Running Header: Drivers and passengers of change

Title: Are invasive species drivers of native species decline or passengers of habitat modification?

A case study of the impact of the common myna (*Acridotheres tristis*) on Australian bird species.

K. Grarock^{*1}, C. R. Tidemann², J.T. Wood³, and D. B. Lindenmayer⁴

Authors:

¹Kate Grarock – Corresponