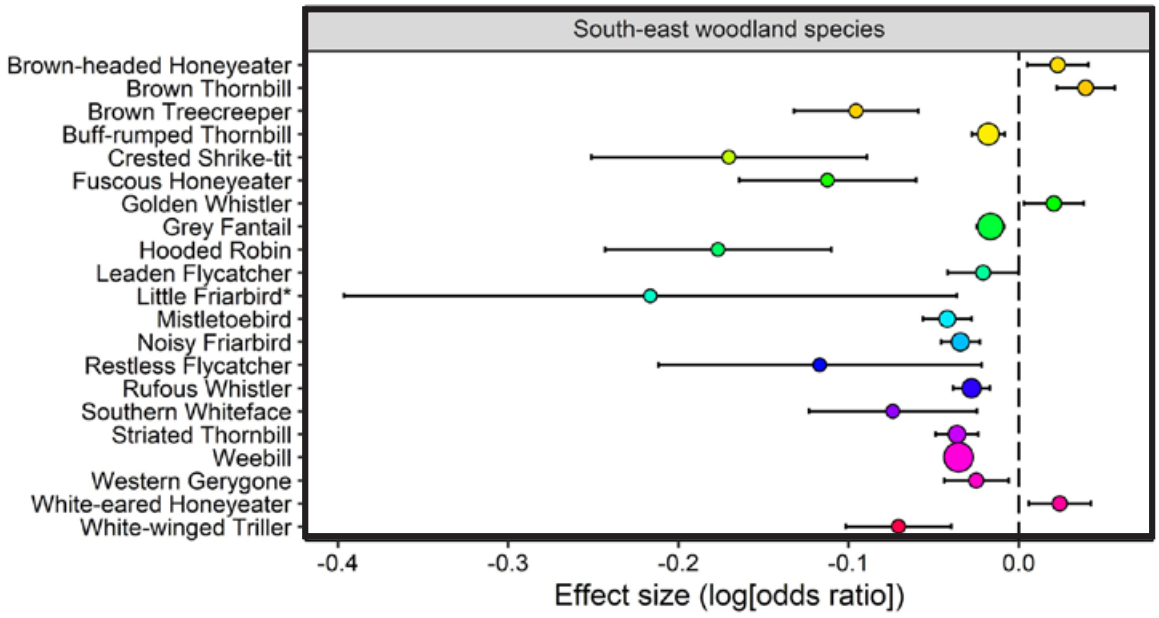
Appendix F: All species Rate of Change Plot

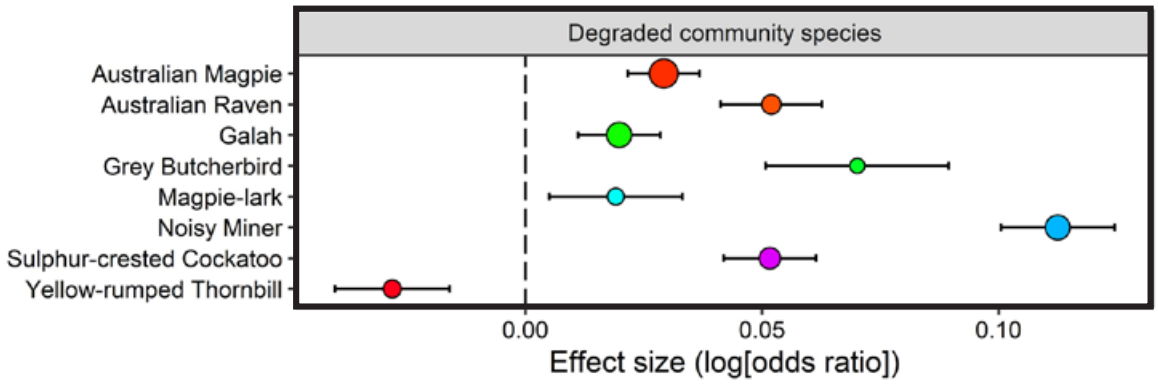
Significant increases (blue) and decreases (red) in the rate of change of the smoothed trends for individual species. Values were calculated for each quarter year. Only significant values were plotted.

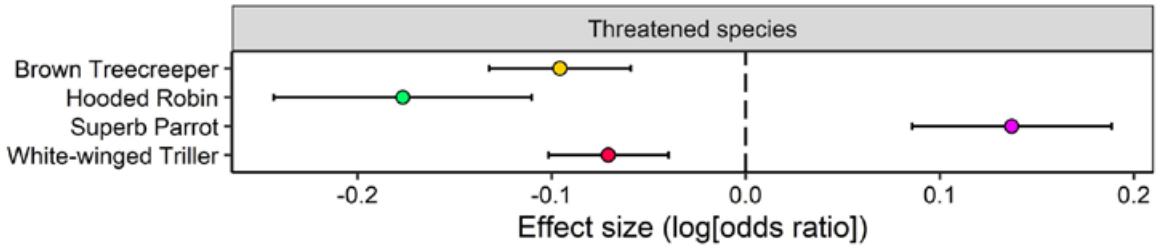
Appendix G: Effect Plots for Individual Species within Bird Groups

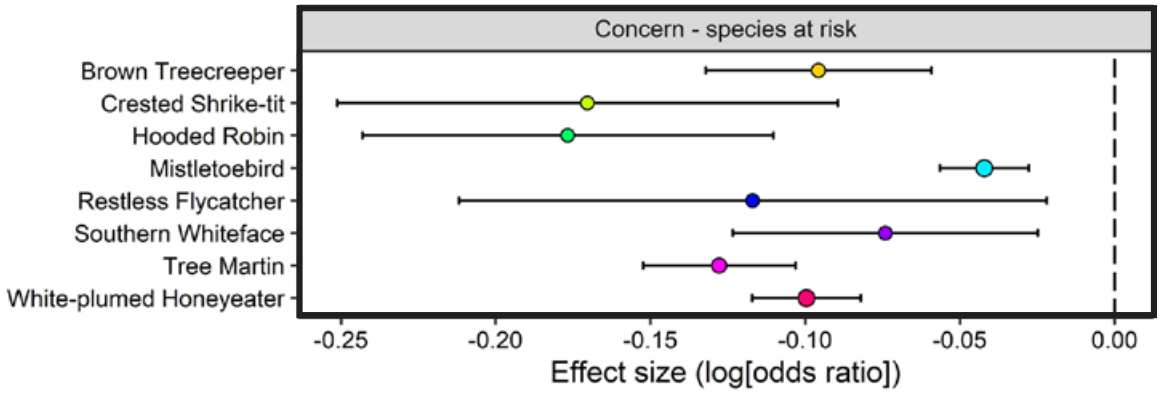
Summary Effect Size plots, showing the magnitude of change in reporting rate for individual bird species arranged by bird groups. The relevant bird group is named at the top of each plot (shaded grey). Groups showing significant, collective, long-term change have a bold black border.

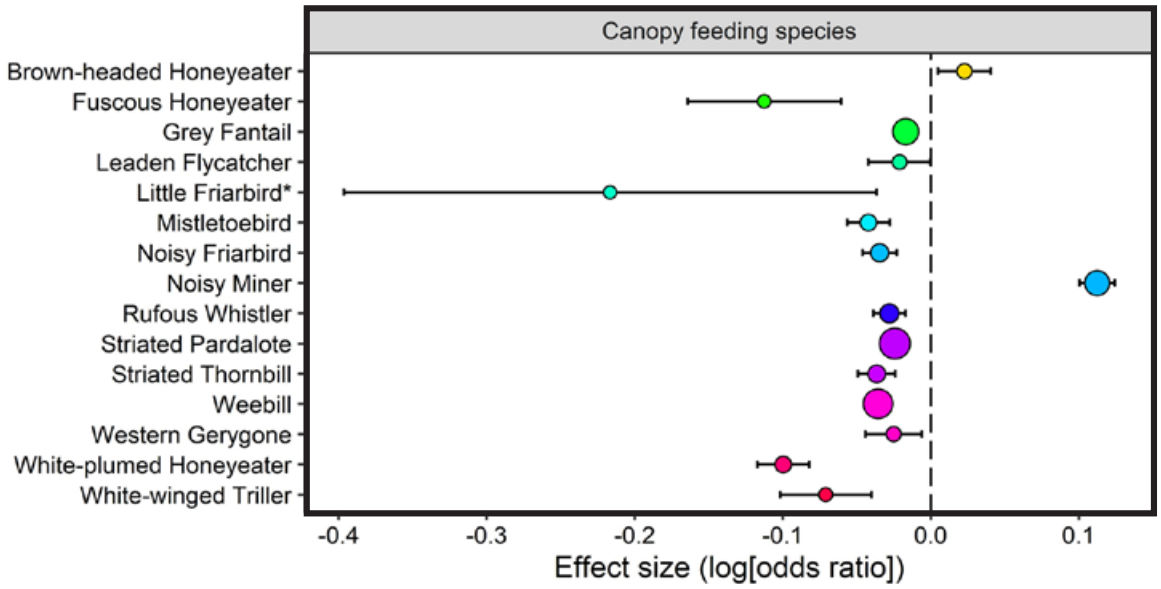


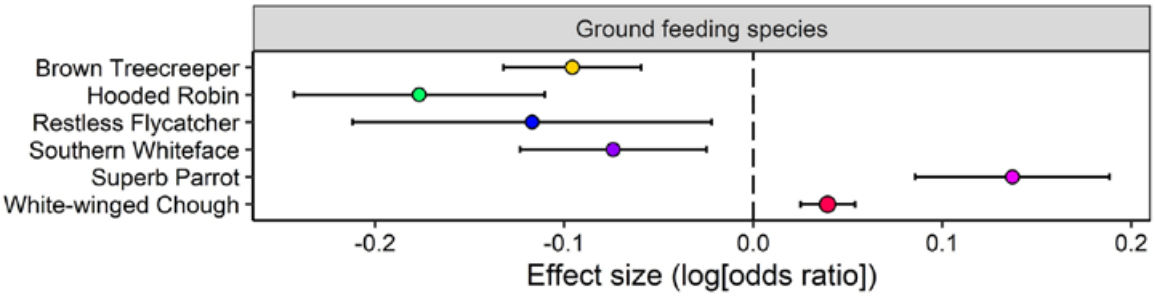


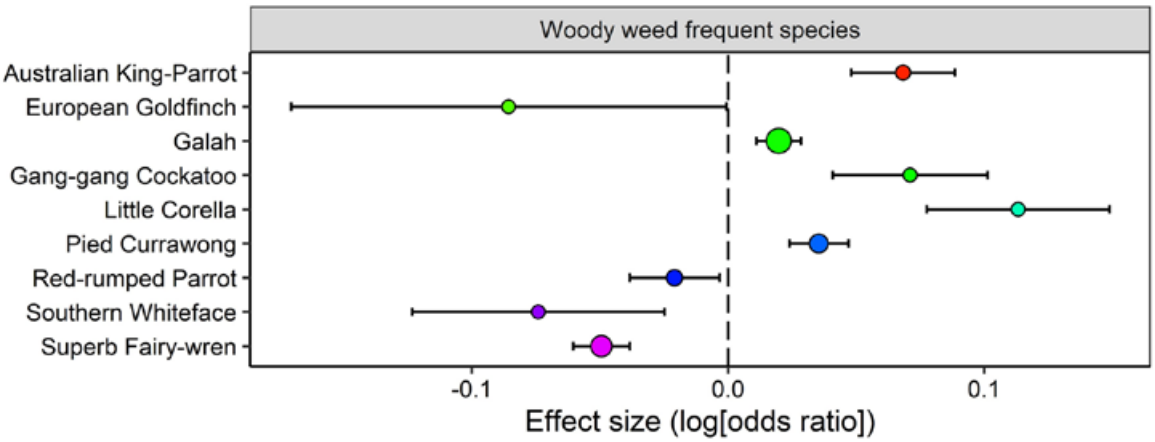

Degraded community species
Australian Magpie
Australian Raven
Galah
Grey Butcherbird
Magpie-lark
Noisy Miner
Sulphur-crested Cockatoo
Yellow-rumped Thornbill
0.00
0.05
0.10
Effect size (log[odds ratio])

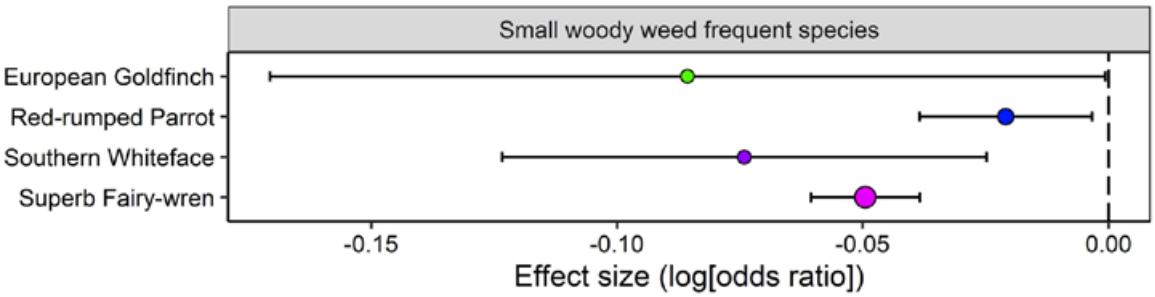

Threatened species
Brown Treecreeper
Hooded Robin
Superb Parrot
White-winged Triller
-0.2
-0.1
0.0
0.1
0.2
Effect size (log[odds ratio])

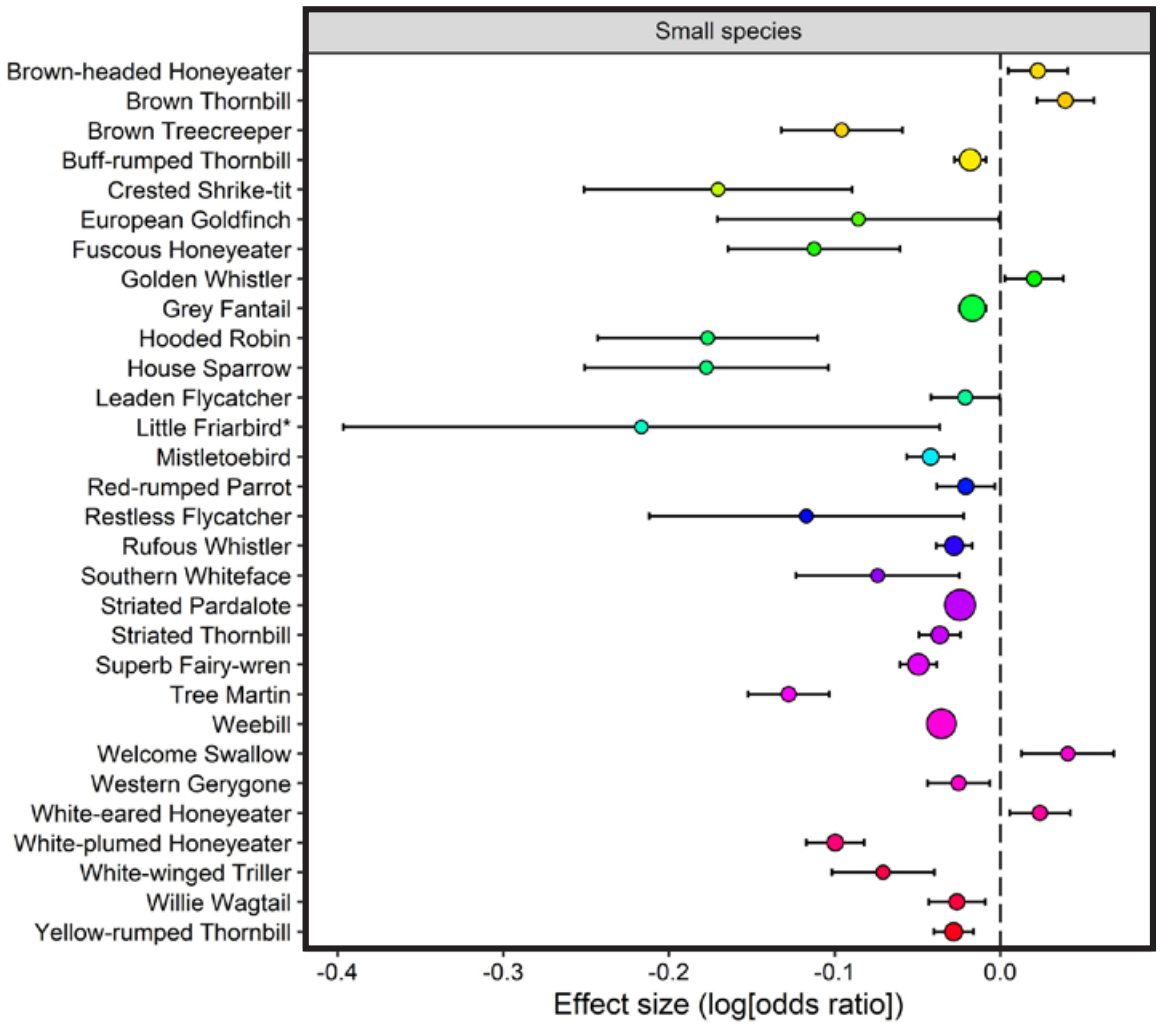

Concern - species at risk
Brown Treecreeper
Crested Shrike-tit
Hooded Robin
Mistletoebird
Restless Flycatcher
Southern Whiteface
Tree Martin
White-plumed Honeyeater
-0.25
-0.20
-0.15
-0.10
-0.05
0.00
Effect size (log[odds ratio])

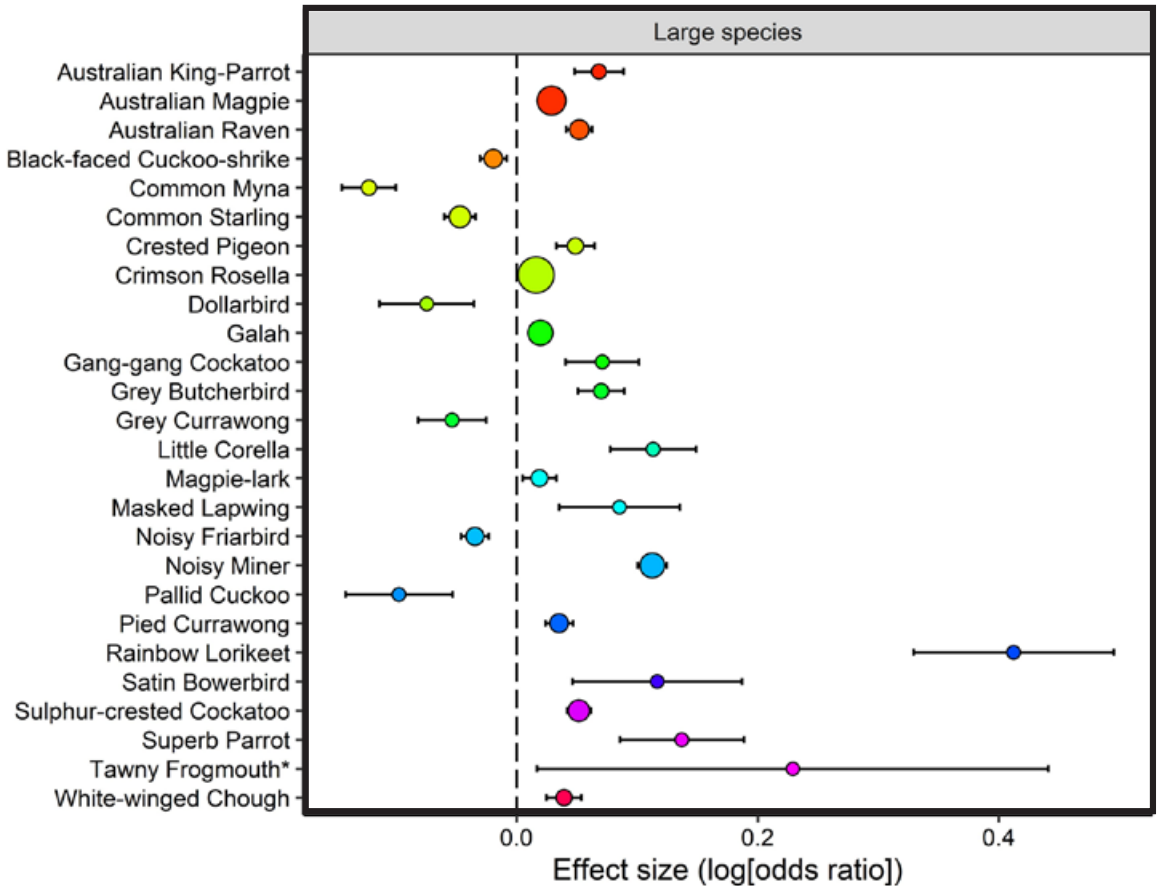

Canopy feeding species
Brown-headed Honeyeater
Fuscous Honeyeater
Grey Fantail
Leaden Flycatcher
Little Friarbird*
Mistletoebird
Noisy Friarbird
Noisy Miner
Rufous Whistler
Striated Pardalote
Striated Thornbill
Weebill
Western Gerygone
White-plumed Honeyeater
White-winged Triller
-0.4
-0.3
-0.2
-0.1
0.0
0.1
Effect size (log[odds ratio])


Ground feeding species
Brown Treecreeper
Hooded Robin
Restless Flycatcher
Southern Whiteface
Superb Parrot
White-winged Chough
-0.2
-0.1
0.0
0.1
0.2
Effect size (log[odds ratio])


Woody weed frequent species
Australian King-Parrot
European Goldfinch
Galah
Gang-gang Cockatoo
Little Corella
Pied Currawong
Red-rumped Parrot
Southern Whiteface
Superb Fairy-wren
-0.1
0.0
0.1
Effect size (log[odds ratio])


Small woody weed frequent species
European Goldfinch
Red-rumped Parrot
Southern Whiteface
Superb Fairy-wren
-0.15
-0.10
-0.05
0.00
Effect size (log[odds ratio])


Small species
Brown-headed Honeyeater
Brown Thornbill
Brown Treecreeper
Buff-rumped Thornbill
Crested Shrike-tit
European Goldfinch
Fuscous Honeyeater
Golden Whistler
Grey Fantail
Hooded Robin
House Sparrow
Leaden Flycatcher
Little Friarbird*
Mistletoebird
Red-rumped Parrot
Restless Flycatcher
Rufous Whistler
Southern Whiteface
Striated Pardalote
Striated Thornbill
Superb Fairy-wren
Tree Martin
Weebill
Welcome Swallow
Western Gerygone
White-eared Honeyeater
White-plumed Honeyeater
White-winged Triller
Willie Wagtail
Yellow-rumped Thornbill
-0.4
-0.3
-0.2
-0.1
0.0
Effect size (log[odds ratio])


Large species
Australian King-Parrot
Australian Magpie
Australian Raven
Black-faced Cuckoo-shrike
Common Myna
Common Starling
Crested Pigeon
Crimson Rosella
Dollarbird
Galah
Gang-gang Cockatoo
Grey Butcherbird
Grey Currawong
Little Corella
Magpie-lark
Masked Lapwing
Noisy Friarbird
Noisy Miner
Pallid Cuckoo
Pied Currawong
Rainbow Lorikeet
Satin Bowerbird
Sulphur-crested Cockatoo
Superb Parrot
Tawny Frogmouth*
White-winged Chough
0.0
0.2
0.4
Effect size (log[odds ratio])

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ORIGINAL ARTICLE

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

Katie A. Lowe · Charlotte E. Taylor · Richard E. Major

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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 Verstädterungs-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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K. A. Lowe · C. E. Taylor
School of Biological Sciences, University of Sydney, Sydney, NSW 2006, Australia

R. E. Major (✉)
Terrestrial Ecology, Australian Museum, 6 College St, Sydney, NSW 2010, Australia
e-mail: richard.major@austmus.gov.au

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 outcompete other birds at food resources and drive other birds out of habitats. These claims are built largely from a body of anecdotal of 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.

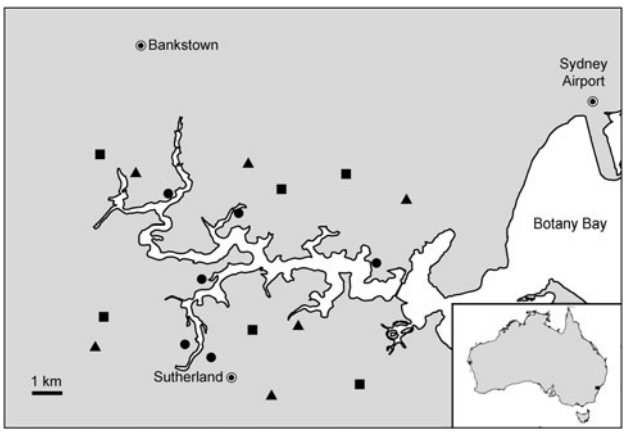

Fig. 1 The study sites in southern Sydney, Australia. *Filled circle* bushland, *filled triangle* residential, *filled square* commercial

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 subsitituted 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%.

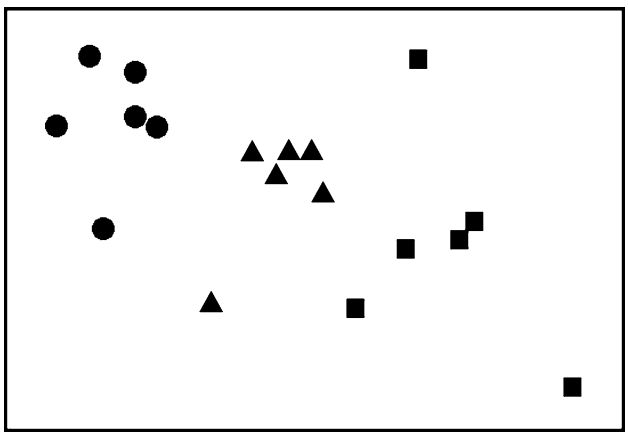

Fig. 2 Ordination of sites based on bird species abundance analysed by non-metric multidimensional scaling using Bray–Curtis similarities between sites. *Filled circle* bushland, *filled triangle* residential, *filled square* commercial

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 (±7.9 SE) in commercial, to 4.9 (±3.9 SE) in residential, to 0.0 in bushland sites.

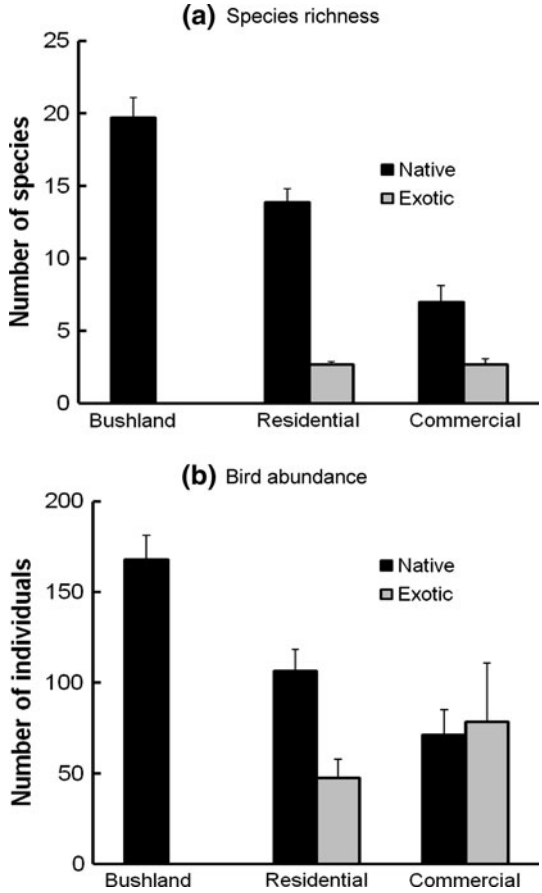

Fig. 3 **a** The mean species richness, and **b** mean abundance, of native and exotic species in each habitat type. *Black bars* native species, *grey bars* exotic species. *Error bars* SEs

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 (±0.28 SE) aggressive interactions per hour occurred in bushland remnants, 1.50 (±0.39 SE) interactions per hour in residential streetscapes, and 0.92 (±0.15 SE) interactions per hour in commercial areas. Twenty-seven of the 39 interactions were interspecific with an average number of 0.42 (±24 SE) interspecific interactions per hour in bushland remnants, 1.17 (±31 SE) in residential streetscapes and 0.67 (±0.17 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.

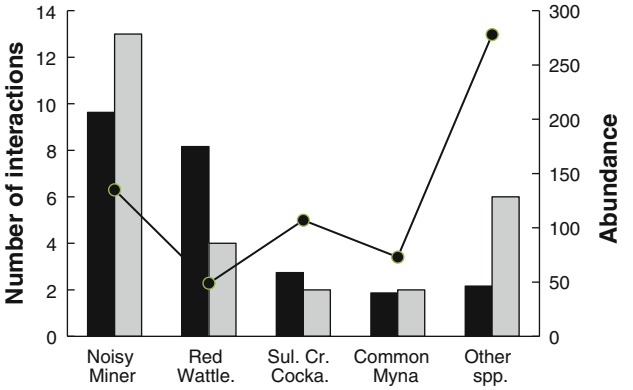

Fig. 4 The number of interspecific interactions initiated by each species of aggressor (*grey bars*) and numerical abundance of each species (*black line*) observed during focal site surveys. The *black bars* show relative levels of interspecific aggression after adjusting to account for abundance

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 surveyors 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 ($\chi^2 = 40.1$, $df = 1$, $P < 0.01$).

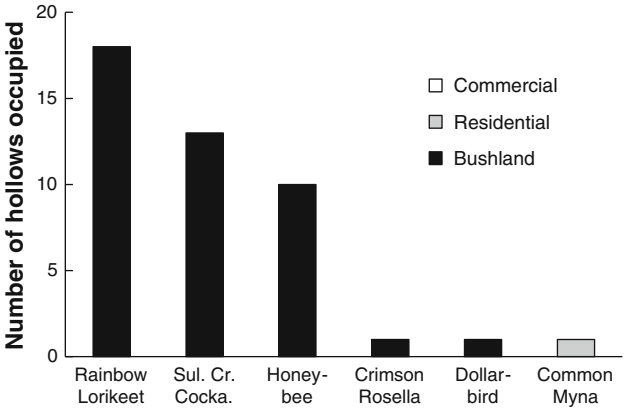

Fig. 6 The total number of tree hollows occupied by each species in bushland (*black bars*), residential (*grey bars*) and commercial (*white bars*) habitats. No occupancy of tree hollows was observed in commercial areas

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).

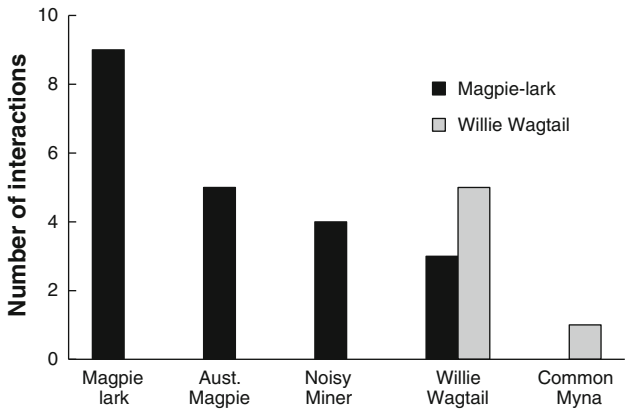

Fig. 5 The total of number of attacks by each species of aggressor on focal Magpie-larks *Grallina cyanoleuca* (*black bars*) and focal Willie Wagtails *Rhipidura leucophrys* (*grey bars*)

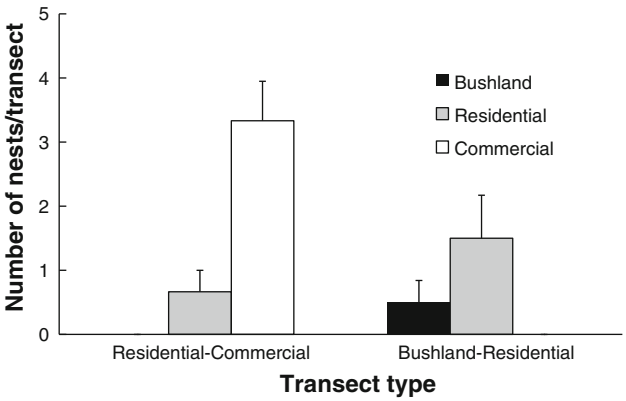

Fig. 7 The mean (+SE) number of Common Myna *Acridotheres tristis* nests detected in bushland (*black bars*), residential (*grey bars*) and commercial (*white bars*) habitats during transect surveys along different habitat interfaces

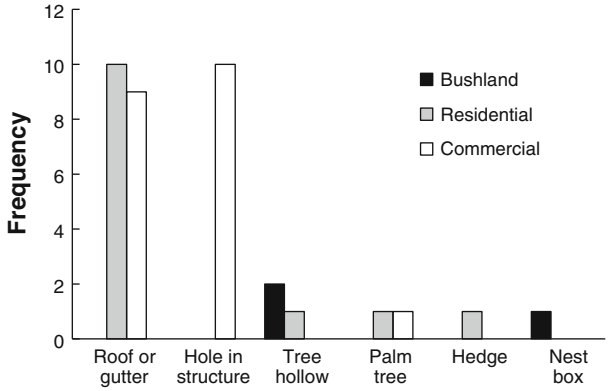

Fig. 8 The frequency of each cavity type used by Common Mynas in bushland (*black bars*), residential (*grey bars*) and commercial (*white bars*) habitats

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 defendable (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 Mynas 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 Counsilman (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 lead 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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CHARLES STURT UNIVERSITY

Competitive behaviour in Common Mynas

An investigation into potential impacts on native fauna and solutions for management

Kathryn Haythorpe B.A. B.Sc. B.Env.Sc.Mgt.(Hons)

August 2013

Common Mynas and an Eastern Rosella at a nest box. Photo: K. Haythorpe.

Thesis submitted to Charles Sturt University in partial fulfilment of the requirements of the Doctor of Philosophy (PhD)

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Certificate of Authorship

I hereby declare that this submission is my own work and that, to the best of my knowledge and belief, it contains no material previously published or written by another person nor material which to a substantial extent has been accepted for the award of any other degree or diploma at Charles Sturt University or any other educational institution, except where due acknowledgment is made in the thesis. Any contribution made to the research by colleagues with whom I have worked at Charles Sturt University or elsewhere during my candidature is fully acknowledged. I agree that this thesis be accessible for the purpose of study and research in accordance with the normal conditions established by the Executive Director, Library Services or nominee, for the care, loan and reproduction of theses.

Kathryn M. Haythorpe

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Ethics Statement

This work was carried out under the following ethics protocols:

Chapter 1:

Newcastle University Animal Ethics Committee ethics protocol A-2008-173

Chapter 2:

Newcastle University Animal Ethics Committee ethics protocol A-2008-173

Newcastle University Animal Ethics Committee ethics protocol A-2011-103

Chapter 3:

Newcastle University Animal Ethics Committee ethics protocol A-2011-103

Charles Sturt University Animal Care and Ethics Committee ethics protocol 11/074

Chapter 4:

Newcastle University Human Ethics Committee ethics protocol H-2011-0217

Abstract

The introduction of organisms to new areas is a source of ongoing concern for environmental managers. Common Mynas *Sturnus tristis* are an introduced bird species found along the East coast of Australia that are thought to aggressively outcompete native species for food and nesting resources. Public attitudes towards Common Mynas are extremely negative, and substantial time and financial resources are allocated towards control and eradication of the species. However, most accounts of aggression from Common Mynas are anecdotal, and empirical evidence of a significant impact on native species in Australia due to the presence of Common Mynas is yet to be found. In this thesis I examined the competitive behaviour of Common Mynas around food and nesting resources to quantify this impact. I found that, contrary to all expectations, Common Mynas were one of the least aggressive species studied. Some native species like Noisy Miners *Manorina melanocephala* and Australian Magpies *Cracticus tibicen* display much higher levels of aggression and have the potential to be far more detrimental to native species, yet are not subject to control efforts. In particular, Common Mynas do not appear to display substantial aggression towards other cavity-nesting species such as rosellas (*Platycercus* spp.) around nest sites, an issue that has been of great concern for anyone providing nest boxes for wildlife. It appears that Common Mynas attempt to outcompete other species not through direct aggression, but through the use of novel behavioural strategies such as the filling of multiple nest sites with nesting material to discourage other birds from using them, although even this strategy appeared to be largely unsuccessful in this study. The overall conclusion of this work is that, contrary to previous thought, Common Mynas probably do not represent a significant threat to native wildlife in most Australian urban environments. However, they continue to be considered with vitriol by the public. I investigated this attitude through a survey asking for the opinions of nest box owners and found that, despite their dislike for the species, uptake of nest boxes by Common Mynas did not generally lead to them removing or disabling their nest box, or decrease their overall enjoyment of the experience, suggesting that Common Mynas were not having an indirect effect on native species in this way. In response to an ardent desire from residents

for a way to exclude Common Mynas from nest boxes, I also investigated a commercially available structure known as the anti-myna baffle design, which is used to prevent Common Mynas from accessing nest boxes while still allowing access by parrot species, and recommendations for the improvement of this structure have been included for the consideration of interested parties. Overall, the collection of works embodied in this thesis are important to our understanding of the behaviour of an introduced species during a low density phase of ongoing invasion in Australia, and have the potential to prevent substantial resources being wasted attempting to control a species that is actually relatively benign.

Introduction

Introduced Species

The Science of Invasion

Introduced species, also known as non-native, non-indigenous, invasive, foreign, or exotic species, are defined by the International Union for the Conservation of Nature as "a species, subspecies, or lower taxon occurring outside of its natural range (past or present) and dispersal potential (i.e. outside the range it occupies naturally or could not occupy without direct or indirect introduction or care by humans) and includes any part, gametes or propagule of such species that might survive and subsequently reproduce" (Invasive Species Specialist Group 2000). Invasion science, the study of the causes and consequences of these introductions (Blackburn *et al.* 2011; Richardson and Ricciardi 2013), has become an increasingly popular field, with an exponential increase in publications occurring over the past few decades (Simberloff 2004).

Invasion science can be a controversial field. Some scientists (e.g. Davis 2009; Thomas 2013) argue that most introduced species are relatively benign and should not be subject to management on the basis of being classed as "introduced" alone; thus these authors suggest that "invasion science" as a discipline is inherently flawed. Davis *et al.* (2011) termed this concept the "native-versus-alien dichotomy", arguing that many species often appear to be "judged" more on their status as native or introduced than on their actual environmental impact, and management action should not follow this path. One hundred and forty-one scientists signed a letter (Simberloff 2011) objecting to the opinions expressed in this paper and claiming that Davis *et al.* (2011) was downplaying the danger of introduced species. Fierce debate on whether "invasion biology" remains a valid discipline has ensued (Simberloff and Vitule 2013; Valéry *et al.* 2013). Others suggest a "middle-ground" approach – exercising caution about introduced species, but also basing management decisions in a wider range of conservation goals (Shackelford *et al.* 2013).

The Problem of Introduced Species

Damage caused by introduced species is commonly cited as one of the major threats to the health and biodiversity of the natural environment worldwide. Introduced species have been declared the second biggest threat to global biodiversity and the survival of threatened or endangered species, after the direct destruction of habitat by humans in the form of land clearing (Walker and Steffen 1997; Wilcove *et al.* 1998; Ewel *et al.* 1999). Introduced species can disrupt the fundamental evolutionary and ecological processes operating within an ecosystem (Bertness 1984), causing massive changes in abundance. In some cases this can lead to extinction of native species through competition (e.g. Petren and Case 1996), predation (Davis 2003), introduction of pathogens (Gurevitch and Padilla 2004), or the alteration of ecosystem dynamics to an extent that threatens the survival of natives (Vitousek 1990). The economic costs associated with introduced species are substantial – the annual cost of damage and control of introduced species was recently estimated at $120 billion in the United States of America (Pimentel *et al.* 2005). The problem is likely to only become worse with time, as increased levels of trade lead to more species being transported to new areas and threatening new environments (Ricciardi *et al.* 2000).

Introduced species clearly present an ecological challenge for managers. However, land clearing, climate change, urban development, and other human activity has drastically altered the habitat characteristics of most ecosystems, to the advantage of some species and the detriment of others (Crooks *et al.* 2004; van Heezik *et al.* 2008). While in many cases it is introduced species that profit most from these disturbances, it is also possible for native species to dominate a modified landscape at the expense of other species. For instance some native species, such as Lake Trout *Salvelinus namaycush* in the United States of America (Kaeding *et al.* 1996) or Eastern Grey Kangaroos *Macropus giganteus* in Australia (Caughley *et al.* 1984) thrive in regions well outside their former range or have increased drastically in abundance as a result of human modification of habitat.

Potential Bias in the Study of Invasion Science

Resources devoted to management goals are likely to be heavily influenced by the level of interest the goal attracts and the extent to which it is considered a priority (Roome 1992). Management of introduced species is a subject that members of the public and scientists alike feel strongly about. It is estimated that more than $22 billion is spent annually on control of introduced species in the United States (Pimentel *et al.* 2005), and would likely be significantly higher globally. This high level of interest and investment in the issue of introduced species is encouraging. However, some scientists have warned against being too quick to target introduced species only and ignore the potential impacts of native species.

The term "biological bias" has been given to situations where detrimental impacts of native species may be ignored under the assumption that the processes surrounding them are "natural" and thus unlikely to be capable of having a negative effect (Davis *et al.* 2011). In some cases there may be a perception among the public that native species are incapable of causing damage because they have evolved with the current environment, and thus should not need to be controlled or managed (Brown and Sax 2004). On the other hand, Davis et al. (2011) argue that introduced species are sometimes unfairly seen as posing a serious environmental threat, even before such a threat has been properly quantified. The authors of this paper suggest that this may have arisen from the small number of extreme cases where introduced species have been able to cause serious social, agricultural or ecological harm, such as Zebra Mussels *Dreissena polymorpha* in the United States of America (Roberts 1990; Ricciardi *et al.* 1998), or Cane Toads *Bufo marinus* in Australia (Shine 2010). However, Davis *et al.* (2011) argues that these cases are exceptions, and that, outside of highly isolated environments like small islands and lakes, introduced species generally don't pose a major extinction threat to most native species (Davis 2009). In fact, introduced species may actually be beneficial in some cases, providing important ecological services such as suitable habitat or food for native species; yet this can be overlooked simply due to their status as introduced (Carroll 2011; Davis 2011). This opinion is disputed by Richardson and Ricciardi (2013), who argue

that even introduced species that appear benign should not be overlooked: important impacts may not have yet been detected, may occur in different regions to those studied, or may occur at a later point in time. Thus we should continue to monitor and study the impacts of introduced species, even if they appear to be harmless or even beneficial in some instances.

It has been suggested that humans are psychologically wired to be biased towards native species and against introduced species (Franklin 2006). This may be a kind of xenophobia stemming from a generalised fear of foreigners "trespassing" where they don't belong (Franklin 2006). We are naturally inclined to try to classify situations into dichotomies, such as right and wrong or good and bad. In Australia, it is argued (Smith 1999; Simpson 2010) that many people hold the view that introduced species are the "bad guys" – characteristically nasty, vicious, greedy, and dirty – while native species are the victimised "good guys", attempting to exist in their natural, pure state. Protection of native species also draws on the psychological tendency of humans to try to protect those they consider vulnerable – in this case defending native species from influence of introduced species (Franklin 2006). These biases can be detrimental if allowed to impinge on scientific process in the form of implicit assumptions and judgements about the species that then influence interpretations of data and practical decisions (Trigger *et al.* 2008). While psychological values such as these should not be ignored, they must form an appropriate part of the framework through which decisions are made on environmental issues.

Introduced Birds

Since the mid-1800s, more than 400 bird species have been moved by people into regions they previously did not inhabit (Long 1981). People move birds for a variety of reasons: food, aesthetic appeal, recreational hunting, or insect control. Some introductions occur accidentally, when birds escape from cages or domestic stock or arrive on ships. Introduced birds present an economic threat to horticultural production (Bomford and Sinclair 2002), costing an estimated $313 million annually in Australia (Tracey *et al.* 2007; Gong *et al.* 2010). Of particular concern to ecologists is the potential threat to native biodiversity, particularly native bird species. Some scientists (e.g. Kumschick and Nentwig 2010) argue

that some introduced birds may have an impact as severe as the worst introduced mammals, although this claim has been disputed (Strubbe *et al.* 2011). Some of the ways introduced birds may affect native birds are through hybridisation (e.g. Chukar *Alectoris chukar* with other Partridges *Alectoris* spp.; Barilani *et al.* 2007), spread of disease, brood parasitism (e.g. Shiny Cowbird *Molothrus bonariensis*; Woodworth 1997), or competition. Competition may be exploitative (indirect competition due to consumption of limited resources to satisfy their own requirements for survival) or through interference (direct interference with another species through aggression or modification of resources to prevent their use; Birch 1957; Park 1962), and is often over food or nesting sites. The three bird species listed as being among the "100 worst invasive species" (Lowe *et al.* 2000) were Common Starlings *Sturnus vulgaris*, Red-Vented Bulbuls *Pycnonotus cafer*, and Common Mynas *S. tristis*. The greatest threat from these species to native species is thought to be through interference competition or competition for food or nesting sites (Baker *et al.* 2013).

Common Mynas

Common Mynas (Aves: Sturnidae, Linnaeus 1766) are members of the Sturnid family and closely related to Common Starlings. They are mostly chocolate-brown in colour, with yellow bills, legs, and peri-orbital skin, white tips on tail undersides, and distinctive white wing patches most easily visible in flight (Fig. 1). They are medium-sized passerines, with an average length of 25cm and weight of 130g for males and 115g for females (Counsilman 1974a). Aside from weight, the sexes are similar, but juveniles are distinguishable from adults by having brown, rather than black, heads.

Fig. 1 A Common Myna *Sturnus tristis*. Photo courtesy of J Fields.

Common Mynas have been known by a variety of names including Common Myna(h), Indian Myna(h), Calcutta Myna(h), House Myna(h), Tickbird, Chocolate Bird, Chocolate Dollar-bird, White Wings, or Thynne Bird.

Origin and introduction to Australia

Common Mynas evolved in the open woodland habitats of India (Sengupta 1968). Their natural range extends east from Afghanistan, throughout Russian

Turkestan, India and Sri Lanka and into south-western China and Indochina (Lever 1987). They have been introduced to every continent in the world and numerous oceanic islands, usually in attempts to control insect pests, for aesthetic purposes, or as accidental cage bird escapees (Long 1981). The introduced range of Common Mynas now extends throughout parts of Asia, Africa, New Zealand, and the east coast of Australia (Lever 1987, Fig. 2).

Common Mynas were first introduced to Australia in Melbourne in 1862 (Ryan 1906; Hone 1978; Long 1981). Multiple introductions followed throughout the Eastern states and Tasmania until as recently as 1971, more than a century later (Gregory-Smith 1985). The number of birds that were officially released numbers in the hundreds (Long 1981; Newsome and Noble 1986; Lever 1987), and the unofficial number is probably much higher. These multiple introductions and their ongoing status as a protected species (Lever 1987) allowed Common Mynas to thrive and increase their range across much of urban Australia. Common Myna numbers increase at a rapid rate, and are quick to replenish following declines in abundance; features common of invasive species (Duncan *et al.* 2003a; Duncan *et al.* 2003b). Common Mynas in Sydney are now the most common and prolific species, being found in 80% of suburban gardens (Parsons *et al.* 2006), and were estimated in 2003 to be in numbers of more than 100 birds per square kilometre in some areas of Canberra (Tidemann 2003).

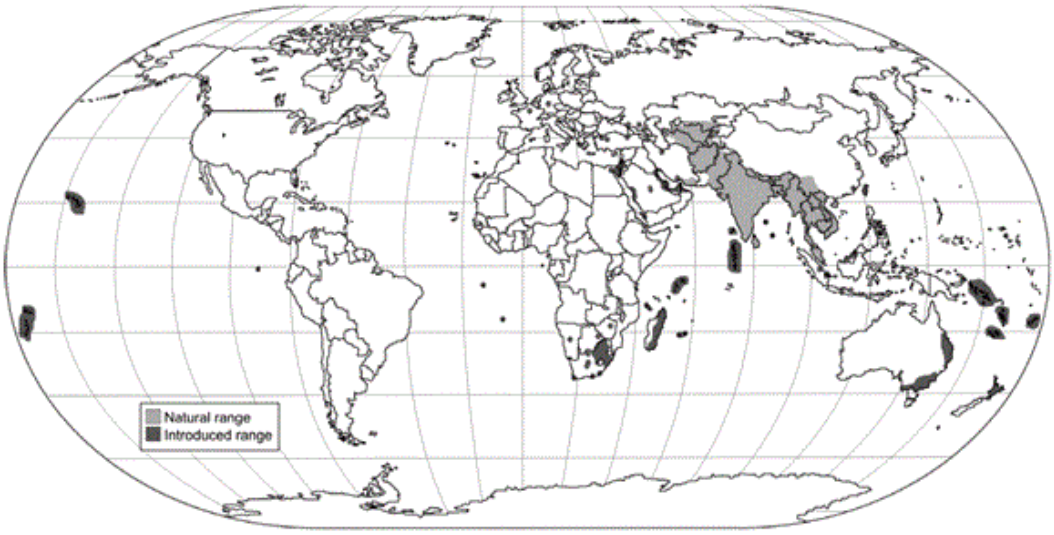

Fig. 2 Common Myna natural range (light grey) and introduced range (dark grey). Image courtesy of Peacock *et al.* (2007).

Distribution and habitat

Common Mynas are found throughout urban centres along the East coast of Australia. Their largest populations are found in Melbourne, Canberra, the Sydney area (including continuous populations extending to Newcastle and Wollongong), Brisbane, and Cairns. Occasional sightings have been recorded in both Adelaide and Tasmania, but so far active control efforts in these areas have prevented their establishment (Lever 1987). The Common Mynas used for the studies that make up this thesis were located in Newcastle (chapters 1 and 2), where they probably spread from Sydney populations in the 1970s (Hone 1978), and Canberra (chapter 3), where they were released in the late 1960s and have since existed as an isolated population (Long 1981).

Common Mynas prefer open habitats with few trees (Parsons *et al.* 2006) such as industrial areas, schools, parks, sporting fields, or gardens. In some regions in Canberra they have been recorded in open woodland near urban centres (Pell and Tidemann 1997a), but they seem unable to penetrate denser bushland found around coastal cities like Newcastle (see chapter 2). They are highly commensal with humans and are rarely found away from human settlements (Sengupta 1968; Hone 1978), although they may extend into some rural areas along roadsides (Hermes 1986). They are probably pre-adapted to the vertical structures and open habitat generally encountered in urban environments due to their evolution in open-woodland habitats (Counsilman 1974a).

Feeding

Common Mynas have an omnivorous, generalist diet, and will consume a wide variety of foods. Their diet consists mostly of arthropods, particularly insects and their larvae, fruit, seeds, food scraps, and some meat or carrion (Wilson 1965; Counsilman 1974a; Moeed 1975; Nee 1992; Crisp and Lill 2006). Outside of Australia, they have also been recorded eating eggs of some species such as Wedge-tailed Shearwaters *Puffinus pacificus* (Byrd 1979) or Seychelles Magpie-Robins *Copsychus sechellarum* (Watson *et al.* 1992), and nestlings of species including Echo Parakeets *Psittacula eques* (Jones 1996), North Island Robins *Petroica Australis longipes* (Armstrong *et al.* 2000), North Island Fantails

Rhipidura fuliginosa placabilis (Blackburn 1966), and Tahiti Flycatchers *Pomarea nigra* (Blanvillain *et al.* 2002); however it is unclear to what extent egg or nestling predation makes up the diet of Common Mynas in Australia. They forage mostly on the ground (Pell and Tidemann 1997a; Crisp and Lill 2006), often in pairs or small flocks (Counsilman 1974b; Hermes 1986).

Reproduction

Breeding occurs during the warmer months, primarily between October and April in the Southern Hemisphere (Counsilman 1974a). Nests are built in cavities, usually found in human-made structures such as walls or ceilings of buildings, bridges, lamps, or posts; and also in natural hollows in branches or tree trunks, or artificial nesting hollows such as nest boxes. Common Mynas are extremely flexible in their choice of nest site, and nests may also be built within an array of additional site types as diverse as traffic lights, tangles of vines, inside a construction crane (Eddinger 1967; Counsilman 1974a; Sengupta 1982), or in the internal walls of shopping centres (pers. obs.). Nests are most often built between two and six metres high (Sengupta 1982), but may be found anywhere between 1.5 and 25 metres off the ground (Counsilman 1974a).

Nests are constructed by both members of a pair and are loose, untidy, bowl-shaped structures with central depressions. They are built with easily obtainable material from the surrounding area, including natural material such as twigs, straw, grass, leaves, feathers, and even snake skins (Ambrose 1982; Sengupta 1982), and artificial material like pieces of paper, plastic, foil, cloth, string, cellophane, or cigarette butts (Wells 1991; Perkins 2000; Fitzsimmons 2001). This material may help young climb up steep cavity walls and provide incubation for eggs or nestlings (Sengupta 1968; McComb and Noble 1981; Harper *et al.* 2005). It has also been suggested that provisioning of nesting material may be part of a competitive strategy used to prevent occupation by other competing species by rendering potential nest sites unusable (Pell and Tidemann 1997b).

Common Mynas are monogamous and are thought to pair for life (Eddinger 1967; Wilson 1973; Sengupta 1982). Both members of the pair incubate and feed nestlings, and several birds in addition to both parents may be involved in the

defence of a nest site or young (Booth 1963; Sengupta 1968; Counsilman 1974a; Fitzsimmons 2001). Nestling mortality is most often due to starvation or predation (Sengupta 1968; Wilson 1973; Counsilman 1974a).

Management and control

Attempts have been made all over the world to remove or control Common Mynas. On a small scale, some measures have been shown to be successful, at least temporarily, in preventing Common Mynas from nesting or roosting in particular areas. Some of the more drastic actions undertaken by authorities include setting off fireworks, broadcasting distress calls or high-pitched soundwaves, and even hoisting a caged cat into the branches of a tree (Laycock 1970). Some reports suggest Common Mynas have been prevented from using nest sites by the installation of a mirror, which serves to trick the nesting bird into thinking the site is already occupied (Eddinger 1967; Wells 1991). Removal of nesting material can delay Common Mynas, but needs to be ongoing as Common Mynas will continually attempt to rebuild destroyed sites (Harper *et al.* 2005).

Significant reduction of numbers is difficult to achieve. Shooting and poisoning of birds has been tried with limited success (Nee *et al.* 1990), and trapping birds using a walk-in baited trap designed specifically for Common Mynas has experienced some success in reducing numbers at a localised level, but is unlikely to be effective on a larger scale (Tidemann 2010). Common Myna numbers are quick to replenish, and reducing their population directly is likely to simply open up vacant niches for new juveniles to establish territories (Phillipps 1994). Attempts at habitat modification have found some success through pruning branches of roost trees (Yap *et al.* 2002), but this approach is unlikely to provide a widespread, long-term solution as Common Mynas can easily move onto unmodified trees or even acclimatise to pruned trees. As with control of any species, these solutions require ongoing and widespread resources and effort (Mack *et al.* 2000).

In August 2011, "competition with native fauna by the Common Myna *Sturnus tristis*" was listed as a "Potentially Threatening Process" in Victoria, a nomination allowing it to be considered for "Key Threatening Process" status. If

found eligible, Common Mynas would be subject to additional regulatory and threat abatement procedures. In August 2012, however, it was deemed ineligible for this listing as it did not satisfy any of the necessary criteria, and was rejected. It was found that this process failed to pose, or to have the potential to pose, “a significant threat to the survival” of either “a range of flora or fauna” or “two or more taxa” (Department of Sustainability and Environment 2012).

Are Common Mynas having an Impact on Native Species?

The focus of this thesis is determining what impact introduced Common Mynas are having on native species in urban centres. Previous literature on this subject is vast, yet findings are often conflicting. I summarise below the current state of our understanding of the impacts of Common Mynas.

Aggression and Competition with Native Species

Aggressive competition over resources has the potential to negatively affect native species at a wide scale, and Common Mynas are frequently reported engaging in aggressive interactions with other species. Blanvillain *et al.* (2002) report "particularly violent" interactions between Common Mynas and Tahiti Monarchs *Pomarea nigra*, involving multiple Common Mynas attacking a single bird or pair of birds, and encounters leading to "both interacting birds falling to the ground, the female (monarch) grasping the head of the Indian Mynah". In Australia, two witnessed accounts (Wright and Wright 1991; Peters and Peters 1993) describe Common Mynas attacking nesting Eastern Rosellas until they abandoned the nest sites. One report (Lindenmayer 1993) recounts an instance of Common Mynas competing with Galahs *Eolophus roseicapillus*, Eastern and Crimson Rosellas, and Common Starlings for access to a desired nesting hollow, and ultimately being successful in utilising it for breeding.

Pell and Tidemann (1997b) conducted a detailed study of competitive behaviour between Common Mynas, Crimson Rosellas, and Eastern Rosellas at nest boxes and natural hollows in savannah and woodland areas near Canberra. They found that Common Mynas were dominant in aggressive encounters with the rosella species over nesting resources, winning more encounters than they lost. Additionally, the authors describe a pair of Common Mynas constructing and subsequently abandoning several nests, with only one of these sites later occupied by a native species, despite high uptake rates at other hollows in the area. The authors suggest that these nests were built as deterrents to other nesting species in order to reduce competition within the territory and maintain territory, and that the strategy appeared to be effective. In further support of this hypothesis, the majority (96%) of nest boxes showed evidence of chewing,

presumed to be an expression of nesting interest by Galahs or rosella species; yet Common Mynas were ultimately the dominant users of the resource.

Grarock *et al.* (2013) report witnessing "interruption" of nesting Crimson Rosellas and Eastern Rosellas by Common Mynas. This involved the occupation of a nest box by a rosella species and subsequent occupation by Common Mynas within a time interval short enough to exclude natural post-fledging vacation of the site by rosellas. Common Mynas interrupted Crimson Rosella nesting 14 times, and Eastern Rosella nesting two times throughout the study. However, the relative frequency of these occurrences, or their comparison to the reverse situation (interruption of Common Mynas by rosellas), is not given.

In a Melbourne study (Crisp and Lill 2006) Common Myna involvement in aggression was found to be "infrequent", with interactions resulting in displacement of only a few metres and Common Mynas being recipients of aggression equally often. A study of the feeding behaviour of New Zealand garden birds in the presence of heterospecifics decoys (Borowske *et al.* 2012) found that decoys representing Common Mynas were considered the least threatening, while fear behaviour was exhibited to Australian Magpies. The authors concluded that Common Mynas were probably not considered a major threat, as instances of aggression were likely to be reasonably rare compared with Australian Magpies.

Nest Predation

Overseas, Common Mynas have been recorded preying on eggs or nestlings of other bird species. In one study Common Mynas in Hawaii were found to take up to 23% of Wedge-Tailed Shearwater *Puffinus pacificus* eggs in one area (Byrd 1979). Destruction of up to 25% of eggs by pecking small holes in the blunt ends has also been recorded in Sooty Terns *Onychoprion fuscatus* on Ascension Island (Hughes *et al.* 2008). A correlation between a high number of encounters between Common Mynas and Tahiti Monarchs around nest sites and nest failure led the authors of another study (Blanvillain *et al.* 2002) to conclude that egg predation was likely, although it was never directly witnessed. Egg predation by Common Mynas was similarly thought to be responsible for nest failure in

Seychelles Magpie-Robins *Copsychus sechellarum* (Watson *et al.* 1992), North Island Fantails *Rhipidura fuliginosa placabilis* (Blackburn 1966), and North Island Robins *Petroica Australia longipes* (Armstrong *et al.* 2000), although again without the incidents being directly witnessed. In New Zealand, Common Mynas have been observed pecking holes in the side of the necks of nestling House Sparrows *Passer domesticus* (Macdonald 1951), and two separate reports describe Common Mynas pulling Common Starling nestlings out of nests by the wing and dropping them to the ground, resulting in their death (Stoddart 1956; Wright 1962). Common Mynas have been observed attacking Black Noddy *Anous minutus* and White Tern *Gygis alba* chicks, and the discovery of chewed Black Noddy chicks were put down to predation by Common Mynas or mice (Grant 1982). Jones (1996) records an instance of a Common Myna attacking a half-grown Echo Parakeet, which later died. Nest predation behaviour in Australia has only been recorded by Pell and Tidemann (1997b), who report seeing common mynas pecking "inside the [nest] box, presumably at the resident chicks or eggs", and later finding all but one of a clutch of six Eastern rosella nestlings had died.

Evidence for Impact on Wildlife

Although aggressive and predatory behaviours have been recorded for Common Mynas, the relative impact of these events at a species or ecosystem level is difficult to ascertain, and evidence seems conflicting. Overseas, negative relationships between Common Myna abundance and abundance of Samoan Starlings *Aplonis atrifusca* led the authors of one study (Freifeld 1999) to suggest that Common Mynas were directly influencing the starlings, probably through competition for nest sites, although it remains equally possible that the trend is merely the result of different habitat preferences. On a New Zealand island (Tindall *et al.* 2007) the removal of Common Mynas coincided with an increase in the populations of 23 other bird species, suggesting their historical decline may have been related to Common Myna introduction. However, rats *Rattus* spp were also removed in this study, and the effect on native species abundances could have been attributed to either removal. In Tahiti, Blanvillain *et al.* (2002) "strongly suggest" that Common Mynas present a particular threat to critically

endangered Tahiti Monarchs, after suspected egg predation by Common Mynas resulted in only approximately a quarter of nests producing young. In Singapore, Huong and Sodhi (1997) speculated that the decline of Oriental Magpie-Robins *C. saularis* in urban areas could be due to competition for nesting cavities with Common Mynas; although as with any correlative study, an association between factors does not provide sufficient evidence of a causative effect.

In Australia, pioneering work was conducted by Pell and Tidemann (1997b). These authors reported on competition between Common Mynas and native parrots in savannah and woodland areas and concluded that, due to "strong circumstantial evidence" of Common Myna success in aggressive encounters with these species, Common Mynas have the potential to reduce their breeding success. However, subsequent studies have shown that these species do not appear to have suffered any decline as a result, and in fact have doubled in some areas of the urban regions they share with Common Mynas over the past century (Loyn and Menkhorst 2011; Davis *et al.* 2012).

Parsons *et al.* (2006) report on surveys of 20 small bird species, including Common Mynas, conducted on 721 gardens in the Greater Sydney region. The authors found that no small bird species were negatively associated with the presence of Common Mynas, and in fact some species, such as Willie Wagtails *Rhipidura leucophrys*, New Holland Honeyeaters *Phylidonyris novaehollandiae* and Superb Fairy-wrens *Malurus cyaneus*, were positively associated with the presence of Common Mynas. The authors concluded that there was "no evidence here to suggest that common mynas are negatively native species" [sic]. Another study in the Sydney region (Lowe *et al.* 2011) found that Common Mynas did not significantly compete with native birds for either food or nesting resources, and did not engage in aggressive interactions more than other species. The authors state that their results "suggest that, in southern Sydney at least, the Common Myna does not have a significant competitive impact on native bird species". Similarly, after finding that Common Mynas did not often engage in interspecific aggression, Crisp and Lill (2006) concluded that they "were not major competitors with co-habiting bird species for food resources through aggressive interference".

A study utilising longitudinal data (Grarock *et al.* 2012) in Canberra over 29 years found a negative relationship between Common Mynas and the long-term abundance of three cavity-nesting species: Sulphur-Crested Cockatoos *Cacatua galerita*, Crimson Rosellas, and Laughing Kookaburras *Dacelo novaeguineae*; as well as seven small bird species: Superb Fairy-wrens, Striated Pardalotes *Pardalotus striatus*, Willie Wagtails, Grey Fantails *Rhipidura albiscapa*, Magpie-larks *Grallina cyanoleuca*, Silvereyes *Zosterops lateralis*, and Common Blackbirds *Turdus merula*. There was no evidence that other cavity-nesting species such as Galahs, Australian King-Parrots *Alisterus scapularis*, Eastern Rosellas, or Common Starlings were affected by Common Mynas. None of the species that appear to be negatively associated with Common Mynas are rare or even threatened, and in fact three are introduced; and no effect on threatened species was found, although this may be due to the limited populations of these species. This study also did not factor in the changes in abundance of Common Mynas, which is particularly relevant given the recent culling of over 45,000 birds (CIMAG 2013) may have created an artificial increase in abundance of other species that was falsely interpreted as a positive relationship with Common Myna establishment. The authors concluded that the effect of Common Mynas on native bird species in the area was not benign, but that "there are still questions regarding the seriousness of this impact and the type of management (if any) that is warranted". A further study by the same authors (Grarock *et al.* 2013) found a negative relationship between Common Myna nest box occupancy and the abundance of Crimson Rosellas and Eastern Rosellas at low tree density sites, and of Crimson Rosellas at high tree density sites. The authors suggest that Common Mynas, in combination with factors such as habitat quality, may limit nesting cavities for native species and contribute to a decline in their abundance.

Public Opinion

The opinion of the public can be instrumental in driving environmental action (Beierle 2005). While many introduced species pose threats to wildlife, Common Mynas appear to be particularly subject to negative attention, and interest in their control and management is high. A poll by the Australian Broadcasting Company found they were considered to be the "most significant

pest/problem", as well as the "pest/problem that needs more control" (http://www.abc.net.au/tv/wildwatch/results/award.htm), ahead of species traditionally considered to pose the greatest danger to native ecosystems such as Cane Toads or European Rabbits *Oryctolagus cuniculus*. The attitude of the public towards Common Mynas is reflected in the variety of nick-names and catchy phrases used, such as "the scavengers of suburbia", "the bullies of the boulevards", the "masked bandits of the metropolis" or the "menace behind the mask" (Phillipps 1994). Other popular references include "flying cane toads" (McCulloch 1997), "garbage birds", "flying rats" (Curtis 2004), or "cane toads with wings" (Dooley 2011). It is not uncommon for exaggerated terminology to be applied to introduced species in cases like this where significant negative opinion exists (Chew and Laubichler 2003).

The reason for this negative attention may be that Common Mynas are highly conspicuous in urban centres due to their large numbers and close association with humans, and their impacts are thus more noticeable to the general public. Common Mynas roost communally (Counsilman 1974b), a behavoiur that results in considerable noise (Eddinger 1967) and faecal mess (Nee *et al.* 1990; Yap *et al.* 2002), as well as the potential health risk of attracting mites, such as the Tropical Fowl Mite *Ornithonyssus bursa* to buildings (Gojrati 1971; Liversidge 1975). In some places they cause agricultural damage, eating or degrading fruit crops, and are known to spread seeds of pest weed species such as Spanish Flag *Lantana camara* (Eddinger 1967) and Olive trees *Olea europaea* (Spennemann and Allen 2000). In addition, the media and various action groups, such as the Canberra Indian Myna Action Group (www.indianmynaaction.org.au), Yarra Indian Myna Action Group (www.yimag.org.au), or Hawkesbury Indian Myna Action Group (www.hawkesbury.nsw.gov.au/environment/natural-environment/indian-myna-control-program-himag), are likely to be responsible for some of the negative image of Common Mynas. CIMAG, for instance, includes on its website a cartoon drawing of a Common Myna kicking a green parrot to the ground and the words "you can have native birds or Indian mynas – but not both" (Fig. 3). HIMAG distributes brochures with highly emotive cartoon images of Common Mynas pulling eggs out of a nest and claiming hollows from possums (Fig. 4).

The media has also shown great interest in publishing stories on the impact of Common Mynas, with one article even misreporting the findings of a study (Grarock *et al.* 2012) to conclude that it provided proof of Common Mynas causing a decline in native species' abundance, when in fact this study showed a correlation between Common Mynas and an increase in native bird abundance, a crucial error pointed out by the paper's authors[1].

Figs. 3 and 4: Examples of depictions of Common Mynas by action groups. Images courtesy of CIMAG and HIMAG.

[1] http://lostinscience.wordpress.com/2012/09/24/myna-inconvenience/

Preface

I began my first study (Chapter 1) with the general intention of documenting aggression by Common Mynas and determining what impact, if any, it had on native species. As results came to light, I was quite surprised to find that Common Mynas displayed comparatively little aggression over food, and that most of the aggression shown was intraspecific (targeting members of their own species) and appeared reasonably harmless. The species responsible for the greatest amount of aggression were native species; Noisy Miners and Australian Magpies. It appeared that the reputation of Common Mynas for aggression, at least when it came to food, was highly exaggerated. These findings generated substantial attention from the media (see Appendix C) and tied in closely to results from a similar study published around the same time (Lowe *et al.* 2011). This study also found that Common Mynas do not display much aggression around food in "natural" foraging situations, as opposed to the more extreme situation of the present study where food was artificially provided and in small, high-density concentrations.

I followed up feeding aggression with an attempt to determine what happens at a more general level (i.e. situations where food or nest resources were not experimentally manipulated) through the use of bird count surveys (Chapter 2). My first aim was to document how much aggressive behaviour Common Mynas engaged in compared with other birds. In addition, I had heard that Common Mynas in Canberra had been shown to penetrate natural remnant habitats, particularly when looking for breeding sites (Pell and Tidemann 1997b), so to investigate this in the Newcastle region I also included surveys of bush sites and edge sites. I was unable to find any evidence of serious aggression from Common Mynas – instead, most noteworthy activity came from Noisy Miners, a native species present in very high numbers. Noisy Miners, unlike Common Mynas, were also frequently recorded penetrating bushland. Noisy Miners are a protected species, and are not currently the subject of control measures, despite their links to significant decline in other native species. This provided an interesting contrast to the Common Myna situation, and led me to explore the concepts of eco-

nationalism and the native-versus-alien dichotomy, a controversial subject that is highly topical in recent literature concerning invasive species.

Having investigated overall behaviours and food related behaviours, I turned my focus towards behaviour exhibited around nest sites (Chapter 3), where most aggression is anecdotally witnessed. Interestingly, I could not find any evidence to indicate significant engagement in aggressive or displacement behaviours by any species at nesting sites, including Common Mynas. It seems likely that, at least in scenarios such as this one where nest sites were not limiting, Common Mynas are not the "bullies" they have been previously thought to be. Taken with the previous two chapters, it seems that the aggressive reputation of Common Mynas has been highly exaggerated, and the vitriol aimed at them by many people is the result of misinformation.

Finally, after an informal chat with Assoc. Prof. Simon Griffith of the Avian Behavioural Ecology Group of Macquarie University, I decided to conduct a survey focussed on the sociological and human-based views of the impacts of Common Mynas (Chapter 4). It was very interesting to document the experience of the public – people often responded with vitriol and hatred towards Common Mynas, with several describing drastic actions they had taken to discourage them from nesting in their nest box, or reporting that Common Mynas had displayed aggression towards native species or driven them away. I wondered if the bias towards Common Mynas was having an indirect effect on native species by preventing homeowners from enjoying the experience of owning a nest box or even leading them to remove the nest box to prevent Common Myna uptake, and ran some analyses to explore this. The results, however, were surprising – those with Common Mynas nesting appeared to get greater enjoyment out of the nest box experience overall, and only a minority of people reported removal of the nest box as a result of Common Mynas. It seemed that, despite overall dislike for the species, attracting Common Mynas was seen as a more interesting alternative to attracting no wildlife at all.

Aims

This project aimed to examine and quantify the impacts of Common Mynas on native species through aggressive competition. This involved examining several aspects: predicting their potential for detrimental impact through studying their range and abundance, and examining their overall aggressive behaviours, competitive aggression over food resources, and aggressive behaviour at nest sites. In addition, I hoped to provide possible management solutions to any impacts discovered.

Due to the varying length of time each study took to complete, the order in which they were submitted for publication does not necessarily follow the logical flow of the below hypotheses. To avoid confusion throughout the body of this thesis I have presented the chapters in the chronological order in which I completed them, rather than the hypothesis order given below, and I have noted in brackets next to each hypothesis the chapter that is concerned with that question.

If Common Mynas were capable of having a serious impact on native wildlife, I predicted that they would:

1) Be present in higher abundance, occupy a greater range of habitats (e.g. suburb, edge, bush), and be observed engaging in interspecific encounters more frequently than other species overall (Chapter 2);
2) Display more interspecific aggression around clumped food resources than other species (Chapter 1);
3) Display more interspecific aggression around nest sites than other species (Chapter 3);

In addition to the above, an analysis of the human and social implications of research on Common Mynas, with particular reference to their behaviour at nest boxes, is explored in Chapter 4.

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Chapter 1: Relative levels of food aggression displayed by Common Mynas when foraging with other bird species in suburbia

Top: Common Myna at food board. Middle: Australian Magpies and Common Mynas at foraging site. Bottom: Australian Magpie and Australian Raven fighting over food. Photos by K. Haythorpe.

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Relative levels of food aggression displayed by Common Mynas when foraging with other bird species in suburbia

Authors: Haythorpe, K.M[A,C]., Sulikowski, D[B]., and Burke, D[A].

[A] University of Newcastle, Ourimbah NSW 2258

[B] Charles Sturt University, Bathurst NSW 2795

[C] Corresponding author: kathryn.haythorpe@newcastle.edu.au

Running head: Food aggression in Common Mynas

Abstract

Invasive species present economic and ecological challenges worldwide. In many cases we are not aware of the full effect they have on the environment, the extent of any damage, or the factors contributing to their success. In this study we examined the foraging aggression of wild Common Mynas (*Sturnus tristis*) as a potential explanation for their invasive success, and quantified the effect of this behaviour on other birds. Common Mynas did not display significantly more aggression than other species, and displayed significantly less aggression than native Australian Magpies (*Cracticus tibicen*). Furthermore, the presence of Common Mynas at a feeding resource had no greater effect on the abundance of heterospecific individuals than the presence of any other species. Presence of each species was negatively correlated with the presence of other species, that is all species were less likely to approach the feeding station if any other species was present there. Common Mynas also did not displace other birds at feeding sites any more frequently than three of the other four species, and less frequently than two other native species. Overall, the findings suggest that Common Mynas do not display more food-related aggression than other species in suburban habitats, suggesting that competitive aggression over food is not likely to be one of the behavioural traits leading to the success of Common Mynas in suburban habitats.

Additional keywords: displacement, foraging behaviour, invasive species, resource competition, *Sturnus*.

Introduction

Damage caused by invasive species continues to be one of the major threats to the health and biodiversity of the natural environment worldwide. Research into invasion biology has previously paid too little attention to the role of behavioural traits in increasing the invasive success of a species. Behavioural strategies, however, are likely play a fundamental role in acquiring resources, which are key to the survival of an introduced species. This may be shown prominently in competition over food, a most important and immediately necessary resource.

Common Mynas (*Sturnus tristis*) were introduced to every continent in the world over the past two centuries (Long 1981) and abound worldwide. Since their introduction to Australia during the 19th century they have successfully colonised most of the Eastern seaboard and are rapidly expanding their range inland (Martin 1996). Their success may be partly attributable to behavioural strategies adopted when competing for resources. However, the extent to which competition for resources influences the success of Common Mynas is presently unknown. Despite their large numbers, and the implementation of various control programs to reduce populations, there are very few quantitative data on the effect of Common Mynas on native ecology. Understanding the effects of the presence of Common Mynas and the mechanisms that contribute to their success are the first steps in developing an effective management plan for this species.

Common Mynas are often thought of as aggressive and 'bullying' (Phillipps 1994). Anecdotal reports abound with accounts of Common Mynas attacking various bird species including White Terns (*Gygis alba*) (Grant 1982), Mauritius Kestrels (*Falco punctatus*) (Feare and Craig 1999), Tahiti Monarchs (*Pomarea nigra*) (Blanvillain *et al*. 2003), and Samoan Starlings (*Aplonis atrifusca*)

(Freifeld 1999). There are even accounts of Common Mynas attacking people, Dogs (*Canis familiaris*), Cats (*Felis catus*) (Booth 1963; Edgar 1975), and a Coati (*Nasua* spp.) (Fitzsimmons 2006). Common Mynas are suspected of driving competing species out of desired nesting hollows and threatening breeding success (e.g. Macdonald 1951; Wright 1962; Counsilman 1974; Wright and Wright 1991; Lindenmayer 1993; Pell and Tidemann 1997).

The large anecdotal record of aggressive tendencies in Common Mynas suggests that aggression could contribute to their invasive success. Numerous other studies have attributed the success of an introduced species at least partly to their ability to compete with native species for local resources (e.g. Petren and Case 1996; Poling and Hayslette 2006; for review see Holway and Suarez 1999). Aggressive exclusion has been shown to be an adaptive and effective means of gaining sole access to a resource, such as food, that may not otherwise be available (Brown 1964). However, information on the aggressiveness of Common Mynas in regards to monopolising food resources is lacking. Counsilman (1974) records some anecdotal information on the aggression of Common Mynas around sources of food, but with insufficient observations to draw firm conclusions. Some authors (e.g. Veerman 2002; Crisp and Lill 2006; Harper *et al*. 2005; Olsen *et al*. 2006; Parsons *et al*. 2006) have suggested that the present focus on eradication of Common Mynas is disproportional to the scientific evidence of the threat they present, and indeed thorough quantification of this threat is so far lacking.

Some studies (e.g. Rotenberry 1980) have shown that species that display aggressive competition are not necessarily the most successful ones. Rather, species that use resources opportunistically and discreetly may have greater success. An invasive species that is lacking experience of its competitors may gain by not entering into physical conflict with them. The cost of resource defence is also high (Brown 1964) and, for an invasive species, with little experience of the long-term value of particular resources, initial success may be more likely if as little energy as possible is expended attacking potentially dangerous natives or defending potentially uncertain resources.

A recent observational study suggests that Common Mynas may adopt such a 'laying-low' strategy. Lowe *et al*. (2011) reports that Common Mynas do not show greater interspecific aggression than other species and rarely interfere with their foraging activity. This survey provides an important indication of normal feeding behaviour, but does not address whether Common Mynas show aggression in more extreme feeding circumstances, where the potential pay-off may be higher. It is important that we determine how effective Common Mynas can be at aggressively outcompeting other species for resources as competition for food can result in dramatic population declines (Coblentz 1978). The potential effect on native birds would be heightened by the remarkably broad diet of Common Mynas, forcing even species with specialist diets to compete with them.

In this study, we looked at the feeding behaviour of wild avian populations containing Common Mynas and native species such as Australian Magpies (*Cracticus tibicen*; hereafter simply Magpies), Noisy Miners (*Manorina melanocephala*), Magpie-larks (*Grallina cyanoleuca*), Australian Ravens (*Corvus coronoides*; hereafter simply Ravens), Crested Pigeons (*Ocyphaps lophotes*), and Silver Gulls (*Larus novaehollandiae*); and introduced species including feral Rock Doves (*Columba livia*), Spotted Turtle Doves (*Streptopelia chinensis*), and Common Starlings (*Sturnus vulgaris*; hereafter simply Starlings). If the success of Common Mynas is partly driven by aggressive competition around valuable food resources, we would predict that Common Mynas (when compared with individuals of other species): (1) would initiate more aggressive interactions towards both other Common Mynas and individuals of other species when foraging at concentrated food sources; (2) displace individuals of other species at food sources at a higher rate than these other species do; and (3) their presence would be associated with a disproportionate reduction in the amount of time other species spend foraging at the food source. A 'laying-low' strategy, however, would not result in such differences.

Method

Procedures

Thirty-eight feeding stations were established in public suburban areas in Newcastle, Australia, spanning an area of 10 400 m^2, with a minimum distance of 256 m between adjacent feeders. Sites included small parks, pathways, street corners, and suburban backyards. Observations were conducted between 0600 and 1100 hours throughout the Common Myna breeding season (September–March) and non-breeding season (April–August). Individual sites were sampled (as described below) between four and seven times over the course of the study.

The feeding stations contained a mix of commercial seed mix (a blend of Sorghum, Wheat, Barley, French White Millet, Maize, Black Sunflower and shell grit), dog pellets, carbohydrate-rich items (cooked white rice, white bread, Madeira cake), finely chopped fruits (sultanas, raisins, and apricots) and live mealworms (*Tenebrio molitor*). Food was placed on a wooden board 30 × 20 × 1 cm. Wild birds were encouraged to feed from it during two pre-feeding periods. For 7 consecutive days after pre-feeding, birds were video-recorded for 5 min of activity, beginning from the time of the first arrival of a bird of any species at the feeding station. On some days no birds arrived, or birds failed to remain for the full 5 min, and data from these days were not used.

For the purpose of analysis we divided the area surrounding the food board into a foraging zone (a circle of 1-m radius surrounding the feeding board) and an outer zone (a circle of 25-m radius surrounding the foraging zone). This allowed us to note birds present in the area but not accessing the food (outer zone), and those interacting in some way with the food or with other birds accessing the food (foraging zone).

As spatial clumping increases aggression (e.g. Goldberg *et al*. 2001; Robb and Grant 1998), we minimised artificially heightened levels of aggression owing to jostling for physical space at the feeding site by providing food on the board and also scattered within the foraging zone. This allowed birds on the periphery of the foraging zone to access food and reduced the effect of clumping.

Analysis

Aggressive acts

Aggressive acts were defined as behaviours directed at another bird that had the potential to cause harm, and included pecking, chasing and swooping, even when such behaviours were ritualistic only. Video-recordings were played back at 0.5× normal speed using VLC Media Player 1.0.3 (VideoLAN, Paris, France) and intraspecific and interspecific aggressive acts were continuously scored for all individuals present in the foraging zone. For each aggressive act we noted the identity of both the initiating and recipient species. This resulted in counts for each species of number of initiated interspecific, initiated intraspecific, and recipient aggressive acts. As the numbers of individuals in the foraging zone fluctuated throughout the study period, an abundance score was calculated for each species at each sampled site by counting the number of individuals of that species at each 30-s interval over 5 min and summing these 10 scores. All aggression data from each site was pooled, regardless of the number of within-site trials, and each site was treated as a replicate.

Species that were present in the foraging zone (of any given site) on fewer than 10 occasions were excluded from that site only for the analysis. We calculated a relative aggression score for each species at each site by first calculating how many aggressive acts we expected each species to be responsible for at each site given the relative abundance of each species and the total number of aggressive acts that occurred at the site. We did this using the formulae:

$$E_{\text{agg}} = (ab_{\text{x}}/AB) \times A$$

and

$$NE_{\text{agg}} = A - E_{\text{agg}}$$

where E_{agg} is the number of aggressive events we would expect species x to initiate; NE_{agg} is the number of aggressive acts we would expect species x *not* to

have initiated; ab_x is the abundance score of species x; *AB* is the sum of the abundance scores of all species present at the site and *A* is the total number of aggressive events observed at the site.

We then calculated the relative aggression score (*RA*) for each species using the formula

$$RA = [(E_{agg} - O_{agg})^2/E_{agg}] + [(NE_{agg} - NO_{agg})^2/NE_{agg}]$$

where O_{agg} is the number of aggressive acts species x was observed to initiate; and NO_{agg} is the number of aggressive acts that species x was *not* observed to have initiated. We also defined the *RA* score as negative if a species was observed to initiate fewer aggressive acts than was expected and positive if a species was observed to initiate more aggressive acts than was expected. The distribution of these scores was similar to the Chi-square distribution and so they were cube-root transformed before analysis to achieve normality.

Relative aggression scores calculated in this manner are not influenced by either the absolute or relative number of a given species at a given site. This means they will not be higher or lower at one site compared with another simply because there are more individuals or a greater proportion of that species at one of the sites. The sign and magnitude of these scores is also meaningful as a score of zero means that a species is behaving exactly as aggressively as the average aggressiveness of other species with which it shares a site. As different combinations of species (and habitat characteristics) can presumably result in situations that are either more or less conducive to aggressive acts, this score also has the benefit that a low *RA* score cannot be attributed to a species simply because it occurs at sites that are relatively unconducive to aggressive acts. This adds validity to between-species comparisons of mean *RA* scores (mean calculated for a species from scores from numerous sites).

We calculated three *RA* scores for each species, reflecting: (1) interspecific acts of initiated aggression; (2) intraspecific acts of initiated aggression; and (3) events in which the species in question was the recipient of aggression. We conducted a one-way analysis of variance (ANOVA) on the cube-root

transformed *RA* values with species as the independent variable. In order to minimise the likelihood of Type-II errors and fully explore the data, we used Fisher's Least Significant Difference (LSD) tests for *post hoc* comparisons. Tukey's Honestly Significant Difference (HSD) tests were also applied to ensure that the use of Fisher's LSD tests did not increase the incidence of Type-I errors.

We then correlated the interspecific *RA* scores and recipient *RA* scores to determine if the species more likely to initiate aggressive acts were less likely to receive them. The analysis was conducted both with and without the *RA* scores of Common Mynas to see if Common Mynas deviated from the trend produced by the other species.

Association

During the 5-minute period of observation, recordings of species present in the foraging and outer zones were made at 30-s time-stops. This method, known as instantaneous time-sampling, is described by Martin and Bateson (1986) and has been widely used and shown to give a very close approximation to actual proportion of time spent in a state as measured by continuous sampling (Dunbar 1976; Leger 1977; Tyler 1979).

For each species pair at each site (e.g. Magpies–Ravens, Magpies–Common Mynas) we first calculated how many of the time-stops (*E*) we would expect to see both species present in the foraging zone if the presence of one species was independent of the presence of the other species. We used the following formula:

$$E = (n_x \times n_y)/n_{xy}$$

where n_x and n_y are the number of time-stops where species x and species y were present in the foraging zone and n_{xy} is the number of time-stops where both species were present in the outer zone. Inherent in this formula is the assumption that if both species are present in the outer zone, they are equally likely to be present in the foraging zone.

We then calculated an association score (*AS*) for each species using the formula:

$$AS = (O - E)/(O + E)$$

where *O* is the number of time-stops at which both species were present in the foraging zone. This created a score that was negative if the species were associating less often than predicted by chance and positive if the species were associating more often than predicted by chance.

We pooled each combination of species pairs by species. Owing to the high proportion of incidences of species pairs never being observed to associate at a given site (resulting in a large proportion of *AS* scores being equal to –1) the data were not suitable for parametric analysis, so we ran a Kruskal–Wallis test by species. Further Kruskal–Wallis pairwise comparisons between different species were conducted *post hoc*.

Displacement

We also recorded displacement events, which involved one or more individual birds leaving the feeding station immediately after the arrival of another bird. This measurement has also been made by Kalinoski (1975) under the name 'avoidance behaviour', and Pryke and Andersson (2003) under the name 'passive supplanting'.

We calculated the number of displacement events we would expect each species to be responsible for based solely on the relative abundance of each species (data were pooled across all sites as the total number of displacement events observed was relatively small, $n = 36$). Expected displacement scores were calculated as:

$$D_{\mathrm{Exp}} = (n/N) \times D$$

where *n* is the total number of individuals of the species in question observed across all sites; *N* is the total number of individuals of *all* species observed across all sites and *D* is the total number of displacement events observed. A Chi-square

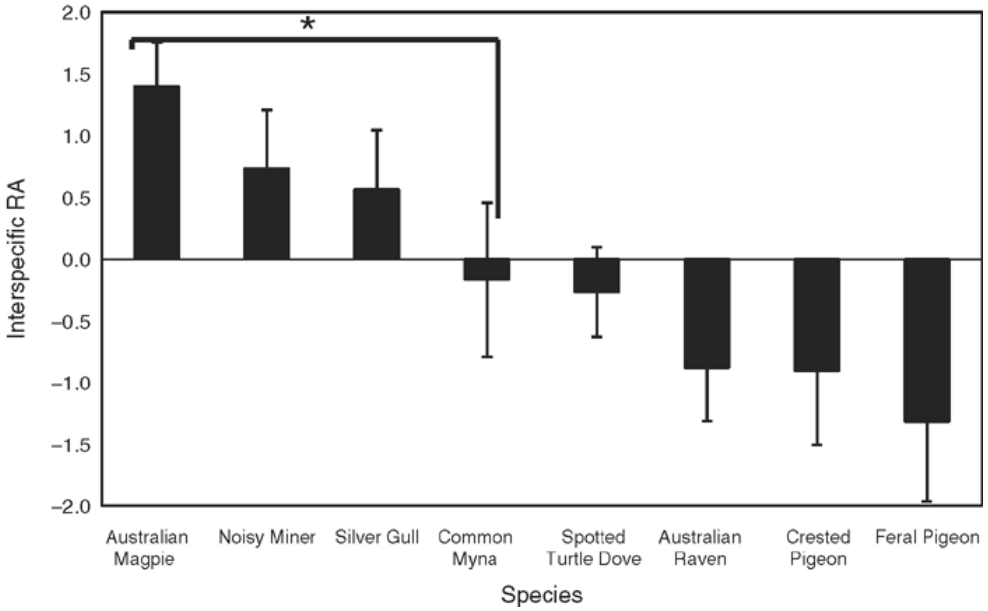
goodness-of-fit analysis was then used to compare the expected number of displacement events with the observed number for each species.

Results

Aggression

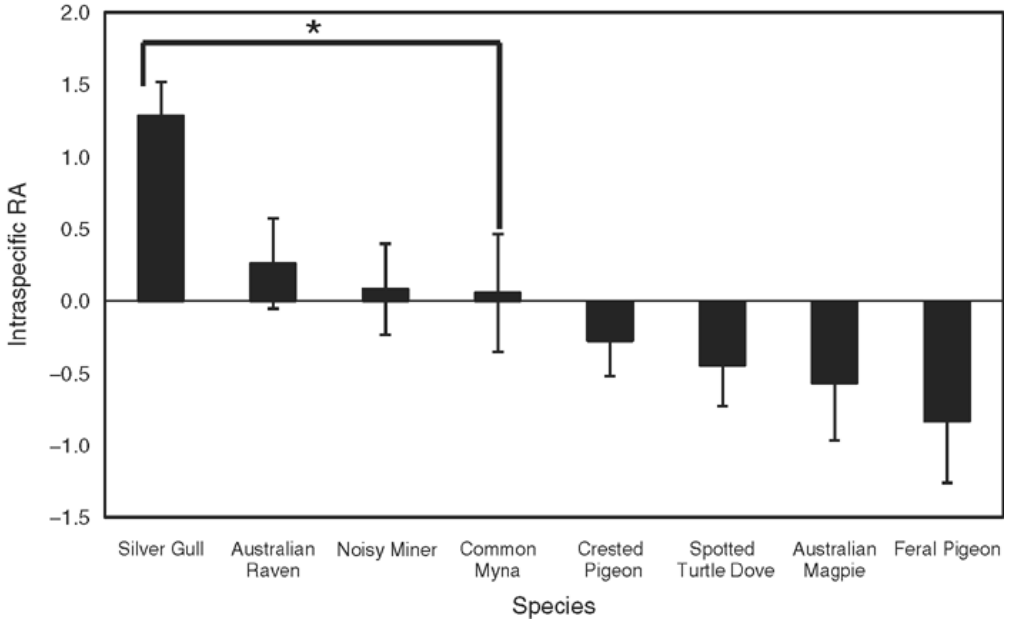
A significant effect of species was found for interspecific ($F_{(7,180)} = 4.382$, $P < 0.001$), intraspecific ($F_{(7,180)} = 3.240$, $P = 0.003$), and recipient ($F_{(1,180)} = 2.292$, $P = 0.029$) aggression scores, suggesting that some species displayed significantly more aggression than others. With respect to interspecific aggression, according to Fisher's LSD, Magpies showed significantly greater levels of aggression than most other species (all $P < 0.025$), including Common Mynas ($P = 0.002$), but not including Silver Gulls and Noisy Miners (all $P > 0.232$). When Tukey's HSD tests were applied, Magpies were still more aggressive than Ravens, Crested Pigeons, feral Rock Doves and Common Mynas (all $P < 0.048$), but no longer more aggressive than any other species (all $P > 0.324$). There were no other significant differences between species (by either *post hoc* test) and Common Mynas ranked fourth overall on aggression, behind Magpies, Noisy Miners and Silver Gulls (Fig. 1).

Fig. 1. Transformed values of interspecific attacks initiated by foraging species. Lines indicate two significantly different values. Only significant differences involving Common Mynas are shown.



With respect to intraspecific aggression, according to Fisher's LSD, Silver Gulls displayed the highest level of aggression, significantly higher than all other species (all $P < 0.044$), including Common Mynas ($P = 0.008$). When Tukey's HSD tests were applied, Silver Gulls were still more aggressive than Magpies, Crested Pigeons and feral Rock Doves (all $P < 0.046$), but no longer more aggressive than Ravens and Noisy Miners (both $P > 0.214$), or Common Mynas ($P = 0.141$). Common Mynas ranked fourth, behind Silver Gulls, Ravens and Noisy Miners (Fig. 2). Thus, Common Mynas showed intermediate amounts of both intraspecific and interspecific aggression, and did not stand out as the most or least aggressive species in either measurement.

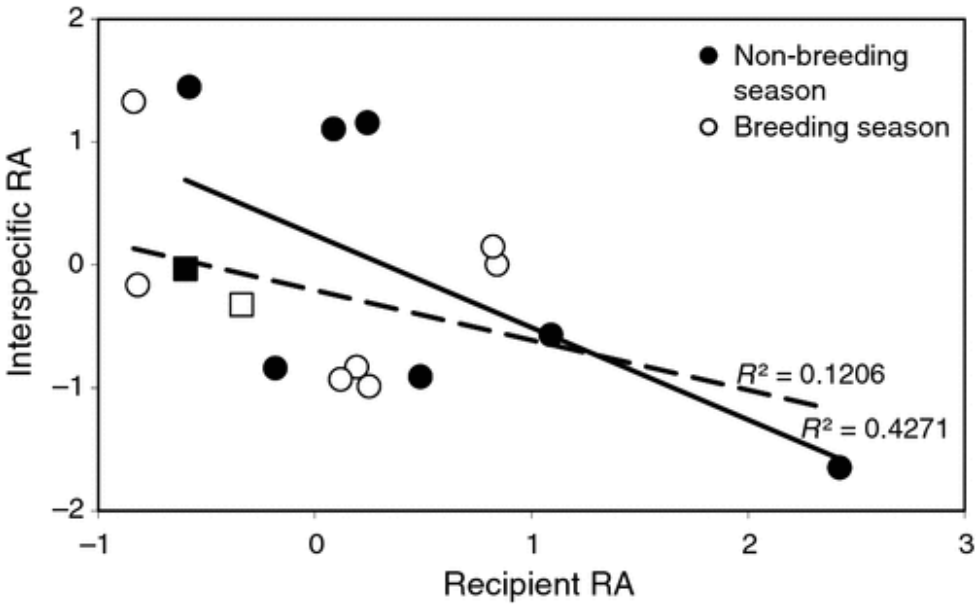
Fig. 2. Transformed values of intraspecific attacks initiated by foraging species. Lines indicate two significantly different values. Only significant differences involving Common Mynas are shown.



With respect to recipient values, feral Rock Doves and Spotted Turtle Doves received the highest levels of aggression respectively. According to Fisher's LSD test both ranked significantly higher (both $P < 0.041$) than Common Mynas, although this effect was lost when Tukey's HSD test was applied. Common Mynas ranked second to last, indicating that they were not one of the highest recipients of interspecific aggression.

For those data collected in the non-breeding season, there was a trend towards a negative correlation between the mean recipient aggression scores and interspecific aggression scores of each species, with a medium to strong effect size ($r = –0.6535$, $n = 8$, $P = 0.079$). The relationship was in the same direction for the breeding season data but did not reach significance ($r = –0.3472$, $n = 8$, $P = 0.399$). These relationships were similar irrespective of whether Common Mynas were included in the analysis or not (with Common Mynas removed; non-breeding: $r = –0.709$, $n = 7$, $P = 0.074$; breeding: $r = –0.370$, $n = 7$, $P = 0.414$; see Fig. 3).

Fig. 3. Negative correlation between interspecific (initiator) and recipient values for breeding and non-breeding seasons. There is a trend indicating that birds receiving fewer attacks are likely to be initiating more attacks, and vice versa. Squares indicate Common Myna breeding season and non-breeding season values.

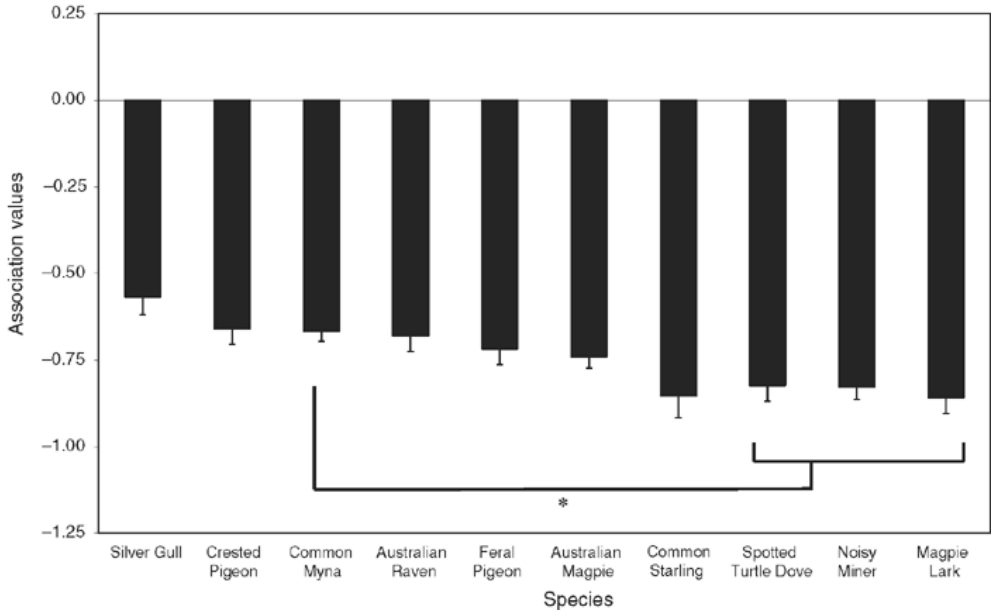


Association patterns

All of the association scores calculated for the species pairs were negative, indicating that, for all species pairs, each species was observed in the foraging zone in the presence of the other species less often than predicted by the number of times both species were present in the outer zone if they were approaching the foraging zone independently. Silver Gulls had the highest association scores, Magpie-larks the lowest, and Common Mynas had the third highest association score. This indicates that Common Mynas were more likely than most of the other species to be found in association with heterospecifics. The Kruskal–Wallis tests revealed a significant effect of species ($P < 0.001$), and the *post hoc* pairwise Kruskal–Wallis comparisons showed that Common Mynas had significantly higher *AS* scores than Spotted Turtle Doves, Magpie-Larks and

Noisy Miners (all $P < 0.002$), and marginally higher than Common Starlings ($P = 0.056$). They were not significantly less likely to associate with heterospecifics than any other species (all $P > 0.220$; Fig. 4).

Fig. 4. Association values of foraging species. Negative association values indicate species are negatively correlated with the presence of other species, and greater negative values indicate decreased likelihood of association with other species. Lines indicate significant differences. Only significant differences involving Common Mynas are shown.



Displacement

Only five species exhibited displacement behaviour, and this behaviour was uncommon. Of 36 displacement events, Magpies were responsible for 58% (21 of 36), and Common Mynas were responsible for only four. Other species displaced eight times (Ravens), twice (Noisy Miners) and once (Silver Gulls). The Chi-square goodness of fit showed that the observed proportion of displacement events by species was significantly different from that predicted by chance ($\chi^2_{(4)} = 52.461$, $P < 0.001$, Yates Chi-square correction applied owing to

some cells having expected value <5). The biggest contributor to this Chi-square value was the Magpie, being responsible for approximately four times as many displacement events as expected.

Discussion

This study examined the importance of behavioural aggression around food resources to the invasive success of Common Mynas, and the potential effect of this on native species in suburban habitats. Contrary to anecdotal suggestions and general public opinion, findings suggest that in situations of high food concentration Common Mynas do not display disproportionately high levels of aggression around food and do not monopolise food sources by displacing or preventing other birds from accessing them, when compared with a variety of native and exotic species with which the Common Myna co-exists. It is, therefore, unlikely that aggressively outcompeting other birds for food resources is the key behavioural trait responsible for the invasive success of Common Mynas.

Our findings complement those of Lowe *et al*. (2011), who reported that Common Mynas rarely interfered with other foraging species, and did not initiate aggressive interactions more than other species. It is important to note that their study was conducted in wild foraging situations, with no artificial manipulation of food resources, thus demonstrating that Common Mynas do not show high levels of aggression even during food scarcity, as would be likely in a natural foraging situation. Our findings show that the other extreme foraging situation, that of providing clumped, artificial food resources where species are foraging in very close proximity to one another, also fails to elicit disproportionately high levels of interspecific aggression from Common Mynas.

Our study also found that, across species, there was a trend towards higher frequencies of initiated aggression being associated with lower frequencies of being the recipient of an aggressive act, a relationship that has been demonstrated in other studies (e.g. Drickamer *et al*. 1999; D'Eath 2002; Løvendahl *et al*. 2005).

Importantly, the behaviour of Common Mynas did not obviously deviate from this trend. Theories of high aggression would predict Common Mynas to fall well above the trendline, indicating a tendency to be involved in disproportionately high levels of interspecific aggression relative to other species. In fact, Common Mynas fell just below the trendline (in both the breeding season and non-breeding season), providing no evidence that they either initiate or elicit aggression more than the other species observed. Common Mynas were only the fifth or fourth most aggressive initiator of the eight species surveyed (depending on whether it was their breeding or non-breeding season), and ranked sixth or eighth of the eight species in likelihood to be the recipient of an aggressive act, in the breeding and non-breeding seasons, respectively. So although Common Mynas may benefit from not being the target of heterospecific aggression, they themselves only exhibit aggression at a rate that is about median when compared with other species with which they occur.

Association scores showed that all species were less likely to be present at the feeder if heterospecific individuals were present. This finding suggests that the birds were actively avoiding heterospecifics, which was somewhat unexpected as it was presumed that birds might use the presence of feeding heterospecifics as cues to the location and profitability of feeding resources. This may still be the case, of course, but with the low association scores being caused by either displacement of one species by another, or passive avoidance of the feeder when heterospecifics are there. Analysis of displacement events revealed that Common Mynas were only responsible for a small proportion of the displacement events, fewer than expected by chance (based on the relative abundance of the various species). Therefore it is unlikely that Common Mynas are actively monopolising food resources through displacement of other birds. Even to the extent that passive avoidance contributed to the latency scores, there was no evidence that Common Mynas were being avoided more than any other species.

Although scientific findings, including ours, have not revealed Common Mynas to be excessively aggressive, anecdotal reports of aggression persist. Possibly, such attacks have occurred mostly while in defence of nesting sites or young. Excessive nestling defence behaviour is common in many species, both native

and introduced (Montgomerie and Weatherhead 1988). Feare and Craig (1999) reported that most Common Myna aggression they witnessed occurred around nesting sites, and only some around food. Other reports of antagonistic behaviour concern aggressive displays from a nesting pair (Booth 1963), and attacks on a Cat during the height of the breeding season in December (Edgar 1975). Airey (1995) reported that Common Mynas attack only during the nesting season, and the rest of the time merely 'scold'.

It may also be that Common Mynas are no more aggressive than other species but that any aggression is more noticeable and obvious to us owing to the strong association Common Mynas have with human activity (e.g. Dean 2000; Peacock *et al*. 2007). Additionally, the status of Common Mynas as an introduced species may lead to a bias in their perceived levels of aggression. Econationalistic ideals held by many Australians decree that introduced, 'outsider' species are unnatural and must necessarily be vicious, greedy and inherently nasty (Trigger *et al*. 2008; Simpson 2010). Native species are seen as gentle and vulnerable, whereas we are quick to expect the worst behaviour of any species seen as foreign to or outside of the natural Australian ecosystem (Franklin 2006). Thus any aggression from Common Mynas is likely to be noted or reported, whereas other native species can exhibit the same behaviour without drawing the same attention from us.

These findings may inform management efforts aiming to assist natives under threat from the expansion of Common Mynas. They suggest that managerial strategies that involve non-targeted or scattered food-provisioning are not likely to assist native species greatly. In fact, given the broad diet of the Common Myna, such activities may result in a net competitive gain for the Common Myna, especially compared with native species with specialist diets, an obviously undesirable outcome. If Common Mynas are negatively affecting native species or causing their decline as suggested (e.g. Pell and Tidemann 1997; Blanvillain *et al*. 2003), they may be doing so through competition for other resources, such as suitable nesting sites.

Conclusion

The findings of this study strongly suggest that Common Mynas do not achieve their success in urban habitats through aggressive monopolisation of food sources, actively displacing native birds or disrupting their ability to access food. Attempts to manage the effect of Common Mynas on native bird populations by provisioning extra food intended for native species may not have the desired outcome. Research efforts should now focus on other ecological factors, such as competition for nesting sites, as the potential ground on which Common Mynas gain their competitive edge over natives.

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Chapter 2: The native versus alien dichotomy: Relative impact of native noisy miners and introduced common mynas

Top: Common Myna, photo by K. Haythorpe. Bottom: Noisy Miner, photo by D. Sulikowski.

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The native versus alien dichotomy:

Relative impact of native noisy miners and introduced common mynas

Authors:

Kathryn M. Haythorpe[A,C]

Darren Burke[B]

Danielle Sulikowski[A]

[A]Charles Sturt University, School of Psychology, Panorama Avenue, Bathurst, NSW 2795, Australia.

[B]University of Newcastle, School of Psychology, 10 Chittaway Road, Ourimbah NSW 2258, Australia.

[C]Corresponding author. Email: kathryn.haythorpe@gmail.com

Abstract

Human activity can dramatically affect biodiversity, often by introducing non-native species, or by increasing the abundance of a small number of native species. Management strategies aimed at conserving biodiversity need to be informed by the actual impacts of highly abundant species, whether native or introduced. In this study we examined characteristics of two bird species, introduced common mynas and native noisy miners, both of which are highly abundant in urbanised areas along the East coast of Australia. Current managerial practices have a strong focus on eradication of common mynas, while noisy miners are largely ignored. However, in this study noisy miners were found in a broader range of habitats, and in greater abundance, than common mynas; displayed more aggressive behaviour; and were linked to a decline in the diversity and abundance of other species where common mynas were not. We suggest that the adaptability of a species and the variety of habitats it can colonise may be a better predictor of its potential impact, than whether it is native or introduced.

Keywords: aggression, *Manorina melanocephala*, species abundance, *Sturnus tristis*
Running head: Common myna and noisy miner impacts

Introduction

Urbanised environments are associated with low species diversity (e.g. Clergeau et al. 2001; Jokimäki et al. 1996; McKinney 2002; Sol et al. 2011) and tend to be dominated by large numbers of a few species that are able to take advantage of the unique conditions experienced in an urban landscape (Bezzel 1985; Kark et al. 2007). These dominant species, termed "urban exploiters" (Blair 1996), typically have generalist diets and habits (Evans et al. 2011) and high sociality (Kark et al. 2007). Urban exploiter birds are often introduced (e.g. McKinney 2002; Orchan et al. 2013), but can also be native species existing in greater abundance than historically recorded (Blair 1996; Emlen 1974; Huhtalo and Jarvinen 1977; Jokimäki et al. 1996; Kark et al. 2007). Management bodies have previously focussed heavily on control of introduced species found in such environments, often attempting to reduce numbers or eradicate the species from entire areas (Gurevitch and Padilla 2004; Orchan et al. 2013; Schlaepfer et al. 2011a; Stromberg et al. 2009). The same is clearly not true for native species, which are generally assumed to have positive effects on the environment (Brown and Sax 2004; Davis et al. 2011). Recent scientific opinion (e.g. Carroll 2011; Davis 2011; Davis et al. 2011; Schlaepfer et al. 2011a; Schlaepfer et al. 2011b), however, suggests managerial bodies shift from the present focus on whether a species is native or not, and towards objective assessment of each species' environmental impact in its particular ecosystem. Objective assessments of relative species impacts are often obtained through investigation into several primary predictors, such as range, abundance, and per-capita or per-biomass effect on parameters such as community structure and population dynamics (Parker et al. 1999).

This paper focuses on the potential impact of two common but unrelated bird species, noisy miners Manorina melanocephala (Meliphagidae) and common mynas Sturnus tristis (Sturnidae). Noisy miners are native, communally-breeding honeyeaters found throughout the southern and eastern parts of Australia (Simpson and Day 2004). Common mynas, which belong to the Sturnid family, were introduced to the eastern coast of Australia in large numbers between 1860 and 1972 (Long 1981), and have become common in cities in this range. Common mynas are obligate hollow-nesting birds, and thrive in regions cleared of natural vegetation. They are heavily associated with human activity, taking advantage of food provided in the form of scraps and rubbish (Crisp and Lill 2006), using gutters and rooves of buildings for nesting (Bomford and Sinclair 2002; Counsilman 1974), and even furbishing their nests with human litter (Counsilman 1974; Sengupta 1968, 1982). Similarly, noisy miners, as edge specialists, have benefited greatly from the clearing of vegetation and segregation of remnant patches of woodland within most of their range in Eastern Australia (Catterall et al. 1991; Clarke and Schedvin 1997a; Low 1994; Loyn 1987), as such habitat appears to be optimal for territorial defence (Taylor et al. 2008); however they require natural vegetation for building cup nests (Dow 1978).

Although common mynas are introduced and noisy miners are native, both species have been accused of creating environmental problems and anthropogenic disturbances. Noisy miners are particularly known for their antagonistic, mobbing behaviour and extreme territorial aggression (Dow 1977, 1979; Grey et al. 1997). Presumably due to these behaviours, noisy miner numbers negatively correlate with the diversity of bird species, particularly small insectivorous birds in suburban gardens (Grey et al. 1997; Grey et al. 1998; Parsons et al. 2006) and woodlands (Major et al. 2001), and their aggressive exclusion of other birds has recently been nominated for inclusion in the EPBC Act list of Key Threatening Processes (DSEWPC 2011). They have even been referred to as a 'reverse keystone' species, as it appears that low abundance or total absence of the species is necessary for high diversity to be sustainable in some areas (Piper and Catterall 2003). They have recently been found to be highly aggressive in competition over artificially provided food sources (Haythorpe et al. 2012; Sol et al. 2011). In addition, they have increased dramatically in abundance over the last few decades (Barrett et al. 2002; Barrett et al. 2007), creating ever greater potential for negative impact on native wildlife.

Common mynas are also thought to pose a threat to native wildlife, and are typically disliked by members of the public (e.g. Nee et al. 1990; Tidemann 2003). They are now one of only three avian species to be listed by the IUCN as among the 'World's 100 worst invasive species' (Lowe et al. 2000), and a poll by the Australian Broadcasting Company found that Australians considered them to be the "most significant pest/problem" (http://www.abc.net.au/tv/wildwatch/results/award.htm). In their native range in India they are known to compete with other hole-nesting species such as ring doves Streptopelia decaocto (Dhanda and Dhindsa 1993) and rose-ringed parakeets Psittacula krameri (Dhanda and Dhindsa

1996), as well as in parts of their introduced range such as Israel (Orchan et al. 2013). On some islands common mynas are thought to be contributing to decreases in abundance of threatened species, such as magpie robins Copsychus spp (Huong and Sodhi 1997; Watson et al. 1992) found on the Seychelles Islands and echo parakeets Psittacula eques (Jones 1996) found on Indian Ocean islands near Madagascar. Artificial removal of common mynas on a New Zealand island was associated with an increase in some native bird species, suggesting they may have played a role in their historical decline (Tindall et al. 2007). However, it does not follow that common mynas would pose the same threat in an ecosystem less fragile than that of a smaller island, and until recently, actual quantitative evidence on their ecological impact, particularly in Australia, has been inconclusive. A long-term study on the impact of common mynas on native species at the population level (Grarock et al. 2012) has recently provided strong evidence to suggest that abundance of some cavity-nesting and small bird species may be threatened by common myna establishment. However, this study also found that the numbers of some species previously thought to be threatened by common mynas, such as Eastern rosellas and common starlings (Pell and Tidemann 1997b), may not be. Given the species thought to be at risk from common mynas are still highly abundant (and three are actually introduced and potential pests themselves), the seriousness of this risk is still to be determined. Others have speculated that the overlap of their range with that of some currently abundant species may cause these species to become threatened in the future (Tracey and Saunders 2003). For instance, Pell and Tidemann (1997b) document a possible threat to native crimson rosellas Platycercus elegans and Eastern rosellas P. eximius through competition for suitable nesting hollows. It should be recognised, however, that while common mynas occupy principally urban environments, Eastern rosellas occupy both urban and woodland habitats, and crimson rosellas in fact prefer to colonise dense forests, including rainforest. Thus far there is no indication that the numbers of crimson or Eastern rosellas is declining in response to common mynas, and in fact both rosella species appear to be thriving (Tidemann 2010; Veerman 2002).

Although both noisy miners and common mynas are considered pests by the general public, considerable attention is given to controlling common mynas in Australia, which are the target of ongoing eradication efforts. Australian governments and local councils have invested funding into developing specialist traps designed to exterminate large numbers of common mynas. Organisations such as the Hawkesbury Indian Myna Action Group (HIMAG), the Canberra Indian Myna Action Group (CIMAG) Inc , the Indian Myna Bird Project (Mid North Coast) , and the Yarra Indian Myna Action Group (YIMAG) Inc , to name just a few, provide information on trapping and humane euthanasia of common mynas and distribute traps to the public. Noisy miners, on the other hand, are a protected species in each state in which they occur in Australia – for example, in New South Wales under the National Parks and Wildlife Act 1974 (Schedule 11), it is illegal to kill, trap or harm them (Section 98). Despite these highly different management strategies, the actual impact of each of these birds is not clear.

There is evidence that both noisy miners and common mynas may potentially be impacting negatively on biodiversity. It is difficult, however, to make judgements about the relative severity of each species' impacts when such judgements need to be based on comparisons between studies, which may employ different methodology and measures and be conducted at different sites. With this consideration in mind, the present study directly compares the three primary indicators of impact – range, abundance, and per-capita effect, in this case focussing on effects on population (abundance) and community (diversity; Parker et al. 1999) – between native noisy miners and introduced common mynas across suburb, edge and bush habitats, and across breeding and non-breeding seasons. We predict that the greatest threat to biodiversity would come from a species that (1) was present in a variety of habitat types; (2) was present in great numbers or was the most abundant species in the area; (3) was linked to a decrease in diversity and/or abundance of other species; and (4) would be frequently observed engaging in aggressive behaviours.

Method

Transect Locations and Descriptions

Forty-five transects were mapped within and around the city of Newcastle, Australia, located approximately 160 kilometres north of Sydney. There were 15 'suburban' transects, situated at least 50 metres from the nearest bushland; 15 'bush' transects that were at least 50 metres into bushland; and 15 'edge' transects placed in regions where bushland of at least 260 hectares met with suburbs (Fig 1). Bushland was found in urban nature reserves, including Awabakal Nature Reserve, Blackbutt Reserve, Glenrock State Conservation Area, Richley Recreation Reserve, Sygna Close Reserve, and Tingira Heights Nature Reserve, and consisted of dense native remnant vegetation, including tall trees and undergrowth. Edge transects ran along the border of these two habitats, extending into the suburbs on one side and into bushland on the other. Selection of transect locations was random in suburban sites, but limited in edge sites to those that were accessible (e.g. not part of a residential property), and in bush sites limited to reserves that allowed public access.

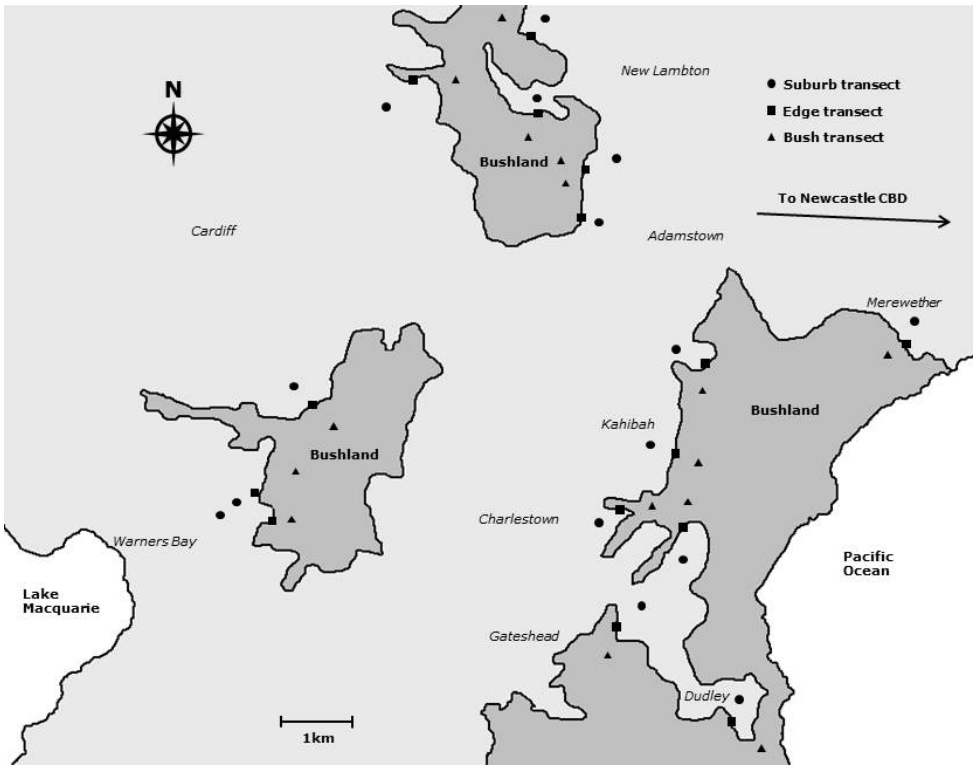

Fig. 1 Map of transect locations across Newcastle, New South Wales. Circles indicate suburban transects, squares indicate edge transects, and triangles indicate bush transects. Italicised names indicate common suburban locations across the city

Each transect was 200m long and contained four observation stops, at 25m, 75m, 125m, and 175m along the transect, respectively. The observation area for each stop was a circle, centred on the transect, with a radius of 25m.

Observation Procedure

Surveys of all 45 transects were conducted every two months for a year, resulting in three surveys (in April, June and August, respectively) being conducted during the non-breeding season and three surveys (in October, December and February, respectively) during the breeding season of most birds. These are referred to throughout this paper as the observations.

Each survey of a transect totalled 20mins of observation, with a sampling time of 5mins at each of the four stops. Sampling time was based on previous work by Blair (Blair 1996, 2004), and initial pilot studies in the study areas determined that practically all observations (>95%) were recorded within the first 2mins of sampling, a trend that continued throughout the rest of data collection; thus 5mins was considered to be sufficiently long to pick up the majority of relevant behavioural observations. From the centre of the stop, the observer recorded the numbers and species of all birds seen and heard within the observation area, excluding those that were only flying overhead and not spending any other time at the stop. For each survey, the observer also recorded whether or not each bird species initiated an aggressive act on another bird. An

aggressive act was defined as a swoop, peck at, physical fight with, or chase of another bird, regardless of whether this resulted in the other bird leaving the area. To avoid bias in number of aggressive interactions due to increased number of any one species, multiple individuals engaging in an aggressive encounter were only counted once, and noisy miners and common mynas were recorded in a yes/no fashion as either engaging in aggressive behaviour or not engaging in aggressive behaviour for each stop, thus multiple attacks in one visit were given the same weight as only one attack.

Overall Abundance and Species Diversity

Species diversity was defined as the total number of species observed and overall abundance was defined as the total number of individuals observed. Both of these measures were calculated separately for each transect (by combining observations over the four stops), during each of the six two-monthly surveys. The scores from the three breeding-season and three non-breeding-season surveys, respectively, were then averaged so that each transect was left with one species diversity score and one overall abundance score for each season.

Transect diversity was defined as the total number of species observed at that transect, tallied over the six surveys of the study. Note that transect diversity is not simply a sum of the six species diversity scores recorded during the six surveys, as a single species observed during four of the surveys, for example, counted as just one species for the purposes of the transect diversity measure, not as four species. Transect abundance was defined as the total number of individuals present at that transect, tallied over all six surveys. This measure is simply a sum of the six overall abundance scores recorded for each transect.

To explore the effect of habitat type on overall abundance and species diversity, we conducted repeated-measures ANOVAs, using each of the 45 transects as the basic replicates of the analyses. Habitat type was entered as a between-transect factor (3 levels: suburb, edge and bush), and season as the within–transect factor (2 levels: breeding and non-breeding).

Core Species

For analyses involving all subsequent dependent variables we included only species that were present in at least 5 of the 15 transects for each habitat type. This elimination process resulted in 15 core species for the suburb (including both noisy miners and common mynas), 19 core species for the edge (which included noisy miners but not common mynas), and 20 core species for the bush (also including noisy miners, and not common mynas).

Absolute Abundance and Relative Abundance Scores

Absolute abundance (the total number of sightings of a given species) was recorded at every transect for each core species of each habitat type. Absolute abundance scores for each core species during the breeding and non-breeding seasons were calculated for each transect by

summing absolute abundance totals observed at that transect during the October, December and February surveys and the April June and August surveys, respectively. An absolute abundance score that combined data from the breeding and non-breeding seasons was also calculated for noisy miners and common mynas only.

Relative abundance scores for each core species at each transect (for the breeding and non-breeding seasons separately, as well as both seasons combined for noisy miners and common mynas as described above) were also calculated using the formula:

$RA_x = AA_x - AA_{\sum}/n$

where RA_x is the relative abundance of species x, AA_x is the absolute abundance of species x, $AA_{\sum}$ is the sum of all core species' absolute abundance scores at that transect, and n is the number of core species observed at that transect. The resulting score was negative if the number of a species was lower than the mean for that transect and positive if the number of a species was higher than the mean for that transect. Species with an absolute abundance score of zero for a given transect (indicating the species was never observed at that transect) were still allocated a relative abundance score for that transect as per the above formula.

Results

Effects of habitat type

Overall abundance did not differ significantly between the habitat types ($F_{(2,42)}$=0.576, p=0.567), or between the breeding and non-breeding seasons ($F_{(1,42)}$=1.069, p=0.307), nor was it affected by an interaction between these variables ($F_{(2,42)}$=1.229, p=0.303; Fig 2a). Species diversity was significantly different across habitat type ($F_{(2,42)}$=3.688, p=0.033). Post hoc contrasts showed both bush (p=0.04953) and edge (p=0.013) habitats had greater diversity than suburban habitats, but bush habitats did not differ from edge habitats (p=0.579). There was also no main effect of season on species diversity ($F_{(1,42)}$=1.409, p=0.242) and no interaction between season and habitat type ($F_{(2,42)}$=0.312, p=0.734; Fig 2b).

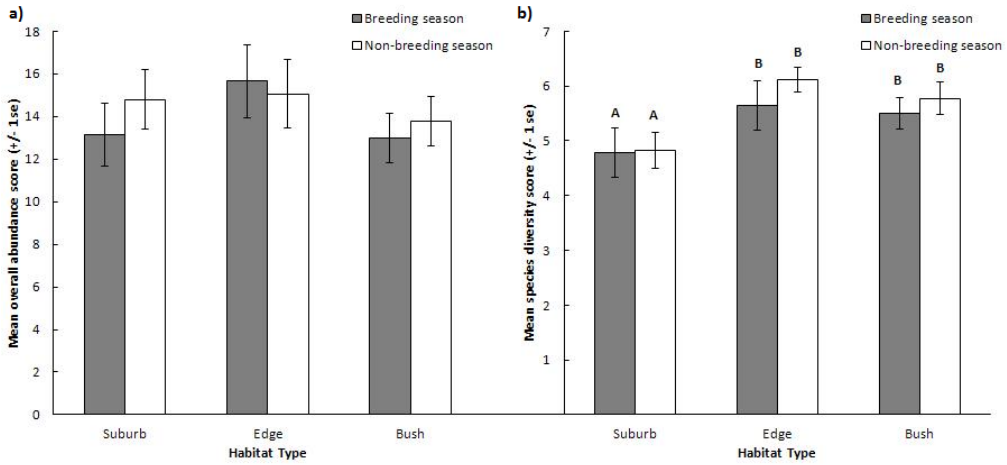

Fig. 2 Shows the overall abundance and species diversity across the different habitat types. Overall abundance was not significantly affected by either habitat or season (a), while species diversity was significantly lower in the suburb than in either the edge or the bush, although this effect was not seasonal (b). Different uppercase letters (A, B) indicate significant differences between variables

Noisy miners were the most abundant species recorded in suburb (333 sightings) and edge (374 sightings) habitats. They were also one of the most abundant species sighted in the bush (157 sightings), outnumbered only by small bush specialist wren species, white-browed scrubwrens *Sericornis frontalis* (193 sightings) and superb fairy-wrens *Malurus cyaneus* (233 sightings). Common mynas by contrast were not present in bush habitats, and were present in only three of 15 edge habitat transects, coming to a total of only six sightings. They were, however the second most abundant species (139 sightings) in suburban habitats after noisy miners.

Repeated-measures ANOVAs were used to investigate the effect of habitat type on absolute abundance and relative abundance of noisy miners. Each transect was the basic replicate of the analysis, and habitat type was entered as a between-transect factor (3 levels: suburb, edge and bush) and season (2 levels: breeding and non-breeding) as a within-transect measure.

Noisy miner absolute abundance was not affected by season ($F(1,42)=1.729$, $p=0.196$) but differed significantly between the habitat types ($F_{(2,42)}=3.229$, $p=0.0496$). *Post hoc* simple contrasts showed a significant drop in numbers occurred in bush habitats compared with edge habitats ($p=0.021$), but did not occur between suburb and bush ($p=0.059$), or between edge and suburb ($p=0.654$; Fig 3a). The relative abundance of noisy miners was not affected by season ($F_{(1,42)}=1.366$, $p=0.249$) nor by an interaction between season and habitat type ($F_{(1,42)}=0.590$, $p=0.559$), but the main effect of habitat type did approach significance ($F_{(2,42)}=3.070$, $p=0.057$). *Post hoc* simple contrasts confirmed that noisy miner relative abundance was significantly higher in the edge compared to the bush ($p=0.021$) habitats and also tended to be higher in the suburb

compared to the bush (p=0.091), but did not differ between the suburb and the edge (p=0.507; Fig 3b).

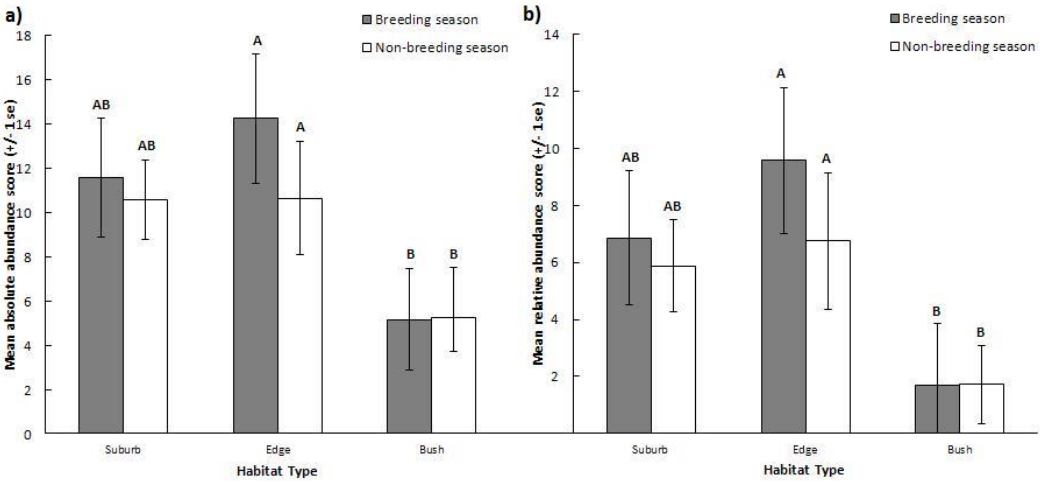

Fig. 3 Shows the noisy miner relative and absolute abundances across habitat types. Both relative (a) and absolute (b) abundances were significantly higher in edge sites than in bush sites, and this was not affected by season. Different uppercase letters (A, B) indicate significant differences between variables

Since 96% of common myna sightings occurred in suburban transects, formal analyses comparing common myna abundance across the three habitat types were not conducted, however, paired-samples t-tests were used to compare absolute and relative abundance of common mynas between the breeding and non-breeding seasons. Neither relative abundance ($t(14)=1.006$, $p=0.331$), nor absolute abundance ($t(14)=1.005$, $p=0.332$) of common mynas differed between the breeding and non-breeding seasons in the suburban habitat.

Transect diversity

Given the nil effect of breeding season in the previous analyses, the species diversity scores from each of the six surveys of the study were combined to produce a transect diversity score, reflecting the total number of species observed at a given transect throughout the study period. Pearson's bivariate correlations were then used to correlate these transect diversity scores with common myna relative abundance and absolute abundance across the 15 suburban transects. The same analyses were conducted with noisy miner relative abundance and absolute abundance across all transects at each of the three habitat types.

With respect to common mynas, neither absolute abundance ($r=0.036$, $n=15$, $p=0.898$) nor relative abundance ($r=0.007$, $n=15$, $p=0.979$) was significantly correlated with transect diversity in suburban habitat. The same was true for noisy miner absolute abundance ($r=-0.034$, $n=15$, $p=0.904$) and relative abundance ($r=-0.066$, $n=15$, $p=0.816$) in the suburban habitat and also for absolute abundance ($r=-0.087$, $n=15$, $p=0.757$) and relative abundance ($r=-0.097$, $p=0.731$) in the

bush habitat. There were, however, significant negative correlations between noisy miner absolute abundance (r=-0.636, n=15, p=0.011) and relative abundance (r=-0.680, n=15, p=0.005), respectively, and transect diversity in the edge habitat, confirming that as noisy miner numbers increased, and they became a larger proportion of the bird life, the number of other species observed dropped.

Transect abundance

As described above for the species diversity scores, the overall abundance scores were combined across the six surveys resulting in each transect having a transect abundance score (which was the total number of individual birds observed at that transect over the entire study period). For the purposes of correlating the transect abundance scores with noisy miner and common myna abundance scores, as described below, the numbers of noisy miners and common mynas, respectively, were subtracted from the transect abundance scores, such that each correlation examined the association between abundance of a focal species (either noisy miners or common mynas) and abundance of all *other* core bird species.

Pearson's bivariate correlations were used to correlate transect abundance scores with noisy miner absolute abundance scores in suburb, edge and bush habitats, and with common myna absolute abundance scores in the suburb only. There was no correlation between noisy miner absolute abundance and transect abundance in suburb (r=0.174, n=15, p=0.534), edge (r=-0.275, n=15, p=0.322) or bush (r=-0.068, n=15, p=0.809) and also no correlation between common myna absolute abundance and transect abundance in the suburb (r=0.413, n=15, p=0.126). The same analyses were conducted using relative (rather than absolute) abundance scores of noisy miners and common mynas and these also failed to reveal any significant correlation for common mynas in the suburb (r=0.267, n=15, p=0.335) or for noisy miners in the suburb (r=0.089, n=15, p=0.752), edge (r=-0.358, n=15, p=0.191) or bush (r=-0.169, n=15, p=0.547).

Core species' relative abundances

To examine the association between the relative abundance of noisy miners and common mynas, respectively, and the relative abundance of other species with which they shared a habitat, we conducted a series of Pearson's bivariate correlations between each of the core species' relative abundance scores, and noisy miner relative abundance scores in each of the three habitat types and common myna relative abundance scores in suburban habitats (see Tables 1 & 2).

Table 1 List of core species of the suburb habitat, and corresponding r-values derived from correlating these species' relative abundance scores with those of the noisy miner and common myna across the 15 suburban transects (n=15 for all correlations). Number in brackets after species name is the total sightings of that species in the suburban habitat

Core species		**Noisy miners (*Manorina melanocephala*)**		**Common mynas (*Sturnus tristis*)**	
Common name	Scientific name	Breeding Season	Non-breeding season	Breeding season	Non-breeding season
Australian magpie (n=122)	*Cracticus tibicen*	-0.614	0.026	-0.384	-0.370
Australian raven (n=42)	*Corvus coronoides*	-0.527	-0.300	0.039	-0.142
Common myna (n=139)	*Sturnus tristis*	0.069	-0.425	-	-
Crested pigeon (n=81)	*Ocyphaps lophotes*	-0.636	-0.503	-0.006	-0.303
Crimson rosella (n=14)	*Platycercus elegans*	-0.728	0.152	-0.261	-0.190
Eastern rosella (n=35)	*Platycercus eximius*	-0.616	-0.002	-0.185	-0.419
Grey butcherbird (n=5)	*Cracticus torquatus*	-0.784	-0.374	-0.288	0.012
Laughing kookaburra (n=42)	*Dacelo novaeguineae*	-0.381	0.214	-0.644	-0.196
Magpie-lark (n=22)	*Grallina cyanoleuca*	-0.850	-0.292	-0.055	-0.056
Noisy miner (n=333)	*Manorina melanocephala*	-	-	0.069	-0.425
Pied currawong (n=21)	*Strepera graculina*	-0.564	-0.144	-0.509	-0.197
Rainbow lorikeet (n=80)	*Trichoglossus haematodus*	-0.352	0.407	0.008	0.211
Red wattlebird (n=37)	*Anthochaera carunculata*	-0.249	-0.227	0.429	0.022
Sulphur-crested cockatoo (n=37)	*Cacatua galerita*	-0.481	-0.259	0.060	-0.409
Spotted dove (n=113)	*Streptopelia chinensis*	-0.675	-0.355	-0.097	0.689

Table 2 List of core species of the edge and bush habitats and corresponding r-values derived from correlating these species' relative abundance scores with those of the noisy miner across the 15 transects of each habitat type (n=15 for all correlations). Missing values are due to not all species being a core species in both habitat types. Numbers in brackets after species name is the total sightings of that species in edge and bush habitats respectively

Core species		**Edge Habitat**		**Bush Habitat**	
Common name	Scientific name	Breeding Season	Non-breeding season	Breeding season	Non-breeding season
Australian magpie (n=107, 29)	*Cracticus tibicen*	0.046	0.000	-0.148	-0.217
Australian raven (n=27, 39)	*Corvus coronoides*	-0.675	-0.202	-0.172	-0.449
Black-faced cuckoo-shrike (n=18, 16)	*Coracina novaehollandiae*	-0.677	-0.104	-0.298	-0.320
Crested pigeon (n=33, -)	*Ocyphaps lophotes*	-0.512	-0.233	-	-
Crimson rosella (n=20, -)	*Platycercus elegans*	-0.341	-0.194	-0.314	-0.374
Eastern rosella (n=75, 23)	*Platycercus eximius*	-0.087	0.262	-0.142	-0.059
Eastern spinebill (n= -, 14)	*Acanthorhynchus tenuirostris*	-	-	-0.339	-0.653
Eastern whipbird (n=15, 20)	*Psophodes olivaceus*	-0.822	-0.337	-0.136	-0.103
Golden whistler (n= -, 18)	*Pachycephala pectoralis*	-	-	-0.396	-0.408
Grey butcherbird (n=18, 14)	*Cracticus torquatus*	-0.633	-0.253	-0.138	0.082
Grey fantail (n=20, 56)	*Rhipidura albiscapa*	-0.803	-0.757	-0.324	-0.580
Lewin's Honeyeater (n=9, 40)	*Meliphaga lewinii*	-0.738	-0.594	-0.507	-0.555
Laughing kookaburra (n=59, 51)	*Dacelo novaeguineae*	-0.081	-0.147	0.216	-0.057
Pied currawong (n=32, 30)	*Strepera graculina*	-0.639	-0.055	-0.271	0.067
Rainbow lorikeet (n=92, -)	*Trichoglossus haematodus*	0.049	0.348	-	-
Red-browed finch (n= -, 33)	*Neochmia temporalis*	-	-	-0.362	-0.399
Red wattlebird (n=41, 9)	*Anthochaera carunculata*	-0.733	-0.383	-0.307	-0.190
Spotted dove (n=72, 6)	*Streptopelia chinensis*	-0.484	-0.336	-0.220	-0.592
S'-crested cockatoo (n=19, 17)	*Cacatua galerita*	-0.664	-0.370	-0.182	-0.200

Superb fairy wren (n=118, 233)	*Malurus cyaneus*	-0.694	-0.700	-0.442	-0.111
White-browed scrub wren (n=66, 193)	*Sericornis frontalis*	-0.705	-0.599	-0.210	-0.116

The resulting sets of r-values were converted to z'-values (to achieve normality) using Fisher's formula:

$z' = 0.5 \times (\log(1+r)/(1-r))$

To determine whether the relative abundance of either noisy miners or common mynas exhibited a stronger association with the relative abundance of the other core species across the 15 suburban habitat transects (and whether the strength of the associations were affected by season), we conducted a repeated-measures ANOVA on the resulting suburban habitat z' scores with season (2 levels: breeding and non-breeding) and species (2 levels: noisy miner and common myna) both entered as repeated-measures). The z' value derived from the direct correlation between noisy miner and common myna relative abundance was excluded.

Noisy miner z' scores were significantly more negative than common myna z' scores ($F_{(1,12)}$=7.845, p=0.016) and breeding season z' scores were significantly more negative than non-breeding season z' scores ($F_{(1,12)}$=14.013, p=0.003). Both these main effects were qualified by a significant species by season interaction ($F_{(1,12)}$=17.879, p=0.001) with *post hoc* paired comparisons confirming that the noisy miner breeding season z' average was significantly more negative than the noisy miner non-breeding season z' average (t(12)=6.488, p<0.001) and the common myna breeding (t(12)=5.075, p<0.001) and non-breeding (t(12)=4.690, p=0.001) averages. No other post-hoc comparisons were significant (all p>0.6). One-sample t-tests also confirmed that only the noisy miner breeding season z' average was significantly less than zero (t(12)=8.893, p<0.001), with no difference found for noisy miner non-breeding (t(12)=1.714, p=0.112) common myna breeding (t(12)=1.856, p=0.088) or non-breeding (t(12)=1.032, p=0.322). Overall, the results demonstrate that, during the breeding season only, higher numbers of noisy miners were significantly associated with lower numbers of other species of birds, with no such association demonstrated for common mynas (Fig 4a).

To further investigate the association between noisy miner abundance and the abundance of other core species, we conducted another repeated-measures ANOVA on the z' scores calculated for associations between noisy miners and all other core species in the suburb, edge and bush habitats. Season (2 levels: breeding and non-breeding) was entered as a repeated-measure and habitat (3 levels: suburb, edge and bush) was entered as a fixed factor. No significant main effect of habitat was observed ($F_{(2,48)=}$2.270, p=0.114) but there was a significant main effect of season ($F_{(1,48)}$=36.709, p<0.001) and a significant season by habitat interaction ($F_{(2,48)}$=13.856, p<0.001). *Post hoc* paired comparisons confirmed that the negative association between abundance of noisy

miners and other core species was significantly stronger in the breeding season for the suburb ($t(13)=4.414$, $p=0.001$) and edge ($t(17)=5.341$, $p<0.001$) habitats only, with no difference between seasons in the strength of this association found in the bush ($t(18)=0.821$, $p=0.422$). One-sample t-tests confirmed that the z' average was significantly less than zero (signifying a significantly negative average association between the abundance of noisy miners and that of other core species) across all three habitats in the breeding season (suburb: $t(13)=7.067$, $p<0.001$; edge: $t(17)=6.906$, $p<0.001$; bush: $t(18)=6.722$, $p<0.001$) and in the edge ($t(17)=3.536$, $p=0.003$) and bush ($t(18)=5.042$, $p<0.001$), though not the suburb ($t(13)=2.056$, $p=0.060$) in the non-breeding season (Fig 4b).

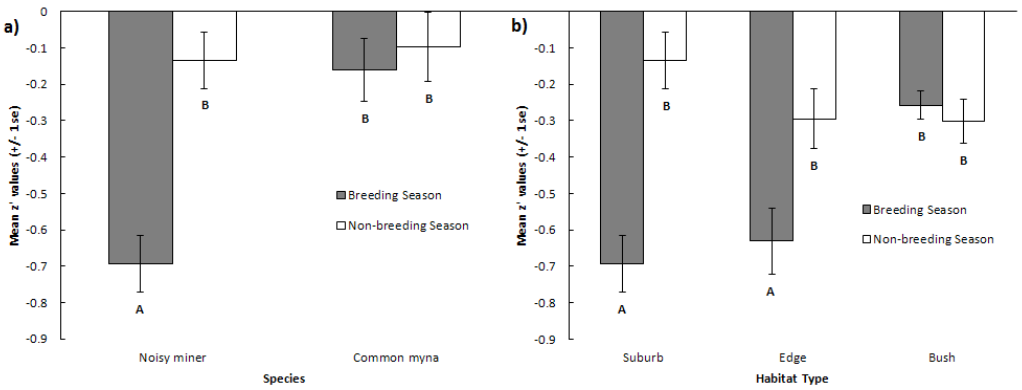

Fig. 4 Shows the z'-values for noisy miners and common mynas in suburban habitats only (a), and the z'-values for noisy miners only across all habitat types (b). Noisy miner z'-values were significantly lower during the breeding season than during the non-breeding season, and during the breeding season were significantly lower than common myna z'-values in either season (a). Noisy miner z'-values were also significantly lower in both suburb and edge habitats during the breeding season than during the non-breeding season, and lower in both edge and suburban habitats during the breeding season than bush habitats in either season. Different uppercase letters (A, B) indicate significant differences between variables

Aggression

All sightings of common mynas and noisy miners in suburban habitats and the respective number of aggressive acts conducted by each species in this habitat were tallied. Common mynas were recorded conducting an aggressive act on two occasions out of 139 sightings (1.4%), while noisy miners conducted aggressive acts on 53 out of 333 (15.9%) sightings in suburban habitats. A Fisher's 2x2 Exact Test (with $\alpha=0.05$) demonstrated a significant association between species and incidence of aggression ($p<0.001$) with noisy miners significantly more likely to display aggression than common mynas within the suburban habitats.

As noisy miners occurred across all habitats, all sightings in each of the three habitat types and respective numbers of aggressive acts conducted by noisy miners in these habitats were also

tallied. In addition to the 53 aggressive acts (of 333 sightings, 15.9%) in suburban habitats, 57 aggressive acts out of 374 (15.2%) sightings in edge habitats, and 14 aggressive acts out of 157 sightings in bush habitats (8.9%) were recorded. A Pearson's chi-square test of contingencies (with α=0.05) was used to determine whether habitat type was associated with the number of aggressive acts conducted by noisy miners. There was no association between frequency of noisy miner aggression and habitat type ($\chi^2_{(2)}$=4.7, p=0.097).

Discussion

This study examines the potential threat posed by common mynas, an introduced Sturnid, and noisy miners, a native honeyeater, by looking at their range and abundance, and investigating their potential and actual per-capita effect through examination of behaviour and relationship to the diversity and abundance of other species (Parker et al. 1999), across habitat types and seasons. Although mathematically quantifying these parameters as indicators of a species' total impact is difficult (e.g. Thiele et al. 2010; Thomsen et al. 2011), a broad examination could be expected to reveal important trends. Direct comparison of these two species within the one study area allows us to make judgements about the relative severity of the impacts of each species without the confounds introduced by trying to compare across sites and across studies.

In this study overall abundance did not differ by habitat type, however species diversity was found to be higher in bush and edge habitats than in suburban habitats. These results were not affected by season. Greater levels of diversity in remnant or bushland areas compared to urbanised areas have been found by numerous other studies (e.g. see Blair 1996; Chace and Walsh 2006; McKinney 2008 for reviews).

Noisy miners were found in all three habitat types in high numbers across seasons. They were the most abundant species in both suburb and edge habitats, and third most abundant in bush habitats. These findings are not surprising, as an affinity for edge areas and human modified habitats has been shown by numerous other studies (e.g. Catterall et al. 1991; Grey et al. 1997; Hastings and Beattie 2006; Major et al. 2001) and their ability to penetrate even dense bushland is well known (Clarke and Oldland 2007; Eyre et al. 2009; Howes and Maron 2009). In parts of southern Queensland they are even regularly recorded penetrating more than 20km into bushland (Maron 2009).

Common mynas by contrast were found primarily in suburban habitats, where they were the second most abundant species after noisy miners, and only rarely in edge habitats, where they probably occupied the more suburban side of the transect. This finding is not surprising, as common mynas are well known as urban habitat specialists that rely heavily on human activity (e.g. Counsilman 1974; Crisp and Lill 2006; Pell and Tidemann 1997a; Sengupta 1968). Contrary to predictions made by Pell and Tidemann (1997b), common mynas in this study did not appear

to be capable of penetrating bushland. It should be noted that the bushland surrounding the Newcastle area in which this study was conducted differs dramatically from the sparse, open woodland found around the Canberra region where studies by Pell and Tidemann (1997b) and Grarock et al. (2012) took place, and may explain the difference in these findings.

A negative correlation between noisy miner abundance and species abundance and diversity has been shown by numerous other studies across Australia, including the coastal city of Sydney (Parsons et al. 2006), the inland wheat belt of New South Wales (Major et al. 2001), the foothills of the Great Dividing Range in Victoria (Grey et al. 1997), and a multitude of sites ranging from Victoria to Queensland (Mac Nally et al. 2012). Noisy miner abundance in this study was negatively associated with species diversity in edge habitats. During the breeding season in both the suburb and edge habitats, an increase in noisy miner relative abundance was associated with a decrease in abundance of other species. This effect was not observed during the non-breeding season. Noisy miners are well known for aggressive nest defence behaviour via group mobbing (Arnold 2000; Maron 2009), and it may be that the prominence of this effect during the breeding season is due to differences in territorial behaviour in the presence of nestlings or fledglings. As noisy miners were less abundant in bush habitats than edge habitats, and associated with decreased species diversity in edge habitats, it may be that noisy miners are primarily using edge habitat for breeding, and aggressively excluding other species most prominently in this location. We did not observe a difference in aggression levels across habitats, however, which is inconsistent with this interpretation. We examined aggression only at a fairly coarse level, though, and more detailed recording of the specifics of aggressive acts, such as their nature and efficacy across various habitats, might be informative.

We found no evidence that common myna abundance was associated with diversity or abundance of other species in the suburban habitats where they were found, but no comparable analysis could be conducted in bush or edge habitats as presence of common mynas in these was extremely low. In a study of suburban gardens in Sydney, Parsons et al. (2006) also found common mynas were not associated with a decrease in species diversity. However, this finding is contradicted by more recent findings from Canberra (Grarock et al. 2012). Several possibilities exist here to explain this difference in results. First, it may be that the habitats of Newcastle and Sydney are less conducive to effective competition or exclusion by common mynas than that of Canberra due to different environmental factors. For example, Newcastle and Sydney are both coastal cities that receive high levels of rainfall, which supports substantially more vegetative growth than the inland city of Canberra. It may be that this increase in vegetation gives native birds in Newcastle and Sydney a competitive edge that birds in Canberra do not have. Canberra also frequently experiences lower temperatures during winter than either of the coastal cities, and common mynas may be better able to withstand this than some native birds, again providing a competitive edge. Aside from potential differences in environmental factors, the different methodologies of the studies need to be considered. Grarock et al.'s (2012) findings consider an

extensive dataset, spanning 29 years and including over 74,000 surveys, while our study and that of Parsons et al. (2006) are much smaller and span one year or less. A finding of impact by common mynas over a lengthy period of time and using a large amount of data, but not during shorter-term studies, indicate that the impact of this species may be gradual and subtle. On the other hand, the impact shown by noisy miners is readily picked up in studies spanning even a few months, suggesting it may be far more acute. These impacts are likely to become even more apparent as the range and abundance of noisy miners continues to increase (Barrett et al. 2007).

The high levels of aggressive behaviour in noisy miners found by this study have been documented elsewhere (e.g. Clarke 1984; Dow 1977, 1979; Loyn 1987). Other closely related miner species, such as bell miners Manorina melanophrys are known to exhibit similar behaviour, and are even capable of completely altering habitat characteristics through aggressive exclusion of all other insectivorous birds, resulting in an increase in abundance of sap-sucking plant parasites such as psyllid insects (Homoptera: Psyllidae) and a corresponding decrease in tree health (Clarke 1984; Clarke and Schedvin 1997b; Dare et al. 2007). Common mynas on the other hand have a long, but mostly anecdotal history of displaying aggression, which was not reflected in the results of this study. Recent studies on common myna feeding behaviour suggests that reports of aggression may have been exaggerated (Haythorpe et al. 2012; Lowe et al. 2011), and it has also been suggested (Haythorpe et al. 2012) that their reputation may be in part due to similarities in the names and appearances of common mynas and noisy miners, leading to identification issues surrounding observations of aggressive interactions by the general public. Previous anecdotal reports of common myna aggression primarily relate to nesting behaviour, including defence of nestlings and competition for nesting resources. While this specific behaviour is clearly of importance in the consideration of impact by common mynas, any bias in this study towards aggressive behaviour due to the proximity of nest sites in relationship to transect locations should actually favour more aggressive behaviour in common mynas than noisy miners, as common mynas are prone to nesting in modified habitats (Counsilman 1974) while noisy miners prefer to nest in natural vegetation, and aggressive behaviour was compared across suburban locations only. However, even with this potential bias noisy miners were significantly more aggressive than common mynas.

To determine the impact of any species we must consider in which range or areas they could pose a threat, which species they have the potential to affect, the possible strength or magnitude of this effect, and the method by which it may come about. The inability of common mynas to penetrate dense bushland calls into question their predicted threat to native wildlife and diversity, at least in the Newcastle area. Although the importance of urban regions to biodiversity is clearly higher than historically thought (Goddard et al. 2010), any potential threat common mynas pose appears to be to the species in this region only – typically either other suburban habitat specialists, or habitat generalists.

Noisy miners, in contrast to common mynas, were found to be highly abundant in all habitat types. Threatening behaviour from noisy miners is thus capable of affecting both habitat specialists and habitat generalists. In addition, it is clear that noisy miners do not just exist in these habitats, but thrive in them, as they were the most abundant species recorded in this study. A typical characteristic of species considered to be 'invasive' – in this case, capable of having a widespread impact on a habitat – is the tendency to be present in comparatively high numbers (Colautti and MacIsaac 2004; Kolar and Lodge 2001; Richardson et al. 2000). In such cases, even relatively benign behaviours or processes may pose serious threats when amplified due to a species' overabundance. Although common mynas may appear to be highly abundant, they were found in substantially lower numbers than noisy miners. Their localised nature has perhaps led to overestimation of numbers, while high abundance of noisy miners in less frequented bushland areas goes relatively unobserved.

There is substantial evidence to suggest that noisy miner overabundance is correlated with a decrease in the diversity and abundance of other bird species (e.g. Grey et al. 1997; Grey et al. 1998; Mac Nally et al. 2012; Major et al. 2001; Maron et al. 2011; Parsons et al. 2006). In addition the findings of this study suggest that this effect fluctuates under various conditions, such as habitat type and breeding status. While it is possible that this effect could be coincidental, as bird species are affected by the same habitat modifications that favour noisy miners (rather than directly being affected by the noisy miners themselves), a substantial number of experimental studies have addressed this question, and generally conclude that a causal relationship does exist (Debus 2008; Grey et al. 1997; Grey et al. 1998; Kath et al. 2009; Mac Nally et al. 2012; Maron et al. 2011; Piper and Catterall 2003). The most likely method for noisy miners to negatively affect the abundance of small bird species is through aggressive exclusion (Dow 1977, 1979). Aggressive behaviour is likely to cause most impact in locations and during times when breeding is occurring, when noisy miners are aggressively defending nestlings through group mobbing behaviour (Arnold 2000). In support of this, we found noisy miners had an impact on the diversity and abundance of other species in primarily the edge habitat type during the breeding season, although aggressive behaviour was consistent across habitat types, suggesting that actual instances of aggression may not be the only part of a strategy designed to exclude other species.

Common mynas, by contrast, were not correlated with a decrease in diversity or abundance of other species in this study, and other studies (Parsons et al. 2006) have also been unable to find such a correlation. In addition, common mynas in this study were only infrequently observed engaging in aggressive behaviours, and a recent study (Lowe et al. 2011) has similarly found that common mynas rarely initiated interspecific aggressive attacks, and did not interfere with other foraging birds, a finding also supported by our previous research in this area (Haythorpe et al. 2012).

Conclusions

This study provides data on the range, abundance, and behaviours of noisy miners and common mynas and their associations with the abundance and diversity of other bird species in the Newcastle region. Noisy miners in this study were present in all three habitat types, while common mynas were only present in one habitat type (suburbs). While it remains possible that common mynas are negatively affecting some rare species, their restricted distribution suggests that this is unlikely; by contrast a range of rare species may be impacted by noisy miners. Noisy miners were highly abundant in all three habitat types, being the most abundant in suburb and edge habitats, and third most abundant in bush habitats. Common mynas were second most abundant in suburban habitats, the only habitat in which they were found. Noisy miners were associated with a decrease in diversity in edge habitats, and a decrease in abundance in edge and suburban habitats during the breeding season only, while common mynas were not associated with a decrease in diversity or abundance at any time. This effect may have been driven by aggressive exclusion – noisy miners were observed initiating aggressive attacks significantly more often than common mynas, and did so consistently across habitat types. We conclude that noisy miners may have the potential to have a greater impact on wildlife than common mynas in this area.

Perhaps because habitat alteration is so often accompanied by the spread of introduced species (Hobbs and Huenneke 1992; Vitousek et al. 1997), there is a tendency to focus attention on these species when assessing likely impacts on native assemblages. In reality, in today's increasingly urbanised environment, virtually all species are to some extent introduced, as few exist now in the same environments in which they evolved competitively. For some species this has provided an advantage, allowing them to increase in number dramatically and leading to detrimental impacts on the environment. These are of even greater concern if that species is also capable of subsequently migrating into areas previously unaffected, as this will impact not just the urban environment, which is already highly degraded by other anthropogenic factors, but more diversity-rich remnant bushland areas as well, which are more likely to contain rare or threatened species. Species usually considered to be 'introduced' frequently cannot survive outside urban centres, and thus their ability to impact the regions of greatest conservation value is limited.

As the current study suggests, greater risks might be posed by native species with artificially increased numbers than by introduced species with ranges restricted to highly urbanised environments. It may be that in environments that have been dramatically altered by human activity the species with the greatest capacity to adapt to these alterations and colonise a range of different habitats are the most likely to negatively impact on other species. As in the current system, these species need not be introduced.

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Chapter 3: Nesting behaviour in Common Mynas **Sturnus tristis** *in an urban environment, and potential effects on competing rosella species.*

Top: A Common Myna bringing artificial nesting material to a nest box. Middle: Eastern Rosellas at a nest box. Bottom: Common Myna pair calling. Photos by K. Haythorpe.

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Nesting behaviour in Common Mynas *Sturnus tristis* in an urban environment, and potential effects on competing rosella species.

Authors: Haythorpe, K.M[A,C], Sulikowski, D[A], and Burke, D[B]

[A] Charles Sturt University, Bathurst NSW 2795

[B] University of Newcastle, Ourimbah NSW 2258

[C] Corresponding author: kathryn.haythorpe@gmail.com

Abstract

This paper examines the behaviours of Common Mynas nesting at artificial nest boxes, with a particular focus on their interactions with native cavity nesting rosella species, Crimson Rosellas *Platycercus elegans* and Eastern Rosellas *P. eximius*. Common Mynas (Aves: Sturnidae; *Sturnus tristis*) have long been suspected of ousting competing cavity-nesting species, including use of competitive strategies such as nest filling. However, evidence of these competitive behaviours is still scarce. We found that, contrary to predictions, all species including Common Mynas generally avoided approaching nest boxes if another species was present, and found no evidence to indicate that Common Mynas are aggressively displacing other species. We present data on the range of materials Common Mynas bring to nest boxes and compare amount and type of materials with type of species occupying the nest box. In addition, we report on differences between species in behavioural use of current nest boxes and make suggestions for improving the efficacy of anti-myna "baffle" designs.

Keywords: aggression, avoidance, displacement, interspecific competition, introduced, invasive, management, *Platycercus*, rosella, Sturnid

Introduction

Common Mynas were introduced to Australia throughout the 19th century (Long 1981; Lever 1987) and now thrive in urban, suburban, and some rural regions.

They are cavity-nesters and require natural or artificial hollows for breeding, and take readily to nest boxes. In some cities Common Mynas have been found to occupy more than a third of nest boxes provided for native wildlife: 38 per cent in suburban Melbourne (Harper *et al.* 2005) and 45 per cent in Canberra (Pell and Tidemann 1997a). Common Mynas are considered so problematic that some authorities (e.g. Grant 1997; Birds Australia 2010) recommend attaching an "anti-myna" design to nest boxes to prevent their entry. This design employs the use of a solid, opaque "baffle" attached to the lip of the nest box roof and extending down to cover the nest box entrance, with grooves or wire mesh on the side of the box to be used as a "ladder" rather than a perch (Birds Australia 2010), and preliminary trials have found it successful in reducing Common Myna nesting (Homan 2000). However, installation of baffle designs can be time-consuming and expensive, and their suitability to native species is still largely unknown.

Common Mynas in urban regions compete for nest boxes with native species such as rosellas, and may be capable of aggressively outcompeting them, leading to a decline in their diversity and abundance (Pell and Tidemann 1997b). Several anecdotal reports (Wright and Wright 1991; Peters and Peters 1993) describe Common Mynas aggressively evicting Eastern Rosellas from nest sites, and one report (Lindenmayer 1993) recounts an instance of Common Mynas succeeding in competition with Galahs *Eolophus roseicapillus*, Eastern Rosellas, Crimson Rosellas, and Common Starlings *S. vulgaris* for access to a nesting hollow. Pell and Tidemann (1997b) document aggressive competition between Common Mynas, Eastern Rosellas and Crimson Rosellas over nest boxes and natural hollows, and conclude that Common Mynas demonstrate the potential to reduce the breeding success of these native parrots. However, recent literature has suggested that previous reports may have been exaggerated, and that clear evidence of a negative effect on native species by Common Mynas has not yet been found (e.g. Parsons *et al.* 2006; Tidemann 2010). A recent study (Haythorpe *et al.* 2013) has shown that Common Mynas rarely display aggression towards other species compared with native species such as Noisy Miners *Manorina melanocephala*. Other findings indicate that Common Mynas display limited aggression over food resources (Crisp and Lill 2006; Lowe *et al.* 2011), even

when forced into close proximity with other birds by artificial clumping of resources (Haythorpe *et al.* 2012). Another study (Grarock *et al.* 2012) found a negative relationship between Common Myna establishment and the abundance of three cavity-nesting species: Sulphur-Crested Cockatoos *Cacatua galerita*, Crimson Rosellas, and Laughing Kookaburras *Dacelo novaeguineae*. However, this study also found a positive association between Common Myna establishment and the abundance of four other cavity-nesting species, including Galahs, Australian King-Parrots *Alisterus scapularis*, Eastern Rosellas, and Common Starlings. Overseas, an impact on native species such as Sooty Terns *Onychoprion fuscata* (Hughes *et al.* 2008), Mauritius Parakeets *Psittacula echo* (Jones 1996), and Seychelles Magpie-Robins (Watson *et al.* 1992; Huong and Sodhi 1997) has been attributed to Common Mynas. In one instance, the removal of Common Mynas coincided with an increase in the populations of 23 other bird species, suggesting their historical decline may have been related to Common Myna introduction (Tindall *et al.* 2007). However, these situations relate to much smaller, isolated islands (all less than 2000km^2 in size). The potential of Common Mynas to affect native wildlife on larger continents like Australia is unlikely to be comparable, and requires investigation.

Interestingly, one study (Pell and Tidemann 1997b) describes an instance of a pair of Common Mynas constructing and subsequently abandoning a number of nests. Despite a high uptake rate by native parrots at other hollows in the area, only one of the ten hollows filled with nesting material was later occupied by a pair of Eastern Rosellas. The authors conclude that the construction of the additional nests may be part of a competitive strategy employed by Common Mynas to act as a deterrent to other nesting species and reduce competition within the territory. There is good evidence to suggest a strategy of building extra nest sites has the potential to be effective in excluding other species – hollows with old nesting material have a greater chance of becoming infested with pathogens (Wasylik 1971; Powlesland 1978; Rendell and Verbeek 1996a), decreasing the likelihood of nestling survival (e.g. Moss and Camin 1971; Duffy 1983), and this is thought to play a large role in the nest site selection of many birds (Rendell and Verbeek 1996b; Loye and Carroll 1998). In addition the presence of nesting material may indicate hollow occupation by another bird, and

avoidance may decrease the risk of confrontation (Muldal *et al.* 1985). However, further documentation of this behaviour is needed before conclusions as to its purpose can be made. For example, one reasonable alternative suggestion might be that construction of multiple nest sites is not part of a competitive strategy at all, but merely represents an insurance policy against site failure.

This study sought to provide observations on the manner in which nest boxes are used by Common Mynas and two competing rosellas species, Crimson Rosellas and Eastern Rosellas. Specifically, we investigated three aspects: Use of nest box structure, provisioning of nesting material by Common Mynas, and interspecific interactions. We used the principles embodied in the anti-myna "baffle" design to inform predictions on how such a design may influence the nesting behaviour of each species, considering the three major components of the design: removal of the perch; installation of a baffle extending from the upper lip of the roof and; extension of the baffle down past the entrance hole (such that birds within the box cannot see directly out, and birds outside cannot enter directly into the nest box). We predicted that the rosella species would make less use of a perch, and less use of the roof lip (to travel between the roof and entrance of the box), than Common Mynas, and also that Common Mynas would make more use of the entrance as a vantage point than the rosella species, and the rosella species would make flying approaches to the nest box entrance from below it more often than Common Mynas. We also aimed to document Common Myna nest material provisioning behaviour to investigate the possibility that Common Mynas use multiple nest building as a means of competition. If nest building was intended only for sites used by the nest builders themselves, then we would expect Common Mynas to bring nesting material only to nest boxes they were attempting to use for breeding, and this should occur either before, or well after, birds of another species used the nest site. On the other hand, if this were not the case then we hypothesised that Common Mynas would bring nesting material not only to their own nest boxes, but to those occupied by other species, and this nest building would occur during, or exclusively during, occupation by the competing species. Additionally, we hypothesised that Common Mynas might provide different quantities and types of nesting material to nest boxes they used for breeding compared to those they did not use for breeding. Finally, if Common

Mynas were aggressively competing for nest sites we hypothesised that they would show greater levels of displacement behaviour (i.e. arrival at a nest box immediately after the departure of the species initially present in that location in a manner that could indicate the first species were forced to vacate the space due to the imminent arrival of the second) towards Eastern Rosellas and Crimson Rosellas, than these species displayed to them in return. If this aggression represented an attempt to disrupt the nesting behaviour of other species we hypothesised that displacement would be higher at sites owned by other species than by the Common Mynas themselves (which would indicate attack on another individual rather than defence of the nest site), and would occur at higher levels during the breeding season. If Common Myna displacement represented organised, strategic attacks rather than defensive responses we might also expect more displacement to occur in pairs or groups rather than individually.

Methods

Site Locations and Descriptions

Between 2009 and 2010, before the commencement of the present study, 206 nest boxes were installed throughout Canberra by the Australian National University as part of a separate research project (Tidemann *et al.* 2010). We selected 20 nest boxes from these based on existing data on their previous levels of use by wildlife and their ease of access for camera installation and monitoring. Of these sites, one failed to attract any wildlife at all and was excluded. The remaining 19 nest boxes were located in the suburbs of Aranda (1), Chifley (1), Cook (2), Kaleen (4), Kambah (4), Theodore (6) and Torrens (1; Fig. 1).

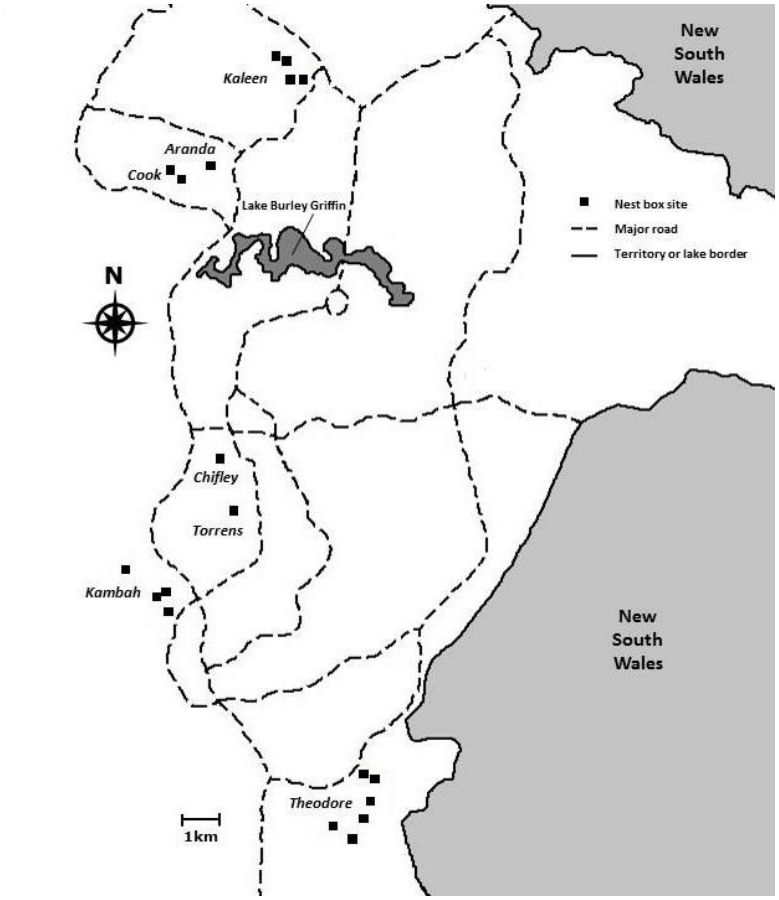

Fig 1. Map showing site locations in Canberra, ACT. Squares indicate nest box sites, dashed lines indicate major roads, and solid lines indicate state/territory borders or lake outline.

Nest boxes were attached to tree trunks between 2.5 and 3.8 metres from the ground using strips of sheet metal 2 x 20 centimetres nailed to the tree, and one bolt suspended from the back of the nest box. Each nest box was constructed of 15 millimetres untreated construction plywood and measured 24 x 27 x 40 centimetres (l x w x h), with an internal volume of 19 litres. The nest box entrance was circular with a diameter of 6.5 centimetres. A 1 x 5 x 24 centimetres (l x w x h) "perch" was attached to the front of the nest box 5 centimetres under the entrance. Nest box rooves overhung the front of the nest box by 7 centimetres, and the nest box floors were detachable and held in place with a screw or clip. Internal wire-mesh "ladders" were fitted to the inside of the nest box underneath the entrance to assist climbing (Fig. 2).

Fig 2. A nest box with some important features labelled. Photo: K. Haythorpe.

Procedure

Monitoring took place from January 2011 to December 2012. Each nest box was monitored continuously using a motion-sensing surveillance camera (Scoutguard SG550V, Gold Coast, Australia), which took still 3-megapixel images. The cameras had a shutter response time of one second and a shutter recovery time of five seconds, and images in low light (e.g. early morning or late evening) were also recorded using IR LED no-flash technology. Once every two months camera batteries were changed and nest boxes were inspected for eggs or nestlings using a wireless inspection camera (Explorer Premium, Goscam, Shenzhen, China). Breeding species occupation was determined by the visual confirmation of eggs or chicks in the nest box at least once during the period of the study by the researcher (see Table 1). There were no instances recorded in this study of nest boxes being used by different species in subsequent breeding seasons.

Site number	Suburb	Breeding species
1	Kaleen	No breeding
2	Kaleen	Eastern Rosellas
3	Kaleen	No breeding
4	Kaleen	Common Mynas
5	Aranda	No breeding
6	Cook	No breeding
7	Cook	No breeding
8	Chifley	Common Mynas
9	Torrens	No breeding
10	Kambah	Common Mynas
11	Kambah	Common Mynas
12	Kambah	No breeding
13	Kambah	No breeding
14	Theodore	Crimson Rosellas
15	Theodore	Eastern Rosellas
16	Theodore	Crimson Rosellas
17	Theodore	No breeding
18	Theodore	No breeding
19	Theodore	No breeding

Table 1. List of nest box sites, their suburban location, and what species used the nest box for breeding.

A total of 732 941 images were taken, with 51 164 images (approximately 7.0%) containing pictures of birds. The remaining 681 777 images were triggered by wind moving tree branches (or, rarely, by movement from humans or a non-avian animal) and were labelled “blank” photos as they did not contain birds. Of the bird photos, 28 235 (55.2%) contained Common Mynas, 11 973 (23.4%) contained Crimson Rosellas, and 5 448 (10.7%) contained Eastern Rosellas. The remaining images contained photos of 23 other bird species, but as these consisted of species that either do not nest in hollows (e.g. Australian Magpies *Cracticus tibicen*), species that would normally nest in hollows but could not use these nest boxes as the entrance diameter was too small for them (e.g. Galahs), or species that were capable of using the nest boxes but were present only on very rare occasions (e.g. Common Starlings, 55 images), these species were excluded from analysis.

For each image with at least one of the target species present (Common Mynas, Crimson Rosellas or Eastern Rosellas) and located within five metres of the nest box, details were recorded of the species, its location, the date and time (to the

second) of the photograph, and details on its activity. Images usually contained only one individual, but some images of multiple birds of the same or different species were taken, and this was noted. Identification of individual birds could not be determined.

For birds located at the entrance of the nest box (defined as more than 50% of their body located in front of the wall containing the entrance hole, with at least one non-wing part of the body touching the nest box), the position of each foot was noted as either on the entrance itself, on the perch, on the side of the box, on the roof lip, or hanging free. All instances where at least one foot was on the perch were counted as "perch used" and all instances where at least one foot was on the roof lip were counted as "roof used". All other cases where the bird was at the entrance but not with at least one foot on the perch or roof were classed as "perch not used" or "roof not used" respectively. The two counts were independent of one another, thus it was possible, for example, for an image of a bird to count towards both "perch used" and "roof used" if it had one foot on the perch and the other on the roof.

Birds located inside the nest box itself but sitting at the entrance facing outwards were classed as "entrance used", and all birds that were using the nest box in some other way (e.g. standing on the roof) were classed as "entrance not used". All images of birds flying towards the entrance hole within one nest box width of the entrance were recorded as approaching from below if more than half of their body was below the halfway point of the entrance hole, or above if at the level of the entrance hole or above it.

Common Mynas were photographed bringing nest material to eight nest boxes. At these locations a total of 42 835 images of birds were taken, of which 31 514 (73.6%) were Common Mynas, and of these images, 1 335 (4.2%) were carrying material in their beaks. Material was classified as either plant matter, feathers, invertebrates, artificial materials, or unidentified.

Analysis

Nest box use

A series of Pearson's chi-square tests of contingencies (with $\alpha=0.05$) were used to evaluate whether the three target species differed in their tendencies to use the perch or roof while at the entrance, use the entrance as a vantage point, or approach the entrance hole on the wing from above or below.

Nest Material Provisioning

For analyses on nest material provisioning, data was pooled within nest box sites by breeding species occupation as either Common Mynas breeding (sites 4, 8, and 11), rosellas breeding (Crimson Rosellas at site 14 and Eastern Rosellas at site 15), or nothing breeding (sites 3, 12, and 18). Data was also separately pooled according to classification by suburban location as Chifley (site 8), Kaleen (sites 3 and 4), Kambah (sites 11 and 12), or Theodore (sites 14, 15 and 18). Pearson's chi-square tests of contingencies (with $\alpha=0.05$) were used to evaluate whether breeding species occupation was related to amount of nesting material brought to nest boxes, and whether location or breeding species occupation were related to proportion of artificial nesting material brought to nest boxes. Pearson's bivariate correlations were calculated to examine relationships between number of rosella visits in each month and yearly proportion of artificial material provided by Common Mynas during that month.

Image Periods (IP), Inter-Image Periods (IIP) and Observed Putative Displacement Events (OBS_{PDE})

To help us determine whether two consecutive images of different species ought to be considered representative of a possible displacement event, we first needed to define the time period that a single image of a given species would nominally represent – known as that species' image period (IP). Since the cameras were motion sensitive, behavioural differences between species (including swiftness and vigour of movement) could have affected the likelihood with which the

camera would be triggered multiple times by a species remaining at the nest box for an extended period. For this reason, image periods were calculated separately for each species: Common Mynas, Eastern Rosellas, and Crimson Rosellas; using the following method.

At each camera, on the 20th day of each month from January 2011 to December 2012, we recorded the number of seconds between consecutive images (interval times) of a bird of the same species for each of the three target species. As no individual identification was possible, images containing a bird of the same species were assumed to be the same individual. The majority of these consecutive images were assumed to have been taken during the same visit – breeze moving the branches resulted in fairly regular images being captured with no bird present, making it likely that consecutive images of the same bird were taken from the same visit. As such, the distribution of these inter-image intervals was strongly positively skewed, with the long tail of the distribution presumably representing those inter-image intervals that occurred between the last image of one visit and the first image of the next. To exclude these long intervals, we removed from each species' list all values greater than two standard deviations above the mean. This resulted in the removal of a total of 23 interval times: 13 of 604 (2.2%) from Common Mynas, 3 of 249 (1.2%) from Crimson Rosellas, and 7 of 151 (4.6%) from Eastern Rosellas. The remaining interval times (pooled across all months and sites) were then averaged to produce each species' image period.

Images taken due to wind moving the branches, rather than bird movement, could be a source of error in the calculation of these image periods and could introduce bias into subsequent analyses, especially if some species happened to occur more often at windier sites than other species. To test whether this was likely to be a problem in our study we correlated the number of blank photos taken at each camera on each 20^{th} day (as a measure of the amount of wind interference for that day) with the average inter-image interval calculated (for a given species) for that day. For none of the three species (Common Mynas: $r = -0.157$, $n = 40$, $p = 0.335$; Crimson Rosellas: $r = -0.127$, $n = 38$, $p = 0.448$; Eastern Rosellas: $r = -0.100$, $n = 18$, $p = 0.693$), was there a significant correlation between the inter-

image interval and the number of blank images taken each day, suggesting that wind factors did not contribute substantially to the image periods calculated.

We defined the maximum time period allowed between two consecutive images of different species in order for that pair of images to be considered an observed putative displacement event (OBS_{PDE}) as equal to the mean of the image-periods (IP) of the two species in question. This measure is represented as IIP_{DIS} (inter-image period of displacement). Table 2 lists the IP for each species and the IIP_{DIS} for each focal species pair.

Image Periods (IP)		**Inter-image Periods of Displacement (IIP_{DIS})**	
Species	IP value (seconds)	Focal Species Pair	IIP_{DIS} value (seconds)
Common Myna (n=591)	92	Common Myna / Crimson Rosella	133.5
Crimson Rosella (n=246)	175	Common Myna / Eastern Rosella	77.5
Eastern Rosella (n=144)	63	Crimson Rosella / Eastern Rosella	119

Table 2. List of calculated Image Periods for the three target species and Inter-image Periods of Displacement for the three focal species pairs. Numbers in brackets after species names represent the number of interval times used to calculate the IP for that species.

Expected Frequencies of Putative Displacement Events (EXP_{PDE})

The frequencies of finite time intervals between independent events that occur at a fixed rate (λ), with equal probability at any point in time, follow the negative exponential distribution. We used this distribution to model the number of putative displacement events we would expect to observe between any given species pair, if in fact all visits were independent events (the null situation in which individuals are neither more or less likely to visit a nest box if another individual of the other species is already present). We then compared the number of expected and observed putative displacement events between each species pair to see if the latter was

higher (potentially indicating aggressive displacement) or lower (potentially indicating avoidance).

Using the image-periods, we first calculated the aggregate observation time (T) in seconds, represented by the images of birds of each focal species pair for each camera and each month:

$$T = (IP_X.N_X) + (IP_Y.N_Y)$$

where IP_X and IP_Y represent the image-periods for the two species in question and N_X and N_Y are the number of images of each of those species, respectively.

We then calculated the rate parameter (λ), in units of images per second for each camera for each month as:

$$\lambda = (N_X + N_Y) / T$$

We then computed the exponential function:

$$F(x) = \lambda \exp(-\lambda.x)$$

for the appropriate value of λ for each focal species pair, for each camera-month. This function, when solved for $x = IIP_{DIS}$, gave us the value of the exponential which, when integrated to find the area under the curve to the left of that value, represented the proportion of expected inter-image periods shorter than IIP_{DIS} (see Fig. 3).

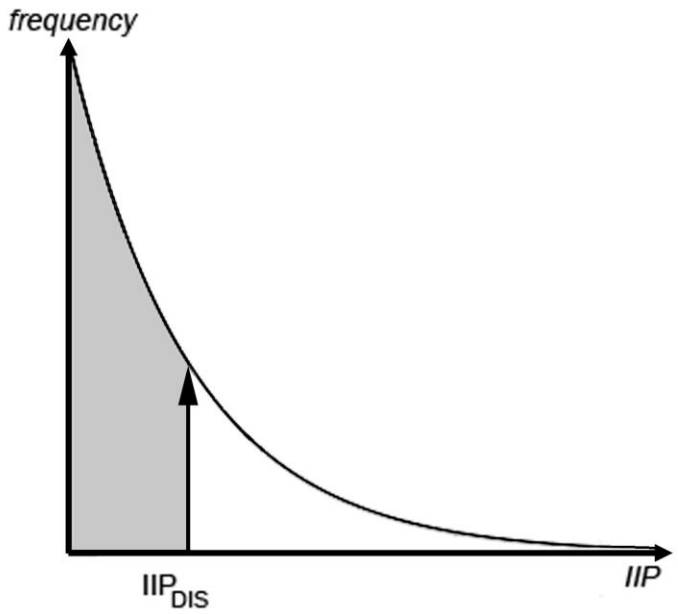

Fig 3. The negative exponential function showing frequency distribution of inter-image period (IIP) lengths. The grey area under the curve represents the proportion of inter-image periods that were short enough to be classified as putative displacement events (as they were shorter than the IIP_{DIS} value for that focal species pair).

Now that we had the proportion of IIPs for each month that would be expected to be shorter than IIP_{DIS}, we then counted how many of the IIPs occurred between images of birds of the same species (and thus could not be defined as putative displacement events) and images of birds of different species (which may have been defined as observed putative displacement events, OBS_{PDE} as long as the IIP was shorter than IIP_{DIS}). We then calculated the expected number of putative displacement events (EXP_{PDE}) as:

$$EXP_{PDE} = PROP_{IIP<DIS}.IIP_{XY} / 2$$

where $PROP_{IIP<DIS}$ is the proportion on inter-image periods expected to be shorter than IIP_{DIS} and IIP_{XY} is the number of inter-image periods which occurred between images of the two species of each species pair (not between images of the same species). The entire equation is divided by two, giving the number of expected putative displacement events that would occur in either direction (species X displaying species Y, or species Y displacing species X).

Comparisons of Expected (EXP_{PDE}) and Observed (OBS_{PDE}) Putative Displacement Events

From the previous steps described, we had calculated (separately for each camera, for each month, for each pair-combination of the four focal species) the number of expected putative displacement events (EXP_{PDE}) and the number of observed putative displacement events (OBS_{PDE}) for each camera. Firstly, for each pair of species, we excluded any camera month (for example, camera 7 in July) for which both the expected and observed number of putative displacement events was equal to zero (this could only occur if the two species being considered were never both photographed by the camera in question on the same day all throughout that month).

Following the methodology adopted by Haythorpe et al (2012), we then calculated a displacement/avoidance (DA) score for the remaining camera months as follows:

$$DA_{XY} = (OBS_{XY} - EXP_{XY})^2 / EXP_{XY}$$

where DA_{XY} is the displacement/avoidance score for species X displacing species Y, OBS_{XY} is the number of observed putative displacement events in which species X displaced species Y, and EXP_{XY} is the expected number of putative displacement events for species X displacing species Y. Furthermore, if there were more observed than expected putative displacement events, DA_{XY} was defined as positive (indicating evidence of actual displacement of species Y by species X) and if there were fewer observed than expected putative displacement events, DA_{XY} was defined as negative (indicating evidence of avoidance of species Y by species X). The DA_{XY} scores were positively skewed and so were ninth-root transformed prior to analysis (back-transformed means and 95 per cent confidence intervals are quoted and illustrated throughout).

Displacement or Avoidance?

To determine whether either species of a given focal pair was more likely to appear at a nest box if the other species was already present (a tendency to displace) or less likely to appear (a tendency to avoid) we compared each set of DA_{XY} scores to a hypothesized mean of 0 using one-sample t-tests. We also used paired-samples t-tests (with samples matched for camera-month) to test for asymmetries in potential displacement/avoidance tendencies within each focal species pair.

Defence or Attack?

We also considered that the tendency to displace or avoid could be contingent on whether a species occupied (or was attempting to occupy) the nest box in question, or whether they were intruding into a nest site that was already being defended. To test this, we conducted independent samples t-tests, splitting each set of DA_{XY} scores into two samples, depending upon whether species X or species Y was the most photographed during that camera-month (on the assumption that if either species could be considered to be occupying or defending a nest site, that species would likely be present the most). Where the assumption of homogeneity of variance was violated (as determined by a Levene's test), an adjusted degrees of freedom was applied.

Effects of Season

Similar to above, we split each set of DA_{XY} scores into two samples based on whether the camera-month occurred in the breeding or non-breeding season (with the breeding season being defined as September-February and non-breeding as March-August). Independent-samples t-tests were then used to compare breeding and non-breeding DA_{XY} scores.

Effects of Group number

We also wanted to examine whether putative displacement events may be more likely to be initiated by groups of 2 or more birds, rather than a single bird. We used a Chi-squared Goodness-of-Fit test to compare the proportions of OBS_{PDE} (observed putative displacement events) that were initiated by multiple con-specific birds with the proportions of all observation of initiating species that included more than one individual, respectively.

Results

Nest box use

Perch

The Pearson's chi-square test of contingency examining an association between species type and tendency to use the perch was statistically significant ($\chi^2_{(2)}$=101.35, p<0.001). Crimson Rosellas used the perch to enter or exit the nest box on 479 out of 2 084 occasions (23.0%), which was significantly more than both the 532 out of 3 824 occasions (13.9%) shown by Common Mynas ($\chi^2_{(1)}$=78.28, p<0.001, or the 105 out of 931 occasions (11.3%) shown by Eastern Rosellas ($\chi^2_{(1)}$=56.47, p<0.001). A further chi-square test of contingencies was performed to examine the difference between Common Mynas and Eastern Rosellas, and it was found that Common Mynas used the perch significantly more than Eastern Rosellas ($\chi^2_{(1)}$=4.48, p=0.034; Fig. 4a).

Roof

The Pearson's chi-square test of contingency examining an association between species type and tendency to use the roof lip to travel between the roof and entrance hole was statistically significant ($\chi^2_{(2)}$=2483.68, p<0.001). Eastern Rosellas used the roof lip on 643 of 931 occasions (69.1%), which was significantly more than the 803 of 2 084 occasions (38.5%) shown by Crimson Rosellas ($\chi^2_{(1)}$=240.39, p<0.001), and in turn Crimson Rosellas used the roof

significantly more than Common Mynas (54 of 3 824 occasions, or 1.4%; $\chi^2_{(1)}$=1498.66, p<0.001; Fig. 4b).

Entrance

The Pearson's chi-square test of contingency examining an association between species type and tendency to crouch at the entrance to the nest box facing outwards was statistically significant ($\chi^2_{(2)}$=535.64, p<0.001). Common Mynas crouched in the entrance on 2 833 of 25 110 occasions (11.3%), which was significantly more than either rosella species (Eastern Rosellas: $\chi^2_{(1)}$=68.66, p<0.001; Crimson Rosellas: $\chi^2_{(1)}$=511.39, p<0.001). Eastern Rosellas crouched at the entrance on 387 of 5 228 occasions (7.4%), which was significantly more than the 487 of 11 903 occasions (4.1%) shown by Crimson Rosellas ($\chi^2_{(1)}$=82.25, p<0.001; Fig. 4c).

Approach method

The Pearson's chi-square test examined association between species type and tendency to approach the entrance hole on the wing from below, and was not statistically significant ($\chi^2_{(2)}$=1.40, p=0.497). Common Mynas approached from below 120 of 213 occasions (56.3%), which was similar to that of Eastern Rosellas (7 of 12 occasions, or 58.3%), but substantially lower than the 14 of 20 occasions (70.0%) shown by Crimson Rosellas (Fig. 4d). The low number of observations of rosella species in flight may have contributed to a failure to find a significant association between species type and approach method.

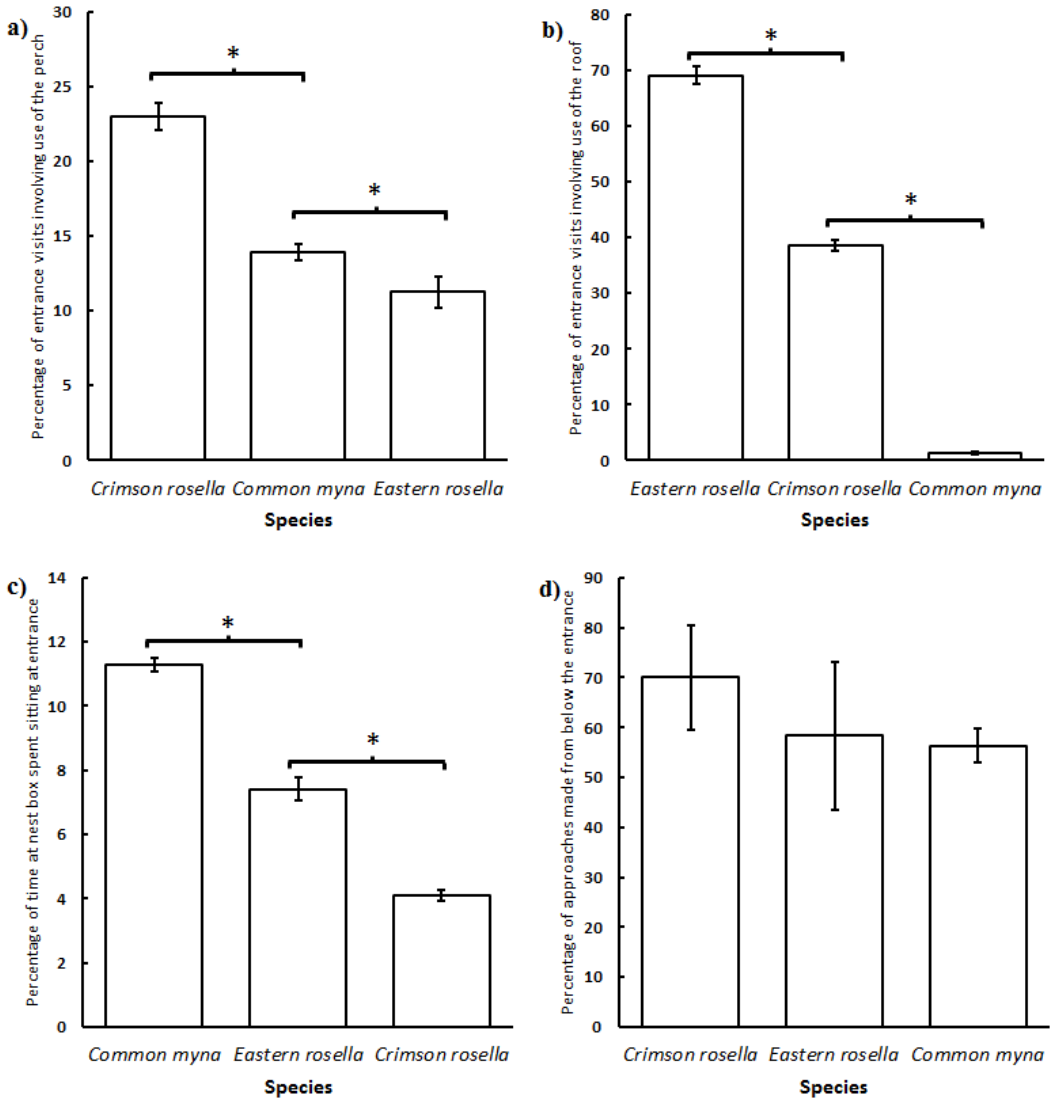

Fig 4. Shows the differences between use of various nest box features by Crimson Rosellas, Common Mynas and Eastern Rosellas. Crimson Rosellas used the perch significantly more than Common Mynas and Eastern Rosellas, and Common Mynas used the perch significantly more than Eastern Rosellas (a). Both rosella species used the roof significantly more than Common Mynas, with Eastern Rosellas using it significantly more often than Crimson Rosellas (b). Common Mynas spent significantly more time sitting at the entrance to the nest box facing out than either of the rosella species, with Eastern Rosellas also sitting at the entrance more often than Crimson Rosellas (c). No significant differences between species were found in the proportion of above or below flying approaches to the nest box entrance (d).

Nest material provisioning

The primary material brought to nest sites was plant matter (64.6%), consisting of twigs, leaves, bark, small branches, and flowers. Artificial material made up the second largest group (16.1%) and consisted of pieces of plastic or paper, occasionally with distinguishable writing or brand names visible on them. Feathers, the third most common material brought to nest boxes (15.1%), originated mostly from species other than Common Mynas, as the majority (69.2%) were either fully white primary feathers, which Common Mynas do not grow, or contained colours not found on Common Mynas (blue, grey or yellow). Invertebrates made up the remaining 4.3% of identified material brought to nest boxes (See Fig. 5).

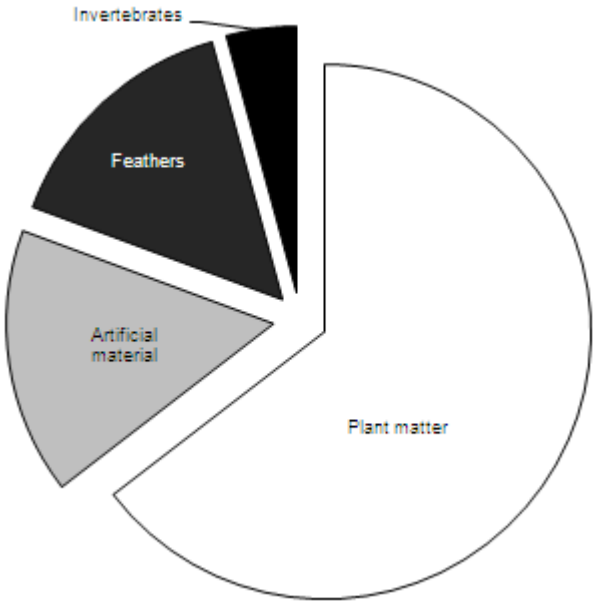

Fig 5. Shows the relative proportion of each type of material seen being carried to nest boxes by Common Mynas (n=1 292).

The amount of material brought to nest boxes occupied by rosella species, Common Mynas, or not occupied by any species were summed and compared (Fig. 6a). A Pearson's chi-square test was used to see if the total amount of material brought to nest boxes was related to the identity of the species occupying the nest box. The chi-square test was significant ($\chi^2_{(2)}$=150.73, p<0.001), indicating an effect of breeding species occupation on provisioning of

nesting material. Further chi-square tests revealed a trend towards a difference ($\chi^2_{(1)}$=3.28, p=0.070) between the amount of nesting material brought to nest boxes Common Mynas were breeding in (768 items in 14 150 visits, or 5.4%) and those rosellas were breeding in (310 items in 6 431 visits, or 4.8%), but significantly less nesting material was brought to boxes that were not used at all (257 items in 10 933 visits, or 2.4%) than to nest boxes occupied by either rosellas ($\chi^2_{(1)}$=78.19, p<0.001) or Common Mynas ($\chi^2_{(1)}$=148.98, p<0.001).

The type of material (artificial or natural) was also summed and compared across nest boxes occupied by rosellas, Common Mynas, or not occupied (Fig. 6b). A Pearson's chi-square test was used to determine if the amount of artificial material (as a proportion of the total amount of material brought by Common Mynas) was related to the identity of the species occupying the nest box. This test also revealed a significant effect of occupying species on proportion of artificial material ($\chi^2_{(2)}$=47.15, p<0.001), with Common Mynas bringing significantly more artificial material to sites where nothing was breeding (72 of 241 items, or 29.9%) than sites that were occupied by Common Mynas (109 of 744 items, or 14.7%; $\chi^2_{(1)}$=28.13, p<0.001) or rosellas (27 of 307 items, or 8.8%; $\chi^2_{(1)}$=40.54, p<0.001), and significantly less artificial material to sites where rosellas were breeding than sites where Common Mynas were breeding ($\chi^2_{(1)}$=6.62, p=0.010).

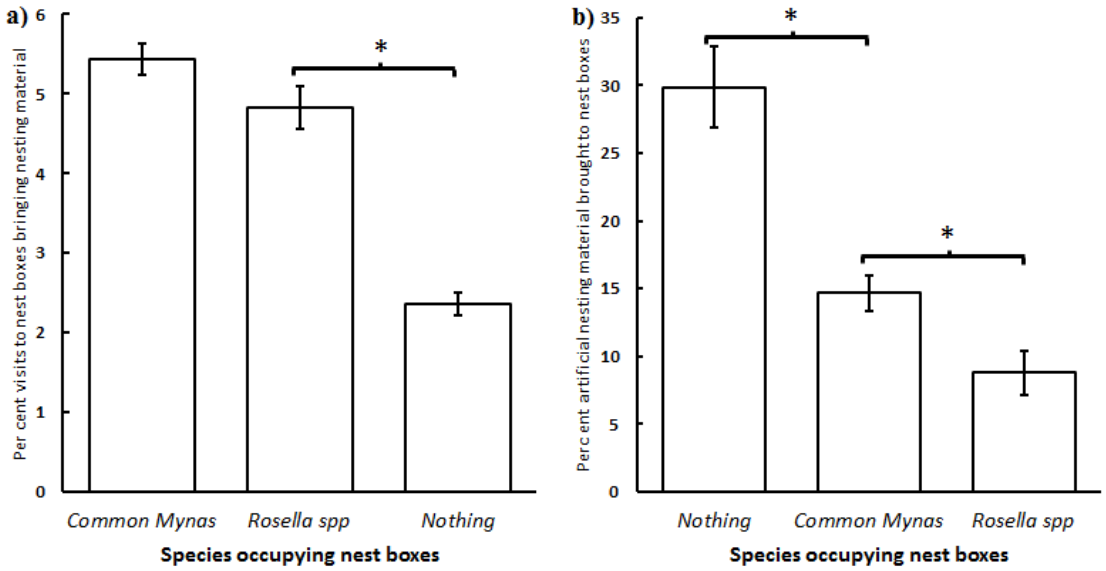

Fig 6. Shows the percentage of Common Myna visits that involved bringing nesting material to nest boxes based on their breeding species occupation (a), and the percentage of that material that was artificial (human rubbish such as plastic wrappers), rather than natural (sticks, leaves, feathers, invertebrates, etc.) brought to nest boxes based on their breeding species occupation (b).

Interspecific interactions

Observed interactions

Two instances were recorded of a Common Myna pair standing on top of a nest box while a rosella species was inside (Crimson Rosellas at site 14 and Eastern Rosellas at site 15). In both cases no evidence was captured of the encounter escalating or becoming aggressive. In a separate incident, an image of a Common Myna swooping a juvenile Crimson Rosella was captured at site 14. The Crimson Rosella did not appear to be displaced by the attack. No other incidents of aggression or nest box change of ownership were recorded. In cases where Common Mynas provided material to nest boxes occupied by rosella species, the nesting birds did not remove material but continued to nest on top of it.

Displacement or Avoidance?

All of the mean DA_{XY} were negative, indicating a general tendency for all species to be less likely to approach a nest box if a member of a different species was present (see Fig. 7).

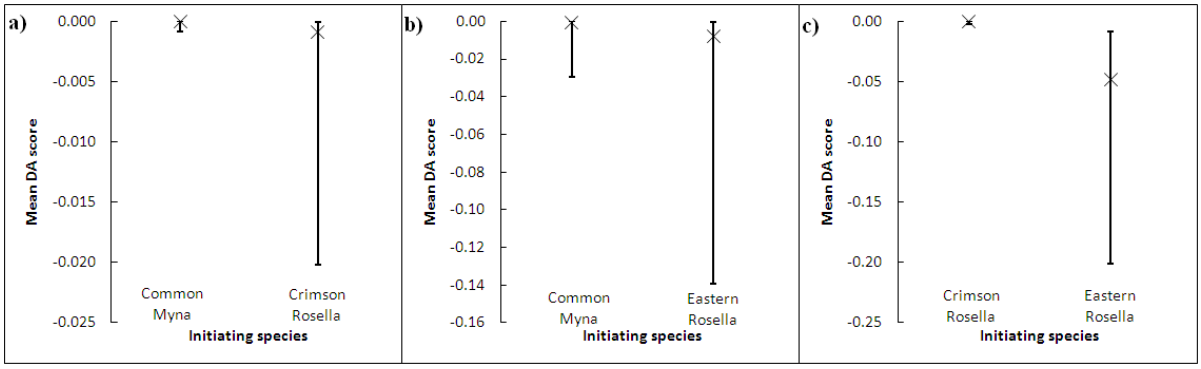

Fig 7. Back-transformed mean DA_{XY} scores with 95 per cent confidence intervals for each initiating species within the focal species pairs of Common Mynas and Crimson Rosellas (a), Common Mynas and Eastern Rosellas (b), and Crimson Rosellas and Eastern Rosellas (c).

Common Mynas displayed a significant bias to avoid Eastern Rosellas (t_{24} = 3.087, p = 0.005) and a marginally significant bias to avoid Crimson Rosellas (t_{52} = 1.874, p = 0.066). Both Eastern Rosellas (t_{24} = 5.120, p < 0.001) and Crimson Rosellas (t_{52} = 4.597, p < 0.001) significantly avoided Common Mynas. There were no differences in the strength of the avoidance bias within either of these focal species pairs (all t < 1.662, all p > 0.102).

Crimson Rosellas (t_{57} = 2.908, p = 0.005) and Eastern Rosellas (t_{57} = 11.394, p < 0.001) significantly avoided each other, with this tendency being significantly stronger for Eastern Rosellas (t_{57} = 3.282, p = 0.002).

Defence or Attack?

Overall, there were no significant effects on the strength of the avoidance bias exhibited by any species toward any other species as a function of which species may have had nest box occupancy (as determined by which species was

photographed most often at the nest box each month). There was no effect of nest-site occupancy on the avoidance bias of Common Mynas ($t_{51} = 0.691$, $p = 0.493$) and Crimson Rosellas ($t_{51} = 0.849$, $p = 0.400$) toward each other. There were similar null results for Common Mynas ($t_{19.166} = 1.618$, $p = 0.122$) and Eastern Rosellas ($t_{22} = 1.006$, $p = 0.325$) as a focal species pair. Nest-site occupancy had no effect on the avoidance bias of Crimson Rosellas ($t_{56} = 0.113$, $p = 0.910$) or Eastern Rosellas ($t_{56} = 1.011$, $p = 0.316$) toward each other.

Effects of Season

Overall, there were no effects of season on any species' tendency to avoid. Detailed results follow. There was no effect of season on the strength of the avoidance bias exhibited by Common Mynas toward Eastern Rosellas ($t_{22} = 0.999$, $p = 0.329$) or Crimson Rosellas ($t_{51} = 0.400$, $p = 0.691$). Similarly, there was no effect of season on the strength of the avoidance bias exhibited toward Common Mynas by Eastern Rosellas ($t_{22} = 0.295$, $p = 0.771$) or Crimson Rosellas ($t_{51} = 1.122$, $p = 0.267$). There were also no effects of breeding season on the extent to which Crimson Rosellas ($t_{56} = 0.907$, $p = 0.368$) and Eastern Rosellas ($t_{56} = 0.751$, $p = 0.456$) avoided each other.

Displacement by Groups of Common Mynas

Of all observations of Common Mynas across the dataset 15.6 per cent (4 401 / 28 235) involved more than one bird. Of all putative displacement events initiated by Common Mynas, 25.0 per cent (27 / 108) involved more than one Common Myna. A Chi-squared Goodness-of-Fit test confirmed that Common Mynas were more likely to be observed in pairs when engaged in a putative displacement event, than at other times ($\chi^2_{(1)} = 6.55$, $p = 0.011$), suggesting that despite the overall low rates of putative displacement events observed, some of these events may have represented actual displacement of other species by pairs of Common Mynas. By contrast, Crimson Rosellas were observed in pairs or groups on 6.4 per cent (764 / 11 973) of occasions, which was not significantly different to the rate of putative displacement events involving more than one

Crimson Rosella (5 / 88, or 5.7%; $\chi^2_{(1)} < 0.001$, $p = 0.999$). A Pearson's Chi-squared test of contingency examined frequencies of involvement of multiple birds in putative displacement events by Common Mynas and by Crimson Rosellas relative to the frequencies with which multiple birds were observed across the data set. It was found that Common Mynas were relatively more likely to initiate putative displacement events while in groups (typically pairs) than Crimson Rosellas ($\chi^2_{(1)} = 13.247$, $p < 0.001$). Numbers of putative displacement events initiated by more than one Eastern Rosella were too low for analysis.

Discussion

We investigated the behaviours of, and interactions between, Common Mynas, Crimson Rosellas, and Eastern Rosellas at 19 artificial nest sites over the course of two years. Although a great deal of speculation exists on the impact of Common Mynas, few studies have documented it in detail over a large time span. Contrary to predictions, we found no evidence to suggest that Common Mynas were aggressively competing with rosella species in this system.

Use of Nest Box Features

Unsurprisingly, we found differences between Common Mynas, Crimson Rosellas and Eastern Rosellas in the manner of nest box utilisation. Rosellas were less likely to crouch in the nest box entrance than Common Mynas, supporting the use of a baffle to deter the latter species. However, Crimson Rosellas used the perch significantly more than the smaller Common Mynas or Eastern Rosellas, and the rosella species were more likely to use the roof lip to climb between the roof and the entrance to the nest box; suggesting that the current baffle design could be improved upon. For instance, the baffle design currently suggests not providing a perch, however perches may prove particularly beneficial to larger Crimson Rosellas, as landing without a perch requires a greater degree of precision to avoid colliding with the top of the entrance hole. By contrast, more than five-sixths of the occasions in which Common Mynas were present at the entrance they were not making contact with the perch at all,

despite being in its immediate vicinity. Common Mynas thus appear capable of making very precise landings at the nest box entrance, and can do so without the aid of a perch, and removing it is unlikely to affect them as much as it will affect Crimson Rosellas.

In particular, the use of the nest box roof lip by rosellas for the majority of entry or exit occasions – more than two-thirds for Eastern Rosellas, and more than one-third for Crimson Rosellas – should be considered. The current baffle design relies on the principle that rosellas prefer to fly beneath the baffle and climb up to it to reach the nest box entrance, and thus involves attaching the baffle to the roof lip, effectively preventing birds from climbing down to reach the entrance. However, the hooked beak and strong claws of rosellas make them well suited to this kind of climbing. Common Mynas, on the other hand, do not possess such equipment, and barely made use of the roof lip at all, using it on only a tiny proportion of occasions. It may be worth considering an "open-top" modification to the baffle design, for instance, by substantially shortening the roof lip and attaching it to the baffle only at the sides, (Fig. 8a). Alternately, in areas exposed to heavy wind or rain where leakage into the nest box is a concern, the roof could be left in place and the sides of the entrance extended at the front to the same distance, with the open-top baffle then located in this position (Fig. 8b). Either of these designs would allow rosellas to reach the nest box entrance directly from the roof, but retain the need for them to climb in order to exclude Common Mynas. It would be of great importance in any such design to ensure that the modifications did not expose the entrance hole directly to the outside at the top, to make sure Common Mynas could not simply fly in from above, and to be sure that a degree of climbing from all approach directions would still be required of any species attempting to reach the nest box entrance.

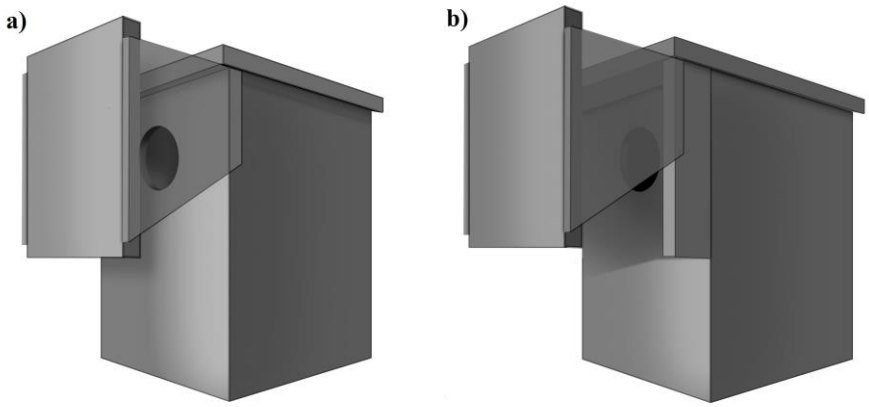

Fig 8. Shows a possible "open-top" variation of the anti-myna baffle design, where the roof is shortened and the baffle is held in place at the sides (a); and shows another possible variation where the roof is kept in place, the sides are extended to be in line with the roof, and the "open-top" baffle is located at this forward position (b). Drawings courtesy of Dr Andrew Howells, School of Design Communication and IT, University of Newcastle.

Nest Material Provisioning

Pell and Tidemann (1997b) were the first to suggest that Common Mynas may be employing a strategy of filling not only the nest site they are actively using for breeding, but other potential nest sites in the local area with large amounts of nesting material, rendering them unusable to other species. In this study we captured footage of Common Mynas filling a total of eight nest boxes with nesting material, but using only three of these for breeding. Of the remaining five, two were used for breeding by other birds, both native rosellas. This ratio (40.0%) is higher than that found by Pell and Tidemann (1997b), who report only one of 10 (10.0%) filled nest sites being subsequently used by a rosella species. Our findings are further evidence to support the hypothesis that Common Mynas do indeed fill nest sites and then fail to use them for breeding, however, questions on the strategic function of this behaviour remain. Multiple nest building has been reported in other bird species, including closely related Common Starlings (Brouwer and Komdeur 2004), as well as other, unrelated species like Barn Swallows *Hirundo rustica* (Soler *et al.* 1998b) and Marsh Wrens *Cistothorus palustris* (Verner 1965). In these cases multiple nests are thought to serve a role

in mate choice (Soler *et al.* 1998a), or possibly as a decoy to predators (Leonard and Picman 1987). These possibilities remain for multiple nest building in Common Mynas, although the former is probably less likely as Common Mynas are monogamous and thus presumably under less selective pressure to impress mates (Eddinger 1967; Wilson 1973), and in any case both sexes construct the nest (Counsilman 1974).

Interestingly, Common Mynas in this study constructed their own nest sites differently to those they did not use. They brought significantly less nesting material to nest boxes that did not have any species breeding in them than those where Common Mynas or rosella species were breeding, but a greater proportion of this material was artificial. On the other hand, less artificial material was brought to sites where rosellas were nesting, even compared to sites used by Common Mynas. In addition, Common Mynas usually provided materials to sites occupied by rosellas primarily during, and slightly after, rosellas had begun to use the nest site for breeding. In fact, significantly more nesting material was brought to sites that were occupied by rosellas than sites that were not occupied at all. If Common Mynas were attempting to use the site themselves we would expect them to begin filling the nest box well before another species arrived, and cease the behaviour either once this species had clearly established itself, or once the Common Myna pair had established a nest elsewhere, but this was not the case. This lends support to the hypothesis that nest material provisioning serves a function beyond merely construction of multiple nests for personal use or as an insurance policy against site failure. However, further data is needed before multiple nest construction can be confirmed as a competitive strategy. Indeed if it is, it did not appear to be a successful one in this study, as both rosella pairs successfully reared young from nest boxes provided with additional nesting material. Examination of multiple nest building behaviour in a much larger number of nest boxes is needed.

Interspecific interactions

Contrary to expectations, we found that rather than displacing or attacking one another, the birds in this study, including Common Mynas, tended to avoid

approaching a nest box if a bird of another species was present. This contradicts several anecdotal reports (Wright and Wright 1991; Lindenmayer 1993; Peters and Peters 1993; Pell and Tidemann 1997b) describing high levels of aggressive behaviour by Common Mynas. Nest sites in this study area had been artificially increased over the past two to three years by the instalment of over 200 nest boxes, with more than half of these not yet taken up by any species (K. Grarock, p. comm.), suggesting that nesting resources were essentially unlimited in this area. It is likely that aggressive behaviour in Common Mynas is reserved primarily for situations where resources are limited. Additionally, the nest sites in this study were newly established, and behaviours observed may not be representative of the patterns of interspecific interactions at more established (and potentially more valuable) locations. Interspecific interactions in areas where nest sites are well established and/or limited should certainly be investigated; however, this study demonstrates that where ample resources are available Common Mynas are unlikely to engage in unnecessary aggression. This strategically makes sense: where alternative resources can be found, avoiding sites occupied by other individuals decreases the risk of potentially costly confrontation (Muldal *et al.* 1985). Indeed, other studies have found that the most successful invasive species are usually not the most aggressive ones (Sol *et al.* 2011), and the tendency of Common Mynas to avoid other species at nest boxes is also similar to their reported avoidance of heterospecifics at food resources (Lowe *et al.* 2011; Haythorpe *et al.* 2012).

Interestingly, although overall displacement events were low, Common Mynas were more likely to be in pairs while initiating these interactions than at other times, and initiated more displacement events while in pairs than other species. When confrontation is necessary, co-ordination with conspecifics so as to engage in encounters while in pairs or groups may reduce overall risk, and increase the chance of a successful attack. This, in the first place, is evidence to suggest that some of these putative displacements events may be instances of actual displacement occurring, rather than images of separate bird species coincidentally occurring in close temporal proximity. Second, this could indicate that Common Mynas are employing a deliberate strategy to increase their chance of success when engaging in aggressive encounters. Species such as Noisy

Miners frequently engage in group mobbing, and this has been shown to be an effective strategy for excluding other species (Dow 1977) or defending nest sites (Arnold 2000). However, it should be remembered that the overall levels of displacement in this study were still very low (and may not have represented displacement at all), and further work is needed on this behaviour in Common Mynas, especially when engaging in competition over limited resources, to confirm this hypothesis.

Acknowledgements

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Appendix 1: Table of all species recorded at nest boxes and their image frequency. As some images contained more than one species, the per cent sum is greater than 100.

Species	Scientific name	Image number	Per cent
Australian King-Parrot	*Allisterus scapularis*	26	0.05
Australian Magpie	*Cracticus tibicen*	49	0.10
Common Blackbird	*Turdus merula*	67	0.13
Common Myna	*Sturnus tristis*	28 235	55.19
Common Starling	*Sturnus vulgaris*	55	0.11
Crested Pigeon	*Ocyphaps lophotes*	56	0.11
Crimson Rosella	*Platycercus elegans*	11 973	23.40
Eastern Rosella	*Platycercus eximius*	5 448	10.65
Galah	*Eolophus roseicapillus*	3 506	6.85
Gang-Gang Cockatoo	*Callocephalon fimbriatum*	5	0.01
Grey Butcherbird	*Cracticus torquatus*	1	0.00
Grey Fantail	*Rhipidura albiscapa*	1	0.00
House Sparrow	*Passer domesticus*	76	0.15
Magpie-Lark	*Grallina cyanoleuca*	52	0.10
Noisy Miner	*Manorina melanocephala*	5	0.01
Pied Currawong	*Strepera graculina*	1 057	2.07
Red Wattlebird	*Anthochaera caranculata*	382	0.75
Rock Dove	*Columba livia*	8	0.02
Satin Bowerbird	*Ptilonorhynchus violaceus*	222	0.43
Silvereye	*Zosterops lateralis*	5	0.01
Spotted Dove	*Streptopelia chinensis*	3	0.01
Spotted Pardalote	*Pardalotus punctatus*	14	0.03
Striated Thornbill	*Acanthiza lineata*	1	0.00
Sulfur-Crested Cockatoo	*Cacatua galerita*	97	0.19
Superb Fairy-Wren	*Malurus cyaneus*	1	0.00
Willie Wagtail	*Rhipidura leucophrys*	1	0.00

Appendix 2: List of interval times with the number of occurrences of that interval time beside it, each of the three target species, following the removal of all values two standard errors above the means.

Common Mynas		**Crimson Rosellas**		**Eastern Rosellas**	
Interval times (Seconds)	Occurrences	Interval times (Seconds)	Occurrences	Interval times (Seconds)	Occurrences
4	1	5	3	5	6
5	11	6	46	6	29
6	69	7	39	7	17
7	83	8	18	8	8
8	59	9	14	9	8
9	41	10	14	10	3
10	18	11	7	11	1
11	27	12	6	12	3
12	9	13	2	13	3
13	14	14	3	14	1
14	14	15	3	16	4
15	15	16	1	17	3
16	8	17	5	18	1
17	8	18	2	19	2
18	9	19	4	20	2
19	9	20	3	21	1
20	3	21	2	22	2
21	8	23	2	23	1
22	4	24	2	24	3
23	5	25	4	25	1
24	3	26	1	26	1
25	7	27	2	27	1
26	2	28	1	29	2
27	1	29	1	34	2
28	4	30	1	38	1
29	4	31	1	43	1
30	6	32	2	44	1
31	2	36	1	49	1
32	4	44	1	51	1
33	3	45	1	56	1
34	3	49	2	62	2

35	2	50	1	63	1
36	3	52	1	65	1
37	2	53	1	66	1
38	1	55	2	69	1
39	3	56	1	72	1
40	3	57	1	73	2
41	3	60	1	74	1
42	2	61	1	77	1
44	5	63	1	78	1
45	2	64	1	97	1
46	2	65	1	112	1
47	2	66	1	120	1
49	2	67	1	123	1
50	2	71	1	130	1
51	1	73	3	131	1
52	1	79	1	140	1
54	2	86	1	152	1
55	3	92	1	153	1
56	1	93	1	213	1
57	2	94	1	215	1
58	1	110	1	224	1
60	1	148	1	248	1
61	2	154	1	253	1
64	2	169	1	282	1
66	1	172	1	322	1
67	1	181	1	454	1
70	1	196	1	654	1
71	1	203	1	896	1
73	2	207	1	898	1
76	3	244	1	1002	1
84	2	252	1		
86	1	265	1		
88	1	374	1		
89	1	386	1		
90	1	416	1		
95	1	526	1		
97	1	747	1		
99	1	837	1		
102	1	1142	1		
103	1	1790	1		
104	1	2208	1		
109	1	2469	1		
115	2	2564	1		
118	1	2608	1		
127	1	3647	1		
128	1	3722	1		

134	1	4111	1
136	1	4589	1
139	1	4828	1
140	1		
142	1		
144	1		
145	1		
146	1		
150	2		
156	1		
172	1		
174	1		
175	1		
180	1		
181	1		
183	1		
192	1		
195	1		
199	1		
204	1		
228	1		
232	1		
237	1		
245	1		
248	1		
251	1		
295	2		
324	1		
342	1		
346	1		
407	1		
451	2		
511	1		
558	1		
571	1		
577	1		
638	1		
758	1		
779	1		
824	1		
825	1		
1004	1		
1021	1		
1141	1		
120			
4	1		

1254	1
1485	1
1490	1
1495	1
2145	1
2319	1
2344	1
2477	1
2521	1
2667	1
2988	1
3056	1

Chapter 4: The use of nest boxes in suburban backyards: homeowner perspectives

Top: Eastern Rosellas at a nest box in a suburban back yard, photo by B. Walters. Bottom: Eastern Rosella nestlings in a homeowner-provided nest box, photo by G. Smith.

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Title: The use of nest boxes in suburban backyards: homeowner perspectives

Authors: Kathryn M. Haythorpe[A,C], Danielle Sulikowski[A], Darren Burke[B].

[A] School of Psychology, Charles Sturt University, Panorama Avenue, Bathurst, NSW, Australia

[B] School of Psychology, Newcastle University, 10 Chittaway Road, Ourimbah, NSW, Australia

[C] Corresponding author: kathryn.haythorpe@gmail.com

Abstract

Nest boxes are a critical resource for hundreds of species of wildlife. In this paper we explore the experience of homeowners with artificial nest boxes and document attitudes and opinions, particularly with regards to uptake by common mynas, a cavity-nesting bird species that has been introduced to every inhabited continent in the world. We wanted to determine whether common myna presence may be indirectly affecting some native species by influencing homeowners' decisions regarding whether or not to provide nest boxes on their properties. We surveyed homeowners about their nest box experiences and attitudes. Uptake of nest boxes by common mynas generally did not prevent the nest box experience from being a positive one, nor did it decrease the level of enjoyment felt by residents at owning a nest box, and did not significantly contribute towards removal of nest boxes. The primary factor preventing the experience from being a positive one was failure to attract any wildlife, and only to a much lesser extent the attraction of non-target species, which included common mynas, but also other native and non-native species. On the whole, owning a nest box was considered a positive experience, with nest boxes giving residents a sense of pride, connection to the natural world, and education about wildlife.

Keywords: common myna, local conservation, rosella, public attitudes, *Sturnus tristis*

Introduction

The importance of nest boxes for cavity-nesting species displaced by land clearing is increasingly being recognised (Durant et al. 2009; Gibbons and Lindenmayer 2002; Goldingay and Stevens 2009). In Australia, up to 300 native vertebrate species use hollows for breeding and may become reliant on some form of artificial hollow or nest box as urbanisation increases (Gibbons and Lindenmayer 2002; Newton 1994). Where native habitat has been degraded, nest boxes provide a viable nesting alternative for many species worldwide (e.g. Ardia et al. 2006; Beyer and Goldingay 2006; Kerth et al. 2001; Lindenmayer et al. 2009; Menkhorst 1984; Newton 1994; Pogue and Schnell 1994; Smith and Agnew 2002). Nest boxes have been used as management tools for more than two decades (Olsen 1996), can be used for population control (e.g. Tidemann et al. 2010), and can aid in research on species that may otherwise be difficult to observe (Beyer and Goldingay 2006; Goldingay and Stevens 2009; Lambrechts et al. 2010). For individuals, nest boxes can also provide an important sense of connection to wildlife, particularly for people living in urban centres who may otherwise have limited exposure to the natural environment. It is well known that a feeling of personal connection to nature is key to fostering responsible environmental behaviour (e.g. Pyle 2003; Wilson 1984).

Cost is a major issue associated with providing nest boxes as part of a conservation strategy. Harper et al. (2005) estimated the cost of installing 120 boxes at approximately $5,500, and monitoring at an additional $2,500 per year. Many homeowners, however, currently provide and maintain nest boxes for wildlife at their own personal cost. Harnessing this resource could prove fruitful for managerial bodies and provide a positive way for homeowners to connect with local wildlife and the scientific community. The current study explores this possibility and examines what role local residents may play in such undertakings, and the factors that might either motivate or discourage them.

Uptake of urban nest boxes intended for native wildlife by undesired, or "non-target" species, presents a major challenge. Common mynas (Aves: Sturnidae, *Sturnus tristis*) are South-East Asian cavity-nesting birds, and have been

introduced to every continent in the world (Long 1981), where they now thrive in urban centres. Common mynas have been recorded preying on eggs and nestlings of some overseas birds (e.g. Byrd 1979; Grant 1982; Hughes et al. 2008), and are thought to be contributing to a decline in the abundance of endangered species including Tahiti Monarchs *Pomarea nigra* (Blanvillain et al. 2002) and Echo Parakeets *Psittacula eques* (Jones 1996). In Australia, concern has been raised over the ability of common mynas to outcompete native parrot species for hollow resources (Pell and Tidemann 1997). The Australian public generally consider common mynas as pests, and concern exists about their substantial noise levels, potential impact on biodiversity, and risk to public health due to faecal mess at communal roosts (Nee et al. 1990; Tidemann 2003; 2010; Yap et al. 2002). A poll by the Australian Broadcasting Company found Australians considered them to be the "most significant pest/problem", as well as the "pest/problem that needs more control" (http://www.abc.net.au/tv/wildwatch/results/award.htm). Uptake of urban nest boxes by common mynas is substantial: 38% in suburban Melbourne (Harper et al. 2005) and 45% in Canberra (Pell and Tidemann 1997), and might be expected to be an undesirable outcome for private nest box owners. Should it cause residents to remove, or deter them from installing, nest boxes this could indirectly impact urban populations of native species by lessening the overall number of available local nest sites. One of the primary motivations of the current study, therefore, is to determine what impact, if any, nest box use by common mynas has on the attitudes and behaviours of resident nest box owners. The purpose of this paper is to gain an insight into the experiences of residents who own a nest box, and in so doing discover what factors affect the quality of the experience. Rather than test direct hypotheses, we sought to examine the factors that affected residents' subjective experience of their nest box and attempted to find explanations of these by qualitatively analysing the responses of the residents.

Method

Survey

We administered a brief, 15-20 minute survey, hosted online by Survey Monkey, and also provided hard copies to some respondents. The only requirement for participation was that homeowners must own, or have previously owned, a nest box, and no minimum length of time of ownership was required. Questions were broken up into five sections: nest box experience; nest box characteristics; host tree or structure characteristics; uptake species; and observations of aggression (see Table 1). Participants also provided their postcode.

Category
Question/s
Responses options, if appropriate
Nest box experience
Overall, has your experience with your nest box been:
A positive one / a negative one / neutral / not sure
What is the reason for your answer to question 3 above?
How much has having a nest box increased your awareness of species using your property?
How much has having a nest box helped you to learn about the species on your property?
How much has having a next box increased your appreciation and affinity for the wildlife on your property?
A lot / Somewhat / Slightly / Not at all
Nest box characteristics
What distance is the nest box from the ground (in metres)?
What direction does the nest box face? Pick two if necessary, e.g. North and East for North-East
North / East / South / West / Don't know
Host tree characteristics
Is the tree species...?
Native / Not native / Don't know
Is the tree a Eucalypt?
Which description best fits your tree?
Shrub / Small tree / Medium tree / Large tree
Uptake species
What species have you seen using the nest box (actually going in and out of the entrance or bringing nesting material or food to the nest box)?
Aggression

Have you ever noticed fighting or aggression in close proximity to the nest box?

Yes / No

What species were involved in the encounter you observed?

Did any appear to 'win' or 'lose', and if so, which ones?

Are there any other details you can provide about these encounters? Every piece of information is important.

Have you ever observed common mynas (sometimes known as Indian mynas; pictured below) interacting with nesting species at your nest box in any way? If yes, please describe the encounter.

Do you have any other comments or interesting observations to share?

Table 1. List of questions provided to survey respondents.

To avoid confusion with noisy miners *Manorina melanocephala*, a native species with a similar name that common mynas are often mistaken for, a picture of a common myna and the words "Note: common mynas are a brown-coloured bird, not the commonly confused noisy miner, which is grey in colour", was included at question 25. Responses to these questions were excluded if it was unclear if a proper distinction had been made between common mynas and noisy miners, e.g. the use of terms like "common miner".

Participants

A total of 153 participants were recruited through online invitations on bird interest group websites, postings in council newsletters, random letterbox drops of hard copies throughout the Greater Newcastle region, and distribution to members of the Great Nestbox Project, a study conducted by a separate group of researchers at the Australian National University in Canberra. The majority of participants were located in the Canberra (n = 74, 48.4%) or Greater Newcastle (n = 35, 22.9%) regions, with other participants recruited from Sydney (n = 9, 5.9%), regional New South Wales (n = 10, 6.5%), Queensland (n = 11, 7.2%), Victoria (n = 9, 5.9%), South Australia (n = 4, 2.6%), and Tasmania (n = 1, 0.7%).

Results

Homeowner attitudes and opinions

Of the 153 residents, 151 provided valid responses to the last three questions of the nest box experience section of the survey relating to how much the nest box experience increased residents' awareness, appreciation and understanding of wildlife. These were scored from 4 (a lot) to 1 (not at all) and summed to give each respondent an overall quantitative enjoyment score ranging from 3 to 12, termed the "mean enjoyment score". The mean enjoyment scores were slightly bimodally distributed, with a higher number of scores in the lower and upper ranges, although the upper ranges had the greatest number of scores overall. The highest score possible, 12, was recorded for more participants than any other score (29 occasions; Figure 1).

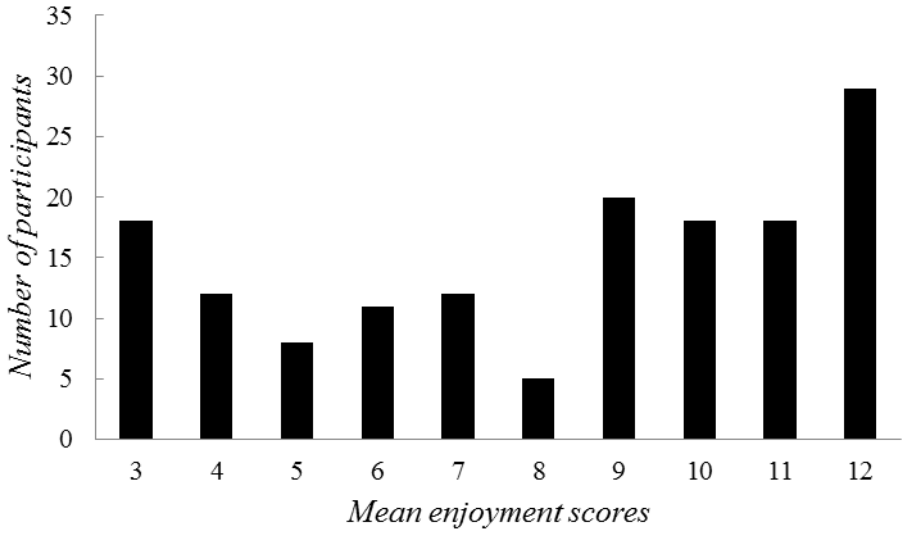

Figure 1. Shows the mean enjoyment scores of participants who own a nest box (n = 151).

The experience of owning a nest box was rated as positive by 97 of 153 respondents (63.4%). A further 44 (28.8%) rated the experience as neutral/not sure, with the remaining 12 (7.8%) rating it as negative. The 97 respondents who rated the experience as positive gave a total of 105 reasons for doing so.

Enjoyment at seeing wildlife using the box accounted for 40 of the 105 (38.1%) reasons, and specifically enjoyment at watching chicks, which accounted for a further 28 (26.7%) reasons. Other reasons given were feeling that the nest box attracted beneficial wildlife to the garden (n = 7 or 6.7%), or helped drive away pest species (n = 7 or 6.7%); provided a sense of satisfaction at the feeling of helping wildlife (n = 6 or 5.7%) or contributing to wildlife research (n = 5 or 4.8%); helped facilitate education on wildlife for themselves (n = 4 or 3.8%) and their children (n = 4 or 3.8%); provided a point of interest for neighbours and friends (n = 3 or 2.9%) and prevented possums from nesting in their roof (n = 1 or 1.0%; Figure 2).

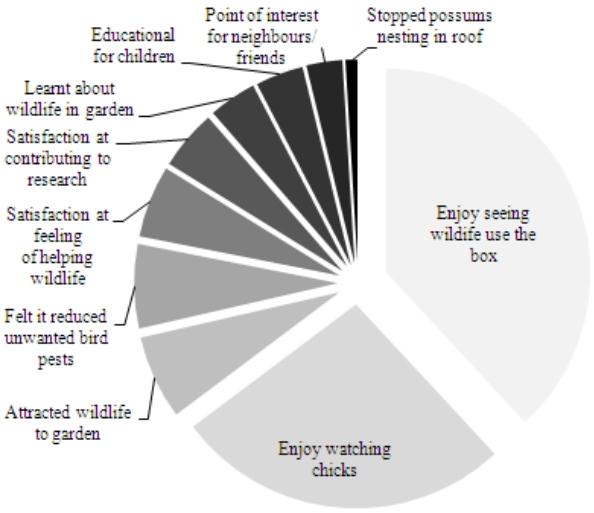

Figure 2. Shows the reasons given by homeowners for rating the experience of owning a nest box as a positive one.

The 56 respondents who did not rate the experience as positive gave a total of 63 reasons for not doing so. Attracting common mynas accounted for 7 of the 63 (11.1%) reasons, and witnessing common myna aggression accounted for a further 3 (4.8%) reasons, totalling 10 reasons (15.9%; Figure 3a). Other reasons given include failure to attract any wildlife to the nest box (n = 30 or 47.6%), attracting other species the residents considered undesirable (referred to henceforth as non-target species), excluding common mynas (n = 17 or 27.0%), witnessing traumatic chick deaths through factors such as overheating (n = 3 or 4.8%), witnessing aggression from common starlings *S. vulgaris* (n = 1 or 1.6%),

or damage caused to the host tree or other garden vegetation (n = 2 or 3.2%; Figure 3b).

Non-target species reported were “bees”, which were most likely European honey bees *Apis mellifera* (n = 10), common mynas (n = 7), “possums”, which were most likely common brushtail possums *Trichosurus vulpecula* or common ringtail possums *Pseudocheirus peregrinus* (n = 3), common starlings (n = 1), dollarbirds *Eurystomus orientalis* (n = 1), rainbow lorikeets *Trichoglossus haematodus* (n = 1), and black soldier flies *Hermetia illucens* (n = 1; Figure 3c).

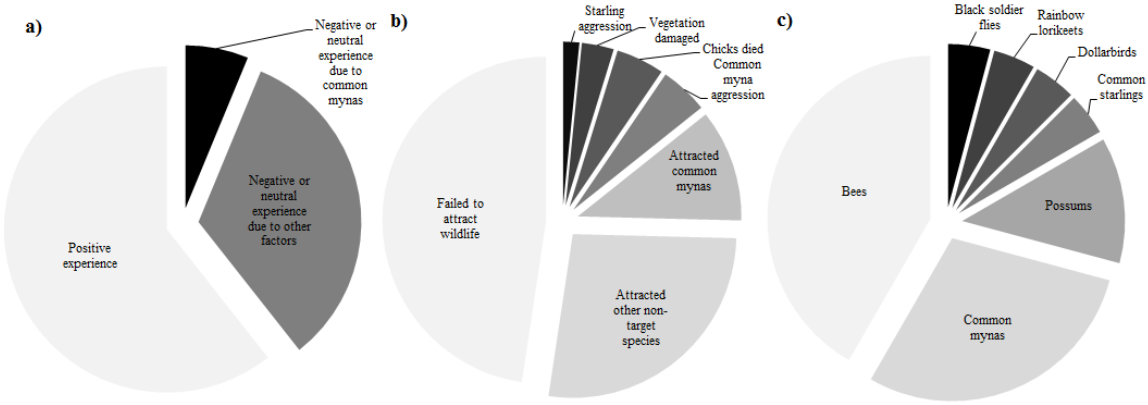

Figure 3. Shows the proportion of homeowners who rated the experience of owning a nest box as positive, or negative/neutral/not sure, and whether this was due to common mynas or other factors (a), the reasons given by homeowners for not having a positive experience (b), and the non-target species cited as preventing the experience being a positive one (c).

Bee occupation appeared undesirable to many people. In addition to 10 respondents claiming bees prevented the experience being a positive one, a further six respondents commented negatively on bees in some way. Five people stated that they removed their nest box, at least temporarily, due to bee occupation, and two respondents claimed bees destroyed nesting rosella eggs or chicks. However, six other respondents expressed satisfaction at having bees occupy nest boxes, or even attempted to encourage it. Responses to possums using nest boxes were also mixed, but more positive overall. Six respondents commented negatively on the existence of possums in their nest boxes, with three of these citing it as a reason for preventing their experience being a positive one.

However, 23 respondents expressed some level of satisfaction at having possums using nest boxes, with 13 of these stating that they had installed nest boxes specifically to attract them.

A total of 43 out of 141 (30.5%) people reported that common mynas nested in their nest box, with the remaining 12 not providing sufficient detail to determine either way. Responses to common myna uptake were mostly negative, with 33 (76.7%) residents expressing dissatisfaction at witnessing common mynas using their nest box, and five people (11.6%) stating they removed or disabled their nest box after it was occupied by common mynas to prevent any further use. However, despite having common mynas using their nest box, the overall experience of owning a nest box was still reported as positive by 28 of these 43 (65.1%), a proportion that is actually slightly higher than both the overall rating (63.4%) and the rating for those that did not have common mynas nesting (61.2%). Six people spoke positively of common myna uptake of their nest box and four others mentioned it in neutral terms, or did not mention it at all.

To compare attitude towards common mynas, respondents were split into two groups: those that reported common myna uptake of their next box and those that did not. An independent samples t-test compared the mean enjoyment scores from each of these two groups. Uptake by common mynas did not reduce reported enjoyment; in fact there was a non-significant trend ($t_{(139)} = 1.846$, $p = 0.067$) for those who did not report having common mynas nest to have lower enjoyment scores (n = 98, M = 7.82, SD = 3.34) than those who did (n = 43, M = 8.88, SD = 2.70).

Discussion

Reported attitudes towards common mynas were generally negative as expected, however, only a small number of people removed or disabled the nest box as a result of common myna presence, and common myna uptake did not prevent the experience of owning a nest box from being a positive one, or affect reported enjoyment ratings. Some concern was expressed over attracting other non-target species, but overwhelmingly the scenario most responsible for preventing the

experience of owning a nest box being a positive one was the failure to attract any wildlife at all. We discuss the benefits of nest box provisioning for people and wildlife and recommend encouragement of this practice, even in environments that may not seem suitable or likely to attract uptake.

Effect of common mynas

Interestingly, despite receiving a large number of negative comments, having common mynas nest did not appear to affect the overall experience of owning a nest box, and in fact may have improved it. The proportion of experiences rated as positive were slightly higher for those residents who had common mynas nesting, and there was also a trend towards a higher mean enjoyment score for these people than for those that did not have common mynas nesting. This is not surprising when it is considered that the greatest contributor towards dissatisfaction with the nest box experience was failure to attract any wildlife at all. Attracting common mynas provides more interest value than attracting nothing, and so for some people may be seen as a preferred option, even in spite of a potential dislike of the species. For instance, one resident described being interested in watching the daily lives of the common mynas and learning about how they raised their young, even though they later chose to have the birds removed. For some people, living in environments that do not readily attract native species, common mynas may be a more interesting and educational alternative to nothing at all. Importantly, common myna uptake led to removal or disabling of the nest box in only approximately one tenth of cases. It appears that, on the whole, common myna use of nest boxes is not affecting wildlife indirectly by turning residents off the experience or reducing their willingness to provide nest boxes.

Other non-target species

Second to attracting nothing, the next largest reason given for dissatisfaction with the experience of owning a nest box was attracting non-target species including bees, possums, common mynas, common starlings, dollarbirds, rainbow lorikeets and black soldier flies. Mirroring attitudes towards nesting common mynas and

consistent findings from other studies (Eymann et al. 2006), homeowners did not always agree on whether a given nesting species was welcome or not. The most common unwelcome species were bees. Residents expressed concern over their own safety and that of their family, as well as of any nesting birds, and were unhappy to be contributing to an increase in the numbers of a species they perceived as an introduced pest. However, some residents appeared happy with the presence of bees in the nest box, claiming it provided interest and beneficial pollination for their garden. Possums were also a concern for some residents, and were described as a nuisance and removed. The majority of people who commented on possums expressed a fondness towards them, however, and were excited to see they were using their nest box, with several people installing nest boxes specifically for the purpose of attracting possums. These people saw possums as interesting and amusing animals, and were additionally glad that the possums were using nest boxes rather than the roof of their house.

The disagreement amongst homeowners about whether or not a species is welcome in their nest box may be an important issue to consider for those seeking to make use of urban residents as an inexpensive way to increase nest sites for some species in urban areas. To ensure that homeowners volunteering to be part of such an effort have realistic expectations and avoid disappointment, it may be prudent to advise homeowners of the variety of species present in any local area which may choose to nest and provide advice on anything the homeowner can do to increase or decrease the likelihood of attracting or deterring certain species. It might also be beneficial to provide homeowners with information about how to safely and humanely remove certain unwanted species. These simple steps could help ensure that homeowners are not disappointed with their nest box experience and remain motivated to maintain it.

Benefits of nest box provisioning

As the proportion of the world's population living in cities continues to grow (United Nations 2012), connecting with wildlife in urban environments is becoming difficult for many people. It has been argued (Kahn 2002) that children raised with limited exposure to wildlife and the natural environment are unlikely

to develop a realistic understanding of the costs of environmental degradation, or appreciate the importance of preserving the natural world. This psychological process, termed environmental generational amnesia (Kahn 1997; Kahn and Friedman 1995), can lead to complacency about environmental issues, and may ultimately result in a generation disinclined to support or engage in conservation activities (Pyle 2003; Turner et al. 2004). Owning a nest box is one way of getting close to wildlife that is available to many people, even those in heavily cleared urban centres, as it requires little space and can be installed in locations readily available to everyone, like the side of a house or a pole erected in a garden. Enjoyment provided by the provisioning of a nest box was high, with homeowners agreeing that the experience had increased their affinity and appreciation for the natural world and helped them learn about wildlife. A number of benefits of owning a nest box were discussed, including the joy of watching wildlife (particularly young chicks), a feeling of taking positive action to create a desired ecological space (attracting target species while discouraging non-target species), increasing education and knowledge, and a feeling of satisfaction at being able to help wildlife and contribute towards scientific research. A fruitful avenue of further research, beyond the scope of the current study, might be to compare the positive attributes discussed here between people who own nest boxes and those who do not.

It was clear from the homeowners' responses that many had developed a feeling of connection with the wildlife in their backyard, or even a sense of responsibility towards them, for example referring to them as "my rosellas", and providing them with food, shelter, and protection. Several participants also talked of clapping to scare off pied currawongs or common mynas when they approached the nest box, and one participant guarded an Eastern rosella clutch from interfering lorikeets by turning the garden hose on them. These kinds of direct, intimate encounters are enormously important in fostering biophilia in a society often cut off from nature and natural processes (Sanderson 2002). The pride showed by these residents, and sense of wanting to protect wildlife on a local scale, could easily develop into an understanding of the need to support conservation efforts, and more importantly, a desire to do so. Canvassing public support and engagement can be a vital step in swaying managerial or

governmental decisions on issues such as allocation of resources to conservation (Roome 1992).

One of the surprising findings of this study was the passion and enthusiasm with which homeowners responded to the questionnaire. Very few participants left open-answer questions blank or responded with only a single word, and many contained over 100 words per answer. Most people had clearly put a great deal of time and thought into their response, with a handful of people even contacting the researcher at a later time to send through more information they had thought of, pictures they had taken, or updates. It seems that most people with nest boxes have a great deal to say about them and are more than happy to have an interested ear to receive the information. The concept of helping natives appears to be a fashionable one, and many people are happy to be seen as contributing to this effort. Given the high cost of installing and monitoring nest boxes at a managerial level, canvassing public enthusiasm to assist in this task seems like a viable option. In Canberra, more than 200 people in nine suburbs responded to an offer to host a nest box (Kate Grarock, p.comm.), suggesting that many people are willing to consider the idea if prompted or provided with an incentive (in this case a nest box installed free of charge). A large proportion of the general population have never even heard of nest boxes, but express interest when the concept is mentioned to them (pers. obs.). It is likely that with some simple campaigning, particularly if an incentive was offered such as free installation or a discount on materials, many thousands of people would be willing to install and provide basic maintenance and/or monitoring for a nest box. Although relying on homeowner monitoring alone can at times be problematic (Tidemann et al. 2010), a combination of official and unofficial monitoring would decrease the total resources required of managerial bodies, and provide residents with a feeling of giving back to the community. Canvassing support from local homeowners to provide and monitor nest boxes should be considered particularly in areas where nest box provisioning is being used as part of a directed conservation strategy (e.g. Crowley et al. 1998).

Conclusions

Nest box provisioning by homeowners is an activity that brings a great deal of joy to most homeowners, with the main cause for concern being the inability to attract any wildlife and, to a lesser extent, attracting only non-target species. There was no evidence from our study that the use of nest boxes by introduced common mynas had a disproportionately negative effect on homeowners' attitudes towards their nest box or their willingness to maintain it. The benefits of nest box provisioning for both people and wildlife are clear, and canvassing of public enthusiasm to assist managerial bodies with this resource-costly activity that is so beneficial to numerous native species is recommended. Future work in this field should look at comparing the attitudes towards wildlife expressed by those people who own nest boxes and those without nest boxes, in order to determine if the positive trend reported by this study is linked to nest box ownership.

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Conclusions

Specific contributions

Perhaps one of the most rewarding aspects of scientific research is the feeling of breaking new ground, developing knowledge, modifying current understanding, and applying it all to people, animals, and real-world situations. Below I highlight some of the main contributions of the research embodied in this thesis.

1) **Move towards a more realistic view of the impact of Common Mynas on wildlife**. Prior to this research, very little empirical data had been collected on the behaviour of Common Mynas in Australia, and the possible impact of this behaviour on native wildlife. Overseas, particularly on small islands, studies suggested that Common Mynas were capable of displaying high levels of aggression and posing a threat to several native species. With little evidence to the contrary in Australia, Common Mynas were assumed to play a similar role here, and likely to pose an immediate and dire threat to Australia's native wildlife. However, the research reported in this thesis has not supported that view. Common Mynas in the study areas appeared to be relatively unconfrontational around food or nesting resources, and were just as likely to receive aggression as to initiate it. In addition, their inability (or unwillingness) to penetrate bushland (chapter 2) limits their ability to pose a threat to native wildlife. While Common Mynas should certainly continue to be monitored closely, as should all introduced species, their impact on wildlife does not appears to be as severe as previously thought.

The findings of this research are supported by ongoing work by other researchers, who are coming to similar conclusions. Parsons *et al.* (2006) found Common Mynas were not negatively correlated with any small bird species in suburban gardens and concluded that they were probably not affecting native wildlife. Lowe *et al.* (2011) found Common Mynas rarely interfered with the foraging activity of other birds and did not appear to compete with them for

nesting sites, and concluded that they were probably having minimal competitive impact on native species. My findings (chapter 1; Haythorpe *et al.* 2012) supported this study, showing that even under extreme competitive conditions Common Mynas still did not display significant competition for food resources, and were in fact often the target of aggression from native species. Grarock *et al.* (2012) analysed long-term data (spanning 29 years) and found a negative relationship between Common Mynas and some native species, but also positive relationships with other native species, and the authors conclude by questioning the seriousness of the impact of Common Mynas and whether management is needed. Indeed, this is the first study to demonstrate an impact of Common Mynas at the population level, and this impact required several decades to become apparent, suggesting that this effect is gradual and subtle. Further, chapter 2 (Haythorpe *et al.* 2013) and chapters 3, and 4 are supportive of these results, indicating that Common Mynas are not very aggressive in "every day" situations, but do occasionally display competitive behaviours at nest sites, although these are rare and their impact is likely to be minimal. Findings like these are gradually helping scientists, managerial bodies, and the public alike to develop a realistic view of the relative impact of Common Mynas and requirements for management. This kind of data will help channel limited resources allocated to wildlife management into the areas where it is most needed.

2) **Investigation into entry and exit behaviours of birds at nest boxes and suggestions for improvement of anti-myna baffle design**. This work (chapter 3) provides information useful for forming predictions on the efficacy of the anti-myna "baffle" design for nest boxes. It is not yet clear whether the baffle does indeed exclude Common Mynas; or, perhaps more importantly, if it allows the natural behaviour of native rosellas to be maintained and continues to provide them with a desirable breeding resource. Yet despite this lack of information, the baffle design is now officially recommended by Birds Australia, is commercially available, and is frequently in use by the public. Although empirical testing in the real world is most needed, the findings of this study at least contribute to our theoretical understanding of the mechanisms of the design from the perspective of avian behaviour. Importantly, we have been able to reveal limitations that had

not been previously considered. These include consideration of the need for perches, so larger species such as Crimson Rosellas can more easily gain access, and the need for a design that does not stifle the climbing behaviours of rosella species. It was previously thought that Common Mynas made more use of perches than rosella species and so perches should not be provided, but our work found this was not the case. The tendency of rosellas to climb down (in this case from a roof) to reach an entrance hole also does not appear to have been previously considered, and this behaviour has been documented and addressed by our work. In addition to highlighting these issues, this work involved developing potential design solutions based on these findings, and these are being made available to interested bodies through publication. Exploration of these designs, and further consideration of the needs of nesting native birds, is expected to follow from this work.

3) **Further data on multiple nest filling behaviour in Common Mynas**. Common Mynas have long been accused of sabotaging nest sites by filling them with nesting material to prevent their use by other species. However, despite the widespread belief that this happens, there is surprisingly little mention of this behaviour in the scientific literature. One study (Pell and Tidemann 1997) describes a possible instance of this occurring, and another (Ambrose 1982) provides speculation that it could occur. The work embodied in chapter 3 discusses another instance of this behaviour, and further empirical data on its occurrence. We found evidence to suggest that the type of nesting material brought to nest sites by Common Mynas is affected by the identity of the species attempting to use the nest site, an interesting finding that may have implications on competitive strategies used by Common Mynas. These findings pave the way for further research on this behaviour, both from a conservation and management perspective and as an issue informing our ideas of the cognition of introduced species. It also provides some evidence that nest filling by Common Mynas does not appear to be successful in completely deterring competing species from using nest boxes, which should alleviate some of the concerns of residents with nest boxes who witness this behaviour.

4) **Involvement of non-scientists and examination of social impact**. This work (chapter 4) gives a "voice" to non-scientific nest box owners, allowing them to express their thoughts and ideas on issues relevant to provisioning of nest boxes for wildlife. Given the high number of residents willingly providing nest boxes at their own expense, and the potential for this number to grow or be utilised by managers, these opinions are increasingly important. Knowledge and understanding of the concerns of non-scientists with an interest in wildlife may be vital to the success of projects that wish to involve communities on this level. Indeed, even having the knowledge that a community of interested people exists, and could potentially be called upon, is highly beneficial. The data-focussed approach of ecology can often preclude this kind of study into psycho-social attitudes, and this kind of study is important in uniting science with society. Solid scientific process is highly desirable, but even scientifically sound projects can still fall short of achieving their goals if public enthusiasm is low or negative.

Limitations and future directions

As is often the case in scientific research, the findings of this thesis have provided as many questions as they have answers. I present here some of the main shortcomings of the present research, some suggested improvements, and avenues for future research.

1) **Experimental manipulation of Common Mynas and Noisy Miners to determine impact**. The studies presented in this thesis largely involve examining what is already occurring within the system, particularly in chapters 2 and 3. The evidence, while valuable in its own right, is necessarily correlative in nature. To truly show causation – for example, to show that Noisy Miner abundance is having an impact on wildlife, or that Common Mynas are not significantly displacing native species from nest sites – manipulation experiments need to be conducted. These could involve removal of a species from a certain area, and pre- and post-measurements of important indicators of species impact.

2) **Nest site displacement and avoidance**. Common Mynas in this study displayed limited aggression towards other species and in fact avoided nest sites when another species was present. However, birds in this study were presented with a surplus of nesting sites and so did not face high competition. It would be useful to know if avoidance behaviour continues as competition increases, or if in the presence of limited nesting resources Common Mynas switch to a strategy of displacing or attacking other species. In this study nest boxes were newly established and less than half were used for nesting, but uptake rate could be expected to increase as new generations of birds continue to take advantage of this resource and increase in number. Examination of these same nesting resources at a later point in time, using the same or similar methods employed in this study, could provide a very useful comparison. Alternatively, nest site availability could be experimentally manipulated by the provisioning of different numbers of nest boxes to certain areas, and aggression levels compared between areas limited or not limited by availability of suitable nesting hollows. In addition, exploitation competition was not considered by this study. The effect of Common Myna use of the available nesting resources on competing native species, regardless of whether Common Mynas are aggressively driving away other species or not, needs to be considered.

3) **Testing of anti-myna baffle design**. Testing of the modifications for anti-myna baffle designs on nest boxes was planned for this thesis, but unfortunately did not eventuate due to time constraints. Without creating and using these designs, their effectiveness can never be more than speculative. Nest boxes with and without open-top designs and/or perches need to be tested for efficiency in excluding Common Mynas and encouraging rosella species and other wildlife. These could be monitored in the field through motion-sensing cameras, or alternately through the use of nest box choice experiments with captive populations of both Common Mynas and a relevant native competitor such as Crimson Rosellas or Eastern Rosellas. In particular, the effect of this design in discouraging rosella species needs to be documented in order to be weighed up against the relative benefit of exclusion of Common Mynas.

4) **Investigation into use of nesting material**. The data collected on nest filling behaviour in Common Mynas (chapter 3) was incidental to the main purpose of the study, which was to investigate competitive behaviours such as acts of aggression. Ideally, data on this behaviour would need to be collected for a far greater number of nest boxes (only 8 in this study). Nest sites should also be checked continuously (perhaps even monitored through internal cameras) so the breeding status of the occupying birds is always known, as the bi-monthly checks conducted in this study left a high margin for error in determining breeding status. To do so in this study would have been logistically difficult, as nest boxes were located in a different city and internal checks required expensive field trips that were not practical on a limited budget.

Future work on this behaviour needs to focus specifically on its purpose. It is now clear that Common Mynas fill multiple nest sites with material then fail to nest in them, but the real question is, what is the adaptive benefit? Do Common Mynas "intend" to use a particular nest site and choose to fill up others to prevent competing species nesting nearby, or do multiple nest sites represent a more innocuous possibility, such as "showing off" to attract mates, or an insurance strategy against possible nest site failure? Further, what is the benefit of using artificial material rather than natural substances? Is this simply a matter of practicality (this material is readily available), or does this material discourage native species from nesting? One possibility is that this material is more likely to harbor sugary substances that may attract insects, creating unpleasant nesting conditions or even rendering a nest site temporarily unusable. The employment of such a strategy would represent a particular threat to native wildlife that could be managed and controlled by homeowners, for example using pesticides or cleaning out nest sites.

These questions could be addressed either through investigation of wild breeding pairs or through the establishment of pairs of Common Mynas in captivity. When several nest boxes and a wide variety of nesting material is presented, do Common Mynas fill one favoured nest box with different materials? Will they actively avoid filling "their" nest box with sugary materials that attract insects? Is the presence of other species required to trigger this

behaviour (suggesting a competitive function)? Substantial scope exists for research in this area.

5) **Egg predation**. The results of these studies and others suggest that egg predation by Common Mynas does indeed occur. The next logical question is the function of this behaviour. Does it serve as a competitive strategy, or merely opportunistic predation? This could again be addressed through either field studies or a captive study of mated pairs presented with nest box options and the use of quail or budgerigar eggs to simulate a real-world nesting scenario. Will birds with ample food resources still choose to consume eggs from the nest site of another bird? Will they do so even if there are several other available nesting options? Such questions should form the basis of future research in this area.

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Appendix A: PhD publications

This appendix contains peer-reviewed articles authored by the candidate arising from the research conducted for this doctorate.

1. Haythorpe, K. M., Sulikowski, D. and Burke, D. (2012). Relative levels of food aggression displayed by Common Mynas when foraging with other bird species in suburbia. *Emu*, 112, 129-136.

2. Haythorpe, K. M., Burke, D. and Sulikowski, D. (2013). The native versus alien dichotomy: Relative impact of native noisy miners and introduced common mynas. *Biological Invasions*, doi 10.1007/s10530-013-0598-5.

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Relative levels of food aggression displayed by Common Mynas when foraging with other bird species in suburbia

K. M. Haythorpe[A,C], *D. Sulikowski*[B] *and D. Burke*[A]

[A]Newcastle University, School of Psychology, 10 Chittaway Road, Ourimbah NSW 2258, Australia.
[B]Charles Sturt University, School of Psychology, Panorama Avenue, Bathurst, NSW 2795, Australia.
[C]Corresponding author. Email: kathryn.haythorpe@newcastle.edu.au

Abstract. Invasive species present economic and ecological challenges worldwide. In many cases we are not aware of the full effect they have on the environment, the extent of any damage, or the factors contributing to their success. In this study we examined the foraging aggression of wild Common Mynas (*Sturnus tristis*) as a potential explanation for their invasive success, and quantified the effect of this behaviour on other birds. Common Mynas did not display significantly more aggression than other species, and displayed significantly less aggression than native Australian Magpies (*Cracticus tibicen*). Furthermore, the presence of Common Mynas at a feeding resource had no greater effect on the abundance of heterospecific individuals than the presence of any other species. Presence of each species was negatively correlated with the presence of other species, that is all species were less likely to approach the feeding station if any other species was present there. Common Mynas also did not displace other birds at feeding sites any more frequently than three of the other four species, and less frequently than two other native species. Overall, the findings suggest that Common Mynas do not display more food-related aggression than other species in suburban habitats, suggesting that competitive aggression over food is not likely to be one of the behavioural traits leading to the success of Common Mynas in suburban habitats.

Additional keywords: displacement, foraging behaviour, invasive species, resource competition, *Sturnus*.

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Introduction

Damage caused by invasive species continues to be one of the major threats to the health and biodiversity of the natural environment worldwide. Research into invasion biology has previously paid too little attention to the role of behavioural traits in increasing the invasive success of a species. Behavioural strategies, however, are likely play a fundamental role in acquiring resources, which are key to the survival of an introduced species. This may be shown prominently in competition over food, a most important and immediately necessary resource.

Common Mynas (*Sturnus tristis*) were introduced to every continent in the world over the past two centuries (Long 1981) and abound worldwide. Since their introduction to Australia during the 19th century they have successfully colonised most of the Eastern seaboard and are rapidly expanding their range inland (Martin 1996). Their success may be partly attributable to behavioural strategies adopted when competing for resources. However, the extent to which competition for resources influences the success of Common Mynas is presently unknown. Despite their large numbers, and the implementation of various control programs to reduce populations, there are very few quantitative data on the effect of Common Mynas on native ecology. Understanding the effects of the presence of Common Mynas and the mechanisms that contribute to their success are the first steps in developing an effective management plan for this species.

Common Mynas are often thought of as aggressive and 'bullying' (Phillipps 1994). Anecdotal reports abound with accounts of Common Mynas attacking various bird species including White Terns (*Gygis alba*) (Grant 1982), Mauritius Kestrels (*Falco punctatus*) (Feare and Craig 1999), Tahiti Monarchs (*Pomarea nigra*) (Blanvillain *et al.* 2003), and Samoan Starlings (*Aplonis atrifusca*) (Freifeld 1999). There are even accounts of Common Mynas attacking people, Dogs (*Canis familiaris*), Cats (*Felis catus*) (Booth 1963; Edgar 1975), and a Coati (*Nasua* spp.) (Fitzsimmons 2006). Common Mynas are suspected of driving competing species out of desired nesting hollows and threatening breeding success (e.g. Macdonald 1951; Wright 1962; Counsilman 1974; Wright and Wright 1991; Lindenmayer 1993; Pell and Tidemann 1997).

The large anecdotal record of aggressive tendencies in Common Mynas suggests that aggression could contribute to their invasive success. Numerous other studies have attributed the success of an introduced species at least partly to their ability to compete with native species for local resources (e.g. Petren and Case 1996; Poling and Hayslette 2006; for review see Holway and Suarez 1999). Aggressive exclusion has been shown to be an adaptive and effective means of gaining sole access to a resource, such as food, that may not otherwise be available (Brown 1964). However, information on the aggressiveness of Common Mynas in regards to monopolising food resources is lacking. Counsilman (1974) records some anecdotal information on the aggression of

Common Mynas around sources of food, but with insufficient observations to draw firm conclusions. Some authors (e.g. Veerman 2002; Crisp and Lill 2006; Harper *et al.* 2005; Olsen *et al.* 2006; Parsons *et al.* 2006) have suggested that the present focus on eradication of Common Mynas is disproportional to the scientific evidence of the threat they present, and indeed thorough quantification of this threat is so far lacking.

Some studies (e.g. Rotenberry 1980) have shown that species that display aggressive competition are not necessarily the most successful ones. Rather, species that use resources opportunistically and discreetly may have greater success. An invasive species that is lacking experience of its competitors may gain by not entering into physical conflict with them. The cost of resource defence is also high (Brown 1964) and, for an invasive species, with little experience of the long-term value of particular resources, initial success may be more likely if as little energy as possible is expended attacking potentially dangerous natives or defending potentially uncertain resources.

A recent observational study suggests that Common Mynas may adopt such a 'laying-low' strategy. Lowe *et al.* (2011) reports that Common Mynas do not show greater interspecific aggression than other species and rarely interfere with their foraging activity. This survey provides an important indication of normal feeding behaviour, but does not address whether Common Mynas show aggression in more extreme feeding circumstances, where the potential pay-off may be higher. It is important that we determine how effective Common Mynas can be at aggressively outcompeting other species for resources as competition for food can result in dramatic population declines (Coblentz 1978). The potential effect on native birds would be heightened by the remarkably broad diet of Common Mynas, forcing even species with specialist diets to compete with them.

In this study, we looked at the feeding behaviour of wild avian populations containing Common Mynas and native species such as Australian Magpies (*Cracticus tibicen*; hereafter simply Magpies), Noisy Miners (*Manorina melanocephala*), Magpie-larks (*Grallina cyanoleuca*), Australian Ravens (*Corvus coronoides*; hereafter simply Ravens), Crested Pigeons (*Ocyphaps lophotes*), and Silver Gulls (*Larus novaehollandiae*); and introduced species including feral Rock Doves (*Columba livia*), Spotted Turtle Doves (*Streptopelia chinensis*), and Common Starlings (*Sturnus vulgaris*; hereafter simply Starlings). If the success of Common Mynas is partly driven by aggressive competition around valuable food resources, we would predict that Common Mynas (when compared with individuals of other species): (1) would initiate more aggressive interactions towards both other Common Mynas and individuals of other species when foraging at concentrated food sources; (2) displace individuals of other species at food sources at a higher rate than these other species do; and (3) their presence would be associated with a disproportionate reduction in the amount of time other species spend foraging at the food source. A 'laying-low' strategy, however, would not result in such differences.

Methods

Procedures

Thirty-eight feeding stations were established in public suburban areas in Newcastle, Australia, spanning an area of 10 400 m^2, with a minimum distance of 256 m between adjacent feeders. Sites included small parks, pathways, street corners, and suburban backyards. Observations were conducted between 0600 and 1100 hours throughout the Common Myna breeding season (September–March) and non-breeding season (April–August). Individual sites were sampled (as described below) between four and seven times over the course of the study.

The feeding stations contained a mix of commercial seed mix (a blend of Sorghum, Wheat, Barley, French White Millet, Maize, Black Sunflower and shell grit), dog pellets, carbohydrate-rich items (cooked white rice, white bread, Madeira cake), finely chopped fruits (sultanas, raisins, and apricots) and live mealworms (*Tenebrio molitor*). Food was placed on a wooden board 30 × 20 × 1 cm. Wild birds were encouraged to feed from it during two pre-feeding periods. For 7 consecutive days after pre-feeding, birds were video-recorded for 5 min of activity, beginning from the time of the first arrival of a bird of any species at the feeding station. On some days no birds arrived, or birds failed to remain for the full 5 min, and data from these days were not used.

For the purpose of analysis we divided the area surrounding the food board into a foraging zone (a circle of 1-m radius surrounding the feeding board) and an outer zone (a circle of 25-m radius surrounding the foraging zone). This allowed us to note birds present in the area but not accessing the food (outer zone), and those interacting in some way with the food or with other birds accessing the food (foraging zone).

As spatial clumping increases aggression (e.g. Goldberg *et al.* 2001; Robb and Grant 1998), we minimised artificially heightened levels of aggression owing to jostling for physical space at the feeding site by providing food on the board and also scattered within the foraging zone. This allowed birds on the periphery of the foraging zone to access food and reduced the effect of clumping.

Analysis

Aggressive acts

Aggressive acts were defined as behaviours directed at another bird that had the potential to cause harm, and included pecking, chasing and swooping, even when such behaviours were ritualistic only. Video-recordings were played back at 0.5× normal speed using VLC Media Player 1.0.3 (VideoLAN, Paris, France) and intraspecific and interspecific aggressive acts were continuously scored for all individuals present in the foraging zone. For each aggressive act we noted the identity of both the initiating and recipient species. This resulted in counts for each species of number of initiated interspecific, initiated intraspecific, and recipient aggressive acts. As the numbers of individuals in the foraging zone fluctuated throughout the study period, an abundance score was calculated for each species at each sampled site by counting the number of individuals of that species at each 30-s interval over 5 min and summing these 10 scores. All aggression data from each site was pooled, regardless of the number of within-site trials, and each site was treated as a replicate.

Species that were present in the foraging zone (of any given site) on fewer than 10 occasions were excluded from that site only for the analysis. We calculated a relative aggression score for each species at each site by first calculating how many aggressive acts

we expected each species to be responsible for at each site given the relative abundance of each species and the total number of aggressive acts that occurred at the site. We did this using the formulae:

$$E_{agg} = (ab_x/AB) \times A$$

and

$$NE_{agg} = A - E_{agg}$$

where E_{agg} is the number of aggressive events we would expect species x to initiate; NE_{agg} is the number of aggressive acts we would expect species x *not* to have initiated; ab_x is the abundance score of species x; AB is the sum of the abundance scores of all species present at the site and A is the total number of aggressive events observed at the site.

We then calculated the relative aggression score (RA) for each species using the formula

$$RA = [(E_{agg} - O_{agg})^2/E_{agg}] + [(NE_{agg} - NO_{agg})^2/NE_{agg}]$$

where O_{agg} is the number of aggressive acts species x was observed to initiate; and NO_{agg} is the number of aggressive acts that species x was *not* observed to have initiated. We also defined the RA score as negative if a species was observed to initiate fewer aggressive acts than was expected and positive if a species was observed to initiate more aggressive acts than was expected. The distribution of these scores was similar to the Chi-square distribution and so they were cube-root transformed before analysis to achieve normality.

Relative aggression scores calculated in this manner are not influenced by either the absolute or relative number of a given species at a given site. This means they will not be higher or lower at one site compared with another simply because there are more individuals or a greater proportion of that species at one of the sites. The sign and magnitude of these scores is also meaningful as a score of zero means that a species is behaving exactly as aggressively as the average aggressiveness of other species with which it shares a site. As different combinations of species (and habitat characteristics) can presumably result in situations that are either more or less conducive to aggressive acts, this score also has the benefit that a low RA score cannot be attributed to a species simply because it occurs at sites that are relatively unconducive to aggressive acts. This adds validity to between-species comparisons of mean RA scores (mean calculated for a species from scores from numerous sites).

We calculated three RA scores for each species, reflecting: (1) interspecific acts of initiated aggression; (2) intraspecific acts of initiated aggression; and (3) events in which the species in question was the recipient of aggression. We conducted a one-way analysis of variance (ANOVA) on the cube-root transformed RA values with species as the independent variable. In order to minimise the likelihood of Type-II errors and fully explore the data, we used Fisher's Least Significant Difference (LSD) tests for *post hoc* comparisons. Tukey's Honestly Significant Difference (HSD) tests were also applied to ensure that the use of Fisher's LSD tests did not increase the incidence of Type-I errors.

We then correlated the interspecific RA scores and recipient RA scores to determine if the species more likely to initiate aggressive acts were less likely to receive them. The analysis was conducted both with and without the RA scores of Common Mynas to see if Common Mynas deviated from the trend produced by the other species.

Association

During the 5-minute period of observation, recordings of species present in the foraging and outer zones were made at 30-s time-stops. This method, known as instantaneous time-sampling, is described by Martin and Bateson (1986) and has been widely used and shown to give a very close approximation to actual proportion of time spent in a state as measured by continuous sampling (Dunbar 1976; Leger 1977; Tyler 1979).

For each species pair at each site (e.g. Magpies–Ravens, Magpies–Common Mynas) we first calculated how many of the time-stops (E) we would expect to see both species present in the foraging zone if the presence of one species was independent of the presence of the other species. We used the following formula:

$$E = (n_x \times n_y)/n_{xy}$$

where n_x and n_y are the number of time-stops where species x and species y were present in the foraging zone and n_{xy} is the number of time-stops where both species were present in the outer zone. Inherent in this formula is the assumption that if both species are present in the outer zone, they are equally likely to be present in the foraging zone.

We then calculated an association score (AS) for each species using the formula:

$$AS = (O - E)/(O + E)$$

where O is the number of time-stops at which both species were present in the foraging zone. This created a score that was negative if the species were associating less often than predicted by chance and positive if the species were associating more often than predicted by chance.

We pooled each combination of species pairs by species. Owing to the high proportion of incidences of species pairs never being observed to associate at a given site (resulting in a large proportion of AS scores being equal to −1) the data were not suitable for parametric analysis, so we ran a Kruskal–Wallis test by species. Further Kruskal–Wallis pairwise comparisons between different species were conducted *post hoc*.

Displacement

We also recorded displacement events, which involved one or more individual birds leaving the feeding station immediately after the arrival of another bird. This measurement has also been made by Kalinoski (1975) under the name 'avoidance behaviour', and Pryke and Andersson (2003) under the name 'passive supplanting'.

We calculated the number of displacement events we would expect each species to be responsible for based solely on the relative abundance of each species (data were pooled across all sites as the total number of displacement events observed was relatively small, $n = 36$). Expected displacement scores were calculated as:

$$D_{Exp} = (n/N) \times D$$

where n is the total number of individuals of the species in question observed across all sites; N is the total number of individuals of *all* species observed across all sites and D is the total number of displacement events observed. A Chi-square goodness-of-fit analysis was then used to compare the expected number of displacement events with the observed number for each species.

Results

Aggression

A significant effect of species was found for interspecific ($F_{(7,180)} = 4.382$, $P < 0.001$), intraspecific ($F_{(7,180)} = 3.240$, $P = 0.003$), and recipient ($F_{(1,180)} = 2.292$, $P = 0.029$) aggression scores, suggesting that some species displayed significantly more aggression than others. With respect to interspecific aggression, according to Fisher's LSD, Magpies showed significantly greater levels of aggression than most other species (all $P < 0.025$), including Common Mynas ($P = 0.002$), but not including Silver Gulls and Noisy Miners (all $P > 0.232$). When Tukey's HSD tests were applied, Magpies were still more aggressive than Ravens, Crested Pigeons, feral Rock Doves and Common Mynas (all $P < 0.048$), but no longer more aggressive than any other species (all $P > 0.324$). There were no other significant differences between species (by either *post hoc* test) and Common Mynas ranked fourth overall on aggression, behind Magpies, Noisy Miners and Silver Gulls (Fig. 1).

With respect to intraspecific aggression, according to Fisher's LSD, Silver Gulls displayed the highest level of aggression, significantly higher than all other species (all $P < 0.044$), including Common Mynas ($P = 0.008$). When Tukey's HSD tests were applied, Silver Gulls were still more aggressive than Magpies, Crested Pigeons and feral Rock Doves (all $P < 0.046$), but no longer more aggressive than Ravens and Noisy Miners (both $P > 0.214$), or Common Mynas ($P = 0.141$). Common Mynas ranked fourth, behind Silver Gulls, Ravens and Noisy Miners (Fig. 2). Thus, Common Mynas showed intermediate amounts of both intraspecific and interspecific aggression, and did not stand out as the most or least aggressive species in either measurement.

With respect to recipient values, feral Rock Doves and Spotted Turtle Doves received the highest levels of aggression respectively. According to Fisher's LSD test both ranked significantly higher (both $P < 0.041$) than Common Mynas, although this effect was lost when Tukey's HSD test was applied. Common Mynas ranked second to last, indicating that they were not one of the highest recipients of interspecific aggression.

For those data collected in the non-breeding season, there was a trend towards a negative correlation between the mean recipient aggression scores and interspecific aggression scores of each species, with a medium to strong effect size ($r = -0.6535$, $n = 8$, $P = 0.079$). The relationship was in the same direction for the breeding season data but did not reach significance ($r = -0.3472$, $n = 8$, $P = 0.399$). These relationships were similar irrespective of whether Common Mynas were included in the analysis or not (with Common Mynas removed; non-breeding: $r = -0.709$, $n = 7$, $P = 0.074$; breeding: $r = -0.370$, $n = 7$, $P = 0.414$; see Fig. 3).

Association patterns

All of the association scores calculated for the species pairs were negative, indicating that, for all species pairs, each species was observed in the foraging zone in the presence of the other species less often than predicted by the number of times both species were present in the outer zone if they were approaching the foraging zone independently. Silver Gulls had the highest association scores, Magpie-larks the lowest, and Common Mynas had the third highest association score. This indicates that Common Mynas were more likely than most of the other species to be found in association with heterospecifics. The Kruskal–Wallis tests revealed a significant effect of species ($P < 0.001$), and the *post hoc* pairwise Kruskal–Wallis comparisons showed that Common Mynas had significantly higher *AS* scores than Spotted

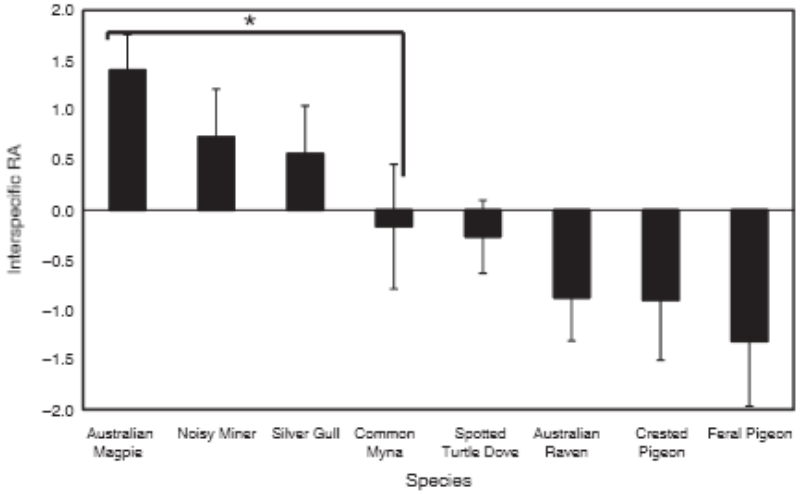

Fig. 1. Transformed values of interspecific attacks initiated by foraging species. Lines indicate two significantly different values. Only significant differences involving Common Mynas are shown.

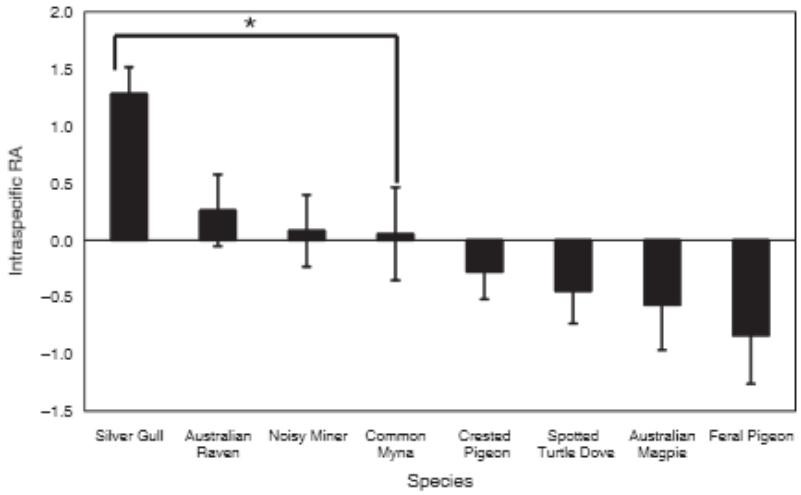

Fig. 2. Transformed values of intraspecific attacks initiated by foraging species. Lines indicate two significantly different values. Only significant differences involving Common Mynas are shown.

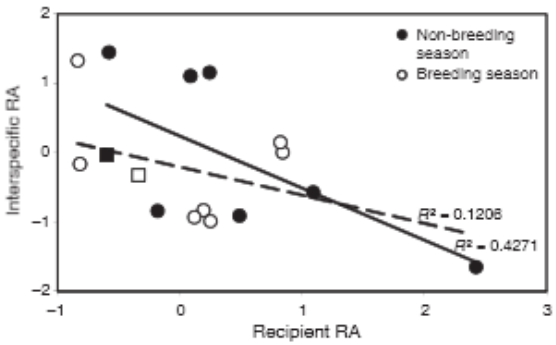

Fig. 3. Negative correlation between interspecific (initiator) and recipient values for breeding and non-breeding seasons. There is a trend indicating that birds receiving fewer attacks are likely to be initiating more attacks, and vice versa. Squares indicate Common Myna breeding season and non-breeding season values.

Turtle Doves, Magpie-Larks and Noisy Miners (all $P < 0.002$), and marginally higher than Common Starlings ($P = 0.056$). They were not significantly less likely to associate with heterospecifics than any other species (all $P > 0.220$; Fig. 4).

Displacement

Only five species exhibited displacement behaviour, and this behaviour was uncommon. Of 36 displacement events, Magpies were responsible for 58% (21 of 36), and Common Mynas were responsible for only four. Other species displaced eight times (Ravens), twice (Noisy Miners) and once (Silver Gulls). The Chi-square goodness of fit showed that the observed proportion of displacement events by species was significantly different from that predicted by chance ($\chi^2_{(4)} = 52.461$, $P < 0.001$, Yates Chi-square correction applied owing to some cells having expected value <5). The biggest contributor to this Chi-square value was the Magpie, being responsible for approximately four times as many displacement events as expected.

Discussion

This study examined the importance of behavioural aggression around food resources to the invasive success of Common Mynas, and the potential effect of this on native species in suburban habitats. Contrary to anecdotal suggestions and general public opinion, findings suggest that in situations of high food concentration Common Mynas do not display disproportionately high levels of aggression around food and do not monopolise food sources by displacing or preventing other birds from accessing them, when compared with a variety of native and exotic species with which the Common Myna co-exists. It is, therefore, unlikely that aggressively outcompeting other birds for food resources is the key behavioural trait responsible for the invasive success of Common Mynas.

Our findings complement those of Lowe *et al.* (2011), who reported that Common Mynas rarely interfered with other foraging species, and did not initiate aggressive interactions more than other species. It is important to note that their study was conducted in wild foraging situations, with no artificial manipulation of food resources, thus demonstrating that Common Mynas do not show high levels of aggression even during food scarcity, as would be likely in a natural foraging situation. Our findings show that the other extreme foraging situation, that of providing clumped, artificial food resources where species are foraging in very close proximity to one another, also fails to elicit disproportionately high levels of interspecific aggression from Common Mynas.

Our study also found that, across species, there was a trend towards higher frequencies of initiated aggression being associated with lower frequencies of being the recipient of an aggressive act, a relationship that has been demonstrated in other studies

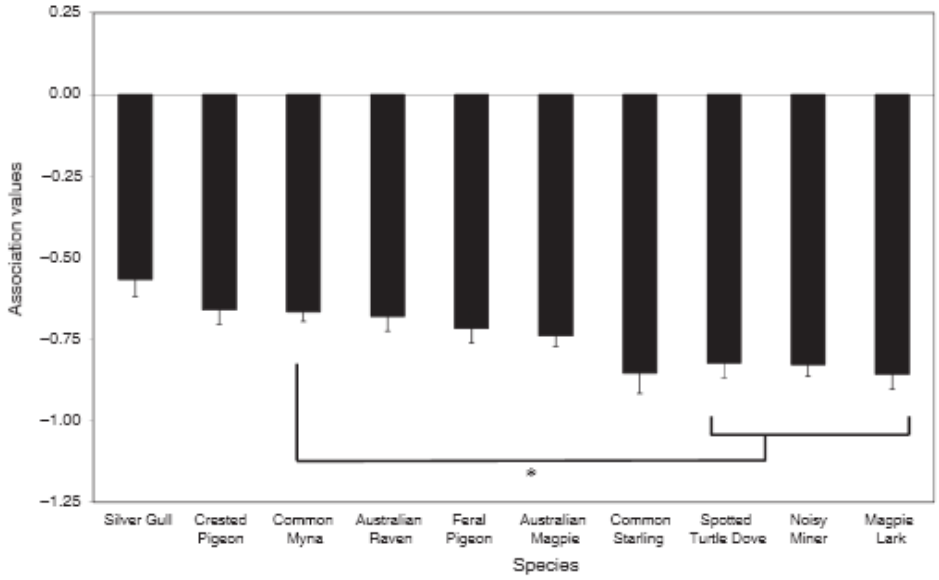

Fig. 4. Association values of foraging species. Negative association values indicate species are negatively correlated with the presence of other species, and greater negative values indicate decreased likelihood of association with other species. Lines indicate significant differences. Only significant differences involving Common Mynas are shown.

(e.g. Drickamer *et al.* 1999; D'Eath 2002; Løvendahl *et al.* 2005). Importantly, the behaviour of Common Mynas did not obviously deviate from this trend. Theories of high aggression would predict Common Mynas to fall well above the trendline, indicating a tendency to be involved in disproportionately high levels of interspecific aggression relative to other species. In fact, Common Mynas fell just below the trendline (in both the breeding season and non-breeding season), providing no evidence that they either initiate or elicit aggression more than the other species observed. Common Mynas were only the fifth or fourth most aggressive initiator of the eight species surveyed (depending on whether it was their breeding or non-breeding season), and ranked sixth or eighth of the eight species in likelihood to be the recipient of an aggressive act, in the breeding and non-breeding seasons, respectively. So although Common Mynas may benefit from not being the target of heterospecific aggression, they themselves only exhibit aggression at a rate that is about median when compared with other species with which they occur.

Association scores showed that all species were less likely to be present at the feeder if heterospecific individuals were present. This finding suggests that the birds were actively avoiding heterospecifics, which was somewhat unexpected as it was presumed that birds might use the presence of feeding heterospecifics as cues to the location and profitability of feeding resources. This may still be the case, of course, but with the low association scores being caused by either displacement of one species by another, or passive avoidance of the feeder when heterospecifics are there. Analysis of displacement events revealed that Common Mynas were only responsible for a small proportion of the displacement events, fewer than expected by chance (based on the relative abundance of the various species). Therefore it is unlikely that Common Mynas are actively monopolising food resources through displacement of other birds. Even to the extent that passive avoidance contributed to the latency scores, there was no evidence that Common Mynas were being avoided more than any other species.

Although scientific findings, including ours, have not revealed Common Mynas to be excessively aggressive, anecdotal reports of aggression persist. Possibly, such attacks have occurred mostly while in defence of nesting sites or young. Excessive nestling defence behaviour is common in many species, both native and introduced (Montgomerie and Weatherhead 1988). Feare and Craig (1999) reported that most Common Myna aggression they witnessed occurred around nesting sites, and only some around food. Other reports of antagonistic behaviour concern aggressive displays from a nesting pair (Booth 1963), and attacks on a Cat during the height of the breeding season in December (Edgar 1975). Airey (1995) reported that Common Mynas attack only during the nesting season, and the rest of the time merely 'scold'.

It may also be that Common Mynas are no more aggressive than other species but that any aggression is more noticeable and obvious to us owing to the strong association Common Mynas have with human activity (e.g. Dean 2000; Peacock *et al.* 2007). Additionally, the status of Common Mynas as an introduced species may lead to a bias in their perceived levels of aggression. Econationalistic ideals held by many Australians decree that introduced, 'outsider' species are unnatural and must necessarily be vicious, greedy and inherently nasty (Trigger *et al.* 2008; Simpson 2010). Native species are seen as gentle and vulnerable, whereas we are quick to expect the worst behaviour of any species seen as foreign to or outside of the natural Australian ecosystem

(Franklin 2006). Thus any aggression from Common Mynas is likely to be noted or reported, whereas other native species can exhibit the same behaviour without drawing the same attention from us.

These findings may inform management efforts aiming to assist natives under threat from the expansion of Common Mynas. They suggest that managerial strategies that involve non-targeted or scattered food-provisioning are not likely to assist native species greatly. In fact, given the broad diet of the Common Myna, such activities may result in a net competitive gain for the Common Myna, especially compared with native species with specialist diets, an obviously undesirable outcome. If Common Mynas are negatively affecting native species or causing their decline as suggested (e.g. Pell and Tidemann 1997; Blanvillain *et al.* 2003), they may be doing so through competition for other resources, such as suitable nesting sites.

Conclusion

The findings of this study strongly suggest that Common Mynas do not achieve their success in urban habitats through aggressive monopolisation of food sources, actively displacing native birds or disrupting their ability to access food. Attempts to manage the effect of Common Mynas on native bird populations by provisioning extra food intended for native species may not have the desired outcome. Research efforts should now focus on other ecological factors, such as competition for nesting sites, as the potential ground on which Common Mynas gain their competitive edge over natives.

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ORIGINAL PAPER

The native versus alien dichotomy: relative impact of native noisy miners and introduced common mynas

Kathryn M. Haythorpe · Darren Burke · Danielle Sulikowski

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Abstract Human activity can dramatically affect biodiversity, often by introducing non-native species, or by increasing the abundance of a small number of native species. Management strategies aimed at conserving biodiversity need to be informed by the actual impacts of highly abundant species, whether native or introduced. In this study we examined characteristics of two bird species, introduced common mynas and native noisy miners, both of which are highly abundant in urbanised areas along the East coast of Australia. Current managerial practices have a strong focus on eradication of common mynas, while noisy miners are largely ignored. However, in this study noisy miners were found in a broader range of habitats, and in greater abundance, than common mynas; displayed more aggressive behaviour; and were linked to a decline in the diversity and abundance of other species where common mynas were not. We suggest that the adaptability of a species and the variety of habitats it can colonise may be a better predictor of its potential impact, than whether it is native or introduced.

Keywords Aggression · *Manorina melanocephala* · Species abundance · *Sturnus tristis*

K. M. Haythorpe (✉) · D. Sulikowski
School of Psychology, Charles Sturt University, Panorama Avenue, Bathurst, NSW 2795, Australia
e-mail: kathryn.haythorpe@gmail.com

D. Burke
School of Psychology, University of Newcastle, 10 Chittaway Road, Ourimbah, NSW 2258, Australia

Introduction

Urbanised environments are associated with low species diversity (e.g. Clergeau et al. 2001; Jokimäki et al. 1996; McKinney 2002; Sol et al. 2011) and tend to be dominated by large numbers of a few species that are able to take advantage of the unique conditions experienced in an urban landscape (Bezzel 1985; Kark et al. 2007). These dominant species, termed "urban exploiters" (Blair 1996), typically have generalist diets and habits (Evans et al. 2011) and high sociality (Kark et al. 2007). Urban exploiter birds are often introduced (e.g. McKinney 2002; Orchan et al. 2013), but can also be native species existing in greater abundance than historically recorded (Blair 1996; Emlen 1974; Huhtalo and Jarvinen 1977; Jokimäki et al. 1996; Kark et al. 2007). Management bodies have previously focussed heavily on control of introduced species found in such environments, often attempting to reduce numbers or eradicate the species from entire areas (Gurevitch and Padilla 2004; Orchan et al. 2013; Schlaepfer et al. 2011a; Stromberg et al. 2009). The same is clearly not true for native species, which are generally assumed to have positive effects on the environment (Brown and Sax 2004; Davis et al. 2011). Recent scientific opinion (e.g. Carroll 2011;

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Davis 2011; Davis et al. 2011; Schlaepfer et al. 2011a, b), however, suggests managerial bodies shift from the present focus on whether a species is native or not, and towards objective assessment of each species' environmental impact in its particular ecosystem. Objective assessments of relative species impacts are often obtained through investigation into several primary predictors, such as range, abundance, and per-capita or per-biomass effect on parameters such as community structure and population dynamics (Parker et al. 1999).

This paper focuses on the potential impact of two common but unrelated bird species, noisy miners *Manorina melanocephala* (Meliphagidae) and common mynas *Sturnus tristis* (Sturnidae). Noisy miners are native, communally-breeding honeyeaters found throughout the southern and eastern parts of Australia (Simpson and Day 2004). Common mynas, which belong to the Sturnid family, were introduced to the eastern coast of Australia in large numbers between 1860 and 1972 (Long 1981), and have become common in cities in this range. Common mynas are obligate hollow-nesting birds, and thrive in regions cleared of natural vegetation. They are heavily associated with human activity, taking advantage of food provided in the form of scraps and rubbish (Crisp and Lill 2006), using gutters and rooves of buildings for nesting (Bomford and Sinclair 2002; Counsilman 1974), and even furbishing their nests with human litter (Counsilman 1974; Sengupta 1968, 1982). Similarly, noisy miners, as edge specialists, have benefited greatly from the clearing of vegetation and segregation of remnant patches of woodland within most of their range in Eastern Australia (Catterall et al. 1991; Clarke and Schedvin 1997a; Low 1994; Loyn 1987), as such habitat appears to be optimal for territorial defence (Taylor et al. 2008); however they require natural vegetation for building cup nests (Dow 1978).

Although common mynas are introduced and noisy miners are native, both species have been accused of creating environmental problems and anthropogenic disturbances. Noisy miners are particularly known for their antagonistic, mobbing behaviour and extreme territorial aggression (Dow 1977, 1979; Grey et al. 1997). Presumably due to these behaviours, noisy miner numbers negatively correlate with the diversity of bird species, particularly small insectivorous birds in suburban gardens (et al. Grey 1997, 1998; Parsons et al. 2006) and woodlands (Major et al. 2001), and their aggressive exclusion of other birds has recently been nominated for inclusion in the EPBC Act list of Key Threatening Processes (DSEWPC 2011). They have even been referred to as a 'reverse keystone' species, as it appears that low abundance or total absence of the species is necessary for high diversity to be sustainable in some areas (Piper and Catterall 2003). They have recently been found to be highly aggressive in competition over artificially provided food sources (Haythorpe et al. 2012; Sol et al. 2011). In addition, they have increased dramatically in abundance over the last few decades (Barrett et al. 2002; Barrett et al. 2007), creating ever greater potential for negative impact on native wildlife.

Common mynas are also thought to pose a threat to native wildlife, and are typically disliked by members of the public (e.g. Nee et al. 1990; Tidemann 2003). They are now one of only three avian species to be listed by the IUCN as among the 'World's 100 worst invasive species' (Lowe et al. 2000), and a poll by the Australian Broadcasting Company found that Australians considered them to be the "most significant pest/problem" (http://www.abc.net.au/tv/wildwatch/results/award.htm). In their native range in India they are known to compete with other hole-nesting species such as ring doves *Streptopelia decaocto* (Dhanda and Dhindsa 1993) and rose-ringed parakeets *Psittacula krameri* (Dhanda and Dhindsa 1996), as well as in parts of their introduced range such as Israel (Orchan et al. 2013). On some islands common mynas are thought to be contributing to decreases in abundance of threatened species, such as magpie robins *Copsychus* spp. (Huong and Sodhi 1997; Watson et al. 1992) found on the Seychelles Islands and echo parakeets *Psittacula eques* (Jones 1996) found on Indian Ocean islands near Madagascar. Artificial removal of common mynas on a New Zealand island was associated with an increase in some native bird species, suggesting they may have played a role in their historical decline (Tindall et al. 2007). However, it does not follow that common mynas would pose the same threat in an ecosystem less fragile than that of a smaller island, and until recently, actual quantitative evidence on their ecological impact, particularly in Australia, has been inconclusive. A long-term study on the impact of common mynas on native species at the population level (Grarock et al. 2012) has recently provided strong evidence to suggest that abundance of

some cavity-nesting and small bird species may be threatened by common myna establishment. However, this study also found that the numbers of some species previously thought to be threatened by common mynas, such as Eastern rosellas and common starlings (Pell and Tidemann 1997b), may not be. Given the species thought to be at risk from common mynas are still highly abundant (and three are actually introduced and potential pests themselves), the seriousness of this risk is still to be determined. Others have speculated that the overlap of their range with that of some currently abundant species may cause these species to become threatened in the future (Tracey and Saunders 2003). For instance, Pell and Tidemann (1997b) document a possible threat to native crimson rosellas *Platycercus elegans* and Eastern rosellas *P. eximius* through competition for suitable nesting hollows. It should be recognised, however, that while common mynas occupy principally urban environments, Eastern rosellas occupy both urban and woodland habitats, and crimson rosellas in fact prefer to colonise dense forests, including rainforest. Thus far there is no indication that the numbers of crimson or Eastern rosellas is declining in response to common mynas, and in fact both rosella species appear to be thriving (Tidemann 2010; Veerman 2002).

Although both noisy miners and common mynas are considered pests by the general public, considerable attention is given to controlling common mynas in Australia, which are the target of ongoing eradication efforts. Australian governments and local councils have invested funding into developing specialist traps designed to exterminate large numbers of common mynas. Organisations such as the Hawkesbury Indian Myna Action Group (HIMAG),[1] the Canberra Indian Myna Action Group (CIMAG) Inc,[2] the Indian Myna Bird Project (Mid North Coast),[3] and the Yarra Indian Myna Action Group (YIMAG) Inc,[4] to name just a few, provide information on trapping and humane euthanasia of common mynas and distribute traps to the public. Noisy miners, on the other hand, are a protected species in each state in which they occur in Australia—for example, in New South Wales under the National Parks and Wildlife Act 1974 (Schedule 11), it is illegal to kill, trap or harm them (Section 98). Despite these highly different management strategies, the actual impact of each of these birds is not clear.

There is evidence that both noisy miners and common mynas may potentially be impacting negatively on biodiversity. It is difficult, however, to make judgements about the relative severity of each species' impacts when such judgements need to be based on comparisons between studies, which may employ different methodology and measures and be conducted at different sites. With this consideration in mind, the present study directly compares the three primary indicators of impact—range, abundance, and per-capita effect, in this case focussing on effects on population (abundance) and community (diversity; Parker et al. 1999)—between native noisy miners and introduced common mynas across suburb, edge and bush habitats, and across breeding and non-breeding seasons. We predict that the greatest threat to biodiversity would come from a species that (1) was present in a variety of habitat types; (2) was present in great numbers or was the most abundant species in the area; (3) was linked to a decrease in diversity and/or abundance of other species; and (4) would be frequently observed engaging in aggressive behaviours.

Methods

Transect locations and descriptions

Forty-five transects were mapped within and around the city of Newcastle, Australia, located approximately 160 km north of Sydney. There were 15 'suburban' transects, situated at least 50 m from the nearest bushland; 15 'bush' transects that were at least 50 m into bushland; and 15 'edge' transects placed in regions where bushland of at least 260 hectares met with suburbs (Fig. 1). Bushland was found in urban nature reserves, including Awabakal Nature Reserve, Blackbutt Reserve, Glenrock State Conservation Area, Richley Recreation Reserve, Sygna Close Reserve, and Tingira Heights Nature Reserve, and consisted of dense native remnant vegetation, including tall trees and undergrowth. Edge transects ran along the border of these two habitats, extending into the suburbs on one side and into bushland on the other. Selection of

[1] http://www.hawkesbury.nsw.gov.au/environmental-services/natural-environment/indian-myna-control-program-himag.

[2] www.indianmynaaction.org.au.

[3] www.indianmyna.org.

[4] www.yimag.org.au.

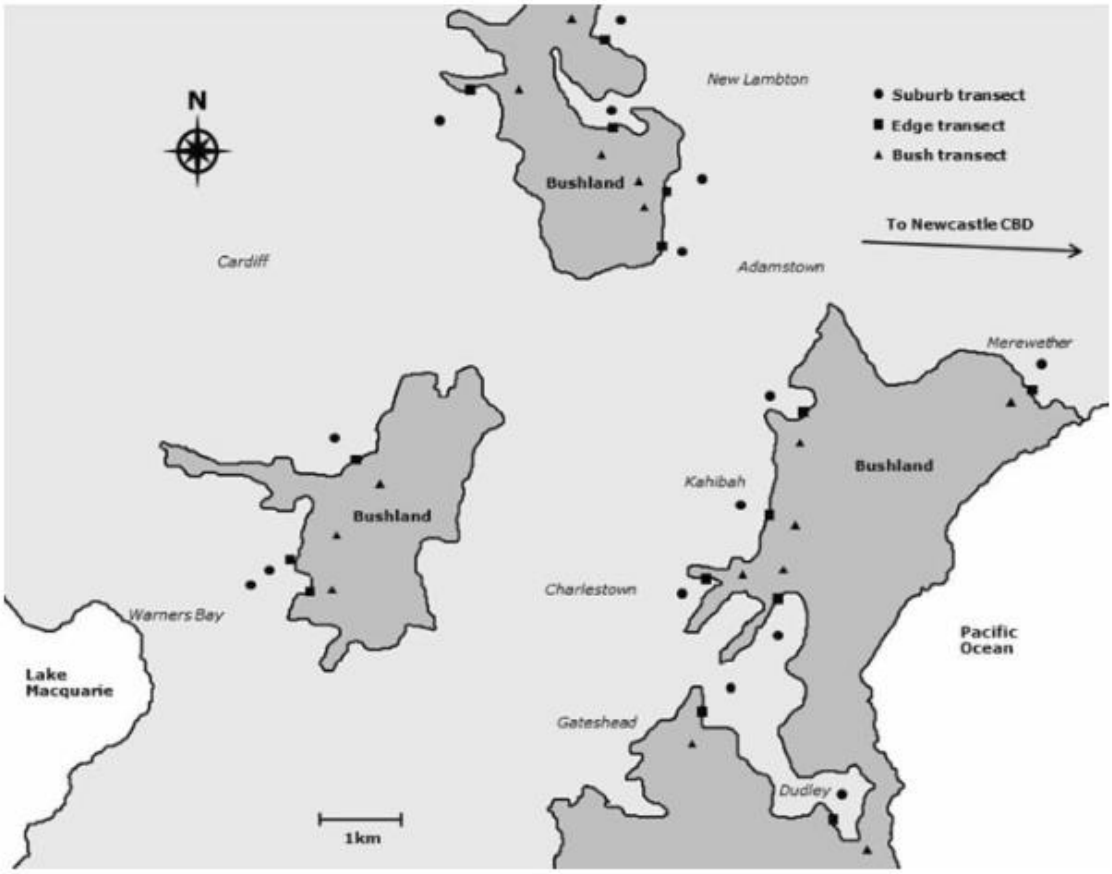

Fig. 1 Map of transect locations across Newcastle, New South Wales. *Circles* indicate suburban transects, *squares* indicate edge transects, and *triangles* indicate bush transects. *Italicised* names indicate common suburban locations across the city

transect locations was random in suburban sites, but limited in edge sites to those that were accessible (e.g. not part of a residential property), and in bush sites limited to reserves that allowed public access.

Each transect was 200 m long and contained four observation stops, at 25, 75, 125, and 175 m along the transect, respectively. The observation area for each stop was a circle, centred on the transect, with a radius of 25 m.

Observation procedure

Surveys of all 45 transects were conducted every 2 months for a year, resulting in three surveys (in April, June and August, respectively) being conducted during the non-breeding season and three surveys (in October, December and February, respectively) during the breeding season of most birds. These are referred to throughout this paper as the observations.

Each survey of a transect totalled 20 min of observation, with a sampling time of 5 min at each of the four stops. Sampling time was based on previous work by Blair (1996, 2004), and initial pilot studies in the study areas determined that practically all observations (>95 %) were recorded within the first 2 min of sampling, a trend that continued throughout the rest of data collection; thus 5 min was considered to be sufficiently long to pick up the majority of relevant behavioural observations. From the centre of the stop, the observer recorded the numbers and species of all birds seen and heard within the observation area, excluding those that were only flying overhead and not spending any other time at the stop. For each survey,

the observer also recorded whether or not each bird species initiated an aggressive act on another bird. An aggressive act was defined as a swoop, peck at, physical fight with, or chase of another bird, regardless of whether this resulted in the other bird leaving the area. To avoid bias in number of aggressive interactions due to increased number of any one species, multiple individuals engaging in an aggressive encounter were only counted once, and noisy miners and common mynas were recorded in a yes/no fashion as either engaging in aggressive behaviour or not engaging in aggressive behaviour for each stop, thus multiple attacks in one visit were given the same weight as only one attack.

Overall abundance and species diversity

Species diversity was defined as the total number of species observed and overall abundance was defined as the total number of individuals observed. Both of these measures were calculated separately for each transect (by combining observations over the four stops), during each of the six two-monthly surveys. The scores from the three breeding-season and three non-breeding-season surveys, respectively, were then averaged so that each transect was left with one species diversity score and one overall abundance score for each season.

Transect diversity was defined as the total number of species observed at that transect, tallied over the six surveys of the study. Note that transect diversity is not simply a sum of the six species diversity scores recorded during the six surveys, as a single species observed during four of the surveys, for example, counted as just one species for the purposes of the transect diversity measure, not as four species. Transect abundance was defined as the total number of individuals present at that transect, tallied over all six surveys. This measure is simply a sum of the six overall abundance scores recorded for each transect.

To explore the effect of habitat type on overall abundance and species diversity, we conducted repeated-measures ANOVAs, using each of the 45 transects as the basic replicates of the analyses. Habitat type was entered as a between-transect factor (3 levels: suburb, edge and bush), and season as the within–transect factor (2 levels: breeding and non-breeding).

Core species

For analyses involving all subsequent dependent variables we included only species that were present in at least 5 of the 15 transects for each habitat type. This elimination process resulted in 15 core species for the suburb (including both noisy miners and common mynas), 19 core species for the edge (which included noisy miners but not common mynas), and 20 core species for the bush (also including noisy miners, and not common mynas).

Absolute abundance and relative abundance scores

Absolute abundance (the total number of sightings of a given species) was recorded at every transect for each core species of each habitat type. Absolute abundance scores for each core species during the breeding and non-breeding seasons were calculated for each transect by summing absolute abundance totals observed at that transect during the October, December and February surveys and the April June and August surveys, respectively. An absolute abundance score that combined data from the breeding and non-breeding seasons was also calculated for noisy miners and common mynas only.

Relative abundance scores for each core species at each transect (for the breeding and non-breeding seasons separately, as well as both seasons combined for noisy miners and common mynas as described above) were also calculated using the formula:

$$RA_x = AA_x - AA_{\sum}/n$$

where RA_x is the relative abundance of species x, AA_x is the absolute abundance of species x, $AA_{\sum}$ is the sum of all core species' absolute abundance scores at that transect, and n is the number of core species observed at that transect. The resulting score was negative if the number of a species was lower than the mean for that transect and positive if the number of a species was higher than the mean for that transect. Species with an absolute abundance score of zero for a given transect (indicating the species was never observed at that transect) were still allocated a relative abundance score for that transect as per the above formula.

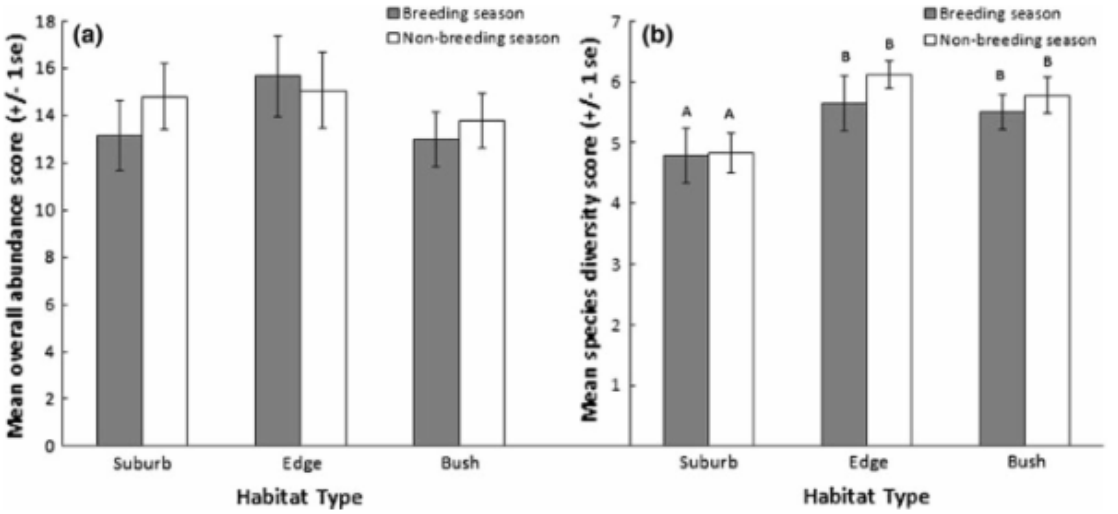

Fig. 2 The overall abundance and species diversity across the different habitat types. Overall abundance was not significantly affected by either habitat or season (**a**), while species diversity was significantly lower in the suburb than in either the edge or the bush, although this effect was not seasonal (**b**). Different uppercase letters (*A*, *B*) indicate significant differences between variables

Results

Effects of habitat type

Overall abundance did not differ significantly between the habitat types ($F_{(2,42)} = 0.576$, $p = 0.567$), or between the breeding and non-breeding seasons ($F_{(1,42)} = 1.069$, $p = 0.307$), nor was it affected by an interaction between these variables ($F_{(2,42)} = 1.229$, $p = 0.303$; Fig. 2a). Species diversity was significantly different across habitat type ($F_{(2,42)} = 3.688$, $p = 0.033$). Post hoc contrasts showed both bush ($p = 0.04953$) and edge ($p = 0.013$) habitats had greater diversity than suburban habitats, but bush habitats did not differ from edge habitats ($p = 0.579$). There was also no main effect of season on species diversity ($F_{(1,42)} = 1.409$, $p = 0.242$) and no interaction between season and habitat type ($F_{(2,42)} = 0.312$, $p = 0.734$; Fig. 2b).

Noisy miners were the most abundant species recorded in suburb (333 sightings) and edge (374 sightings) habitats. They were also one of the most abundant species sighted in the bush (157 sightings), outnumbered only by small bush specialist wren species, white-browed scrubwrens *Sericornis frontalis* (193 sightings) and superb fairy-wrens *Malurus cyaneus* (233 sightings). Common mynas by contrast were not present in bush habitats, and were present in only three of 15 edge habitat transects, coming to a total of only six sightings. They were, however the second most abundant species (139 sightings) in suburban habitats after noisy miners.

Repeated-measures ANOVAs were used to investigate the effect of habitat type on absolute abundance and relative abundance of noisy miners. Each transect was the basic replicate of the analysis, and habitat type was entered as a between-transect factor (3 levels: suburb, edge and bush) and season (2 levels: breeding and non-breeding) as a within-transect measure.

Noisy miner absolute abundance was not affected by season ($F_{(1,42)} = 1.729$, $p = 0.196$) but differed significantly between the habitat types ($F_{(2,42)} = 3.229$, $p = 0.0496$). *Post hoc* simple contrasts showed a significant drop in numbers occurred in bush habitats compared with edge habitats ($p = 0.021$), but did not occur between suburb and bush ($p = 0.059$), or between edge and suburb ($p = 0.654$; Fig. 3a). The relative abundance of noisy miners was not affected by season ($F_{(1,42)} = 1.366$, $p = 0.249$) nor by an interaction between season and habitat type ($F_{(1,42)} = 0.590$, $p = 0.559$), but the main effect of habitat type did approach significance ($F_{(2,42)} = 3.070$, $p = 0.057$). *Post hoc* simple contrasts confirmed that noisy miner relative abundance was significantly higher in the edge

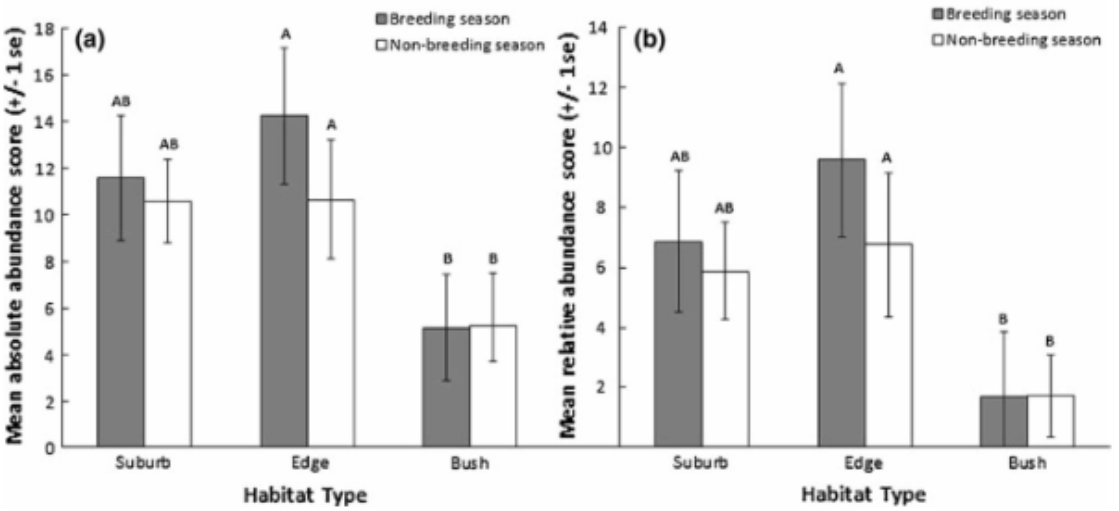

Fig. 3 The noisy miner relative and absolute abundances across habitat types. Both absolute (**a**) and relative (**b**) abundances were significantly higher in edge sites than in bush sites, and this was not affected by season. Different uppercase letters (*A*, *B*) indicate significant differences between variables

compared to the bush ($p = 0.021$) habitats and also tended to be higher in the suburb compared to the bush ($p = 0.091$), but did not differ between the suburb and the edge ($p = 0.507$; Fig. 3b).

Since 96 % of common myna sightings occurred in suburban transects, formal analyses comparing common myna abundance across the three habitat types were not conducted, however, paired-samples t-tests were used to compare absolute and relative abundance of common mynas between the breeding and non-breeding seasons. Neither relative abundance ($t(14) = 1.006$, $p = 0.331$), nor absolute abundance ($t(14) = 1.005$, $p = 0.332$) of common mynas differed between the breeding and non-breeding seasons in the suburban habitat.

Transect diversity

Given the nil effect of breeding season in the previous analyses, the species diversity scores from each of the six surveys of the study were combined to produce a transect diversity score, reflecting the total number of species observed at a given transect throughout the study period. Pearson's bivariate correlations were then used to correlate these transect diversity scores with common myna relative abundance and absolute abundance across the 15 suburban transects. The same analyses were conducted with noisy miner relative abundance and absolute abundance across all transects at each of the three habitat types.

With respect to common mynas, neither absolute abundance ($r = 0.036$, $n = 15$, $p = 0.898$) nor relative abundance ($r = 0.007$, $n = 15$, $p = 0.979$) was significantly correlated with transect diversity in suburban habitat. The same was true for noisy miner absolute abundance ($r = -0.034$, $n = 15$, $p = 0.904$) and relative abundance ($r = -0.066$, $n = 15$, $p = 0.816$) in the suburban habitat and also for absolute abundance ($r = -0.087$, $n = 15$, $p = 0.757$) and relative abundance ($r = -0.097$, $p = 0.731$) in the bush habitat. There were, however, significant negative correlations between noisy miner absolute abundance ($r = -0.636$, $n = 15$, $p = 0.011$) and relative abundance ($r = -0.680$, $n = 15$, $p = 0.005$), respectively, and transect diversity in the edge habitat, confirming that as noisy miner numbers increased, and they became a larger proportion of the bird life, the number of other species observed dropped.

Transect abundance

As described above for the species diversity scores, the overall abundance scores were combined across the six surveys resulting in each transect having a transect

Table 1 List of core species of the suburb habitat, and corresponding r-values derived from correlating these species' relative abundance scores with those of the noisy miner and common myna across the 15 suburban transects (n = 15 for all correlations)

Core species		Noisy miners (*Manorina melanocephala*)		Common mynas (*Sturnus tristis*)	
Common name	Scientific name	Breeding Season	Non-breeding season	Breeding season	Non-breeding season
Australian magpie (n = 122)	*Cracticus tibicen*	−0.614	0.026	−0.384	−0.370
Australian raven (n = 42)	*Corvus coronoides*	−0.527	−0.300	0.039	−0.142
Common myna (n = 139)	*Sturnus tristis*	0.069	−0.425	–	–
Crested pigeon (n = 81)	*Ocyphaps lophotes*	−0.636	−0.503	−0.006	−0.303
Crimson rosella (n = 14)	*Platycercus elegans*	−0.728	0.152	−0.261	−0.190
Eastern rosella (n = 35)	*Platycercus eximius*	−0.616	−0.002	−0.185	−0.419
Grey butcherbird (n = 5)	*Cracticus torquatus*	−0.784	−0.374	−0.288	0.012
Laughing kookaburra (n = 42)	*Dacelo novaeguineae*	−0.381	0.214	−0.644	−0.196
Magpie-lark (n = 22)	*Grallina cyanoleuca*	−0.850	−0.292	−0.055	−0.056
Noisy miner (n = 333)	*Manorina melanocephala*	–	–	0.069	−0.425
Pied currawong (n = 21)	*Strepera graculina*	−0.564	−0.144	−0.509	−0.197
Rainbow lorikeet (n = 80)	*Trichoglossus haematodus*	−0.352	0.407	0.008	0.211
Red wattlebird (n = 37)	*Anthochaera carunculata*	−0.249	−0.227	0.429	0.022
Sulphur-crested cockatoo (n = 37)	*Cacatua galerita*	−0.481	−0.259	0.060	−0.409
Spotted dove (n = 113)	*Streptopelia chinensis*	−0.675	−0.355	−0.097	0.689

Number in brackets after species name is the total sightings of that species in the suburban habitat

abundance score (which was the total number of individual birds observed at that transect over the entire study period). For the purposes of correlating the transect abundance scores with noisy miner and common myna abundance scores, as described below, the numbers of noisy miners and common mynas, respectively, were subtracted from the transect abundance scores, such that each correlation examined the association between abundance of a focal species (either noisy miners or common mynas) and abundance of all *other* core bird species.

Pearson's bivariate correlations were used to correlate transect abundance scores with noisy miner absolute abundance scores in suburb, edge and bush habitats, and with common myna absolute abundance scores in the suburb only. There was no correlation between noisy miner absolute abundance and transect abundance in suburb ($r = 0.174$, $n = 15$, $p = 0.534$), edge ($r = -0.275$, $n = 15$, $p = 0.322$) or bush ($r = -0.068$, $n = 15$, $p = 0.809$) and also no correlation between common myna absolute abundance and transect abundance in the suburb ($r = 0.413$, $n = 15$, $p = 0.126$). The same analyses were conducted using relative (rather than absolute) abundance scores of noisy miners and common mynas and these also failed to reveal any significant correlation for common mynas in the suburb ($r = 0.267$, $n = 15$, $p = 0.335$) or for noisy miners in the suburb ($r = 0.089$, $n = 15$, $p = 0.752$), edge ($r = -0.358$, $n = 15$, $p = 0.191$) or bush ($r = -0.169$, $n = 15$, $p = 0.547$).

Core species' relative abundances

To examine the association between the relative abundance of noisy miners and common mynas, respectively, and the relative abundance of other species with which they shared a habitat, we conducted a series of Pearson's bivariate correlations between each of the core species' relative abundance scores, and noisy miner relative abundance scores in each of the three habitat types and common myna relative abundance scores in suburban habitats (see Tables 1, 2).

The resulting sets of r-values were converted to z′-values (to achieve normality) using Fisher's formula:

$$z' = 0.5 \times (\log(1 + r)/(1 - r))$$

Table 2 List of core species of the edge and bush habitats and corresponding r-values derived from correlating these species' relative abundance scores with those of the noisy miner across the 15 transects of each habitat type (n = 15 for all correlations)

Core species		Edge habitat		Bush habitat	
Common name	Scientific name	Breeding season	Non-breeding season	Breeding season	Non-breeding season
Australian magpie (n = 107, 29)	*Cracticus tibicen*	0.046	0.000	−0.148	−0.217
Australian raven (n = 27, 39)	*Corvus coronoides*	−0.675	−0.202	−0.172	−0.449
Black-faced cuckoo-shrike (n = 18, 16)	*Coracina novaehollandiae*	−0.677	−0.104	−0.298	−0.320
Crested pigeon (n = 33, –)	*Ocyphaps lophotes*	−0.512	−0.233	–	–
Crimson rosella (n = 20, –)	*Platycercus elegans*	−0.341	−0.194	−0.314	−0.374
Eastern rosella (n = 75, 23)	*Platycercus eximius*	−0.087	0.262	−0.142	−0.059
Eastern spinebill (n = –, 14)	*Acanthorhynchus tenuirostris*	–	–	−0.339	−0.653
Eastern whipbird (n = 15, 20)	*Psophodes olivaceus*	−0.822	−0.337	−0.136	−0.103
Golden whistler (n = –, 18)	*Pachycephala pectoralis*	–	–	−0.396	−0.408
Grey butcherbird (n = 18, 14)	*Cracticus torquatus*	−0.633	−0.253	−0.138	0.082
Grey fantail (n = 20, 56)	*Rhipidura albiscapa*	−0.803	−0.757	−0.324	−0.580
Lewin's Honeyeater (n = 9, 40)	*Meliphaga lewinii*	−0.738	−0.594	−0.507	−0.555
Laughing kookaburra (n = 59, 51)	*Dacelo novaeguineae*	−0.081	−0.147	0.216	−0.057
Pied currawong (n = 32, 30)	*Strepera graculina*	−0.639	−0.055	−0.271	0.067
Rainbow lorikeet (n = 92, –)	*Trichoglossus haematodus*	0.049	0.348	–	–
Red-browed finch (n = –, 33)	*Neochmia temporalis*	–	–	−0.362	−0.399
Red wattlebird (n = 41, 9)	*Anthochaera carunculata*	−0.733	−0.383	−0.307	−0.190
Spotted dove (n = 72, 6)	*Streptopelia chinensis*	−0.484	−0.336	−0.220	−0.592
S′-crested cockatoo (n = 19, 17)	*Cacatua galerita*	−0.664	−0.370	−0.182	−0.200
Superb fairy wren (n = 118, 233)	*Malurus cyaneus*	−0.694	−0.700	−0.442	−0.111
White-browed scrub wren (n = 66, 193)	*Sericornis frontalis*	−0.705	−0.599	−0.210	−0.116

Missing values are due to not all species being a core species in both habitat types. Numbers in brackets after species name is the total sightings of that species in edge and bush habitats respectively

To determine whether the relative abundance of either noisy miners or common mynas exhibited a stronger association with the relative abundance of the other core species across the 15 suburban habitat transects (and whether the strength of the associations were affected by season), we conducted a repeated-measures ANOVA on the resulting suburban habitat z′ scores with season (2 levels: breeding and non-breeding) and species (2 levels: noisy miner and common myna) both entered as repeated-measures). The z′ value derived from the direct correlation between noisy miner and common myna relative abundance was excluded.

Noisy miner z′ scores were significantly more negative than common myna z′ scores ($F_{(1,12)} = 7.845$, $p = 0.016$) and breeding season z' scores were significantly more negative than non-breeding season z' scores ($F_{(1,12)} = 14.013$, $p = 0.003$). Both these main effects were qualified by a significant species by season interaction ($F_{(1,12)} = 17.879$, $p = 0.001$) with post hoc paired comparisons confirming that the noisy miner breeding season z' average was significantly more negative than the noisy miner non-breeding season z' average ($t(12) = 6.488$, $p < 0.001$) and the common myna breeding ($t(12) = 5.075$, $p < 0.001$) and non-breeding ($t(12) = 4.690$, $p = 0.001$) averages. No other post hoc comparisons were significant (all $p > 0.6$). One-sample t-tests also confirmed that only the noisy miner breeding season z' average was significantly less than zero ($t(12) = 8.893$, $p < 0.001$), with no difference found for noisy miner non-breeding ($t(12) = 1.714$, $p = 0.112$) common myna breeding ($t(12) = 1.856$, $p = 0.088$) or non-breeding ($t(12) = 1.032$, $p = 0.322$). Overall, the results demonstrate that, during the breeding season only, higher numbers of noisy miners were significantly associated with lower numbers of other species

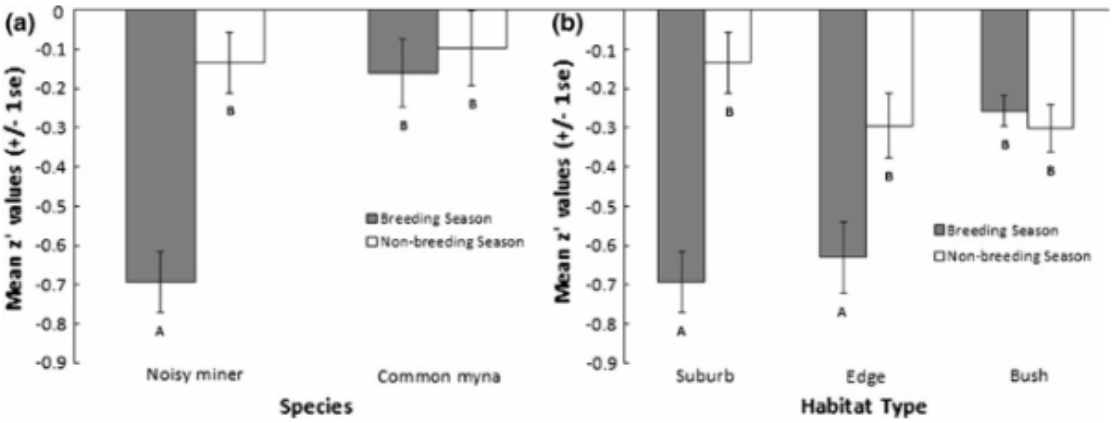

Fig. 4 The z′-values for noisy miners and common mynas in suburban habitats only (**a**), and the z′-values for noisy miners only across all habitat types (**b**). Noisy miner z′-values were significantly lower during the breeding season than during the non-breeding season, and during the breeding season were significantly lower than common myna z′-values in either season (**a**). Noisy miner z′-values were also significantly lower in both suburb and edge habitats during the breeding season than during the non-breeding season, and lower in both edge and suburban habitats during the breeding season than bush habitats in either season. Different uppercase letters (*A*, *B*) indicate significant differences between variables

of birds, with no such association demonstrated for common mynas (Fig. 4a).

To further investigate the association between noisy miner abundance and the abundance of other core species, we conducted another repeated-measures ANOVA on the z′ scores calculated for associations between noisy miners and all other core species in the suburb, edge and bush habitats. Season (2 levels: breeding and non-breeding) was entered as a repeated-measure and habitat (3 levels: suburb, edge and bush) was entered as a fixed factor. No significant main effect of habitat was observed ($F_{(2,48)}$=2.270, $p = 0.114$) but there was a significant main effect of season ($F_{(1,48)} = 36.709$, $p < 0.001$) and a significant season by habitat interaction ($F_{(2,48)} = 13.856$, $p < 0.001$). Post hoc paired comparisons confirmed that the negative association between abundance of noisy miners and other core species was significantly stronger in the breeding season for the suburb ($t(13) = 4.414$, $p = 0.001$) and edge ($t(17) = 5.341$, $p < 0.001$) habitats only, with no difference between seasons in the strength of this association found in the bush ($t(18) = 0.821$, $p = 0.422$). One-sample t-tests confirmed that the z′ average was significantly less than zero (signifying a significantly negative average association between the abundance of noisy miners and that of other core species) across all three habitats in the breeding season (suburb: $t(13) = 7.067$, $p < 0.001$; edge: $t(17) = 6.906$, $p < 0.001$; bush: $t(18) = 6.722$, $p < 0.001$) and in the edge ($t(17) = 3.536$, $p = 0.003$) and bush ($t(18) = 5.042$, $p < 0.001$), though not the suburb ($t(13) = 2.056$, $p = 0.060$) in the non-breeding season (Fig. 4b).

Aggression

All sightings of common mynas and noisy miners in suburban habitats and the respective number of aggressive acts conducted by each species in this habitat were tallied. Common mynas were recorded conducting an aggressive act on two occasions out of 139 sightings (1.4 %), while noisy miners conducted aggressive acts on 53 out of 333 (15.9 %) sightings in suburban habitats. A Fisher's 2 × 2 Exact Test (with $\alpha = 0.05$) demonstrated a significant association between species and incidence of aggression ($p < 0.001$) with noisy miners significantly more likely to display aggression than common mynas within the suburban habitats.

As noisy miners occurred across all habitats, all sightings in each of the three habitat types and respective numbers of aggressive acts conducted by noisy miners in these habitats were also tallied. In addition to the 53 aggressive acts (of 333 sightings, 15.9 %) in suburban habitats, 57 aggressive acts out of 374 (15.2 %) sightings in edge habitats, and 14 aggressive acts out of 157 sightings in bush habitats

(8.9 %) were recorded. A Pearson's χ^2 test of contingencies (with $\alpha = 0.05$) was used to determine whether habitat type was associated with the number of aggressive acts conducted by noisy miners. There was no association between frequency of noisy miner aggression and habitat type ($\chi^2_{(2)} = 4.7, p = 0.097$).

Discussion

This study examines the potential threat posed by common mynas, an introduced Sturnid, and noisy miners, a native honeyeater, by looking at their range and abundance, and investigating their potential and actual per-capita effect through examination of behaviour and relationship to the diversity and abundance of other species (Parker et al. 1999), across habitat types and seasons. Although mathematically quantifying these parameters as indicators of a species' total impact is difficult (e.g. Thiele et al. 2010; Thomsen et al. 2011), a broad examination could be expected to reveal important trends. Direct comparison of these two species within the one study area allows us to make judgements about the relative severity of the impacts of each species without the confounds introduced by trying to compare across sites and across studies.

In this study overall abundance did not differ by habitat type, however species diversity was found to be higher in bush and edge habitats than in suburban habitats. These results were not affected by season. Greater levels of diversity in remnant or bushland areas compared to urbanised areas have been found by numerous other studies (e.g. see Blair 1996; Chace and Walsh 2006; McKinney 2008 for reviews).

Noisy miners were found in all three habitat types in high numbers across seasons. They were the most abundant species in both suburb and edge habitats, and third most abundant in bush habitats. These findings are not surprising, as an affinity for edge areas and human modified habitats has been shown by numerous other studies (e.g. Catterall et al. 1991; Grey et al. 1997; Hastings and Beattie 2006; Major et al. 2001) and their ability to penetrate even dense bushland is well known (Clarke and Oldland 2007; Eyre et al. 2009; Howes and Maron 2009). In parts of southern Queensland they are even regularly recorded penetrating more than 20 km into bushland (Maron 2009).

Common mynas by contrast were found primarily in suburban habitats, where they were the second most abundant species after noisy miners, and only rarely in edge habitats, where they probably occupied the more suburban side of the transect. This finding is not surprising, as common mynas are well known as urban habitat specialists that rely heavily on human activity (e.g. Counsilman 1974; Crisp and Lill 2006; Pell and Tidemann 1997a; Sengupta 1968). Contrary to predictions made by Pell and Tidemann (1997b), common mynas in this study did not appear to be capable of penetrating bushland. It should be noted that the bushland surrounding the Newcastle area in which this study was conducted differs dramatically from the sparse, open woodland found around the Canberra region where studies by Pell and Tidemann (1997b) and Grarock et al. (2012) took place, and may explain the difference in these findings.

A negative correlation between noisy miner abundance and species abundance and diversity has been shown by numerous other studies across Australia, including the coastal city of Sydney (Parsons et al. 2006), the inland wheat belt of New South Wales (Major et al. 2001), the foothills of the Great Dividing Range in Victoria (Grey et al. 1997), and a multitude of sites ranging from Victoria to Queensland (Mac Nally et al. 2012). Noisy miner abundance in this study was negatively associated with species diversity in edge habitats. During the breeding season in both the suburb and edge habitats, an increase in noisy miner relative abundance was associated with a decrease in abundance of other species. This effect was not observed during the non-breeding season. Noisy miners are well known for aggressive nest defence behaviour via group mobbing (Arnold 2000; Maron 2009), and it may be that the prominence of this effect during the breeding season is due to differences in territorial behaviour in the presence of nestlings or fledglings. As noisy miners were less abundant in bush habitats than edge habitats, and associated with decreased species diversity in edge habitats, it may be that noisy miners are primarily using edge habitat for breeding, and aggressively excluding other species most prominently in this location. We did not observe a difference in aggression levels across habitats, however, which is inconsistent with this interpretation. We examined aggression only at a fairly coarse level, though, and more detailed recording of the specifics of

aggressive acts, such as their nature and efficacy across various habitats, might be informative.

We found no evidence that common myna abundance was associated with diversity or abundance of other species in the suburban habitats where they were found, but no comparable analysis could be conducted in bush or edge habitats as presence of common mynas in these was extremely low. In a study of suburban gardens in Sydney, Parsons et al. (2006) also found common mynas were not associated with a decrease in species diversity. However, this finding is contradicted by more recent findings from Canberra (Grarock et al. 2012). Several possibilities exist here to explain this difference in results. First, it may be that the habitats of Newcastle and Sydney are less conducive to effective competition or exclusion by common mynas than that of Canberra due to different environmental factors. For example, Newcastle and Sydney are both coastal cities that receive high levels of rainfall, which supports substantially more vegetative growth than the inland city of Canberra. It may be that this increase in vegetation gives native birds in Newcastle and Sydney a competitive edge that birds in Canberra do not have. Canberra also frequently experiences lower temperatures during winter than either of the coastal cities, and common mynas may be better able to withstand this than some native birds, again providing a competitive edge. Aside from potential differences in environmental factors, the different methodologies of the studies need to be considered. Grarock et al.'s (2012) findings consider an extensive dataset, spanning 29 years and including over 74,000 surveys, while our study and that of Parsons et al. (2006) are much smaller and span 1 year or less. A finding of impact by common mynas over a lengthy period of time and using a large amount of data, but not during shorter-term studies, indicate that the impact of this species may be gradual and subtle. On the other hand, the impact shown by noisy miners is readily picked up in studies spanning even a few months, suggesting it may be far more acute. These impacts are likely to become even more apparent as the range and abundance of noisy miners continues to increase (Barrett et al. 2007).

The high levels of aggressive behaviour in noisy miners found by this study have been documented elsewhere (e.g. Clarke 1984; Dow 1977, 1979; Loyn 1987). Other closely related miner species, such as bell miners *Manorina melanophrys* are known to exhibit similar behaviour, and are even capable of completely altering habitat characteristics through aggressive exclusion of all other insectivorous birds, resulting in an increase in abundance of sap-sucking plant parasites such as psyllid insects (Homoptera: Psyllidae) and a corresponding decrease in tree health (Clarke 1984; Clarke and Schedvin 1997b; Dare et al. 2007). Common mynas on the other hand have a long, but mostly anecdotal history of displaying aggression, which was not reflected in the results of this study. Recent studies on common myna feeding behaviour suggests that reports of aggression may have been exaggerated (Haythorpe et al. 2012; Lowe et al. 2011), and it has also been suggested (Haythorpe et al. 2012) that their reputation may be in part due to similarities in the names and appearances of common mynas and noisy miners, leading to identification issues surrounding observations of aggressive interactions by the general public. Previous anecdotal reports of common myna aggression primarily relate to nesting behaviour, including defence of nestlings and competition for nesting resources. While this specific behaviour is clearly of importance in the consideration of impact by common mynas, any bias in this study towards aggressive behaviour due to the proximity of nest sites in relationship to transect locations should actually favour more aggressive behaviour in common mynas than noisy miners, as common mynas are prone to nesting in modified habitats (Counsilman 1974) while noisy miners prefer to nest in natural vegetation, and aggressive behaviour was compared across suburban locations only. However, even with this potential bias noisy miners were significantly more aggressive than common mynas.

To determine the impact of any species we must consider in which range or areas they could pose a threat, which species they have the potential to affect, the possible strength or magnitude of this effect, and the method by which it may come about. The inability of common mynas to penetrate dense bushland calls into question their predicted threat to native wildlife and diversity, at least in the Newcastle area. Although the importance of urban regions to biodiversity is clearly higher than historically thought (Goddard et al. 2010), any potential threat common mynas pose appears to be to the species in this region only—typically either other suburban habitat specialists, or habitat generalists.

Noisy miners, in contrast to common mynas, were found to be highly abundant in all habitat types.

Threatening behaviour from noisy miners is thus capable of affecting both habitat specialists and habitat generalists. In addition, it is clear that noisy miners do not just exist in these habitats, but thrive in them, as they were the most abundant species recorded in this study. A typical characteristic of species considered to be 'invasive'—in this case, capable of having a widespread impact on a habitat—is the tendency to be present in comparatively high numbers (Colautti and MacIsaac 2004; Kolar and Lodge 2001; Richardson et al. 2000). In such cases, even relatively benign behaviours or processes may pose serious threats when amplified due to a species' overabundance. Although common mynas may appear to be highly abundant, they were found in substantially lower numbers than noisy miners. Their localised nature has perhaps led to overestimation of numbers, while high abundance of noisy miners in less frequented bushland areas goes relatively unobserved.

There is substantial evidence to suggest that noisy miner overabundance is correlated with a decrease in the diversity and abundance of other bird species (e.g. Grey et al. 1997, 1998; Mac Nally et al. 2012; Major et al. 2001; Maron et al. 2011; Parsons et al. 2006). In addition the findings of this study suggest that this effect fluctuates under various conditions, such as habitat type and breeding status. While it is possible that this effect could be coincidental, as bird species are affected by the same habitat modifications that favour noisy miners (rather than directly being affected by the noisy miners themselves), a substantial number of experimental studies have addressed this question, and generally conclude that a causal relationship does exist (Debus 2008; Grey et al. 1997, 1998; Kath et al. 2009; Mac Nally et al. 2012; Maron et al. 2011; Piper and Catterall 2003). The most likely method for noisy miners to negatively affect the abundance of small bird species is through aggressive exclusion (Dow 1977, 1979). Aggressive behaviour is likely to cause most impact in locations and during times when breeding is occurring, when noisy miners are aggressively defending nestlings through group mobbing behaviour (Arnold 2000). In support of this, we found noisy miners had an impact on the diversity and abundance of other species in primarily the edge habitat type during the breeding season, although aggressive behaviour was consistent across habitat types, suggesting that actual instances of aggression may not be the only part of a strategy designed to exclude other species.

Common mynas, by contrast, were not correlated with a decrease in diversity or abundance of other species in this study, and other studies (Parsons et al. 2006) have also been unable to find such a correlation. In addition, common mynas in this study were only infrequently observed engaging in aggressive behaviours, and a recent study (Lowe et al. 2011) has similarly found that common mynas rarely initiated interspecific aggressive attacks, and did not interfere with other foraging birds, a finding also supported by our previous research in this area (Haythorpe et al. 2012).

Conclusions

This study provides data on the range, abundance, and behaviours of noisy miners and common mynas and their associations with the abundance and diversity of other bird species in the Newcastle region. Noisy miners in this study were present in all three habitat types, while common mynas were only present in one habitat type (suburbs). While it remains possible that common mynas are negatively affecting some rare species, their restricted distribution suggests that this is unlikely; by contrast a range of rare species may be impacted by noisy miners. Noisy miners were highly abundant in all three habitat types, being the most abundant in suburb and edge habitats, and third most abundant in bush habitats. Common mynas were second most abundant in suburban habitats, the only habitat in which they were found. Noisy miners were associated with a decrease in diversity in edge habitats, and a decrease in abundance in edge and suburban habitats during the breeding season only, while common mynas were not associated with a decrease in diversity or abundance at any time. This effect may have been driven by aggressive exclusion—noisy miners were observed initiating aggressive attacks significantly more often than common mynas, and did so consistently across habitat types. We conclude that noisy miners may have the potential to have a greater impact on wildlife than common mynas in this area.

Perhaps because habitat alteration is so often accompanied by the spread of introduced species (Hobbs and Huenneke 1992; Vitousek et al. 1997), there is a tendency to focus attention on these species when assessing likely impacts on native assemblages.

In reality, in today's increasingly urbanised environment, virtually all species are to some extent introduced, as few exist now in the same environments in which they evolved competitively. For some species this has provided an advantage, allowing them to increase in number dramatically and leading to detrimental impacts on the environment. These are of even greater concern if that species is also capable of subsequently migrating into areas previously unaffected, as this will impact not just the urban environment, which is already highly degraded by other anthropogenic factors, but more diversity-rich remnant bushland areas as well, which are more likely to contain rare or threatened species. Species usually considered to be 'introduced' frequently cannot survive outside urban centres, and thus their ability to impact the regions of greatest conservation value is limited.

As the current study suggests, greater risks might be posed by native species with artificially increased numbers than by introduced species with ranges restricted to highly urbanised environments. It may be that in environments that have been dramatically altered by human activity the species with the greatest capacity to adapt to these alterations and colonise a range of different habitats are the most likely to negatively impact on other species. As in the current system, these species need not be introduced.

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Appendix B: Non-PhD publications

This appendix contains peer-reviewed articles authored or co-authored by the candidate during the period of candidature that did *not* contribute towards the doctorate degree.

3. Haythorpe, K. M. and McDonald, P. (2010). Non-lethal foraging by bell miners on a herbivorous insect: potential implications for forest health. *Austral Ecology,* 35**,** 444-450.

4. Griffin, A. S. and Haythorpe, K. M. (2011). Learning from watching alarmed demonstrators: does the cause of alarm matter? *Animal Behaviour,* 81**,** 1163-1169.

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Non-lethal foraging by bell miners on a herbivorous insect: Potential implications for forest health

KATHRYN M. HAYTHORPE[1,2] AND PAUL G. M°DONALD[2]*

[1]*School of Environmental and Life Sciences, Newcastle University, Newcastle, New South Wales, and* [2]*Department of Brain, Behaviour and Evolution, Macquarie University, Sydney, NSW 2109, Australia (Email: paul.mcdonald@mq.edu.au)*

Abstract Tree health is often negatively linked with the localized abundance of parasitic invertebrates. One group, the sap-sucking psyllid insects (Homoptera: Psyllidae) are well known for their negative impact upon vegetation, an impact that often culminates in the defoliation and even death of hosts. In Australia, psyllid-infested forest in poor health is also frequently occupied by a native honeyeater, the bell miner (*Manorina melanophrys*; Meliphagidae), so much so that the phenomenon has been dubbed 'bell miner-associated dieback' (BMAD). Bell miners are thought to be the causative agent behind BMAD, in part because the species may selectively forage only upon the outer covering (lerp) exuded by psyllid nymphs, leaving the insect underneath to continue parasitizing hosts. As bell miners also aggressively exclude all other avian psyllid predators from occupied areas, these behavioural traits may favour increases in psyllid populations. We examined bell miner foraging behaviour to determine if non-lethal foraging upon psyllid nymphs occurred more often than in a congener, the noisy miner (*M. melanocephala*; Meliphagidae). This was indeed the case, with bell miners significantly more likely to remove only the lerp covering during feeding, leaving the insect intact underneath. This arose from bell miners using their tongue to pry off the lerp cases, whereas noisy miners used their mandibles to snap at both the lerp and insect underneath. Furthermore, psyllids left behind following a bell miner foraging event were significantly more likely to be viable and regrow a lerp covering than those exposed by noisy miners. Together, this behaviour supports the theory that non-lethal foraging behaviour of bell miners may contribute to high psyllid abundance, consistent with the mechanisms by which BMAD is thought to develop.

Key words: bell miner-associated dieback, group living, lerp, *Manorina*, psyllid.

INTRODUCTION

Insect-based herbivory can affect every aspect of a plant's performance, leading to a decrease in defences and a vulnerability to declining health and subsequent plant death, a phenomenon known as 'dieback'. Symptoms of tree dieback can be varied, but are typically characterized by the wholesale defoliation of leaves and thus canopy cover, a process that often progresses through to tree death if left unchecked (Landsberg & Wylie 1983). Forests of almost any composition of species worldwide can be affected by insect infestations and subsequent dieback, leading to devastating results (e.g. Vejpustková & Holuša 2006; Carus 2009). The impact of insect herbivory looms as a significant threat to the health of large areas of forest worldwide, as habitats are placed under ever increasing levels of anthropogenic stress (Crawley 1989). In Australia many threatened and endangered species are restricted to eucalypt-dominated woodland (Yates & Hobbs 1997). Threatened taxa in these habitats face further challenges in the near future given the grim changes forecast by most climate change models (Dale *et al.* 2001). Given this, it is important that the factors surrounding insect infestation and the ensuing dieback be determined promptly, so that attempts to redress the issue can be made in stands set aside for both conservation and harvesting purposes around the globe. Numerous insects have been implicated in diminishing tree health and dieback of eucalypts in Australia (for a review see Ohmart & Edwards 1991); however, psyllids (Homoptera: Psyllidae) appear to be among the most detrimental (Stone *et al.* 2008).

The most commonly observed psyllid genus in dieback areas is *Glycaspis*, both within Australia (Stone *et al.* 2008) and in plantations where these insects have been introduced (Brennan *et al.* 2001). These parasitic invertebrates are small (adults usually <5 mm; Moore 1961) and resemble aphids (Hemiptera: Aphidoidea) in appearance. Adults are winged and mobile, with females laying their eggs on the leaves of eucalypts that later hatch into wingless nymphs. Nymphs suck sap directly out of leaves via a specially adapted

*Corresponding author.
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mouthpiece called a stylet. For protection, they exude a solid, carbohydrate-rich substance that covers the insect in a dome-like structure, termed 'lerp' (Moore 1961). This sugary lerp is a favoured component of the diet of many bird species, particularly passerines (Woinarski *et al.* 1989; Kavanagh & Stanton 2003). One species in particular, the bell miner *Manorina melanophrys* (Meliphagidae), is heavily dependent upon lerp as a food source both as adults and nestlings (te Marvelde *et al.* 2009). This species, native to Australia, is an obligate cooperative breeder (e.g. Clarke 1989; M^{c}Donald *et al.* 2008; Wright *et al.* 2009) that forms large colonies of several hundred individuals. Individuals cooperate in a variety of contexts within the colony (Pacheco *et al.* 2008), but are best known for their despotic nature. Aggression directed at other species is so severe and effective in this species that potential avian predators and food competitors are typically excluded from the entire area that colonies occupy (Loyn *et al.* 1983; Poiani 1991).

This exclusion of other potential psyllid predators from colonized areas of forest has been suggested to be one mechanism by which psyllids may become overly abundant, leading to subsequent eucalypt dieback. Bell miners are thought to consume mainly the lerp covering of psyllid insects, leaving the insect sheltering underneath intact. This is in contrast to more typical avian predators that consume both the insect and lerp (Loyn 1987). The combination of this specialized foraging strategy and extreme territoriality may then lead to psyllid numbers increasing to the point where tree health declines and canopy dieback occurs. The correlation between bell miners and eucalypt dieback has long been suggested (Chandler 1922) and has come to be known as bell miner-associated dieback (BMAD). This process has the potential to affect up to 2.5 million hectares of forest in the Australian state of New South Wales (NSW) alone (Wardell-johnson *et al.* 2005). These areas likely to be affected are home to over 40 threatened species of flora and fauna (NSW Scientific Committee 2008), a significant portion of the regions biodiversity.

However, the mechanism/s driving BMAD are not well understood. While it has long been proposed that selective consumption of lerp casings by bell miners exists (Loyn *et al.* 1983), empirical data are lacking and it remains a matter of some debate (Poiani 1993; Loyn 1995; Wardell-johnson *et al.* 2005). Field observations of this phenomenon by foraging birds using binoculars (e.g. Woinarski 1985; Poiani 1996) are typically considered unreliable because of the difficulty of recording such small insects in a moving tree canopy (Woinarski *et al.* 1989). Earlier work examined the feeding behaviour of captive bell miners, but results were inconclusive because of small sample sizes (Tyers 1981; Robinson 1982). Gut contents of birds have been analysed, with a greater ratio of nymphs to lerp coverings observed (Poiani 1993), contrary to that expected if foraging habits of the birds are driving BMAD. However, the digestion speed of lerp relative to nymphs in bell miners' guts was not quantified. Moreover, psyllid nymphs retain previously moulted exoskeletons under their lerp coverings, and these additional exoskeletons may have potentially been recorded as entire predated nymphs, complicating interpretation of these results. In any case the key point to the relationship is whether nymphs remaining following bell miner predation are able to continue to parasitize leaves, a factor that has yet to be investigated in any study. Given this, while there is correlational evidence linking bell miner presence with high numbers of psyllids and eucalypt dieback (e.g. Wardell-johnson *et al.* 2005; Stone *et al.* 2008) and some experimental evidence (e.g. Loyn *et al.* 1983; Loyn 1987, but see also Clarke & Schedvin 1999), the causative mechanism has yet to be established (Dare *et al.* 2007).

We attempted to clarify bell miner foraging practices, by assessing the feeding behaviour of captive bell miners on leaves with known numbers of psyllids. If bell miners do forage in a specialized manner, we anticipated that they would leave more nymphs behind than their congener, the noisy miner (*M. melanocephala*; Meliphagidae). This species has a similar dietary niche and habitat requirements to bell miners, yet its association with dieback is less widespread and is more apparent in small isolated remnants (e.g. Grey *et al.* 1998) as opposed to BMAD that can be found in areas of forest largely undisturbed by humans (Wardell-johnson *et al.* 2005). To verify the veracity of psyllid nymphs remaining on leaves, we further examined the regrowth potential of nymphs following removal of their lerp coverings.

METHODS

Source of animals and husbandry

Bell miners (n = 16) were temporarily removed from a local population at Mount Wilberforce Lookout Reserve, NSW, Australia (151°02′51.04″E, 33°44′44.38″S). Noisy miners (n = 16) were sourced from a colony located on the Macquarie University campus (151°06′48.43″E, 33°46′09.22″S). Birds were housed at Macquarie University in cages measuring 2 × 1 × 1.8 m (l × w × h: hereafter cage A), with a smaller cage measuring 75 × 44 × 44 cm (hereafter cage B) attached to one side. A sliding door separated the two cages. Birds were fed Wombaroo lorikeet and honeyeater mix (Wombaroo Food Products, Adelaide, Australia), nutritional supplements and water every morning in cage B, to allow them to grow accustomed to feeding from this cage. Trials were carried out in the morning (07.00–12.00 hours), with food withheld prior to testing to ensure high levels of motivation. Following

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completion of experimental work birds were released back at the point of capture.

Selective foraging trials

Psyllid-infested leaves were collected from the canopies of spotted gums (Myrtaceae: *Corymbia maculata*) inhabited by bell miners in the Cumberland State Forest, Sydney, NSW, Australia (151°02′15.54″E, 33°44′27.64″S) and on private property in Cooranbong, NSW, Australia (151°25′11.85″E, 33°05′12.15″S). Stalks of removed branches were immediately placed in water until presented in trials. For each experiment, a single miner was first isolated in cage B. Birds were only ever fed in this cage, so they were easily enticed into this area and showed no overt signs of alarm when the door to cage A was closed. Two sets of data were then taken. The first was a measurement of restricted feeding behaviour and involved allowing the bird to have 10 foraging attempts (pecks) at the lerp on leaves presented (hereafter referred to as short-term trials). Following this leaves were then covered with a plastic container (170 × 120 × 55 mm) that simultaneously protected leaves used in short-term trials from further predation and exposed a second set of psyllid-infested leaves. Birds were then allowed unrestricted access on this second set of leaves for 5 min from the beginning of their first foraging attempt (long-term trials). This period was ample time for birds to consume all of the lerps presented if so desired. Trials were recorded with a video camera (Sony HDR-HC9E, Tokyo, Japan) for later review.

Before presenting individually marked leaves, the number of lerp presented was noted, with all psyllids offered from the *Glycaspis* genus (C. Stone, pers. comm., 2008). Leaves were inspected immediately after the birds had completed both foraging trials, with the number of lerp–psyllid complexes scored that were (i) removed in their entirety; (ii) had only the lerp casing removed, with nymphs remaining on leaves; or (iii) untouched. Leaves were then individually bagged and frozen to preserve the insects *in situ*. Each bird completed two short- and two long-term trials, with means taken for each individual bird in analyses.

Assessment of nymph viability following predation

Psyllids that had had their lerp coverings removed were examined under a dissecting microscope (Olympus SZH-ILLB, Center Valley, America) at 70× magnification. Nymphs were checked for damage to the exoskeleton or appendages. Following this we gently lifted the insect from the surface of the leaf and noted whether the stylet was still within the leaf or not. If the latter, the stylet was dissected from the body of the insect and its condition noted. Nymphs were then classed as either viable: no visible anomalies or damaged: any damage to the stylet, or missing crucial body parts (e.g. head) that would prevent any continued parasitism of leaves. We conservatively scored psyllids that were missing non-crucial body parts (e.g. antennae or legs) as viable in these analyses.

Nymph capacity to regrow lerp casings after predation

To critically assess the impact of bell miner foraging on psyllid numbers relative to other species, it is necessary to establish the viability of nymphs left behind following predation events. We assessed this directly by presenting additional lerp-covered leaves to bell and noisy miners outside of the short- and long-term foraging trials above. These leaves were still attached to twigs and had only recently been removed from the same trees sampled previously. Prior to presentation we inspected every leaf on these twigs and removed any psyllid nymph that did not have a lerp covering. Birds of either species were then permitted to forage upon the leaves for 10 min in cage B. After this the twigs were removed and each leaf inspected. Areas around psyllids that had suffered removal of their lerp covering were marked (Sharpie E07 marker, Oak Brook, America) before twigs were again placed in water and left undisturbed for 20 h. After this time each psyllid nymph was then classified as either: (i) lerp regrown, where the lerp covering had been fully regrown; (ii) lerp not regrown, psyllid nymph present but did not regrow lerp or at least only partially did so; or (iii) psyllid absent, nymph was no longer on leaf (presumably these dropped from leaves of their own accord as no predation occurred outside of trials).

Statistical analyses

Selective foraging trials

Data were analysed according to the proportion of predation events that involved the lerp casing only being removed, as opposed to both the lerp and nymph. These data departed significantly from normality using Shapiro-Wilk tests, so were normalized using an arcsine square-root transformation. Results were then aggregated into a mean proportion per bird per foraging trial (short- *vs.* long-term trials). Univariate ANOVAs with the number of lerp presented as a covariate were used to assess differences across species in both trials according to the number of psyllids presented and, for those psyllid–lerp complexes predated, the proportion where only the lerp casing was removed, leaving the nymph behind.

Nymph viability

Based on the scored damage under the dissecting microscope of predation nymphs, a logistic regression was carried out to determine if viability (yes/no) of predated nymphs was influenced by trial type (short- *vs.* long-term trials), species (bell miner *vs.* noisy miner), individual bird used in the trial or the interaction between trial type and species. Terms were dropped sequentially until only significant factors remained.

Lerp casing regrowth

The proportion of nymphs with regrown lerps following predation by either species was compared using a contingency table with a continuity correction.

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Alpha was set at 0.05 for all analyses, and means are presented as ±1 SE throughout. All data were analysed using the statistical software package *SPSS* v16.0 for Mac (SPSS, California, USA).

RESULTS

Selective foraging trials

As a result of the random nature of infestation levels of leaves used in experiments, we were unable to control for the number of psyllids offered in each trial. This lead to the number of psyllids presented to each species differing significantly in both the short- (10 foraging attempts only: $F_{1,30} = 20.450$, $P < 0.001$) and long-term trials (5 min unrestricted foraging: $F_{1,30} = 13.573$, $P = 0.001$). In both cases noisy miners were offered more psyllids than bell miners. Given this, we subsequently weighted the following analyses by the number of lerp offered in each trial as a covariate. In both the short- ($F_{1,30} = 20.328$, $P < 0.001$) and long-term trials ($F_{1,30} = 31.122$, $P < 0.001$) bell miner predation resulted in more instances where only the lerp casing was removed, leaving the psyllid nymph behind (Fig. 1). The magnitude of species differences in the proportion of psyllids left behind following predation increased from the short- (where only single predation events were recorded) through to long-term trials (where individuals had ample time to revisit predated nymphs).

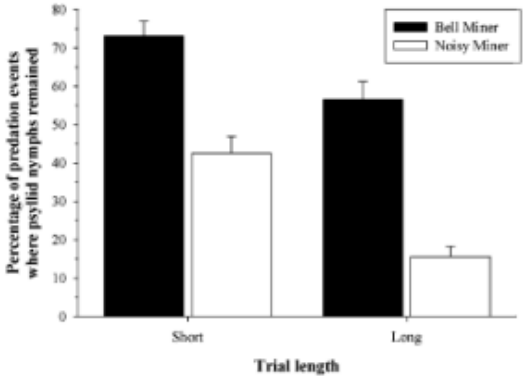

Fig. 1. The proportion of predation events where only the lerp casing was removed, leaving the psyllid nymph behind during foraging bouts of either bell (solid) or noisy miners (open bars). Observations were either short- (10 foraging attempts only) or long-term (5 min of unrestricted foraging). Data represent means ± 1 SE, based on a total of 16 birds of each species in each trial.

Behavioural observation

From perusal of the video footage frame-by-frame, the two species used different foraging techniques to predate psyllid–lerp complexes in all foraging events observed. Bell miners placed one mandible either side of the lerp casing, holding their head at about a 45°C angle, before removing the (primarily) lerp casing by flicking their tongue forward. In contrast, noisy miners held their bill at a more perpendicular angle, snapping at the entire lerp–psyllid complex several times to remove both the lerp casing and often the psyllid underneath with their mandibles alone.

Nymph viability following predation events

We subsequently examined the viability of nymphs left after birds removed their lerp casing via assessment of damage using a dissecting microscope. Again, there was a significant difference between species (Wald = 6.339, d.f. = 1, $P = 0.012$). Nymphs exposed by bell miners were more often classed as viable (89%, $n = 171$) than those of noisy miners (84%, $n = 198$). Neither the interaction between trial length and foraging species (Wald = 1.828, d.f. = 1, $P = 0.176$), nor trial length alone (Wald = 0.591, d.f. = 1, $P = 0.442$), had a significant effect on viability.

Nymph capacity to regrow lerp casings after predation

We further assessed the viability of remaining nymphs that had lost their lerp covering by monitoring the proportion of nymphs that were able to regrow their lerp casing. After a period of 20-h post-predation, regrowth was significantly higher in psyllids predated by bell miners (75.3%; $n = 89$) than those attacked by noisy miners (34.7%; $n = 95$; $\chi^2_1 = 28.832$, $P < 0.001$).

DISCUSSION

Although bell miners have long been associated with the presence of dieback in eucalypt forests, their role as a causative agent has been unclear. We examined the foraging strategies of bell miners as a potential mechanism by which psyllid numbers might increase, and hence exacerbate canopy dieback. Bell miners in this study exhibited the selective foraging behaviour proposed by Loyn (1987) when feeding on psyllids, frequently removing their lerp casings and leaving the nymphs behind intact. This occurred significantly more often following bell as opposed to noisy miner predation, a congener that also regularly takes psyllids

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in the field. It is worth noting that noisy miners also occasionally removed lerp casings only and left the psyllid behind, and this supports data suggesting that noisy miners can sometimes be implicated in much smaller-scale dieback (Grey *et al.* 1998). However, psyllids that had their lerp removed by bell miners were more likely to appear morphologically intact and viable, and subsequently regrow their lerp covering, than those predated by noisy miners. These data therefore suggest a selective foraging strategy is employed by bell miners that is likely to have a reduced impact on prey populations relative to that of other predators. In this way, bell miners may indeed exacerbate or even be the causal agent behind BMAD. Non-lethal bell miner foraging may facilitate psyllid populations increasing to the point where they parasitize eucalypts unsustainably, leading to decreased tree health and significant reductions in habitat quality (Stone *et al.* 2008). This supports the experimental evidence presented by Loyn *et al.* (1983), who showed that psyllid populations were reduced and the populations of other, more effective, avian predators recovered along with tree health after bell miners were removed. Modifications of environmental characteristics directly via the actions of one species have been documented previously, particularly for changes in vegetative structure following herbivory (e.g. Clay *et al.* 1993; Baines *et al.* 1994). However, the apparent indirect, third-party pathway of the eucalypt–psyllid–bell miner interaction herein appears much less common.

An association between the presence of bell miners and eucalypt crown dieback has been documented for many years (Chandler 1922). Although some evidence existed (e.g. Robinson 1982; Loyn 1987) empirical evidence suggesting bell miners are the causative agent of BMAD was sparse, with their presence also likely to simply be a symptom of other ecosystem problems, such as anthropogenic tree stress (Dare *et al.* 2007). Here we present data indicating a mechanism that might well lead to increases in the number of parasitic psyllids present in an ecosystem. Bell miners appear to primarily remove only the lerp covering of psyllids during foraging events by utilizing a specific foraging action, frequently leaving the psyllid underneath viable and able to continue parasitizing leaves, consistent with earlier work (Robinson 1982; Loyn *et al.* 1983). These findings are also in accordance with the 'farming' analogy first proposed by Loyn (1987), and are in contrast to the more indirect findings of gut contents analysis of Poiani (1993).

'Denuded' psyllid nymphs that have had their lerp covering removed are obviously vulnerable to further potential damage, desiccation and subsequent lethal predation (White 1970). This might include additional attacks from the same or other bell miners, or predators not assessed here, such as spiders and parasitic wasps (Stone 1996, 2005). While we assessed regrowth to the 20 h mark in this study, the vast majority of psyllids had completely regrown their covering by 8-h post-predation (K.M. Haythorpe, pers. obs., 2008). However, in the captive trials both bell and noisy miners had the opportunity to predate nymphs that they had exposed by their earlier foraging attempts. Noisy miners did so more often, as can be visualized in Figure 1. Note that other insectivores known to routinely take lerp, such as spotted pardalotes *Pardalotus punctatus*, have also been described removing the underlying psyllid insect more often than the lerp casing alone during foraging (e.g. Loyn *et al.* 1983; Woinarski *et al.* 1989), as we observed in noisy miners. This further suggests that the non-lethal foraging of bell miners is distinct from other insectivores that commonly prey upon lerp.

Whether this unique foraging strategy is the primary cause or simply an accelerating factor in the development of BMAD requires further data. Many factors interplay in the spread of this phenomenon (Stone 2005); however, given that we have confirmed bell miners could well play a large role in promoting dieback, pertinent questions for investigation now include: Why has dieback resulting from BMAD only occurred during the last century? Is BMAD a 'natural' phenomenon or the end product of anthropogenic factors? Do bell miners colonize areas already showing signs of increased psyllid numbers or simply preferentially occupy psyllid-rich habitats? Do psyllid populations increase rapidly following colonization? Does removal of bell miners lead to a return in tree vigour? Preliminary work exists in the latter area; however, results have been contradictory. For example, Clarke and Schedvin (1999) found removal of bell miners from a site did not alter tree health, while Loyn *et al.* (1983) and Loyn (1987) reported significant improvement following miner removal. What is now required is larger-scale monitoring of bell miner density, tree health and vegetation characteristics over an ecologically relevant time frame of several years in order to clearly demonstrate that bell miner foraging has an effect upon forest health.

Together, our work demonstrates that bell miners are likely to leave behind a greater number of psyllids from a foraging event than noisy miners. Furthermore, these denuded psyllids are more likely to be intact and capable of regrowing a lerp casing and thus continue to parasitize host trees. These findings indicate that bell miners have the potential to be one of the main causal factors in driving the spread of BMAD and thus modifying the habitat the species occupies. Direct habitat modification through the actions of species has been observed often, for example, the actions of prairie dogs (*Cynomis* spp., Whicker & Detling 1988), moose (*Alces alces*, McInnes *et al.* 1992) or beavers (*Castor canadensis*, Wright *et al.* 2002) all directly modify their surroundings. In contrast, it appears that the BMAD

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pathway here is indirect; bell miners modify psyllid abundance and these in turn affect the health and structure of vegetation. Impacts of this appear to be sufficient to cause the death of large patches of habitat, with bell miner colonies subsequently shifting to new areas (Loyn 1987; Dare *et al.* 2007). This is not dissimilar to the 'slash and burn' method of agriculture used by humans (Loyn 1987, 1995), and the environmental implications of such habitat modification by bell miners warrants further investigation to ameliorate any potential conservation and/or economic impacts of this apparent bell miner–psyllid–dieback interaction.

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Learning from watching alarmed demonstrators: does the cause of alarm matter?

Andrea S. Griffin [a,*], Kathryn Haythorpe [b]

[a] School of Psychology, University of Newcastle
[b] School of Environmental and Life Sciences, University of Newcastle

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Many social-learning opportunities expose animals to the behaviour of conspecifics, but also to causes and consequences of those behaviours. Attending to information over and above social behaviour per se may provide a strategy by which the reliability of social information is ensured. Earlier work in Indian mynahs, *Sturnus tristis*, has demonstrated that observers become more wary in a location in which they are accustomed to foraging after they have viewed a conspecific undergo a 'predator' attack at that site. We determined whether observation of both an alarmed demonstrator and the cause of the conspecific's alarm (capture by a human) were critical to such observational learning. Experimental observers watched a demonstrator mynah display high levels of alarm in response to pursuit and capture by a human, while control mynahs watched a demonstrator express a similar level of alarm to a threatening nearby human, but visual access to the human by observers was blocked. Analysis of observer behaviour at the feeding site both before and after observational training revealed that experimental observers remained wary at the feeding site after training relative to before, relative to control observers that became far less wary, strongly suggesting that both social and causal information were important for observational learning. This result contributes to the growing body of empirical evidence that use of social learning is modulated by a rich variety of contextual information that may help ensure that its use is adaptive.

Observational learning is a form of social learning in which animals acquire information about the world by observing the behaviour of other individuals, typically conspecifics (Mason & Reidinger 1981, 1982; Mason 1988; Zentall & Galef 1988; Mineka & Cook 1993; Heyes 1994; Olsson et al. 2007). For example, animals may acquire an alarm response to a novel predator after watching the alarm responses of other individuals to the predator (Mineka & Cook 1993; Blanchard et al. 2001; Griffin & Evans 2003). Observation of social companions creates opportunities to view not only the behaviour of others, but also the causes and consequences of those behaviours. Theoretical analyses, supported by a handful of empirical studies, have revealed that there are a variety of conditions under which using social information may be maladaptive (Laland & Williams 1998; Pongrácz et al. 2003). Consequently, it has been predicted that the use of social learning should be governed by a collection of strategies that define the circumstances under which its use is adaptive (Laland 2004). In addition to strategies such as copying more experienced individuals, and only copying when individual learning is unsuccessful, only copying social behaviours with causes and consequences may help, under some circumstances, to ensure that learning is worthwhile. For example, copying an alarm response that has no apparent cause, or a foraging technique that fails to produce food, is unlikely to yield an adaptive advantage.

Although observational learning is a well-studied phenomenon, there have been surprisingly few attempts to determine to what extent information other than that provided by demonstrator behaviour per se is necessary for learning to occur. The only systematic work in this area has been in the context of the social transmission of food-finding techniques. Careful behavioural analyses have revealed that visual access to both demonstration of a novel foraging technique (e.g. lid opening) and subsequent food consumption by the demonstrator (consequence) facilitates acquisition of the novel foraging technique in observers, strongly suggesting that observers attend to both the behaviour of the demonstrator and the consequence of that behaviour on the environment (Groesbeck & Duerfeldt 1971; Palameta & Lefebvre 1985; Heyes 1994; Akins & Zentall 1998; Coolen et al. 2005).

To our knowledge, only one study has attempted to examine whether observing the cause of a social companion's behaviour affects the likelihood of observational learning. Observer rats, *Rattus norvegicus*, given the opportunity to watch a demonstrator rat express a fear response after touching a candle with its nose,

* Correspondence: A. S. Griffin, School of Psychology, University of Newcastle, Callaghan, NSW 2308, Australia.
E-mail address: andrea.griffin@newcastle.edu.au (A. S. Griffin).

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and singeing its whiskers, subsequently made fewer nose contacts with a candle than rats that were exposed to a candle and a fearful demonstrator that could not make nose contacts with the candle (Lore et al. 1971). A subsequent analysis indicated that observation of demonstrator contact with the flame played a critical role in observer learning. Indeed, learning was impaired if visual access to the point of contact between the rat's nose and the candle was blocked, suggesting that learning required not only observation of a fearful demonstrator and an external stimulus present at the same time (candle), but also information that allowed one to be related to the other (Bunch & Zentall 1980; Zentall 2006).

Social learning about predators is commonly used to explore mechanisms of social transmission of fear (Griffin 2004). The typical experimental paradigm involves measuring whether the responses of observers to a novel predator increase after they have undergone 'training' sessions in which they view the novel predator in conjunction with social alarm signals. As is, the training protocol combines presentation of social alarm signals and their cause (i.e. the novel predator), making it effective for demonstrating that animals can learn about the cause of a companion's alarm, but, by the same token, making it impossible to examine whether observing the cause of social alarm is necessary for learning. The present study overcame this problem by employing a place-learning paradigm to explore whether observation of the cause of a social companion's alarm plays a determining role in social learning about danger.

The Indian mynah, *Sturnus tristis* (formerly classified as *Acridotheres tristis*, Christidis & Boles 2008; also referred to as the common myna), is a highly opportunistic species of Passerine that has invaded large areas of the east coast of Australia since it was introduced in the 1800s. Indian mynahs are highly social and can be found foraging in groups of two to 20 individuals throughout the day (Pell & Tideman 1997). At night, birds form communal roosts sometimes containing thousands of individuals. The social and highly opportunistic lifestyle of Indian mynahs makes them an ideal system to study mechanisms of social learning about danger (Pell & Tideman 1997; Pizzey & Knight 1998; Tideman 2006; Griffin 2008b, 2009; Griffin & Boyce 2009; Griffin et al. 2010).

We have previously shown that Indian mynahs become more wary in a location in which they are accustomed to foraging after they have observed a human 'predator' chase, catch and remove a social companion from that location, relative to control mynahs that become less wary in the feeding site after they have watched a human perform the same capture gestures at the feeding site, but in the absence of any conspecific mynah (Griffin & Boyce 2009). In that study, the control group became less place-wary after training, suggesting that increased place wariness in experimental birds was not due to observation of a threatening human per se, but rather to social alarm, or alternatively, an interaction between cues from the human and cues from the alarmed demonstrator (Griffin & Boyce 2009). In a parallel study, observer mynahs that watched a demonstrator express a high-level alarm response triggered by a cat that observers could not see subsequently failed to become more place-wary relative to control mynahs that watched a companion mynah feeding at the foraging site; this result pointed to the hypothesis that learning relied upon an interaction, rather than social alarm alone (Griffin et al. 2010). The aim of the present study was to extend this earlier work by specifically testing to what extent place learning is dependent upon viewing both an alarmed demonstrator and capture of the demonstrator by a human, the cause of alarm.

Building on previous work, we used once again capture by a human to trigger place learning. Capture by a human provides the unique opportunity to examine the effects of presenting social alarm with or without an external event (human performing capture gestures), which plays the role of a cause, but at the same time remains neutral in the sense that it does not trigger place learning on its own (Griffin & Boyce 2009). Any acquired place wariness in response to watching a demonstrator be caught is therefore necessarily a consequence of an interaction between human cues and social alarm, or social alarm alone, hypotheses the present experiment was designed to tease apart.

Food-deprived mynahs were trained to move between a holding site and a feeding site through a small pipe. Mynahs allocated to a human-present observer group were then provided with the opportunity to watch a demonstrator mynah located at the feeding site being chased and caught by a human (observational training). Mynahs assigned to a human-absent observer group also watched a demonstrator mynah expressing a high-level alarm response to a nearby threatening human, but visual access to the human by observers was blocked. Consequently, they viewed only the alarmed demonstrator mynah without any information about what was causing the demonstrator's alarm response. As in previous work in our laboratory, and consistent with work on individual learning about dangerous places, we quantified place learning by measuring levels of risk assessment behaviour both before and after observational training in both groups of observer mynahs at the foraging site (Fanselow 1990; Blanchard et al. 2001, 2005; Hubbard et al. 2004; Griffin & Boyce 2009; Griffin et al. 2010). Comparisons between human-present and human-absent observers allowed changes in behaviour that were specifically attributable to differential observational experience (human present versus human absent) to be isolated from those caused by nonassociative learning processes, such as sensitization (Griffin & Evans 2003; Griffin 2009; Griffin & Boyce 2009).

METHODS

Subjects and Husbandry

Fifty-eight Indian mynahs were captured in an urban location in Newcastle, a medium-sized city on the eastern coast of Australia, using a walk-in baited trap designed specifically to trap this species and widely used for population control (Tideman 2006; see Griffin 2008b for a detailed description of the trap). Trapping and transport were identical to earlier work (Griffin 2008b, 2009; Griffin & Boyce 2009). Each bird was weighed, measured and individually identified with a light-weight coloured plastic leg band. Birds were then transported to the Central Animal House at the University of Newcastle and released into a large outdoor group flight aviary (2.25 × 1.25 m and 4.4 m high). Birds were left undisturbed for a minimum of 3 weeks to acclimatize to captivity. All captive mynahs had ad libitum access to water and a mixture of dog pellets, fresh fruit and vegetables.

Twenty-nine randomly selected mynahs were assigned to act as observers and 29 were assigned to act as demonstrators. Of the 29 observers, 15 were assigned to watch a demonstrator undergo capture by a human (human-present observers) and 14 were assigned to watch an alarmed demonstrator in the absence of a human (human-absent observers; see below for more details). Each of the 29 demonstrators was randomly assigned to one of the 29 observers. No attempt was made to control for sex during experiments, as Indian mynahs are not sexually dimorphic. Sample sizes were determined on the basis of extensive previous work on predator recognition and predator avoidance learning by the first author (Griffin 2003, 2008b, 2009; Griffin & Galef 2005; Griffin & Boyce 2009; Griffin et al. 2010).

For testing, each subject was transferred from the outdoor flight aviary to an individual home cage (0.6. × 0.6 × 0.6 m). Cages containing demonstrators were in visual and acoustic contact, while

those containing observers were only in acoustic contact. Observers were maintained in visual isolation to avoid any observational experience acquired in home cages interfering with that acquired during experiments (see below). Each home cage was equipped with a perch, a food bowl, a water bowl and an opaque nestbox (0.3 × 0.2 m and 0.18 m high) that was used to move individuals between their home cage and the test apparatus. Individual home cages were held in a room with complete access to natural light. Following transfer from group to individual housing, birds were left undisturbed for 2 days to acclimatize to their new environment.

All animal care, husbandry and experimental procedures were approved by the University of Newcastle Animal Research Ethics Committee. Indian mynahs are classified as a pest species in Australia and neither the government nor the Animal Care and Ethics Committee allows for release after capture. Consequently, all birds were euthanized by a qualified veterinarian at the end of the experiment using an overdose of CO_2. As previously, the study was undertaken during the nonbreeding season of Indian mynahs (March–May; Griffin & Boyce 2009).

Apparatus

The experiment took place in a separate room to that containing the home cages. The apparatus was identical to the one used in our earlier work on place learning in Indian mynahs with the mere addition of several strategically placed curtains around it (Fig. 1; Griffin & Boyce 2009; Griffin et al. 2010). The apparatus consisted of a long table divided in half by a vertical wooden screen, which could be raised or lowered by the experimenter from behind a curtain (Fig. 1). A cage (0.7 × 0.7 × 0.7 m) was located on each side of the screen. The two cages were connected by a horizontal opaque pipe (8.5 cm diameter, 30 cm long) located at ground height. When the screen was raised, visual contact between the two cages and passage between them through the pipe was possible; lowering the screen blocked visual contact and passage through the pipe.

One cage was referred to as the holding cage, while the other was referred to as the feeding cage. Both cages were equipped with a perch. In addition, a food tray was located on the floor of the feeding cage. Each cage was fitted with a small security camera (0.03 × 0.03 × 0.01 m, camera CCD mini B&W 380TVL Samsung), which was connected to a PC computer running security software (PICO2000 UCL Technologies Inc., http://www.ucltech.com). The set-up was used to view and record the behaviour of the birds during experiments. A curtain hanging alongside the wooden screen and perpendicular to the table could be either opened or closed during observational trials depending on the treatment (Fig. 1; see below). During all experiments, white noise was played back through two loudspeakers at a mean volume of 70 dB to mask the vocalizations of mynahs in the adjacent home cage room, as well as any noise produced accidentally by the experimenter.

Experimental Protocol

Initial training

We began the experiment by training each observer mynah to cross from the holding cage to the feeding cage via the pipe, and to forage in the food tray (Fig. 1). The procedure was identical to that used in previous work on place learning in Indian mynahs (Griffin & Boyce 2009; Griffin et al. 2010). On the evening of the third day after transfer from the flight aviary to individual home cages, observers were food deprived overnight. The next morning, mealworms were placed in the food dish in the feeding cage of the experimental apparatus and several others were placed inside the pipe (Fig. 1). The screen between the holding cage and the feeding cage was lowered (Fig. 1). Each observer mynah was then released into the test apparatus. After lifting the dividing screen, we waited until the subject had crossed into the feeding cage and 10 min had passed. The subject was then returned to its home cage where it was immediately provided with food. That evening, observers were food deprived once again, and the procedure was repeated the next morning, except that no food was placed in the pipe. By the end of

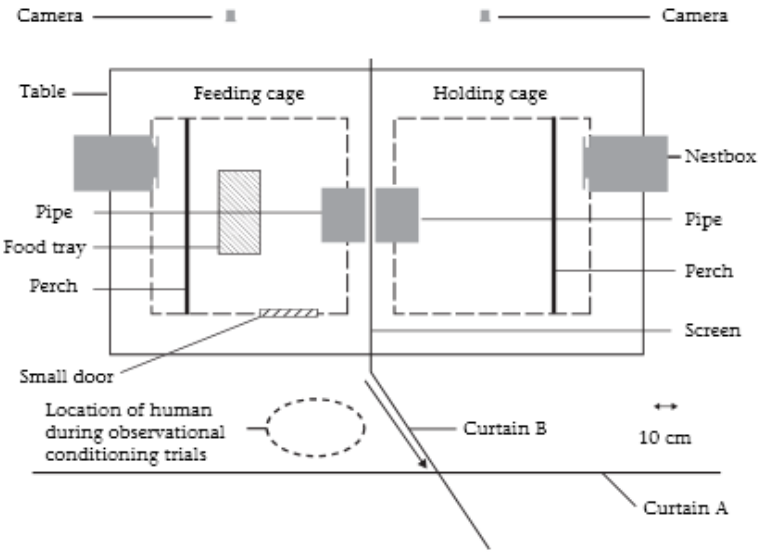

Figure 1. Aerial view of the experimental apparatus. Cameras were connected to a computer (not shown) located behind curtain A. Curtain B was either open (i.e. drawn in the direction of the arrow) to reveal a human chasing and capturing a demonstrator through a small door on the side of the cage to observers in the holding cage (human-present observers) or closed (i.e. drawn in the opposite direction to the arrow) to hide a nearby threatening human from observers (human-absent observers) during observational training. During both types of trial, the human stood within the indicated zone. See text for further details.

the second training session, all observers had learnt to cross through the pipe to access the food in the feeding cage. During this phase of initial training, mynahs assigned to act as demonstrators were left undisturbed in their home cages.

Pre- and post-tests

Each observer mynah first received a pretest during which it was released into the holding cage and given access by lifting the dividing screen to the feeding cage where several mealworms cut in half were available in the food tray. The following day, we conducted two observational training trials during which observers were confined to the holding cage and given the opportunity to watch either a demonstrator express a high-level alarm response to a hidden human (human-absent observers; see below) or a demonstrator undergo capture by a visible human (human-present observers; see below). The day after the two observational training sessions, each observer mynah underwent a post-test, the procedure of which was identical to that of the pretest. For pretests and post-tests mynahs were food deprived overnight, using the same deprivation schedule as during initial training.

Observational training

During observational training trials, passage between the holding cage and the feeding cage was blocked by filling the connecting pipe with a cloth (Fig. 1). Each observer was then released into the holding cage with the screen lowered. Observers in the human-present group were then given the opportunity to watch a human chase, capture and remove a demonstrator from the feeding cage. To this end, a demonstrator was released into the feeding cage and the screen was raised. A human then appeared from behind a black opaque curtain and began to chase the demonstrator with a net through a small door on the side of the cage (Fig. 1). As in previous work, the human stood alongside the rectangular table to avoid making eye contact with the observer mynah, thus reducing the likelihood that it would perceive the capture event as self-directed (Griffin & Boyce 2009). The capture event lasted 2 min, the last 30 s of which involved the actual capture and removal of the demonstrator from the cage, immediately after which the screen was lowered and the observer returned to its home cage.

Observers in the human-absent group also watched a demonstrator express a high-level alarm response to a nearby threatening human for 2 min, but visual access to the human was blocked by closing a black opaque curtain that hung alongside the screen and perpendicular to the table (Fig. 1). Hiding the human behind the curtain allowed us to expose observers in the human-absent group to an alarmed demonstrator, without allowing them to see the event to which the demonstrator was responding. Wild-caught, caged Indian mynahs naturally show high levels of alarm in response to a human standing next to their cage, particularly if they are stared at. However, to ensure that demonstrators in the human-absent group exhibited alarm responses of similar amplitude to demonstrators undergoing pursuit and capture, the nearby human fixated on the demonstrator and made jerky movements towards it to simulate sudden approach. Alarm responses in both groups of demonstrators involved very high levels of locomotion and obvious attempts to escape from the cage (pushing beak between bars), but no alarm calls as caged Indian mynahs do not alarm call to humans (A. S. Griffin, unpublished data).

As in previous work, we conducted two observational training trials to increase the amount of observational experience observers received relative to their individual experience of the feeding cage (initial training and pretest). Consequently, each observer mynah received a second observational training trial, identical to the first, between 60 and 90 min after the end of the first.

Our design incorporated the comparison of learning in a group of observers exposed to an alarmed conspecific experiencing pursuit and capture by a human and another group of observers exposed to an alarmed demonstrator on its own. The experience of both groups of observers with the feeding cage was in all other respects identical. Consequently, differences between the two groups' behaviour after observational training are necessarily attributable to differential observational experience. Between-group comparisons of this kind are the critical parameter demonstrating observational learning, and not within-group comparisons between behaviour before and after training, which might be attributable to nonassociative effects, such as sensitization to the experimental setting (Shettleworth 1998; Griffin 2003).

Data Analysis

All trials were video recorded and behaviour was scored from tape played back at half speed using JWatcher 1.0 (Blumstein et al. 2006) by an experimenter who was unaware of which treatment subjects had undergone.

Earlier work on socially acquired predator recognition, as well as observational place learning, in this system has reliably shown that Indian mynahs respond to danger by increasing locomotion (Griffin 2008b; Griffin & Boyce 2009), while, in the literature, changes in foraging patterns are reliably associated with perceived risk (Krebs & Davies 1997). Consequently, we analysed both of these behavioural variables. Specifically, we quantified the percentage of time allocated to locomotion (walk and flight) and peck rates during a 2 min observation period that began immediately after each observer had entered the feeding cage for the first time during both pre- and post-tests. As in our earlier work on place learning in Indian mynahs (Griffin & Boyce 2009; Griffin et al. 2010), we also measured for both pre- and post-tests the latency of each observer to enter the feeding cage after the screen was raised, as well as the proportion of time spent inside the feeding cage relative to the total observation period. As in previous work, neither of these variables was affected by exposure to a treatment; consequently, results are not included here.

Each dependent variable was either logged or square-root transformed to meet the requirements of normality. We then tested for differential changes in behaviour between observer treatments using a repeated measures ANOVA with time (pretest, post-test) as a within-subject variable and observer treatment (human present, human absent) as a factor. We expected that observationally acquired wariness of the feeding cage would be reflected by an increase in the percentage of time allocated to locomotion and a decrease in peck rates once there during post-tests relative to pretests. In contrast, habituation to the feeding cage would be reflected by a decrease in the percentage of time allocated to locomotion and an increase in peck rate once there.

All statistical analyses were carried out using PASW Statistics 18.0 (SPSS Inc., Chicago, IL, U.S.A.). Two-tailed tests were used throughout and alpha levels were fixed at 0.05.

RESULTS

A two-way repeated measures ANOVA on the percentage of time allocated to locomotion by observers revealed a significant main effect of time ($F_{1,27} = 9.403$, $P = 0.005$) and a significant time*-treatment interaction ($F_{1,27} = 12.230$, $P = 0.002$). The main effect of treatment was not significant ($F_{1,27} = 0.126$, $P = 0.726$). An identical analysis on pecking rate revealed a time*treatment interaction that fell just short of significance ($F_{1,27} = 3.888$, $P = 0.059$) and no main effect of time ($F_{1,27} = 2.150$, $P = 0.154$) or treatment ($F_{1,27} = 2.103$, $P = 0.159$). Overall, observers that watched a demonstrator mynah

display high levels of alarm in response to being pursued, captured in a net and removed from the feeding site by a human maintained their pretest levels of locomotion after training relative to before, relative to control observers, which watched a highly alarmed demonstrator with no apparent cause, and which decreased locomotion at the feeding site after observational training relative to before (Fig. 2). Together, these results suggest that observation of an alarmed demonstrator being chased, caught and removed from the feeding site inhibited a decrease in wariness at the feeding site that occurred in observers that watched a demonstrator express alarm without cause.

To ensure that between-group differences in acquired locomotion were not attributable to differences in the two groups' initial behaviour, we compared the two treatments' initial levels of locomotion using an independent *t* test. This analysis revealed no treatment effect indicating that there were no differences in behaviour between the two treatments before observational training ($t_{27} = 1.636$, $P = 0.113$).

Finally, to examine the role of demonstrator alarm in observer learning, we calculated the mean proportion of time allocated to locomotion, as well as the mean locomotion rate, of each demonstrator across the two observational training sessions. As in previous work (see Methods), very high locomotion was the most obvious behavioural response to nearby human presence in both demonstrator treatments. For each demonstrator, locomotion was scored during a 90 s time period that began as soon as the screen between holding and feeding cages was lifted. These analyses revealed no significant differences in either the total time allocated to locomotion (mean + SE; human present: 69.1 + 6.2%; human absent: 76.8 + 3.5%; independent *t* test: $t_{27} = 1.062$, $P = 0.298$), or locomotion rate (mean + SE; human present: 0.28 + 0.06 [1/s]; human absent: 0.41 + 0.08 [1/s]; independent *t* test: $t_{27} = 1.950$, $P = 0.062$) by the two demonstrator treatments. The effect of demonstrator locomotion on observer learning was further explored by introducing these descriptors of locomotion as covariates into the overall 2 by 2 time by treatment repeated measures ANOVA on observer locomotion. Proportion of time allocated to locomotion and locomotion rate were highly negatively correlated (Pearson correlation coefficient: $r = -0.722$, $P < 0.001$), so each variable was introduced into the overall model separately. These analyses revealed no significant effect of either of the two demonstrator locomotion variables on differential changes in locomotion of observer treatments (locomotion rate: $F_{1,26} = 0.02$, $P = 0.888$; percentage locomotion: $F_{1,26} = 0.004$, $P = 0.951$). Furthermore, the significant time*treatment interaction on observer locomotion remained significant in each case indicating that demonstrator locomotion during observational training did not explain treatment differences in observer behaviour after training.

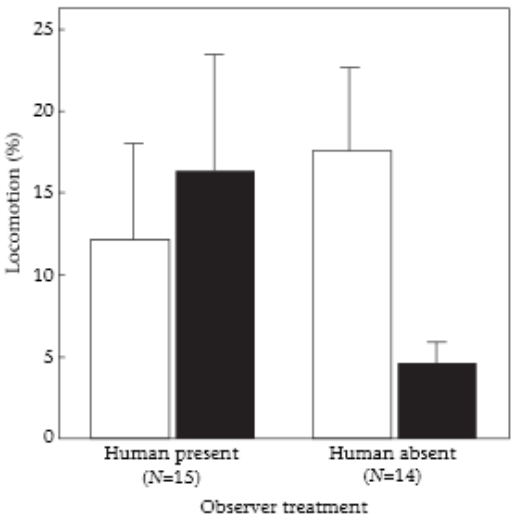

Figure 2. Locomotion expressed in the feeding cage by human-present and human-absent observer mynahs both before (pretest, open bars) and after (post-test, black bars) observational training. The mean + SE percentage of time allocated to locomotion is indicated for a 120 s time period after entering the feeding cage. See text for further details.

DISCUSSION

In the present study, we examined to what extent observational place learning was dependent upon observing both the alarm behaviour of a conspecific and its cause. Results revealed that observers that had visual access to the capture of a companion by a human remained cautious during a subsequent trip to the foraging site, while observers that had watched a demonstrator exhibit an alarm response without any apparent cause became less wary after training relative to before. These findings suggest that visual access to an interaction between cues from the human and cues from the demonstrator played a key role in triggering acquired changes in behaviour.

Rather than enhancing risk assessment behaviour, it might be argued that observing capture of a companion inhibited a decline in risk assessment behaviour that occurred in mynahs that observed an alarmed companion alone (Fig. 2). Reduced wariness in control mynahs is likely to have been mediated by habituation to the feeding cage, but further research will be needed to ascertain this. Comparable directional changes in behaviour of experimental and control groups have been found during earlier work on observational place learning and social learning of predators in Indian mynahs. For example, presenting a novel predator together with social alarm calls inhibits a reduction in visual attention that occurs in mynahs that receive novel predator and alarm calls separately (Griffin 2009). The key point is that between-group comparisons of behavioural changes across observational training are critical to isolating the effects of a specific between-group manipulation and not within-group comparisons of behaviour before and after training. Although it would be of theoretical interest to habituate observers to the feeding cage fully prior to observational learning, and to show that observation of social alarm and human inculcates a significant increase in wariness in human-present mynahs relative to an absence of change in human-absent mynahs, this approach would require providing mynahs with extensive nonaversive individual experience with the feeding cage prior to observational training. Two aversive social-learning experiences may then be insufficient to modify extended previous individual experience (Galef & Whiskin 2001).

The present findings replicate and extend results from our earlier studies of observational place learning in so far that we consistently find that observing a human chase and catch a demonstrator mynah in a net in an area in which observers are accustomed to feeding heightens risk assessment behaviour on a later trip to the foraging site relative to a control treatment (Fig. 3a, b), while observing an alarmed conspecific per se does not (Fig. 3a, c), and, based on earlier work, neither does a human waving a net in an empty feeding site as if to catch a mynah (Fig. 3b). Consequently, we must assume that some interaction between cues from the human and cues from the demonstrator mynah is necessary for alarm behaviour to be copied. Further work will be needed to isolate exactly what it is about the interaction that produces learning. Given that locomotion behaviour of demonstrators did not explain acquired changes in locomotion of

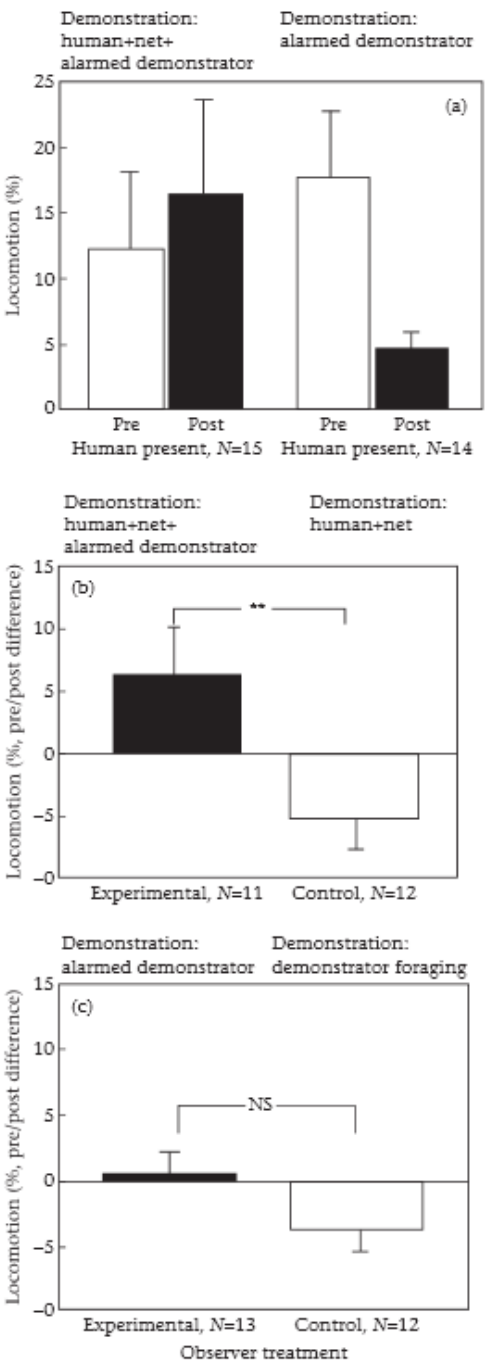

Figure 3. Changes in observer locomotion obtained in (a) the present study, and each of our previous studies on place learning in Indian mynahs: (b) Griffin & Boyce 2009; (c) Griffin et al. 2010. (a) Mean + SE percentage of time allocated to locomotion during pre- and post-tests. (b) and (c) Mean + SE pre/post-test change in locomotion. $^{**}P < 0.01$, nonparametric Mann–Whitney test. To facilitate comparisons, the content of the observational experience of each observer treatment is indicated. (b, c) Reproduced with permission from Elsevier Press.

observers, it is possible that capture by a human is key, rather than merely the presence of both human and demonstrator. As well as providing causal information, capture of a conspecific also encodes predator success, both of which could facilitate learning. Spatial contact between the human and the demonstrator mynah may be sufficient. This hypothesis would be in line with work in rats showing that the presence of a fearful demonstrator and a candle is not sufficient to inculcate candle avoidance. Learning requires observation of the point of contact between the demonstrator's nose and the candle (Bunch & Zentall 1980; Zentall 2006).

However, the interaction effect of human and social alarm may not necessarily be supported by a causal relationship. Human and social alarm cues may interact in such a way that the aversive effect of their combination on observers is greater than that of each of the cues presented alone. These hypotheses could be teased apart by comparing the effect of manipulating the contingent relationship between the two components of the aversive experience (human and social alarm) with the effect of enhancing the intensity of the social alarm demonstration. The former could be achieved by comparing learning in treatments that repeatedly received the two stimuli together with learning in a treatment that repeatedly received the two stimuli separately, while the latter could be achieved by using several demonstrators, for example. In the light of over two decades of learning research showing that animals attend to contingent relationships between physical events (e.g. a light signalling food delivery; Rescorla 1988), it seems reasonable to suggest that they should also have the ability to do so in their social world.

Our results are consistent with detailed work on social transmission of foraging behaviour in pigeons, *Columba livia*, and rats, indicating that observers attend to both the behaviour of social companions and the consequences of those behaviours (Groesbeck & Duerfeldt 1971; Palameta & Lefebvre 1985; Heyes 1994; Akins & Zentall 1998; Coolen et al. 2005). The findings contribute to a growing body of empirical evidence pointing to the fact that social learning is not indiscriminate; animals employ a number of strategies to determine the context in which they should learn from others (Laland 2004). Integration of events over and above social behaviour per se, such as the causes and consequences of the observed behaviour, may assist in ensuring that only reliable social sources are copied. Although field research will be needed to explore the conditions under which causal and outcome information could be used under free-ranging conditions, it will be interesting for future investigations in social learning to entertain the possibility that observational learning may require more complex cognitive integration than proposed by current thinking (Olsson & Phelps 2007; Griffin 2008a).

Acknowledgments

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Appendix C: Media

This appendix contains media publications and appearances regarding the candidate and doctorate work.

1. Stephen Williams, "Revealing the truth about Indian mynahs", *Newcastle Herald,* 11 April 2011.

2. a) Sean Farnham, "Avian home invaders", *Newcastle Herald* Opinions, 12 April 2011.

   b) Chris Skeet, (untitled), *Newcastle Herald* Opinions, 12 April 2011.

   c) Kathryn Haythorpe, "Minor or major problem", *Newcastle Herald* Opinions, 15 April 2011.

3. Jeff Corbett, "A trapped Indian myna", *Newcastle Herald*, 27 April 2011.

4. Paul Bevan, "Show us your PhD", ABC radio interview transcript, 2 May 2011.

5. Damon Cronshaw, "Home to roost in suburbia", *Newcastle Herald*, 7 May 2011.

6. Sean Dooley, "Sympathy for the devil", *Wingspan*, Winter 2011.

7. Paul Callaghan, "Mynas not major worry", *Lakes Mail*, 23 February 2012.

Article in *The Newcastle Herald*: "Revealing the truth about Indian mynahs" by Stephen Williams.

Together Today

The Herald is a proud supporter of Together Today.

RISE

MALIGNED: PhD student Kathryn Haythorpe holding an Indian mynah, left; a native noisy miner, below.

Revealing the truth about Indian mynahs

Kathryn Haythorpe may be proving that these loathed birds are more sinned against than sinning.

How did you become interested in Indian mynahs?
I have always been fascinated by Indian mynahs – they are clearly highly intelligent and doing very well for themselves, and I began to wonder how they have managed to achieve the success they have, and whether it comes at the expense of our natives.

What are the usual assumptions about Indian mynahs and how did they come about?
Most people view Indian mynahs as bullies that aggressively drive out native species. I believe it is likely that Indian mynahs have achieved this reputation because they are a bird that people love to loathe and want justification to eradicate. It is also very likely they are being confused with noisy miners, an aggressive native species.

What led you to think this assumption might need testing?
Despite anecdotal records, and plenty of opinion, I found very little scientific evidence to justify the aggressive reputation of Indian mynahs. The behaviours reported were often no worse than what any bird would do (defending their nests). So I began to wonder: how would Indian mynahs actually compare with natives in terms of aggressiveness?

How are you conducting your research?
I have been filming competitive interactions over food resources to determine which bird species show aggression and if Indian mynahs prevent others from feeding. I have also just begun experiments looking at nesting behaviour. This involves filming nestboxes using motion-detector cameras provided by the Tom Farrell Institute at the university.

What are your preliminary results showing?
So far we have found that Indian mynahs are not one of the most aggressive birds. They are usually beaten by native birds like magpies and noisy miners.

We also found they had no effect on which other birds accessed a food resource. This suggests that Indian mynahs are not affecting the ability of native species to feed.

Behavioural data also indicates that Indian mynahs are rarely seen in aggressive encounters with other birds, while noisy miners are frequently seen attacking other birds.

Some people regularly feed wild birds in their garden. Is this a good idea?
In strict truth the answer is no. If you must feed wild birds, try to read up on what foods are suitable for the particular species.

Bird feeding products are available and are generally a better option than bread. Also don't overfeed. Birds need to learn to forage for themselves, not rely on you.

And of course, be aware that putting out food may generate interest from Indian mynahs.

Kathryn Haythorpe is a PhD student at the University of Newcastle.

Two birds: some minor and major differences

- Indian mynahs were introduced to parts of Victoria, NSW and Queensland in the 1860s to control pest insects. They have since thrived and occupy ever-increasing ranges.
- Noisy miners are native birds known for their loud calls and "mobbing" behaviour. They are protected in NSW and it is illegal to kill them.
- Despite similar sounding names, noisy miners and Indian mynahs are not related – Indian mynahs are sturnids and are related to starlings, while noisy miners are honeyeaters.
- Both noisy miners and Indian mynahs thrive in suburban and urban habitats, although noisy miners tend to prefer areas with flowering vegetation.
- Noisy miners are ⅓ to ½ the weight of Indian mynahs because the bodies of noisy miners are specialised to gather food mostly from trees, and Indian mynahs are specialised to forage on the ground.

Noisy miners

swilliams@theherald.com.au

Responses to above article in the Opinions section of *The Newcastle Herald* by Sean Farnham, Chris Skeet, and myself.

HERALD

Avian home invaders

April 11, 2011, 11:37 p.m.

KATHRYN Haythorpe ("Revealing the truth about Indian mynahs" Herald 11/4) is going to get an unpleasant shock watching the nesting habits of these birds. In my nesting boxes, Eastern Rosellas are almost always kicked out of their nests by the mynahs. We also have a distinct lack of kookaburras, despite swathes of lawn. This is almost certainly due to Indian mynahs taking their nests or the flocks of feral mynahs, starlings and blackbirds eating the kookaburras' food.

The natives have spent a long time successfully competing with the admittedly aggressive noisy miners but just can't compete with the introduced species.

Sean Farnham

Kurri Kurri

Kathryn Haythorpe ("Revealing the truth about Indian mynahs" *Herald* 11/4), obviously has never seen Indian mynahs taking baby rosellas from their nests and killing them. There should be a bounty on Indian mynahs. They are introduced, aggressive and should be eradicated.

Chris Skeet, Whitebridge

HERALD

Minor or major problem

April 14, 2011, 11:44 p.m.

SEAN Farnham and Chris Skeet (Letters 12/4) mention some interesting observations about Indian mynahs evicting rosellas from their nests. My PhD study is focusing on aggressive behaviour from Indian mynahs towards natives in various situations. So far they do not appear to show significant aggression over food.

However, I am presently moving on to a study of nesting behaviour, particularly in regards to rosella-mynah competition, where anecdotal records show evidence of displacement ability. I would be interested to hear from anyone who has witnessed Indian mynah interference at nests: please email me at nestboxstudy@yahoo.com.au.

It is important that we understand all facets of the behaviour of this introduced species, both positive and negative, to best protect our wildlife.

Kathryn Haythorpe

Ourimbah

Blog article by Jeff Corbett in *The Newcastle Herald*.

HERALD

A trapped Indian myna

By Jeff Corbett
April 27, 2011, 7:02 a.m.

Four now, a sight to warm a myna hater's heart.

What is killing the Indian myna flapping about in a trap in my backyard now going to achieve? Yes, I meant to catch it, in a myna-specific trap I made, and I've left it in there to act as a call bird for the other members of its family, or gang as I prefer to see them. But if I don't entice any more into the trap should I let the solitary myna go? And if there is no point in killing the one myna, what point can there be in killing all members of that particular tribe?

I suppose killing all members of the family that lives in palm trees near my backyard would rid my backyard of the strutting rats-on-wings but it would be only brief respite. I read, too, that Maitland City Council is trialling an intensive trapping program aimed at suppressing myna numbers, and unfortunately it, too, can at best offer only brief respite.

I'm not seeking to defend mynas. I don't like them, at all, but I do question whether that alone justifies my trapping and killing them. Yes, I should have pondered this before I built the trap - using, by the way, a design on the Canberra Indian Myna Action Group's website - and before I set it on my chookpen roof and baited it with pet food. The truth is, though, that I didn't expect to catch any and, I have to admit, that I was spurred by the same sense of excitement that drives boys to set traps for anything. I didn't expect to catch any because since a myna-hating friend dragged the first myna I caught from the trap two years ago and wrung its neck the local mynas have been healthily wary of my trap.

In the Herald a couple of weeks ago I read a woman studying mynas for a PhD, Kathryn Haythorpe, saying that people loved to loathe mynas and that they accepted stories about mynas killing native birds as justification for killing them. As you may have read two readers wrote in protest to tell of their witnessing mynas killing rosella fledglings or destroying rosella eggs, and my neck-wringing friend, Al, has seen mynas removing and destroying rosella eggs and fledglings from nesting boxes in his Kahibah backyard many times.

"In the Herald a couple of weeks ago I read a woman studying mynas for a PhD, Kathryn Haythorpe, saying that people loved to loathe mynas and that they accepted stories about mynas killing native birds as justification for killing them. As you may have read two readers wrote in protest to tell of their witnessing mynas killing rosella fledglings or destroying rosella eggs..."

But killing the myna in my trap, and even all its family members, won't save a rosella. Maybe I could justify the killing as rightful retaliation, but is there such a thing as right and wrong in the bird world? The old line about backing natives against the introduced might help, but after 150 years isn't the myna as legitimate a member of our landscape as any other bird?

Yes, I should have sorted these issues before I set the trap, but they do have an urgency now that a myna is flapping about in the trap. And yes, I have given it water and food.

Transcript of radio interview with ABC Newcastle.

ABC Newcastle (Newcastle)
Drive - 2/05/2011 5:44 PM
Paul Bevan

Audience	
Male 16+:	4,000
Female 16+:	3,000
All People:	6,000
Advertising Space Rate	
AUD:	2,860

Regular Segment: Show Us Your PhD Bevan talks to Katherine Haythorpe, PhD Student, University of Newcastle about her PhD on Indian Myna birds. Haythorpe says she found Indian Mynas are actually not very aggressive over food resources to each other, nor to other types of birds. Haythorpe says that the noisy minor is native to Australia, and is protected so it is illegal to kill them. Haythorpe says that the Indian Myna is brown, whereas the native bird is grey. Haythorpe says the Indian Myna was introduced to Australia in order to control insects, but has since failed to do so. She talks about how she did her food aggression study. Haythorpe says the reason behind her research is that she wants to find out exactly what impact Indian Mynas are having on native birds, and what can be done to minimise their impact.

Interviewees: Katherine Haythorpe, PhD Student, University of Newcastle
Duration: 13.25
Summary ID: W00043545768
This program or part thereof is syndicated to the following 1 station(s):-
ABC Upper Hunter (Muswellbrook)

Article published in *The Newcastle Herald*: "Home to roost in suburbia" by Damon Cronshaw.

HERALD

Home to roost in suburbia

Author: By DAMON CRONSHAW
Date: 07/05/2011
Words: 284
Source: NCH

Publication: Newcastle Herald
Section: News
Page: 26

A NEWCASTLE University study will establish ways to discourage Indian myna birds from nesting in suburban gardens.

The study will determine methods to encourage native birds back into yards.

Lake Macquarie City Council will give $7000 to the university for the study.

A council report said the birds cause "considerable noise" and leave large amounts of bird poo below roosts.

"They disperse the seed of exotic weeds and are aggressive towards other bird species in urban areas," the report said.

The study, to be undertaken by university PhD student Kathryn Haythorpe, will use motion-detector cameras to examine bird behaviour.

Ms Haythorpe said the aim of the project was to investigate the nesting behaviour of Indian mynas, including whether they aggressively compete with natives for nesting sites.

"I hope to look at developing strategies to help protect native species at nest sites," she said.

Newcastle University animal behavioural scientist Andrea Griffin runs a separate Indian myna research project.

Dr Griffin said Indian mynas were "certainly the most abundant urban species around".

She could not say whether the birds were a problem for society.

"It depends what you define as a pest," she said. "My main focus is on learning and cognition.

"I'm interested in their brains, the fact they are a very successful species and understanding the behaviours that allow them to be so successful."

Dr Griffin said the birds were good at social learning and "exploiting novel food sources", such as bins.

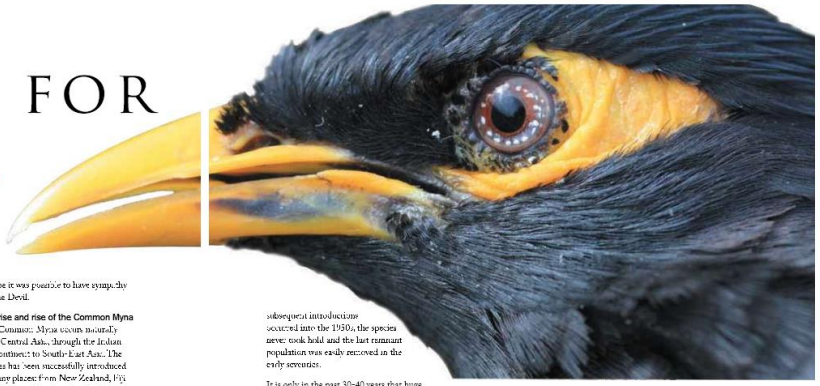
"Some of my experiments have demonstrated Indian mynas learn to avoid traps from watching others get trapped," she said.

Dr Griffin said she was researching ways to control the birds, which were "more ethical than killing them".

"Contraception is one option," she said.

Article published in *Wingspan*: "Sympathy for the Devil" by Sean Dooley.

SYMPATHY FOR THE DEVIL



Just how bad is the Common Myna?

By Sean Dooley

"Send these Indians Back" screamed the headline on the blog of one of Australia's most controversial columnists. It turns out that the story was not about tourists overstaying their visas or foreign students behaving badly; the Indians referred to were Common Mynas.

That this particular tabloid commentator chose to use the old name Indian Myna suggests he had an agenda other than natural history in mind; the headline "Send these Commons Back" would clearly not have had the effect he was seeking, and judging by the flood of racist comments posted on the blog (the most offensive of which were soon removed) it is clear that the scribe succeeded. Yet also in the comments were dozens of people asserting that Common Mynas didn't belong here. The birds were condemned as noisy, brash and annoying; agricultural pests that spread disease and out-compete native species, displacing hollow nesting birds and maliciously killing the young.

This is not just the opinion of a few red-necked readers; such views are part of birdwatching orthodoxy. The International Union for the Conservation of Nature has classified the species as one of the World's 100 Worst Pests. The Canberra Indian Myna Action Group Inc, (CIMAG), who in the past five years have created one of the most successful community environmental groups in our history, state on their website that Common Mynas "are one of the most invasive animal species in the world".

I have to admit that I have firmly been in this camp too, but the blog made me reflect: are Common Mynas really as bad as I've always believed? Maybe my negative attitude towards this species has been little more than ornithological xenophobia.

Maybe it was possible to have sympathy for the Devil.

The rise and rise of the Common Myna

The Common Myna boom recently from Central Asia, through the Indian Subcontinent to South-East Asia. The species has been successfully introduced in many places: from New Zealand, Fiji and Solomon Islands to South Africa, Hong Kong, the Middle East and, of course, Australia.

The first Australian release was in Melbourne in 1863 with other cities such as Sydney following suit. Common Mynas from Melbourne were introduced into North Queensland in the 1880s to control insect pests in the cane fields, and like that species it was spectacularly unsuccessful as a biological pest control agent. Around WWI it was introduced into the Darling Downs for similar reasons. Whilst again not so effective at controlling pests, these birds flourished enough to provide the basis for the species' conquest of Brisbane more than half a century later.

Common Mynas were released in Canberra as late as 1968-71 when, the story goes, an elderly resident of a retirement home in Forrest had family members bring 110 birds down from Sydney, as their singing was a reminder of a childhood spent there. By 1991 it was estimated that there were still only 1,200 birds in the ACT, yet in 2005, before CIMAG began their trapping campaign, estimates were at around 90,000.

This slow build leading to a sudden boom is reflected in many areas. Contrary to its image as an aggressive invader, the Common Myna is naturally a sedentary species and not much of a coloniser. In fact, several introductions failed in places like Tasmania, and in Adelaide, where subsequent introductions occured into the 1950's, the species never took hold and the last remnant population was easily removed in the early seventies.

It is only in the past 30-40 years that huge expansion of their range and numbers has occurred. It took Common Mynas almost 150 years to make the relatively short hop from Melbourne to Ballarat, yet such has been the boom in myna numbers in recent years, that wildlife authorities in Victoria and South Australia are investigating how to keep the westward expansion in check.

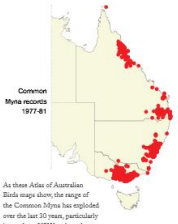
So what has changed? The essential nature of the species hasn't undergone any recent alteration, however the way we use the land has. Over the past few decades we have created ideal conditions for Common Mynas. In the post-war years, the rapid development of suburban sprawl facilitated the outward spread of Common Mynas from the inner-city, industrial suburbs they once almost exclusively favoured. With vast swathes of lawns and roadsides, the outer suburbs were perfect incubators for a thriving myna population.

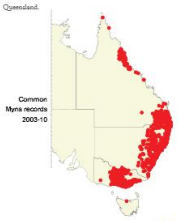
Adding to this was the more recent expansion of freeway networks between major urban centres. With their mown, grassy verges, fast food joints and resultant roadside litter, we have essentially created connectivity to areas of similar urban habitat. This is particularly evident in Victoria where towns like Ballarat, Bendigo and Wangaratta, all major centres that have had highway upgrades to Melbourne within the last two decades, have been swamped by Common Mynas in recent years.

The worst of the worst?

There is surprisingly little scientific evidence on the environmental damage that Common Mynas inflict; most of our opinions on these birds are based on anecdotal evidence. For instance, there were comments on the aforementioned blog from people adamant that Common Mynas had taken over from birds such as Rainbow Lorikeets. These observations were in contrast to my own experience over the past decade, where I have seen Rainbow Lorikeets move into inner city Melbourne neighbourhoods like Carlton and take over Common Myna roosts in palm trees. Assuming that that the observations in the blog comments were accurate, it may be that the drop-off in numbers of native birds was as much a result of local habitat alteration (clearance of local bushland and hollow-bearing vegetation) as due to the arrival of mynas.

I have also noticed whilst birdwatching in the south and east of Melbourne that the very species that are supposed to be suffering the most at the hands of the dastardly mynas—lorikeets, Eastern Rosellas and Red-rumped Parrots—seemed to be doing quite well, even increasing in number. While Atlas data dramatically shows expanding range of the Common Myna, (see maps, right) there is insufficient analysis as to how native species are faring where mynas are present.





As these Atlas of Australian Birds maps show, the range of the Common Myna has exploded over the last 30 years, particularly in northern NSW and southern Queensland.

Main image: The evil eye—the Common Myna has failed to endear itself to the majority of Australians. Photo by Andrew Silcocks

Some valuable research has come out of Canberra. A study in the '90s by Stewart Pell and Chris Tidemann found that in two ACT nature parks Common Mynas successfully excluded native birds from all nesting sites in the area, even though they themselves nested in only a few sites. Perhaps even more persuasive evidence will be forthcoming with the work of PhD student Kate Grarock, who is studying the effects of Common Myna removal carried out by CIMAG in 15 Canberra suburbs. CIMAG is at the forefront of efforts to control Common Mynas and Kate's study will compare control suburbs where no mynas are being removed, with others that have both a high and medium level of trapping. Early indications are that in the suburbs where Common Mynas are removed there is a marked return of species such as Crimson Rosella.

Cane Toads with wings

Even with some evidence emerging that Common Mynas do have an effect, we need to ask just how bad their impact is, particularly when compared to that of other species. Noisy Miners (a native honeyeater totally unrelated to Common Mynas), have time and again been shown to drive out almost all other species from their territories. The Noisy Miner occurs over a far more extensive area than the Common Myna, and in far more natural habitats. In contrast, nowhere in its range has the Common Myna penetrated deeply into timbered habitat; it would be hard to make a case therefore that they threaten the viability of even one species. Noisy Miners represent a threatening process to several threatened species such as Grey-crowned Babbler and Regent Honeyeater, yet there hasn't been a similar groundswell of antipathy to the native species amongst the broader community.

Founding president of the Invasive Species Council, ecologist Barry Traill, says that "there are just some species that people get worked up about." He doesn't believe that Common Mynas should be a high priority for conservation work because, "they are street kids, not bush folk—they like urban and semi-urban areas, roadsides and places like chook farms where there is a lot of rubbish and dropped food scraps," he says.

Traill also believes that mynas may not be as aggressive as we think: "I certainly don't see that they are aggressive to any problematic degree away from nest hollows" he adds, observing that they don't compete for food to any serious degree with any native species. Traill notes that while they do use tree hollows and nest boxes for nesting, he doesn't think competition for nest hollows is likely to seriously affect any populations of native bird species.

Barry's opinion is backed up by an observational study by Kathryn Haythorpe from the University of Newcastle, which suggests that the aggressive nature of Common Mynas has been over-exaggerated, at least when it comes to food gathering behaviour. Further research at Newcastle by Andrea Griffin, Marie Diquelou and others expounds upon myna intelligence, and leads to the conclusion that it may well be that Common Mynas aren't bullying our native birds, so much as outsmarting them.

"Barry's opinion is backed up by an observational study by Kathryn Haythorpe from the University of Newcastle, which suggests that the aggressive nature of Common Mynas has been over-exaggerated, at least when it comes to food gathering behaviour..."

As to their reported ruthless killings of nestlings of other species, I would argue that this is not unique to Common Mynas. Kookaburras, currawongs, ravens and butcherbirds are well known nest robbers, and I imagine inflict far greater damage than the Common Myna. Even seemingly sweet birds such as Grey Shrike-thrush take a substantial toll of nestlings in places like the rainforests of Lamington National Park, while in Tasmania at the last breeding site for the Orange-bellied Parrot, the main nesting competitor is the humble Tree Martin, which has been observed forcibly ejecting OBPs from suitable hollows. Yet once again, there is no campaign against these natives.

Common Mynas have gained such an unsavoury reputation because they occur

Article published in *The Lakes Mail*: "Mynas not major worry" by Paul Callaghan

LAKES MAIL news

Mynas not major worry

Research finds invaders are harshly judged

By PAUL CALLAGHAN

INDIAN mynas have copped a bad rap, a Newcastle University researcher has found.

The Indian or common myna might not be such a villain after all, and native birds display more aggression than mynas when competing for food.

The researcher, PhD student Kathryn Haythorpe, is investigating the common mynas' behaviour and has completed a scientific field study of the birds in the Newcastle region.

Lake Macquarie City Council granted the university $6784 for her study into finding management solutions for the mynas and whether their numbers were expanding in the Hunter.

"Common mynas certainly do seem to be on everyone's radar," Ms Haythorpe said.

Her research is not complete, and she is yet to study the nest competition and usurping behaviour of the birds, but Ms Haythorpe said the mynas' ability to out-compete other bird species for food resources was minimal.

"They (mynas) do not display as much aggression as they are often attributed with," she said.

"In fact, there are native birds that are also doing substantial, if not more damage, such as noisy miners. And birds such as currawongs, kookaburras and magpies display a lot more aggression.

"For sure, common mynas are a problem but I think their tendency to carry out their lives in such close proximity to humans and their status as introduced animals makes us particularly sensitive to any potentially damaging behaviours and likely to overestimate their actual impact."

She said myna attacks on other birds were noticeable because they occurred in suburban backyards.

"Another species may display much greater aggression that goes unseen by us because they live in the bush," she said.

Ms Haythorpe's findings follow a *Lakes Mail* story on February 2 about Sydney's Garry Cunich who is waging war on Indian mynas with his specially designed traps.

Mr Cunich said several councils had bought his traps to hire for public use.

Among them was The Hills Shire which has 20 traps for hire, and a waiting list.

Ms Haythorpe said trapping was only a short-term solution.

"Ultimately if the habitat is suitable for them they will keep coming back," she said.

RESEARCH: Kathryn Haythorpe said Indian mynas did not display as much aggression as the public tended to attribute to them. – *Picture by Peter Morton*

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Does Human-Induced Habitat Modification Influence the Impact of Introduced Species? A Case Study on Cavity-Nesting by the Introduced Common Myna (*Acridotheres tristis*) and Two Australian Native Parrots

Kate Grarock · David B. Lindenmayer · Jeffrey T. Wood · Christopher R. Tidemann

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Abstract Introduced species pose a major threat to biodiversity across the globe. Understanding the impact of introduced species is critical for effective management. Many species around the world are reliant on tree cavities, and competition for these resources can be intense: threatening the survival of native species. Through the establishment of 225 nest boxes, we examined the relationship between tree density and the abundance and nesting success of three bird species in Canberra, Australia. The common myna (*Acridotheres tristis*) is an introduced species in Australia, and the crimson rosella (*Platycercus elegans*) and eastern rosella (*Platycercus eximius*) are native species. We then investigated the impact of common myna nest box occupation on crimson rosella and eastern rosella abundance. Tree density significantly influenced the abundance and cavity-nesting of all three species. Common myna abundance (birds per square kilometer) was greatest at low tree density sites (101.9 ± 22.4) and declined at medium (45.4 ± 10.1) and high (9.7 ± 3.6) tree density sites. The opposite pattern was observed for the crimson rosella, with greater abundance (birds per square kilometer) at high tree density sites (83.9 ± 9.3), declining over medium (61.6 ± 6.4) and low (31.4 ± 3.9) tree density sites. The eastern rosella was more abundant at medium tree density sites (48.6 ± 8.0 birds per square kilometer). Despite the strong influence of tree density, we found a significant negative relationship between common myna nest box occupancy and the abundance of the crimson rosella ($F_{1,13} = 7.548$, $P = 0.017$) and eastern rosella ($F_{1,13} = 9.672$, $P < 0.001$) at some sites. We also observed a slight increase in rosella nesting interruptions by the common myna at lower tree densities (high: 1.3 % ± 1.3, medium: 6.6 % ± 2.2, low: 12.7 % ± 6.2), although this increase was not statistically significant ($F_{2,40} = 2.435$, $P = 0.100$). Our study provides the strongest evidence to date for the negative impact of the common myna on native bird *abundance* through cavity-nesting competition. However, due to the strong influence of habitat on species abundance and nesting, it is essential to investigate the impacts of introduced species in conjunction with habitat variation. We also suggest one component of introduced species management could include habitat restoration to reduce habitat suitability for introduced species.

Keywords Competition · Exotic species · Indian myna · Hollow · Pest management · *Sturnus tristis*

K. Grarock (✉) · D. B. Lindenmayer · J. T. Wood · C. R. Tidemann
Fenner School of Environment and Society, The Australian National University, Canberra, ACT 0200, Australia
e-mail: kate.grarock@gmail.com; kate.grarock@anu.edu.au

D. B. Lindenmayer
e-mail: david.lindenmayer@anu.edu.au

J. T. Wood
e-mail: jeff.wood@anu.edu.au

C. R. Tidemann
e-mail: chris.tidemann@anu.edu.au

K. Grarock
Invasive Animals Cooperative Research Centre, University of Canberra, Canberra, ACT 2617, Australia

D. B. Lindenmayer
Australian Research Council Centre of Excellence for Environmental Decisions, Canberra, ACT, Australia

D. B. Lindenmayer
National Environmental Research Program, Canberra, ACT, Australia

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Introduction

Introduced species and habitat modification pose major threats to biodiversity across the globe (Clavero and García-Berthou 2005; Levin and Crooks 2011; Pimentel and others 2005; Ruhren 2012; Westphal and others 2008). Introduced species can affect native species through predation, competition, herbivory, habitat alteration, disease, and hybridization (Davis 2003; Gurevitch and Padilla 2004; Kumschick and Nentwig 2010; Nentwig and others 2010; Pimentel and others 2005; Ruhren 2012). However, some introduced species can have a devastating impact while others are relatively benign (Davis and others 2011; Shine 2010). Furthermore, an introduced species may have a significant negative impact in one environment and little or no impact in another (Davis and others 2011).

Demonstrating the impact of an introduced species is a complex task, especially when it occurs alongside human-habitat modification (Didham and others 2005; Gurevitch and Padilla 2004; MacDougall and Turkington 2005). Understanding the different impacts on native species, and the interactions between these impacts, is critical for effective native species conservation (Davis and others 2011). For example, introduced species management may not assist threatened species recovery if habitat destruction is the major cause of species decline (Didham and others 2005). Additionally, understanding the impact of a species is essential for managing limited conservation resources (Kumschick and others 2012).

Species distribution and abundance is predominantly determined by resources that are critical for their survival (Elton 1927). Therefore, habitat features can have a large impact on species abundance and distribution (Bradshaw and others 2007; Clergeau and others 1998; Crooks and others 2004; Gardali and Holmes 2011; Munro and others 2009). For example, the availability of tree cavities can be a critical resource for some species (Aitken and Martin 2008; Gibbons and others 2002; Goldingay and Stevens 2009; Newton 1994; Wiebe 2011).

Human modification of landscapes (e.g., habitat clearing, tree removal for public safety) and fire can lead to reductions in cavity availability (Harper and others 2005a; Newton 1994; Wiebe 2011). This reduction can then limit the breeding success and the abundance of native species (Aitken and Martin 2008; Brazill-Boast and others 2010; Wiebe 2011), especially for species that cannot excavate their own cavities (Goldingay and Stevens 2009). In some landscapes, removal of trees with cavities may exceed natural replenishment (Goldingay and Stevens 2009; Lindenmayer and others 2012). As such, there is widespread concern about tree cavity decline, threatening the survival of numerous cavity-nesting species (Goldingay and Stevens 2009; Harper and others 2005a; Lindenmayer and Wood 2010; Newton 1994; Wiebe 2011).

Nest cavity availability can be further reduced by the introduction of new species that compete for these limited resources (Czajka and others 2011; Newson and others 2011; Newton 1994; Strubbe and Matthysen 2009; Wiebe 2011). For example, in the Netherlands, the introduced common starling (*Sturnus vulgaris*) can dominate nest cavities and reduce the population density of great tits (*Parus major*) (van Balen and others 1982). Similarly, the introduction of the ring-necked parakeet (*Psittacula krameri*) in the United Kingdom is believed to cause reductions in nuthatch (*Sitta europaea*) abundance (Strubbe and Matthysen 2009).

In Australia, competition for nest cavities can be especially intense (Gibbons and Lindenmayer 2002). Over 300 species in Australia depend on tree cavities (Gibbons and Lindenmayer 2002) and the development of new cavities takes many years (Lindenmayer and others 2011). No Australian species' create tree cavities, so cavity development is dependent on slow-acting processes of rot and decay (Lindenmayer and others 2000; Lindenmayer and others 2003; Mackowski 1984; Saunders 1979). In conjunction with slow replenishment, existing cavities are often destroyed through human-habitat modification (Harper and others 2005a; Soderquist and Mac Nally 2000). As such, competition from introduced species can have substantial impacts on native cavity-dependent taxa (Gibbons and Lindenmayer 2002; Lindenmayer and others 2009). Despite this, most research on cavity-nesting competition comes from the Northern Hemisphere (Brazill-Boast and others 2010; Czajka and others 2011; Goldingay and Stevens 2009; Newson and others 2011; Newton 1994; Strubbe and Matthysen 2009; van Balen and others 1982; Wiebe 2011).

One introduced species believed to compete with native species for cavity-nest sites is the common myna (*Acridotheres tristis*) (Dhami and Nagle 2009; Harper and others 2005b; Pell and Tidemann 1997a, b). The species has been listed as one of the world's worst invasive species (ISSG 2000). There is global concern that the common myna displaces native species through competitive domination of nest cavities, displacing birds from nest sites and destroying eggs (Byrd 1979; Dhami and Nagle 2009; Feare and Craig 1998; Harper and others 2005b; Pell and Tidemann 1997a, b; Watson and others 1992). The common myna can outcompete the native crimson rosella (*Platycercus elegans*) and eastern rosella (*Platycercus eximius*) in aggressive encounters during the breeding season in Canberra, Australia, potentially reducing their breeding success (Pell and Tidemann 1997b). Harper and others (2005b) found the common myna occupied 37.5 % of nest boxes over a breeding season from late September to March in Melbourne, Australia, possibly limiting the availability of

nesting sites for native species. The common myna can also build nests in multiple cavities, potentially deterring native species from using them (Pell and Tidemann 1997b).

Despite the above examples, there is no conclusive scientific evidence that domination of nest cavities by the common myna reduces the *abundance* of native species. Additionally, our previous research indicates that tree density may influence the abundance and impact of the common myna (Grarock and others 2013).

Many studies have found a strong influence of habitat suitability on species abundance (Didham and others 2005; MacDougall and Turkington 2005; Parsons and others 2006; Ruhren 2012). The common myna is abundant in modified urban landscapes and tends to avoid high-density native woodland areas (Grarock and others 2013). Therefore, in low tree density areas the common myna may become abundant and compete for resources with native species.

Our study investigated the influence of cavity-nesting of the introduced common myna on two native parrot species, the crimson rosella and eastern rosella. Due to the influence of habitat on species abundance, we initially examined variation in the abundance, levels of cavity occupancy and nesting success of these three species, across areas characterized by different tree density. We then investigated the impact of common myna nest box occupation on crimson rosella and eastern rosella abundance. To conduct our investigation, we used artificial nest boxes as a proxy for natural cavities. Artificial nest boxes provide a standardized and repeatable measure of cavity-nesting that can be representative of natural cavity nest use (Beyer and Goldingay 2006).

We developed a series of hypotheses (Table 1). Broadly, we hypothesized that tree density (high, medium, and low) would influence the abundance, rate of cavity-nesting, and nesting success of our three study species. We also hypothesized that common myna nest box occupancy would have a negative impact on the abundance of the crimson rosella and eastern rosella at low tree density sites.

Materials and Methods

Survey Sites

We selected survey sites around Canberra, Australia to investigate cavity-nesting occupancy by the common myna and native rosella species. Each site was located in a nature reserve next to an urban area (residential suburb). Nature reserves ranged from dense woodlands to open grassy woodlands and were dominated by *Eucalyptus* species. Sites extended for 250 m into reserves and followed the suburb edge for 1 km (Fig. 1). We insured adjoining suburbs had been constructed more than 20 years ago so the vegetation was well established. Sites were located at least 2 km apart as the common myna rarely travels further than 2 km from its territory (Dhami and Nagle 2009; Feare and Craig 1998). These criteria limited the potential study sites to 23.

We further refined sites based on tree density, due to the influence of habitat on species abundance (Didham and others 2005; MacDougall and Turkington 2005; Parsons and others 2006; Ruhren 2012). We estimated tree density in the 23 potential sites by walking a 1-km transect through nature reserves and scoring the vegetation cover at 20 m intervals (Fig. 1). We allocated one point for tree cover overhead or zero points for no tree cover. We selected the five sites with the highest vegetation score (>40), the five sites with a medium vegetation score (between 31 and 36) and the five sites with the lowest vegetation score (<26) (Fig. 1). We used an analysis of variance (ANOVA) to test if mean nature reserve tree density varied significantly among the three categories (high, medium, and low) (Fig. 1a).

Nest Box Surveys

We constructed 225 nesting boxes from 15 mm plywood. Boxes had an internal volume of 19 l, and were fitted with a 65-mm diameter entrance hole in the front panel. In July–August 2008, we established 15 nesting boxes in each of the 15 sites. The crimson rosella, eastern rosella and common myna are known to prefer cavities with an entrance diameter of 50–80 mm but will use up to 120 mm (Goldingay and Stevens 2009).

We randomly selected nest box placement by dividing each site into 5 m grid cells. Grid cells were numbered from one to 10,000 and we used a random number generator to select the grid cell location for each nest box. Nest boxes were placed in the tree closest to the center of a selected grid cell. If there were no trees located in the grid cell, we placed the nest box in the closest tree in any direction. Nest boxes were only placed in *Eucalyptus* tree species that had a diameter greater than 20 cm at breast height. We never placed more than one box in a single tree. However, on occasion (if randomly selected), boxes were placed in trees within 5 m of one another. Nest boxes were mounted 3–4 m above the ground on the southern side of trees to insure they were protected from the summer sun.

To check nest boxes we used a bullet-camera surrounded by five light-emitting diodes (32 mm in diameter), that allowed color viewing in total darkness. We mounted the camera on the end of a 3-m pole and connected it to a video camera via a 5-m coaxial cable. To check nest boxes, we placed the camera at the entrance hole of each box, viewing and recording the images on a video camera (Fig. 2). If no

Table 1 Hypotheses for our study on cavity-nesting occupation of the introduced common myna (*Acridotheres tristis*) and the native crimson rosella (*Platycercus elegans*) and eastern rosella (*Platycercus eximius*) in Canberra, Australia

	Literature/rationale	Hypothesis
1	Studies indicate that common myna abundance and nesting are strongly influenced by habitat, with numbers and nesting success increasing as tree density declines (Crisp and Lill 2006; Grarock and others 2013; Lowe and others 2011; Pell and Tidemann 1997a, b; Tracey and others 2007; White and others 2005). However, many native bird species are more abundant and exhibit increased nesting success in dense woodland (Blair 2001; Case 1996; Clergeau and others 1998; Crooks and others 2004; Deng and Gao 2005; Gardali and Holmes 2011; Munro and others 2009; Newton 1994; Sewell and Catteral 1998)	We hypothesized that tree density (high, medium and low) would influence the abundance, rate of cavity-nesting and nesting success for the common myna, crimson rosella and eastern rosella. Specifically, we hypothesized that common myna abundance, rate of cavity-nesting and nesting success would be greater in low tree density sites than in medium and high sites, while crimson rosella and eastern rosella abundance, rate of cavity-nesting, and nesting success would be greater in high tree density sites
2	Artificial nest boxes provide a standardized and repeatable measure of cavity-nesting that can be representative of natural cavity nest use (Beyer and Goldingay 2006; Pell and Tidemann 1997b). Because we used nest boxes as a proxy for natural cavities, we wanted test whether bird abundance was related to nest box occupancy. We used correlations between species nest box occupancy and abundance to determine whether observed trends in nest box occupancy were related to species abundance	We hypothesized that there would be a significant positive relationship between species abundance and nest box occupancy, with greater nest box occupancy by each species in areas where they are more abundant
3	Research suggests that rosella species tend to prefer natural cavities, while the common myna will readily use artificial cavities (Lowe and others 2011). Therefore, we avoided directly comparing nesting rates of common myna to nesting rates of rosella species as these relationships might have been due to preferences for natural cavities over nest boxes. Instead we compared common myna nest box occupancy with rosella abundance. Due to the potentially confounded relationship between species abundance and tree density we investigated the relationship between common myna nest box occupancy and rosella abundance separately over each of the three tree density categories	We hypothesized that there would be a negative impact of common myna nest box occupancy on the abundance of the crimson rosella and eastern rosella at low tree density sites
4	The impact of an introduced species can often increase in severity as it becomes more abundant (Choquenot and Parkes 2001). Therefore, if nest cavities are limited, increases in common myna numbers could lead to greater competition for cavities with native species	We hypothesized that in areas where the common myna is abundant a greater proportion of native birds would be evicted from nest boxes

bird was present, or an adult bird was on the nest potentially obscuring eggs or chicks, we monitored the box for a period of 5–10 min from approximately 20 m away. This enabled us to identify birds when they returned to the box or the number of eggs/chicks present if the adult vacated the box. The procedure was fast and effective, resulting in minimal disturbance to nest box occupants and enabled us to identify the species and the number of eggs in each nest box.

Nest boxes were checked every 4 weeks throughout the breeding season (October–March), over 3 years (2008–2009, 2009–2010, and 2010–2011). The breeding season for many species in Canberra starts later than in other areas of Australia due to the altitude (approximately 605 m above sea level) and comparatively cold weather (Gibbs and others 2011; Lenz 1979). Before each nesting season, we checked and replaced damaged boxes.

We classified nest boxes as 'occupied' if a bird was observed using the box with nesting material and/or eggs at some stage during the breeding season. We classified an egg as 'successful' if it produced a chick that hatched successfully (i.e., no evidence of a dead chick in, or surrounding, the box). However, due to the 4-week survey schedule, on some occasions it was difficult to determine if the eggs hatched successfully as we either did not observe the eggs or the chicks fledged before we returned. Therefore, in the analysis we only included data where we observed eggs on one visit and then observed chicks or unsuccessful eggs (e.g., broken or abandoned), in the following visit. We classified a nesting attempt as 'interrupted' if, after initial inspection, a box was 'occupied' by a native species and the following inspection revealed that the common myna had taken over the box. However, we were careful to avoid identifying an interruption as occurring if the initial species had chicks that were close to fledging and, therefore, may have vacated the box prior to the common myna using it. Using this conservative approach, it is likely that we underestimated the number of native species that were interrupted by the common myna.

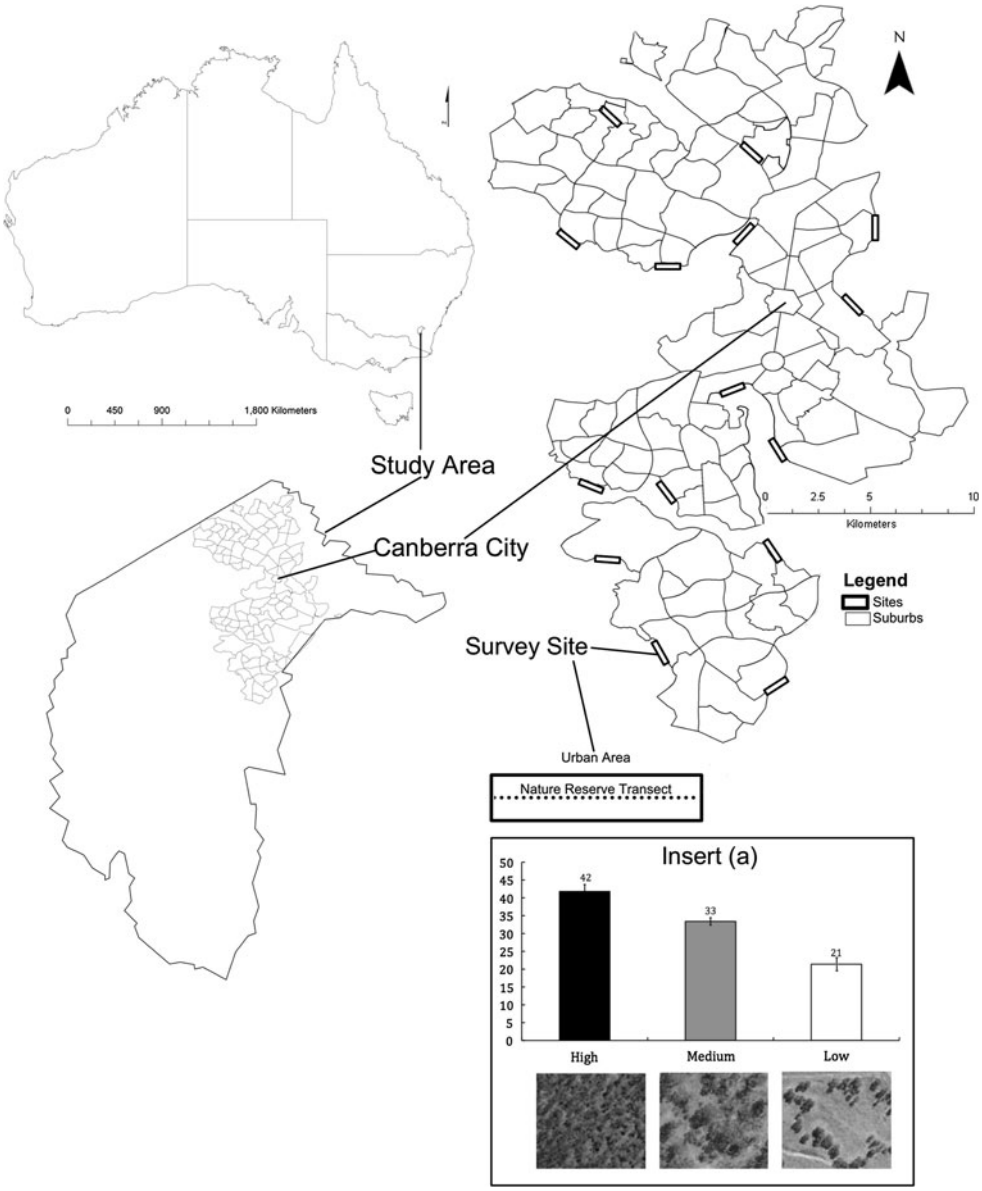

Fig. 1 Study area and location of the 15 survey sites within nature reserves surrounding Canberra, South East Australia. At each site we randomly placed 15 nest boxes and set up a 1-km long and 100-m wide line transect survey. We scored vegetation cover at 20-m intervals along each transect. We then categorized nature reserved as having high, medium or low tree density (± SE) (see inset *a*). Tree density varied significantly among tree categories ($F_{2,12} = 12.5$, $P = 0.001$)

Bird Abundance Surveys

At each of our 15 sites we established a transect survey that was 1 km in length and 100 m wide (Fig. 1). We surveyed bird abundance every second month in the breeding season (November, January, March) from November 2008 to March 2011. Bird observers with more than 20 years bird watching experience identified birds by both sight and call. Observers attempted to sight all birds heard calling to minimize the chance of double counting. We assigned each observer a group of three sites (six transects) to survey. Observers walked transects for 20 min, within 3 h of sunrise. Surveys were only undertaken in good weather conditions when there was little or no rain or wind. During every survey month, each transect was walked two to three times. Fifteen observers completed a total of 310 transect surveys.

Analysis

We used JMP 10® (SAS Institute Inc 2012) statistical software package to complete all statistical analysis. As we found no significant yearly variation in nesting or abundance, we then investigated the effect of tree density. Using the data from the 310 transect surveys we calculated the average number of birds per square kilometer per transect for each year. We also calculated the proportion of nest boxes occupied by each species at each site per year. This produced 45 measures of abundance and nest box occupancy for each species (15 sites over 3 years).

We investigated the influence of tree density on species abundance and nest box occupancy using one-way ANOVAs (Hypothesis 1, see Table 1).

We calculated the proportion of successful eggs from each box where we were able to clearly determine the fate of the eggs. Egg success for each species over the different tree densities was then evaluated using a one-way ANOVA (Hypothesis 1, see Table 1). We also used a one-way ANOVA to test if the average number of common myna eggs laid per clutch was significantly influenced by tree density.

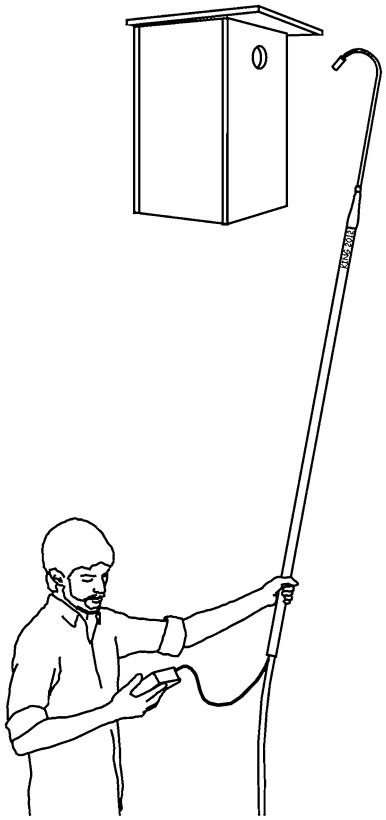

Fig. 2 Diagram of the bullet-camera we used to identify the species and the number of eggs in each nest box. We placed the camera at the entrance hole of each box, viewing and recording the images on the video camera. This procedure was fast and effective, resulting in minimal disturbance to nest box occupants (image credit Daryl King)

We then used linear regression to analyze the relationship between abundance and nest box occupancy, for each species, over the 15 sites for 3 years (45 data points). We wanted to insure there was a significant relationship between bird abundance and nest box occupancy (Hypothesis 2, see Table 1) as we used nest boxes as a proxy for natural cavities.

We investigated the relationship between common myna nest box occupancy and rosella species abundance using linear regression (45 data points). We then investigated this relationship separately over each of the three tree densities (15 data points per tree density) (Hypothesis 3, see Table 1). This helped us to determine if significant correlations were due to habitat preferences between different species or if they were due to nest box competition by the common myna.

Finally, we examined the relationship between tree density and the number of common myna interruptions to rosella nesting using a one-way ANOVA (Hypothesis 4, see Table 1).

Results

Common Myna

The common myna occupied an average of 26.5 % (± 3.5) of nest boxes throughout the survey period (averaged over all sites). The common myna also build apparent 'fake' nests in an additional 6.8 % (± 1.0) of boxes that were not used for egg laying and minimal nesting material was placed in these boxes (Harper and others 2005b). In two nest boxes, we observed that crimson rosella nesting was interrupted and covered over by 'fake' common myna nests. We observed that 'fake' nests built by the common myna had a different appearance to nests built for egg laying. 'Fake' nests were constructed of a thin flat layer of

Table 2 Common myna abundance and nesting in nature reserves surrounding Canberra, Australia

Tree density	Number of nest boxes added	Common myna abundance (birds per km^2)	Nest boxes used by the common myna (%)	Common myna egg success (%)	Average eggs per clutch	Common myna interrupt rosella species nesting (% of boxes)
High	75	9.7 ± 3.6	9.3 % ± 2.1	56.3 % ± 22.7	4.5 ± 0.3	1.3 ± 1.3
Med	75	45.4 ± 10.1	27.1 % ± 2.8	90.0 % ± 4.0	4.3 ± 0.2	6.6 ± 2.2
Low	75	101.9 ± 22.4	43.1 % ± 8.0	91.1 % ± 1.8	4.2 ± 0.1	12.7 ± 6.2
Significance		$F_{2,42} = 10.51$, $P < 0.001$*	$F_{2,42} = 11.29$, $P < 0.001$*	$F_{2,28} = 6.24$, $P = 0.006$*	$F_{2,28} = 0.24$, $P = 0.787$	$F_{2,42} = 1.79$, $P = 0.179$

Analysis of variance was used to test if there was a significant difference in abundance, nest box occupancy, nesting success or nest box interruptions over three tree densities (high, medium, and low)

Significant differences are indicated with an asterisk

twigs and often a large amount of rubbish (e.g., plastic bags, chocolate bar wrappers). Common myna nests built for egg laying had a thick, bowl-shaped layer of twigs with only small pieces of rubbish. We also observed that the common myna placed a layer of green *Eucalyptus* leaves into the nest 2–3 days before laying eggs.

We found that the number of common myna birds per square kilometer declined rapidly with increasing tree density ($F_{2,42} = 10.51$, $P < 0.001$) (Table 2). A similar pattern was observed for common myna nest box occupancy, with higher numbers observed in low tree density sites and observations declining as tree density increased ($F_{2,42} = 11.29$, $P < 0.001$) (Table 2).

The common myna laid one to seven eggs per clutch, with an average of 4.3 (± 0.1) eggs per clutch. Egg success (proportion of eggs laid that hatched) was greater in the low and medium tree density sites than in the high tree density sites ($F_{2,28} = 6.24$, $P = 0.006$) (Table 2). However, we found no significant relationship between the average number of eggs laid per clutch by the common myna and tree density ($F_{2,28} = 0.24$, $P = 0.787$) (Table 2).

We found a significant positive relationship between common myna abundance and the proportion of nest boxes occupied by the species ($F_{1,43} = 131.71$, $P < 0.001$) (Fig. 3).

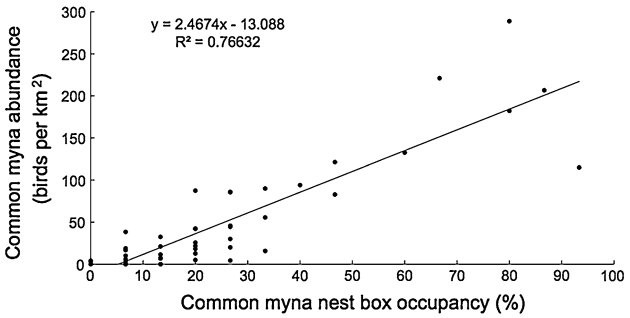

Fig. 3 Relationship between common myna abundance and the proportion of nest boxes occupied by the species ($F_{1,43} = 131.71$, $P < 0.001$)

Crimson Rosella

We observed that crimson rosella abundance at low tree density sites was significantly lower than at medium and high tree density sites ($F_{2,42} = 14.65$, $P < 0.001$) (Table 3).

The crimson rosella occupied an average of 24.3 % (± 2.1) of nest boxes throughout the survey period (averaged over all sites). Crimson rosella box occupancy increased with greater tree density ($F_{2,42} = 3.55$, $P = 0.038$) (Table 3).

Crimson rosella egg success (proportion of eggs laid that hatched) in high tree density sites was greater than egg success at medium or low tree density sites ($F_{2,38} = 4.04$, $P = 0.026$) (Table 3). There was also a significant relationship between crimson rosella abundance and the proportion of nest boxes occupied by the species ($F_{1,43} = 18.48$, $P < 0.001$) (Fig. 4).

Eastern Rosella

We found that eastern rosella abundance at medium tree density sites was higher than eastern rosella abundance at low or high tree density sites ($F_{2,42} = 5.79$, $P < 0.001$) (Table 4). The eastern rosella occupied an average of 8.9 % (± 1.2) of nest boxes throughout the survey period (averaged over all sites). We observed no significant relationship between eastern rosella egg success (proportion of eggs laid that hatched) and tree density ($F_{2,29} = 0.72$, $P = 0.495$) (Table 4). However, eastern rosella nest box occupancy was lower in the high tree density sites than in the medium or low tree density sites ($F_{2,42} = 3.75$, $P = 0.032$) (Table 4). We did not find a significant relationship between eastern rosella abundance and the proportion of nest boxes occupied by the species ($F_{1,43} = 1.75$, $P = 0.192$) (Fig. 5).

Impact on Rosella Nesting

We observed that the common myna interrupted crimson rosella nesting in 14 nest boxes and the eastern rosella in two nest boxes. The number of nest box interruptions by the common myna was slightly higher at sites with lower

Table 3 Crimson rosella abundance and nesting in nature reserves surrounding Canberra, Australia

Tree density	Number of nest boxes added	Crimson rosella abundance (birds per km^2)	Nest boxes used by the crimson rosella (%)	Crimson rosella egg success (%)
High	75	83.9 ± 9.3	28.9 % ± 3.0	65.7 % ± 4.0
Med	75	61.6 ± 6.4	25.8 % ± 3.0	46.1 % ± 5.1
Low	75	31.4 ± 3.9	18.2 % ± 4.3	44.6 % ± 8.5
Significance		$F_{2,42} = 14.65$, $P < 0.001$*	$F_{2,42} = 3.55$, $P = 0.038$*	$F_{2,38} = 4.04$, $P = 0.026$*

Analysis of variance was used to test if there was a significant difference in abundance, nest box occupancy or nesting success over three tree densities (high, medium, and low)

Significant differences are indicated with an asterisk

tree density; however, this relationship was not significant ($F_{2,42} = 1.79$, $P = 0.179$) (Table 2).

There was a significant negative relationship between the proportion of nest boxes occupied by the common myna and the abundance of the crimson rosella ($F_{1,43} = 26.057$, $P < 0.001$) (Fig. 6a). Further investigation revealed this negative relationship was significant at low tree density sites ($F_{1,13} = 7.548$, $P = 0.017$) (Fig. 6c) and high tree density sites ($F_{1,13} = 9.226$, $P < 0.001$) (Fig. 6b), but not at medium tree density sites ($F_{1,13} = 3.256$, $P = 0.094$) (Fig. 6c). At high tree density sites, an increase in the proportion of nest boxes occupied by the common myna, from 10–25 %, was related to a sharp decrease in crimson rosella abundance (Fig. 6b). At low tree density sites the relationship between common myna nest box occupancy and reduced crimson rosella abundance appeared to be less dramatic (Fig. 6d).

We also observed a significant negative relationship between common myna nest box occupancy and eastern rosella abundance ($F_{1,43} = 5.101$, $P = 0.029$) (Fig. 7a). However, further investigation revealed this relationship was only significant at low tree density sites ($F_{1,13} = 9.672$, $P < 0.001$) (Fig. 7d).

Other Species

Other species observed using nest boxes included the European honey bee (*Apis mellifera*), sugar glider (*Petaurus breviceps*), common brushtail possum (*Trichosurus vulpecula*), red-rumped parrot (*Psephotus haematonotus*), Australian owlet-nightjar (*Aegotheles cristatus*), and common starling. The European honey bee occupied 9.9 % (± 1.8) of nest boxes throughout the survey period. However, we found no significant relationship between European honey bee nest box occupancy and tree density ($F_{2,42} = 0.16$, $P = 0.849$). Nest box occupancy by other species was low (<5 % of boxes) and data were insufficient to allow for statistical analysis.

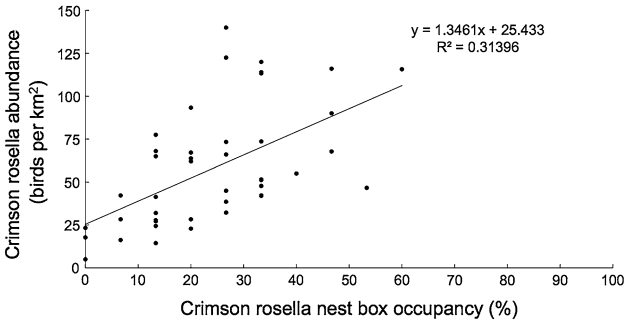

Fig. 4 Relationship between crimson rosella abundance and the proportion of nest boxes occupied by the species ($F_{1,43} = 17.85$, $P < 0.001$)

Discussion

Study Overview

To the best of our knowledge, this study provides the strongest evidence to date of the negative impact of the common myna on native bird *abundance*, through cavity-nesting domination. The key findings of our study were:

(1) The abundance and nesting of the common myna, crimson rosella and eastern rosella were strongly influenced by tree density.
(2) Nest box occupancy for the common myna and crimson rosella was strongly related to their abundance. This indicates that nest boxes provided a good proxy for species cavity-nesting.
(3) Despite the strong influence of tree density on species abundance, at low tree density sites, common myna nest box occupancy had a negative influence on crimson rosella and eastern rosella abundance. At high tree density sites the negative relationship between common myna nest box occupancy and crimson rosella abundance appeared to be more severe.

We did not attempt to quantify the rate of natural cavity-nesting in the study sites. However, nest boxes can provide a standardized and repeatable measure of cavity-nesting (Beyer and Goldingay 2006). Additionally, we believe nest boxes in our study area were representative of natural

Table 4 Eastern rosella abundance and nesting in nature reserves surrounding Canberra, Australia

Tree density	Number of nest boxes added	Eastern rosella abundance (birds per km^2)	Nest boxes used by the eastern rosella (%)	Eastern rosella egg success (%)
High	75	34.5 ± 3.9	4.4 % ± 1.2	63.3 % ± 13.5
Med	75	48.6 ± 8.0	11.1 % ± 2.1	46.8 % ± 10.1
Low	75	22.6 ± 3.0	11.1 % ± 2.4	46.3 % ± 7.5
Significance		$F_{2,42} = 5.79$, $P = 0.006$*	$F_{2,42} = 3.75$, $P = 0.032$*	$F_{2,29} = 0.72$, $P = 0.495$

Analysis of variance was used to test if there was a significant difference in abundance, nest box occupancy or nesting success over three tree densities (high, medium, and low)

Significant differences are indicated with an asterisk

cavity nest use for the common myna and crimson rosella, due to correlations between nest box use and abundance.

The common myna may show a preference for using nest boxes over natural cavities (Lowe and others 2011) and therefore, our observed occupancy and nesting success may be overestimated. The opposite may occur for the crimson rosella with evidence suggesting that they may prefer natural cavities to nesting boxes (Lowe and others 2011).

We have not been able to incorporate all causal variables into our analysis with some unexpected results observed. For example, eastern rosella abundance was not correlated with nest box use. The study species are likely to be influenced by other factors such as potential territorial exclusion by the native noisy miner (*Manorina melanocephala*), predation from the introduced domestic cat (*Felis catus*), land use practices in adjoining suburban areas and the abundance of natural cavities. Additionally, there may be detectability issues across the different observers and different tree densities, with it easier to observe birds in low tree density sites compared to high tree density sites. However, we believe the general trends observed in our study, over replicated sites, provide a firm basis from which to gain an understanding of the influence of habitat and the common myna on two native Australian parrots.

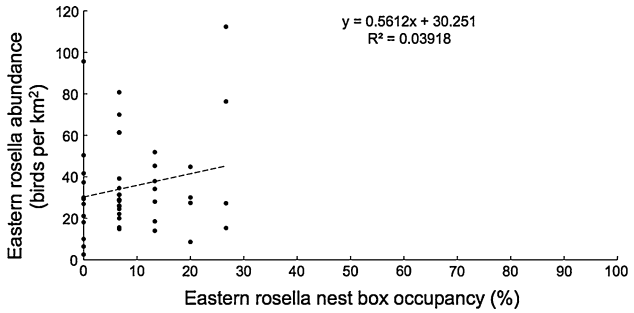

Fig. 5 Relationship between eastern rosella abundance and the proportion of nest boxes occupied by the species ($F_{1,43} = 1.75$, $P = 0.192$)

Influence of Tree Density on Species Abundance and Nesting

Many species are strongly influenced by habitat quality and are more abundant in high quality habitat than low quality habitat (Kajzer and others 2011). These variations in species abundance may occur due to increased breeding success and survival in high quality habitat (Xirouchakis and others 2011). However, high quality habitat for one species may not constitute high quality habitat for another species (Didham and others 2005; Farnsworth 2004; MacDougall and Turkington 2005). For example, the common myna is thought to prefer low tree density areas, while the crimson rosella tends to thrive in higher tree density forests and woodlands (Krebs 1998; Grarock and others 2013; Pell and Tidemann 1997a). In our study tree density significantly influenced the abundance, nest box occupancy, and nesting

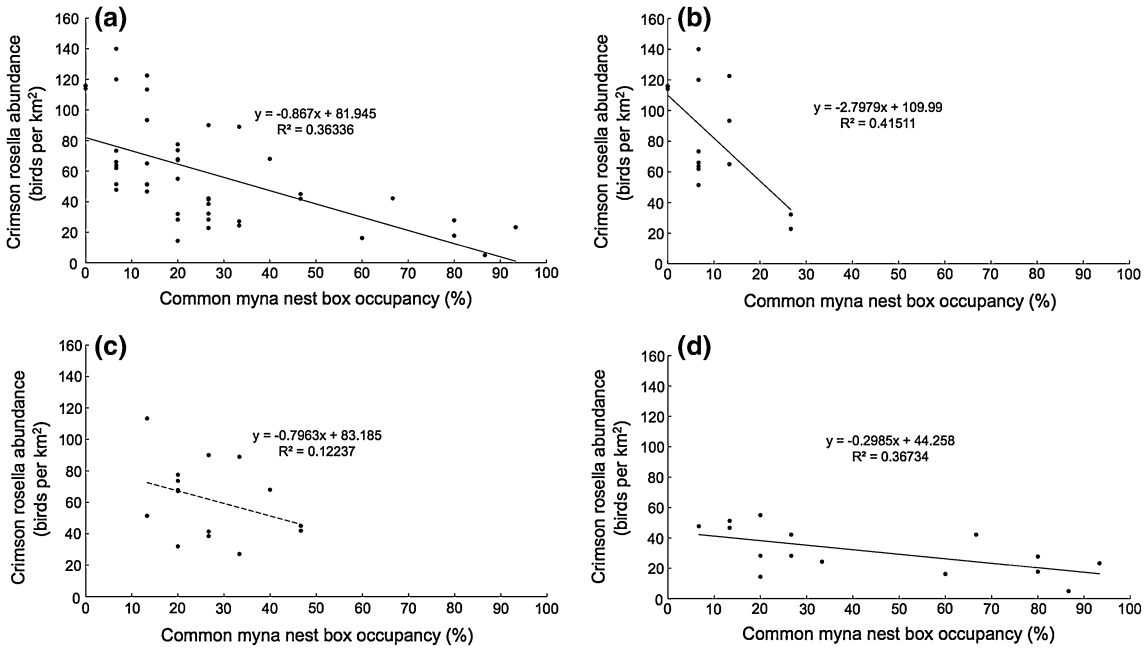

Fig. 6 Relationship between the proportion of nest boxes occupied by the common myna and the abundance of the crimson rosella. **a** The relationship over all sites of differing tree density ($F_{1,43} = 26.057$, $P < 0.001$), **b** high tree density sites only ($F_{1,13} = 9.226$, $P < 0.001$), **c** medium tree density sites only ($F_{1,13} = 3.256$, $P = 0.094$), and **d** low tree density sites only ($F_{1,13} = 7.548$, $P = 0.017$)

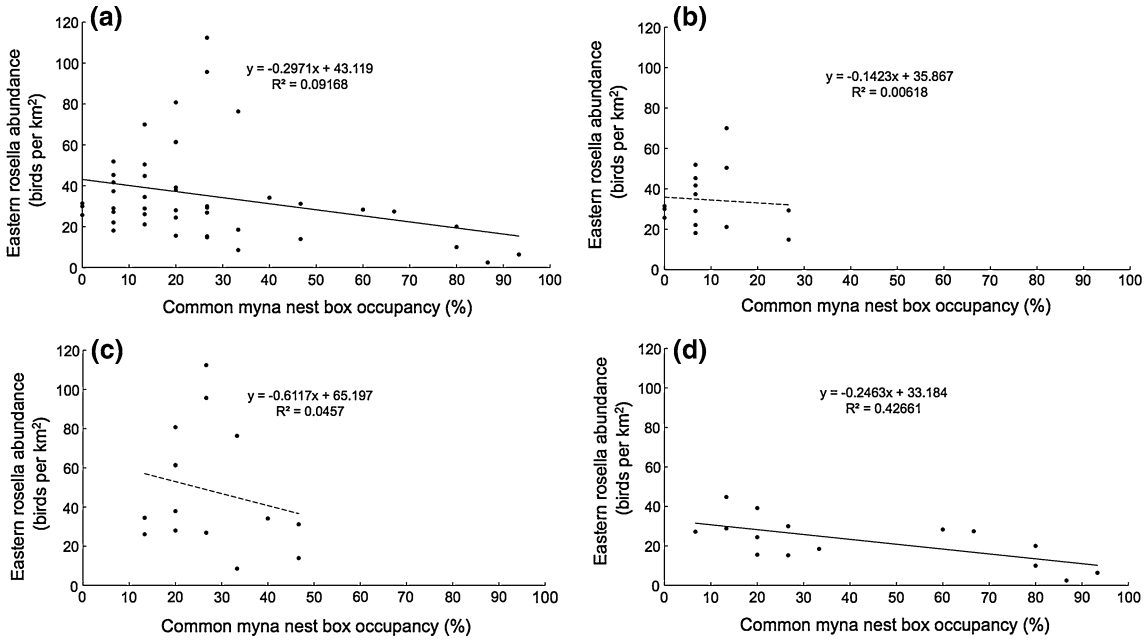

Fig. 7 Relationship between the proportion of nest boxes occupied by the common myna and the abundance of the eastern rosella. **a** The relationship over all sites of differing tree density ($F_{1,43} = 5.101$, $P = 0.029$), **b** high tree density sites only ($F_{1,13} = 0.081$, $P = 0.781$), **c** medium tree density sites only ($F_{1,13} = 1.216$, $P = 0.290$), and **d** low tree density sites only ($F_{1,13} = 9.672$, $P < 0.001$)

success of the common myna and the crimson rosella in opposite ways. Eastern rosella abundance and nest box occupancy were also significantly influenced by tree density. These results supported Hypothesis 1 (see Table 1) that tree density would influence abundance and rate of cavity-nesting of all three study species.

The common myna appeared to prefer low tree density sites, occurring in greater abundance, occupying more nest boxes and having a greater egg success than in other areas (Table 2). The success rate of common myna eggs in low tree density sites was over 90 %. This success rate is very high for a cavity-nesting bird species; however, we were not able to determine the success rate of fledglings. In a review of cavity-nesting species, Nice (1957) found an average egg success rate of 66 %. Therefore, even in high tree density sites, where common myna egg success rate dropped to approximately 56 %, egg success was still relatively high. Low tree density sites may represent high quality habitat for the common myna. These habitats are probably similar to their natural environment in India and central and southern Asia (Feare and Craig 1998; Pell and Tidemann 1997a). Open grassland with low tree density may offer suitable ground foraging areas for the common myna, while providing the species with cavities for nesting.

The crimson rosella appeared to prefer high tree density sites, occurring in greater abundance and nesting in more nest boxes than in other areas (Table 3). We also observed greater egg success rates of 66 % in high tree density sites, while in low tree density sites the egg success rate was 45 %. These egg success rates are similar to the 50 % success rate found in a study of the crimson rosella nesting in a high tree density reserve in Canberra, where the common myna was absent (Krebs 1998). This indicates that the common myna may not negatively impact native parrot egg success; rather the common myna may potentially 'impact' species through reducing the availability of cavities. Alternatively, our observation schedule (every 4 weeks) may not have been frequent enough to fully identify all cases of egg predation or egg failure.

The eastern rosella was more abundant in medium tree density sites, however, relatively high levels of nesting occurred over both medium and low tree density sites (Table 4). The species is known to prefer lightly wooded areas (Pizzey and Knight 2007) that may be representative of the medium and low tree density sites in our study.

Introduced Species Impact in Conjunction with Tree Density

Due to the strong influence of tree density on species abundance, it is essential to investigate the impact of introduced species in conjunction with changes in habitat (Didham and others 2005; Farnsworth 2004; MacDougall and Turkington 2005; Ricciardi 2003). Analysis that encompasses both habitat preferences and the impact of introduced species will assist with discrimination between these two key factors that influence the abundance of native species. This is essential for three reasons:

(1) To clearly understand the impact of introduced species on native species and avoid mistakenly identifying an impact when there is none (negative correlations may be due to habitat preference alone).
(2) To determine if the impact of introduced species on native species is more severe in particular habitats.
(3) To help facilitate effective management and mitigation of the negative impacts of introduced species.

For example, when tree density was not factored into our analysis, we found significant negative relationships between common myna nest box occupancy and the abundance of the crimson rosella and eastern rosella. However, this correlation does not imply causation. We would expect a negative correlation between common myna nest box occupancy and crimson rosella abundance based on habitat preference alone.

When we investigated the impact of the common myna on rosella species separately over different tree densities, we were able to account for some of the influence of habitat on species abundance and nesting. The common myna appeared to have a negative impact on both the crimson rosella and eastern rosella at low tree density sites (Figs. 1d, 7d). This relationship is potentially due to the high number of boxes occupied by the common myna at low tree density sites (over 90 % at one site). This high rate of box occupancy by the common myna and the building of 'fake' nests may limit the availability of nest sites for other cavity-nesting species. We also observed a negative relationship between common myna nest box occupancy and crimson rosella abundance at high tree density sites (Fig. 6b). Indeed the impact of the common myna appeared to be more severe in high tree density sites. Therefore, management of the common myna may be required in these areas that represent 'high quality' habitat for native species, especially in areas where threatened species are nesting, such as the superb parrot (*Polytelis swainsonii*).

Our findings supported Hypothesis 3 (see Table 1), that there would be a negative impact of common myna nest box occupancy on the abundance of the crimson rosella and eastern rosella. We observed higher numbers of native rosella nesting attempts interrupted by the common myna at low tree density sites although this was not statistically significant (Table 2). This finding partially supported Hypothesis 4 (Table 1), that in sites where the common myna is more abundant (low tree density sites), a greater number of native birds would be evicted from nest boxes.

Cavity Limitation

A review of the literature on population limitation in cavity-nesting species by Newton (1994) concluded that evidence exists for cavity limitation in human-modified landscapes but less so in mature, unmanaged forests. More recently, these conclusions have been supported by strong experimental evidence from the Northern Hemisphere (see Wiebe 2011 for a review). All of our sites were human-modified landscapes. Our results indicated that in combination, habitat quality and the common myna might substantially limit cavity nest sites for native species. Therefore, habitat restoration through natural regrowth and tree planting may be a suitable management strategy to increase habitat quality for native species and in the longer-term increase the number of nest cavities. Habitat restoration is also likely to make the habitat less suitable for the common myna, potentially reducing their abundance (Grarock and others 2013).

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ORIGINAL PAPER

Understanding basic species population dynamics for effective control: a case study on community-led culling of the common myna (*Acridotheres tristis*)

Kate Grarock · Christopher R. Tidemann · Jeffrey T. Wood · David B. Lindenmayer

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Abstract Population manipulation of introduced species can be difficult and many widespread eradication or reduction attempts have failed. Understanding the population dynamics of a species is essential for undertaking a successful control program. Despite this, control attempts are frequently undertaken with limited knowledge of the species population dynamics. For example, in Australia, concern over the impact of the introduced common myna (*Acridotheres tristis*) has led to community members culling the species. In this paper, we assessed the impact of community-led common myna culling program over broad and fine-scales in Canberra, Australia. We utilized a basic population model to enhance understanding of common myna population dynamics and the potential influence of various culling regimes. We found a significant negative relationship between common myna abundance and culling at fine-scales (1 km^2). However, over broad-scales the relationship between common myna abundance and culling was not significant. Our population model indicated culling at a rate of 25 birds per km^2 per year would reduce common myna abundance, regardless of initial density. Our results suggest that currently too few individuals are being removed from the Canberra population, and natural reproduction, survival and/or immigration is able to replace the culled individuals. This highlights the value of undertaking basic population modeling to assess if potential control measures are capable of achieving desired outcomes. Our work provides information for researchers, government and community groups interested in controlling not only the common myna, but also other introduced species.

Keywords Community management · Compensatory mortality · Doomed excess · Doomed surplus · Indian myna (*Sturnus tristis*) · Population model

K. Grarock (✉) · C. R. Tidemann · J. T. Wood · D. B. Lindenmayer
Fenner School of Environment and Society, The Australian National University, Canberra, ACT 0200, Australia
e-mail: kate.grarock@gmail.com

C. R. Tidemann
e-mail: chris.tidemann@anu.edu.au

J. T. Wood
e-mail: jeff.wood@anu.edu.au

D. B. Lindenmayer
e-mail: david.lindenmayer@anu.edu.au

K. Grarock
Invasive Animals Cooperative Research Centre, University of Canberra, Canberra, ACT 2617, Australia

D. B. Lindenmayer
Department of Environment, National Environmental Research Program, GPO Box 787, Canberra, ACT 2601, Australia

D. B. Lindenmayer
Australian Research Council Centre of Excellence for Environmental Decisions, Canberra, Australia

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Introduction

Increased trade and transport has led to the introduction of numerous species worldwide and the number of these introductions is increasing (Lockwood et al. 2005; Hulme 2006; Ascunce et al. 2011; Ricciardi 2012) The impact of introduced species can be varied with some species having devastating impacts while others are relatively benign (Davis 2003; Gurevitch and Padilla 2004; Gurevitch et al. 2011; Ricciardi 2012; Ruhren 2012). Due to the negative impacts of some introduced species, often significant effort is made to manage them.

Introduced species management has traditionally focused on eradication (Newton 1998; Davis et al. 2011). However, many widespread eradication or reduction measures have failed, leading to a waste of resources (Caughley 1977; Ward et al. 1979; Feare 1991; Newton 1998; Shine and Doody 2010; Davis et al. 2011). Examples include the failure of 26 of 30 attempted plant eradications in the Galapagos Islands since 1996 (Davis et al. 2011) and the expenditure of more than $AUD20 M that has largely failed to eradicate cane toads (*Rhinella marina*) from Australia (Shine and Doody 2010).

Eradicating an introduced species can be difficult and often only isolated populations or newly established species can be effectively eradicated (Hulme 2006; Zabala et al. 2010). The traits that make a species abundant are often the same traits that make a species difficult to control (for example, a high reproductive rate) (Newton 1998; King and Powell 2011). A population can compensate for culling losses through density dependent changes in reproduction, survival or immigration (Nichols et al. 1984; Conroy and Krementz 1990; Feare 1991; Nichols 1991; Shine and Doody 2010; Lebreton 2005; King and Powell 2011). Therefore, some species can be culled heavily year after year without achieving long-term reductions in their abundance or the damage they cause (Newton 1998; Shine and Doody 2010; King and Powell 2011).

Due to the difficulties associated with introduced species eradication, many programs have now shifted toward impact reduction measures, as they tend to be a more economical and effective way of impact mitigation (Newton 1998; Shine and Doody 2010; Melero et al. 2010). Numerous methods are available to reduce the impact of introduced species, such as the use of physical protection (fencing, netting, exclusion device), repellents (chemical, scaring device), resistance measures (development of resistant crops), and management (changing cropping time, location or planting 'lure crops') (Feare 1991; Newton 1998; Tracey et al. 2007). However, the impact of an introduced species is often related to their abundance and therefore, population control may still be necessary (Braysher 1993). Strategies should aim to reduce species abundance to below a threshold where their 'damage' is minimized (Braysher 1993; Shea and NCEAS 1998; Beggs and Rees 1999; Eiswerth and Johnson 2002). However, defining such thresholds can be extremely difficult (Hone 1995; Choquenot and Parkes 2001; Eiswerth and Johnson 2002; Buckley et al. 2004; Edwards et al. 2004). For detailed assessments of setting thresholds for pest control see Choquenot and Parkes (2001) and Bonesi et al. (2007).

To reduce the size of a population, the number of individuals culled annually needs to be higher than the maximum rate of population growth (Caughley 1977; Sutherland et al. 2004; Zuberogoitia et al. 2010; King and Powell 2011). This rate of culling also needs to be sustained year after year or numbers may recover (Shine and Doody 2010; Zuberogoitia et al. 2010; King and Powell 2011). Therefore, to successfully manipulate the population size of a species, its biology and population dynamics need to be understood (Newton 1998; Zuberogoitia et al. 2010; Zabala et al. 2010; King and Powell 2011; Louette et al. 2013). An effective method for population reduction also needs to be available (Newton 1998; Melero et al. 2010; Zabala et al. 2010; Louette et al. 2013; Tobin et al. 2013).

A key example of a control program, where little is known about species biology and population dynamics, is the culling of the common myna (*Acridotheres tristis*) in Australia. Using this case study, we demonstrated how knowledge of a species population dynamics and biology can be employed to enhance culling outcomes.

Globally, concern over the impact of the common myna is growing rapidly due to it competing with native species for territory and nest cavities (Feare and Craig 1998; ISSG 2000; Dhami and Nagle 2009). In 2005, the Australian community voted the common myna as the 'most significant pest', 'the pest problem seen to be increasing most', and the top 'pest problem that needs more control' (ABC 2011). Concern about the common myna was greater than species that have devastating impacts in Australia such as the cane toad,

red fox (*Vulpes vulpes*), feral cat (*Felis catus*) and European rabbit (*Oryctolagus cuniculus*) (Reddiex and Forsyth 2006). As a result, community-led trapping programs for the common myna have been initiated (CIMAG 2013). These trapping programs often focus on culling as many individuals possible, but are not guided by an understanding of the number of individuals that need to be culled to reduce species abundance. Therefore, it is timely that the effectiveness of community-led culling is assessed.

In this paper, we used culling data and bird survey data to investigate whether community-led culling is reducing common myna abundance in the city of Canberra, Australia. We analyzed changes in common myna abundance over fine-scale (1 km^2) and broad-scale (approximately 70 km^2) regions, in relation to culling by a community group. We hypothesized that community-led culling in Canberra is responsible for apparent reductions in common myna abundance. We then utilized a basic population model to enhance understanding of common myna population dynamics and the potential impact of various culling rates on the abundance of this introduced species.

Methods

Background on the common myna and community-led culling

The common myna is listed in the *world's worst invasive species* (ISSG 2000) and the species has been observed dominating natural nesting cavities, evicting native birds, killing the chicks and destroying eggs (Byrd 1979; Jones 1996; Pell and Tidemann 1997a, b; Feare and Craig 1998; Harper et al. 2005).

In April 2006, the Canberra Indian Myna Action Group (CIMAG) was formed (CIMAG 2013). CIMAG is a community group that aims to reduce the impact of the introduced common myna, through public education (to reducing feeding and breeding opportunities) and a humane culling program. The group uses pet food as bait to lure individuals into valve traps, before humanely euthanizing them (CIMAG 2013). Traps are designed to permit entry to the common myna by means of a valve through which individuals can enter freely but are unable to locate the exit (Tidemann 2010). CIMAG has approximately 1,400 members that have euthanized over 45,000 individuals from the Canberra area over 7 years (CIMAG 2013). The common myna was once the third most common bird species in Canberra but since culling began it has dropped to the thirteenth most common bird species (Canberra Ornithologists Group 2011). The culling program is considered to be 'successful' by CIMAG. Other community groups around Australia and overseas have sought advice and support from CIMAG to implement their own control programs (B Handke 2013 pers. comm. 20 Jan.).

Study site

Canberra is the capital city of Australia with a population of approximately 368,000 people. The city is located in south-east Australia, 150 km inland and at an elevation of 605 m above sea level (Gibbs et al. 2011) (Fig. 1). The city experiences relatively warm to hot summers [mean maximum temperature 27.6 °C (81.7 °F)] and cool to cold winters [mean minimum temperature 0.9 °C (33.6 °F)], when frost and fog are common (BOM 2013). The average annual rainfall is 632 mm (BOM 2013). Canberra is a planned city - known as the 'bush capital' due to the large number of nature reserves and undeveloped hills in and around the city. The main vegetation types in nature reserves are dry sclerophyll forest, open savannah, and woodland.

Broad-scale common myna abundance and impact of culling

We used long-term survey data (see Grarock et al. 2013a) to document common myna abundance over 29 years in Canberra. The Canberra Ornithologists Group (COG) started the Garden Bird Survey (GBS) in 1981. Observers surveyed an area of 3.1 hectares, recording the maximum number of each bird species seen or heard in a site at any one time over a 7-day period. The duration of each observation was unspecified in the GBS procedures. In our analysis, we accounted for this potential variability by estimating average common myna abundance per region per season (see below). A total of 74,492 surveys were undertaken in Canberra from July 1981 to 2010. We used ArcGIS 10® (ESRI 2010) to define four geographic regions in Canberra (Fig. 1). We based regions primarily on geographic location and development history of the city. This enabled grouping of

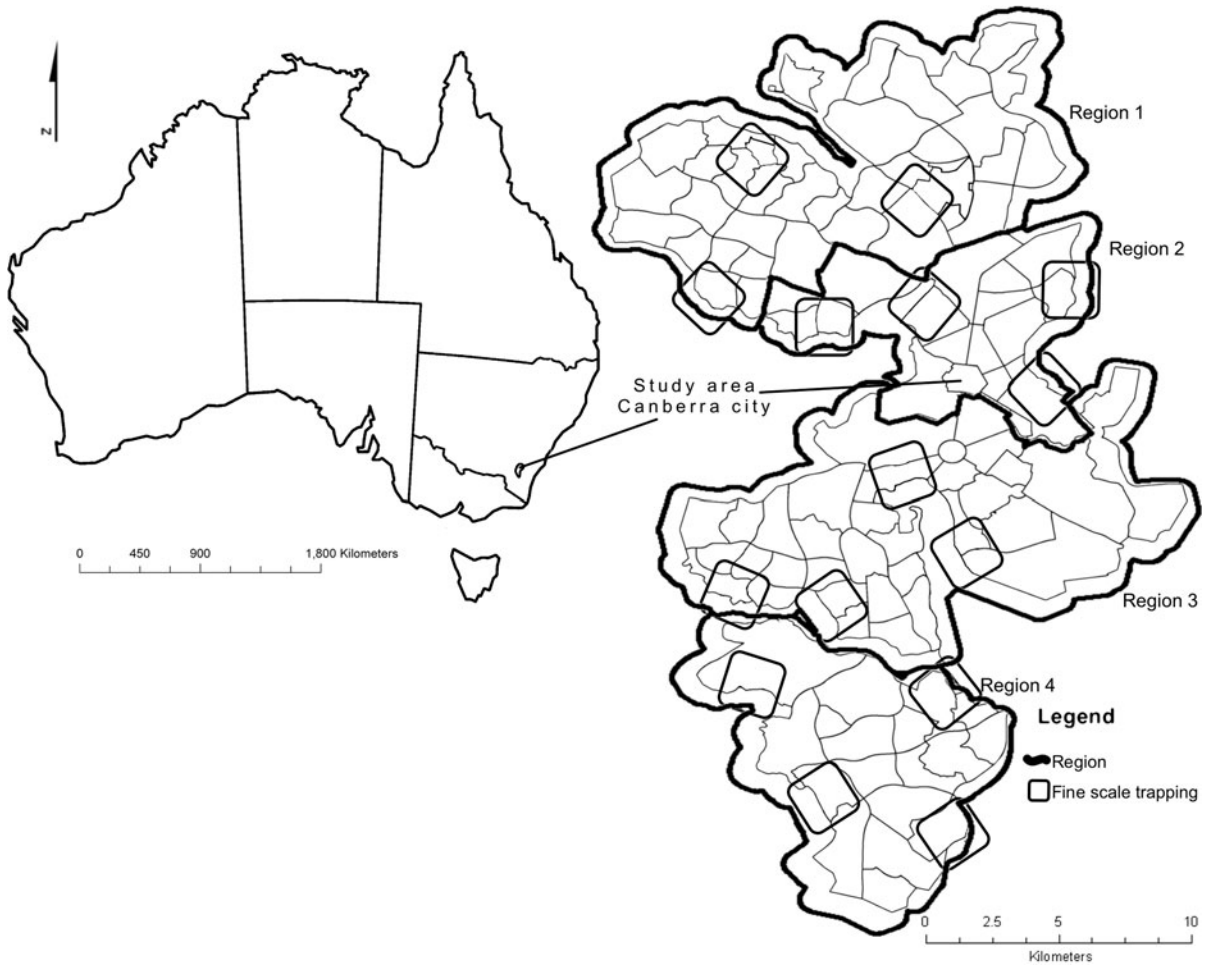

Fig. 1 Study area and location of the four regions and 15 fine-scale culling areas in the suburbs of Canberra, South East Australia

survey sites over a broad area to assess the impact of common myna culling.

Due to seasonal variation in common myna abundance (Pell and Tidemann 1997a), we divided each year into two time periods: Spring–Summer (September to February) and Autumn–Winter (March to August). Common myna abundance is highest in Spring–Summer when the species breeds and young birds fledge. By contrast, common myna abundance declines over Autumn–Winter due to natural mortality (Pell and Tidemann 1997a). We broadly termed these two periods as 'breeding' (Spring–Summer) and 'non-breeding' (Autumn–Winter) seasons.

GenStat 15® (VSN International 2012) was used to conduct all analyses. We fitted a hierarchical generalized linear model (Lee et al. 2006) to raw common myna counts using a quasi-Poisson model with a logarithmic link function. Region (1–4), year (1–29), season (breeding, non-breeding), and their interactions were treated as fixed effects. GBS sites were treated as a random effect with a log-gamma distribution. For each combination of region, year and season, we estimated the average common myna abundance per site, reducing the data from 74,492 individual surveys to 232 estimates. We calculated the bi-annual abundance of the common myna per square kilometer per region.

CIMAG provided information on the location and number of common myna birds trapped and euthanized. Recording of cull numbers commenced in July 2006, with some isolated culling occurring prior to this time. CIMAG send monthly requests to members asking them to report the number of individuals culled. However, some culling may go unreported and no attempt is made to record data on 'trap effort' (the

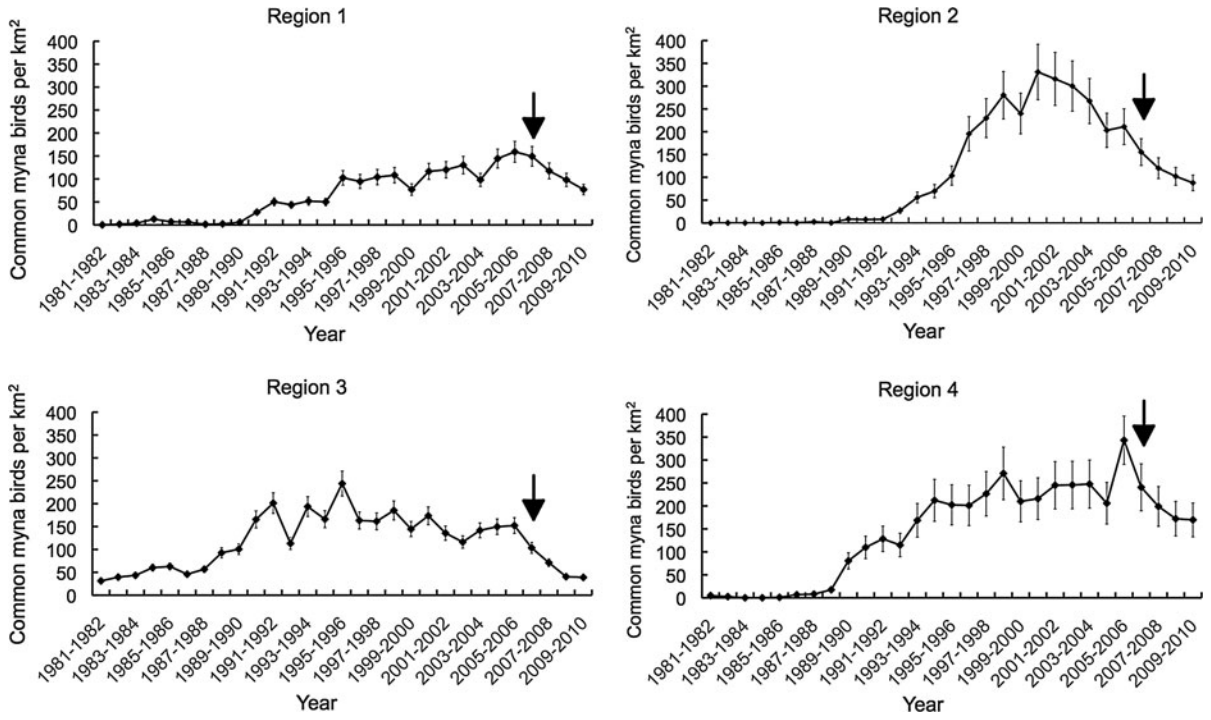

Fig. 2 Long-term abundance of the common myna (*A. tristis*) in four regions in Canberra, South East Australia. *Black arrows* indicate the commencement of a community run common myna culling program

amount of effort required to trap birds). Therefore, we were unable to factor this into our analysis. We defined the yearly culling period as November to October, as we wanted to calculate the number of birds culled prior to November (the peak period of fledgling emergence) (Pell and Tidemann 1997a).

We entered data from CIMAG into ArcGIS 10® (ESRI 2010) with the location and date of each bird caught. We calculated the total numbers of the common myna euthanized in each of the four regions over 12 month culling periods. This provided three culling periods (Nov 2006–Oct 2007, Nov 2007–Oct 2008 and Nov 2008–Oct 2009).

We used estimates of common myna abundance per square kilometer per region in the breeding season to investigate the impact of common myna culling. We focused on the breeding season (Spring–Summer) due to greater abundance and visibility of the species at this time.

We observed that there were some reductions in common myna abundance prior to April 2006, when culling commenced (Fig. 2). Therefore, we wanted to avoid mistakenly identifying culling as reducing common myna abundance when it could potentially be due to other influences (for example, natural population limitation). Therefore, we extended our analysis to include common myna abundance in the 3 years prior to common myna culling (when common myna culling was approximately equal to zero). We used linear regression, to investigate the relationship between changes in common myna abundance per square kilometer and birds culled per square kilometer.

Fine-scale common myna abundance and impact of culling

We selected 15 sites in Canberra to survey the abundance of the common myna. We located each site in a residential suburb that adjoined a nature reserve. Nature reserves ranged from dense woodlands to open grassy woodlands and were dominated by *Eucalyptus* species. We ensured that residential subdivisions were constructed more than 20 years ago so

that the vegetation was well established. Sites were situated at least 2 km apart, as the common myna rarely travels further than 2 km per day (Feare and Craig 1998; Dhami and Nagle 2009).

We surveyed our 15 sites in the breeding season in September, November and January, commencing in September 2008 and concluding in January 2011. We used experienced bird observers to identify the common myna by both sight and call, using line transect surveys. Transects were located through a suburb at a right angle to an adjacent nature reserve. Transects were 1 km in length and 60 m wide. Observers walked transects for a period of 20 min, within 3 h of sunrise. Each transect was walked two to three times per survey month, by two or more observers. We ensured surveys were undertaken in good weather conditions with little or no rain or wind to limit weather effects on detectability (Lindenmayer et al. 2009). Observers attempted to sight all birds heard calling and take into account the movement and direction of birds to minimize the chance of double counting.

A total of 311 transect surveys was conducted by 15 observers. We calculated the mean common myna abundance per square kilometer in each of the 15 sites, each breeding season (2008–2009, 2009–2010, 2010–2011). From this, we calculated the change in common myna abundance from one time period to the next. This provided two measures of the yearly change in common myna abundance for each of the 15 sites.

We used ArcGIS 10® (ESRI 2010) to identify a 1 km buffer around each of the 15 survey transects. We defined this buffer zone as the fine-scale culling area (Fig. 1). We then calculated the number of birds euthanized within each of the 15 culling areas for the two culling periods (Nov 2008–Oct 2009 and Nov 2009–Oct 2010). To assess the impact of culling, we performed linear regression analysis using the change in common myna abundance per square kilometer against birds culled per square kilometer.

Common myna population model

We used a logistic growth model to investigate how a common myna population would react to various levels of culling at different population densities. For simplicity, we utilized a model with no stochastic dynamics (environmental change). We used the generalized logistic model outlined by Sutherland et al. (2004) and derived from Caughley (1977). The change in population from one time period to the next was governed by the following equation (Sutherland et al. 2004):

$$N_{t+1} = N_t + r_{max}N_t(1-N_t/K) - c_tN_t$$

where N_t is population size at $time_t$, r_{max} is the maximum growth rate, K is the carrying capacity, and, c_t is the culling rate for the same time period.

We used the average estimates from Grarock et al. (2013a) for the maximum growth rate and carrying capacity for the common myna over four regions in Canberra. Grarock et al. (2013a) used a composite 41-year data set to reconstruct the invasion sequence of the common myna in Canberra since 1968. The average maximum rate of population growth was 24.1 (±6.4) birds per square kilometer per year and the average maximum population size was 205.9 (±34.6) birds per square kilometer.

We used the logistic growth model to establish an understanding of the number of individuals that need to be culled to reduce the population size. From this model, we identified the level of cull required (per square kilometer per year) at various population sizes to produce a significant (>10 %) reduction in common myna abundance. Ultimately culling strategies should aim to reduce common myna abundance below a predefined threshold, where their 'damage' is minimized (Braysher 1993; Shea and NCEAS 1998; Eiswerth and Johnson 2002). However, in the absence of such research on the common myna, we used a 10 % reduction as a guide to assist with understanding the number of individuals that need to be culled to reduce species abundance.

We then estimated the total annual rate of common myna culling per square kilometer in Canberra required to reduce species abundance. Using ArcGIS 10® (ESRI 2010) we calculated the total area of Canberra suburbs as 326.92 km^2. This area includes only the suburban area of Canberra and excludes nature reserves as well as surrounding urban areas of Queanbeyan, Hall and Hume. We then multiplied the required culling rate per square kilometer per year, by the total area of Canberra suburbs.

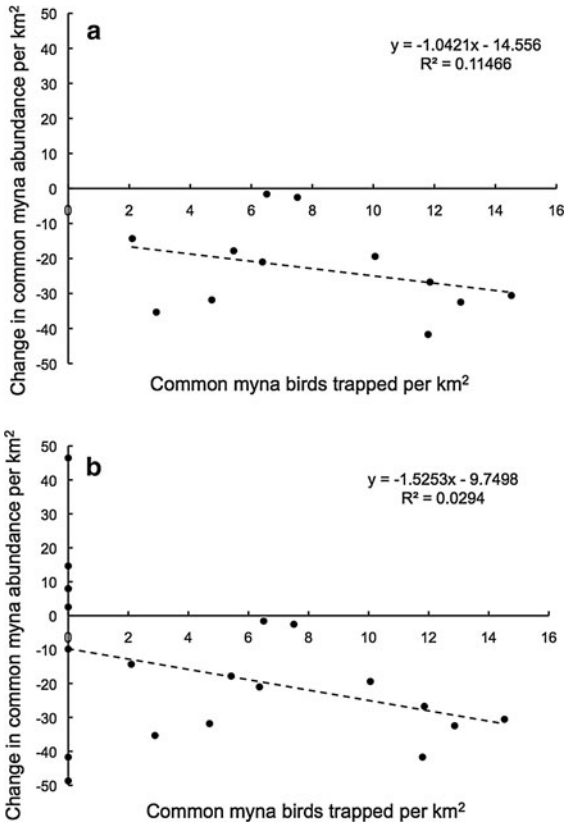

Fig. 3 Relationship between the changes in common myna abundance (*A. tristis*) and the number of individuals culled in four regions in Canberra, South East Australia. **a** Common myna abundance only in the years where culling occurred ($F_{1,10} = 1.30$, $P = 0.28$), **b** common myna abundance with culling and the 3 years prior to common myna culling (when common myna culling was equal to zero) ($F_{1,22} = 0.67$, $P = 0.42$)

Results

Broad-scale common myna abundance and impact of culling

In all four regions in our study, common myna abundance fluctuated over the 29 year period (Fig. 2). In some regions there appeared to be a decline in common myna abundance prior to the commencement of community-led culling (Fig. 2).

We found no significant relationship between the yearly change in common myna abundance per square kilometer and yearly culling per square kilometer ($F_{1,10} = 1.30, P = 0.28$) (Fig. 3a). When we included the 3 years of data prior to common myna culling, the relationship further weakened ($F_{1,22} = 0.67$, $P = 0.42$) (Fig. 3b). Two of the largest observed reductions in common myna abundance (41 and 49 birds per square kilometer) occurred prior to culling (Fig. 3b). This finding indicates that community-led culling in Canberra is *not* responsible for reductions in *broad-scale* common myna abundance.

Fine-scale common myna abundance and impact of culling

We found a significant negative relationship between yearly change in common myna abundance per square kilometer and yearly culling per square kilometer ($F_{1,28} = 20.67, P < 0.001$). This finding indicates that community-led culling in Canberra *is* capable of reducing *fine-scale* common myna abundance.

Common myna population model

The expected outcomes of various culling rates on the abundance of common myna in Canberra are provided in Table 1. The level of cull required to create a significant reduction (>10 %) in population size for each population size is highlighted with an asterisk (Table 1). Italic values are used to highlight populations that have reduced in size (Table 1).

According to our model, the natural rate of population growth is at its greatest when the population is at half carrying capacity (103 birds per square kilometer). A cull effort of approximately 40 birds per square kilometer is required to create significant (10 %) reductions in the population size at this density (Table 1). Fewer birds need to be culled to significantly (10 %) reduce the population size at densities below 103 birds per square kilometer, as the natural rate of population growth decreases. For example, at 20 birds per square kilometer, only 15 birds per square kilometer need to be culled to achieve significant (10 %) reduction in population size. As a population approaches carrying capacity (205.9 birds per square kilometer), the natural rate of population growth slows. When the population is above carrying capacity, the population size reduces naturally without culling. This natural decrease in population size may be responsible for the population reductions observed in some Regions prior to the commencement of the culling program in Canberra (Fig. 2).

Table 1 Expected common myna (*Acridotheres tristis*) abundance after various culling regimes and initial population densities, using a basic logistic growth model

Culling rate (birds per km^2 per year)	0	2	5	10	15	20	24	25	30	40
Initial population size (birds per km^2)										
10	14.5	12.5	*9.5*	*4.5**	*0.0*	*0.0*	*0.0*	*0.0*	*0.0*	*0.0*
20	28.5	26.5	23.5	*18.5*	*13.5**	*8.5*	*4.5*	*3.5*	*0.0*	*0.0*
30	42.0	40.0	37.0	32.0	*27.0**	*22.0*	*18.0*	*17.0*	*12.0*	*2.0*
40	55.1	53.1	50.1	45.1	40.1	*35.1**	*31.1*	*30.1*	*25.1*	*15.1*
50	67.7	65.7	62.7	57.7	52.7	*47.7*	*43.7**	*42.7**	*37.7*	*27.7*
60	79.9	77.9	74.9	69.9	64.9	*59.9*	*55.9*	*54.9*	*49.9**	*39.9*
80	102.9	100.9	97.9	92.9	87.9	82.9	*78.9*	*77.9*	*72.9*	*62.9**
100	124.1	122.1	119.1	114.1	109.1	104.1	100.1	*99.1*	*94.1*	*84.1**
150	169.1	167.1	164.1	159.1	154.1	*149.1*	*145.1*	*144.1*	*139.1*	*129.1**
200	202.7	200.7	*197.7*	*192.7*	*187.7*	*182.7*	*178.7**	*177.7**	*172.7*	*162.7*
250	*224.9**	*222.9*	*219.9*	*214.9*	*209.9*	*204.9*	*200.9*	*199.9*	*194.9*	*184.9*

Italic values indicate populations would be expected to reduce in size and an asterisk indicates populations that would be expected to reduce significantly (>10 %)

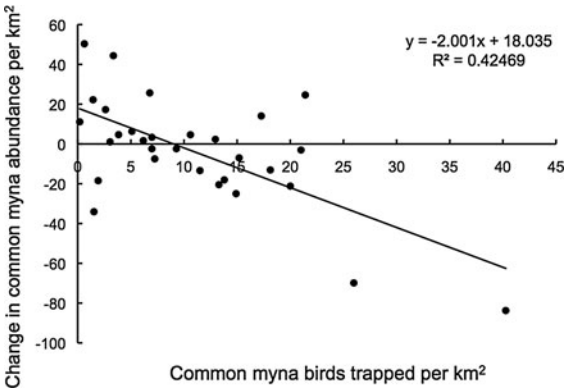

Fig. 4 Relationship between the change in common myna abundance (*A. tristis*) and the number of individuals culled in 15 fine-scale sites (1 km^2) in Canberra, South East Australia ($F_{1,28} = 20.67$, $P < 0.001$)

Our model indicated that culling at a rate of 25 birds per square kilometer would result in reductions in the population size regardless of initial density (Table 1). This corresponds closely with what we observed for culling in Canberra. The culling rate in regions ranged from 0 to 15 birds per square kilometer (Fig. 3) and is lower than the model estimate of 25 birds per square kilometer. This helps to explain the lack of observed correlation between culling and common myna abundance at broad-scale sites. However, culling at fine-scale sites was occasionally >25 birds per square kilometer and we observed a significant negative relationship between culling and common myna abundance (Fig. 4), as would be predicted by our model.

Using this figure of 25 birds per square kilometer, we estimate that approximately 8,173 (25×326.92 km^2 = 8,173) birds would need to be culled *each year* in Canberra to reduce common myna abundance.

Discussion

Study overview

We assessed the impact of community-led culling on common myna abundance, using bird survey data and culling at different spatial scales in Canberra, Australia. To the best of our knowledge, it is the first investigation to demonstrate that community-led culling is capable of reducing the *local* abundance of the common myna. We also utilized a basic population model to enhance the understanding of population dynamics and the potential influence of various culling regimes. Our study provided a unique opportunity to demonstrate how knowledge of species population dynamics and biology can be employed to enhance culling outcomes. This knowledge will enable culling targets to be set and guide planning of methods to achieve these targets. However, thresholds for

common myna abundance still need to be determined and should be a priority. This paper provides valuable approaches and information for researchers, government and community groups interested in controlling not only the common myna but also other introduced species.

We observed that high-intensity culling appears to reduce common myna abundance at a fine-scale (1 km^2). This indicates that community-led culling *is* capable of reducing the *local* abundance of the common myna. However, this high-intensity community-led culling may not be achievable over larger scales. Our results indicate that broad-scale culling of the common myna falls short of the levels required to reduce population numbers. The graphs of broad-scale common myna abundance (Fig. 2) and culling (Fig. 3) and the common myna culling model (Table 1) suggest that too few individuals are currently being removed from the population to achieve a significant (10 %) reduction in the population size. Therefore, we reject our hypothesis that community-led culling in Canberra is responsible for reductions in common myna abundance. However, the apparent success of fine-scale culling indicates that broad-scale culling could become more effective if a greater number of individuals could be trapped. Our findings indicate that both the level of culling and the spatial extent of control operations influence the success of introduced species management.

According to our model, culling at a rate of 25 birds per square kilometer per year should reduce the population size regardless of initial density (although not significantly for population densities between 60 and 150 birds per square kilometer). Culling at 25 birds per square kilometer per year is greater than the maximum rate of increase of 24.1 birds per square kilometer and may be a suitable target rate for culling in Canberra.

Our model and analysis did not take into account environmental stochastisity, nor did it take into account populations in the area surrounding Canberra (nature reserves and other urban areas). Nature reserves are used extensively by the common myna in Canberra (Pell and Tidemann 1997a) and they may act as a source for individuals to reinvade the city areas. Therefore, the actual number of individuals that need to be culled annually would be greater than our estimated 8,173 birds. When culling first began CIMAG culled over 8,000 birds for two consecutive years, but more recently culling rates have ranged from 2,800 to 6,600 birds per year (B Handke 2013 pers. comm. 14 Jun.). These figures fall short of our conservative estimate of 8,173 birds that need to be culled annually.

Our basic population model provided valuable insight into the population dynamics of the common myna. It is by no means a comprehensive assessment of common myna population dynamics and should be seen as providing an initial assessment. This model could be further refined, if required, as more information is gathered, such as estimates of age structure, sex ratio, survival rates and reproduction rates. Additionally, more complex models exist that incorporate the links between life-history dynamics, culling rates and environmental variables (Sutherland et al. 2004).

Species demographics (age structure, sex ratio, birth rate and death rate) are critical in driving a species population size (Hengeveld 1989; Neubert and Caswell 2000; Simberloff and Gibbons 2004). Species are also influenced by density-dependent and density-independent factors. For example, density-dependent population regulation may occur as a population reaches carrying capacity and resources (e.g. food or territory) become scarce (due to overpopulation) (Hengeveld 1989; Rutz 2008). This may result in increased mortality or reduced breeding success and therefore the population size will reduce (Hengeveld 1989). As the population size reduces the pressure on available resources will ease and the population may eventually increase again. This process can lead to fluctuations in population size over time (Hengeveld 1989; Lensink 1998; Rutz 2008). Density-independent factors also influence population size and fluctuations. Examples of density-independent factors include weather, climate, and events such as drought or fires that can cause mortality and affect resource availability (Sher and Hyatt 1999; Richardson and Pysek 2006). Population size can also be influenced by the existence of natural enemies (predators, parasites or disease) that increase mortality (Keane and Crawley 2002; Shea and Chesson 2002).

Due to these natural population fluctuations it can be difficult to clearly identify the influence of culling on a population. Unfortunately we were unable to obtain data on natural population fluctuations from sites without common myna culling. Therefore, without this knowledge we cannot say with certainty that observed changes are due to community-led culling.

However the relationship for both broad and fine scale trapping is consistent with data from our population model, indicating that it is an accurate representation of this system and that high intensity culling is capable of reducing common myna abundance.

Understanding basic species population dynamics for effective control

For the successful management of vertebrate populations, managers need to have an informed and targeted strategy (Beggs and Rees 1999; Zabala et al. 2010; Louette et al. 2013; Tobin et al. 2013). Understanding the factors that influence the population dynamics of a species will assist with making informed decisions and make management more effective. These factors include life history strategy, population density, timing of cull, vulnerability of individuals to culling and species distribution and movements.

Life history strategy

The life history strategy of a species can have a big impact on the effects of a culling program, with some species capable of compensating for culling losses (Conroy and Krementz 1990; King and Powell 2011; Sandercock et al. 2011). For example, the common myna is able to produce up to seven eggs per clutch and up to three clutches per year (Grarock et al. 2013b). This enables the species to compensate for mortality-related losses from culling, through rapid reproduction.

Population density

As seen from our population model, population density had a large influence on the rate of population growth and, therefore, the impact of various culling rates (Eiswerth and Johnson 2002; Zuberogoitia et al. 2010; King and Powell 2011; Sandercock et al. 2011). For example, species population growth tends to be slow initially, reaching a maximum growth rate as the population approaches half the carrying capacity; growth then slows as the density reaches carrying capacity. A species may also be more difficult to cull at lower densities due to difficulty locating or trapping individuals. This increased difficulty in trapping at lower densities has been observed in previous studies on the common myna (Griffin 2008; Griffin and Boyce 2009; King 2010; Tidemann 2010).

Cull timing

The timing of culling can be critical to the success of a management program (Newton 1998; Boyce et al. 1999; Kokko 2001; Ratikainen et al. 2008; Zabala et al. 2010). However, culling is often simply undertaken when the species is easiest to kill (Newton 1998; Shine and Doody 2010). Our data suggests that culling should be undertaken prior to the breeding season when the population is naturally at its lowest point. Common myna cull numbers are currently highest during and just after the breeding season, when young birds are easily trapped. This is not the most effective time for culling, as many of these individuals may be the 'doomed-excess' (individuals that may not have survived over winter) (Clark 1987; Hudson et al. 1997; Newton 1998). We suggest that the lack of reduction in common myna numbers over broad-scales may be caused by culling the 'doomed-excess'. Therefore, common myna culling may be more effective if undertaken prior to the breeding season (approximately September, October and November). Arguably, the common myna has its largest impact during the breeding season (Pell and Tidemann 1997b; Grarock et al. 2013b). Therefore, culling from September to December may also reduce competitive pressure for nest cavities with native species.

Species ecology

Understanding the ecology of a species can help identify traits that may assist with species control (Newton 1998; Melero et al. 2010; King and Powell 2011; Tobin et al. 2013). Culling can target certain individuals in a population and this may reduce the effectiveness of culling operations (Dufour et al. 1993; Lebreton 2005; Melero et al. 2010). For example, valve trapping of the common myna may target young inexperienced birds (King 2010), potentially leaving individuals that cause the most damage and/or are strong breeders (Newton 1998; Brook et al. 2003; Melero et al. 2010). Culling operations that target breeding females may have greater success. Therefore, it is important to understand the age, sex and fitness of individuals being targeted by culling operations (Zabala et al. 2010; King and Powell 2011; Sandercock et al. 2011). Identifying and exploiting vulnerable aspects of a species life history will enhance culling numbers and the impact of culling.

For example, in the evening, common myna individuals group together forming overnight roosts (sometimes containing hundreds of individuals) (Feare and Craig 1998; Tidemann 2010). Therefore, targeting roosts for management actions may be a good strategy. Alternatively, the female common myna remains on the nest over night and if culled, the male will often return with a new female to begin breeding (Tidemann et al. 2011). A highly effective method for euthanizing breeding females in nest boxes and removing their eggs or chicks was developed by Tidemann et al. (2011).

Spatial element

Understanding the spatial structure, dispersal ability and capacity of a species to move from high to low density areas is important for successful management (Edwards et al. 2004; Hampton et al. 2004; Travis and Park 2004; McMahon et al. 2010; Melero et al. 2010; Louette et al. 2013). Attempts to control only a small proportion of the population may fail due to immigration from surrounding areas (Pulliam 1988; Hone 1995; Zuberogoitia et al. 2010) and lead to a waste of management resources (Myers et al. 2000; Zabala et al. 2010). For successful control, management units must be clearly defined (Robertson and Gemmell 2004; Zuberogoitia et al. 2010). Small islands are one example of a naturally occurring management unit (Courchamp et al. 2003; Towns and Broome 2003). Control over a widespread area is possible but logistically difficult (Taylor et al. 2000; Towns and Broome 2003; Courchamp et al. 2003; Zabala et al. 2010). For species that occur on larger islands or continents, management units need to be defined that are both logistically tractable and have a low risk of recolonization (Parkes 1990; Bomford and O'Brien 1995; Hampton et al. 2004). Using molecular genetics can be a powerful tool to understand and identify areas where there is negligible movement of individuals and thus suitable management units for control operations across a landscape (Moritz et al. 1996; Robertson and Gemmell 2004; Sarre et al. 2013).

There are many examples in the literature of species that have been culled in large numbers where there has been little to no overall reduction in their population size (Feare 1991). In Africa, millions of dollars were spent culling hundreds of millions of red billed quelea (*Quelea quelea*) for many years, yet the population size was not affected by this culling (Newton 1998). The species was able to reproduce quickly and control operations (spraying and dynamiting communal roosts) reached only a relatively small number of sites. The species quickly immigrated to, and bred, in areas where there had been heavy culling (Newton 1998). Widespread culling was abandoned with efforts directed towards localized culling for crop protection and altering cropping procedures. Likewise, community-led culling of the cane toad has had "… millions of dollars and thousands of hours …" dedicated to it (Shine and Doody 2010), with "… no evidence that physical removal of cane toads has slowed the invasion" (Peacock 2007). However, in New Zealand community-led trapping of invasive animals, in cooperation with government agencies, appears to be responsible for substantial increases in kiwi (*Apteryx mantelli*) abundance (Glen et al. 2012). The importance of community acceptance and assistance in control programs can be critical to their success (Bremner and Park 2007). Therefore, scientists need to communicate research effectively to ensure community enthusiasm for conservation is harnessed into meaningful projects based on good science.

Common myna management

Suitable management units and abundance thresholds for the common myna still need to be defined. Our study indicated that intense localized culling appears to be effective but that current cull efforts are not high enough to have significant widespread reductions on common myna abundance in Canberra. The common myna appears to be somewhat sedentary and slow at spreading to new areas (Grarock et al. 2013a), potentially enhancing cull effectiveness in the medium term (Sandercock et al. 2011). Due to the species' broad distribution across Australia and other continents, perhaps management actions should be undertaken in localized areas where the species is deemed to have the greatest impact. For example, management of the common myna near threatened species breeding areas, such as the superb parrot (*Polytelis swainsonii*), may be a good strategy. The adaptable nature of the common myna indicates that complimentary methods for controlling the species (such as roost and nest box culling) will likely be required to successfully reduce the abundance of this species (Tidemann 2010; Tidemann et al. 2011).

Conclusion

Our case study shows the value of undertaking basic population modeling before carrying out control programs. A significant amount of effort is currently being directed towards a program that is not fully achieving the desired results. Understanding the population dynamics of a species is essential to undertaking a successful control program. Basic population models can be used to assess whether potential control measures are capable of achieving desired outcomes. These models can help avoid the wasteful allocation of resources to projects that are unlikely to be successful and/or drive innovation into alternative control measures. An effective control program needs to have an informed and targeted strategy with information on the number of individuals that need to be culled and the duration and timing of culling. Central to this understanding is defining the spatial distribution of management units and a target population size where the species impact is minimized.

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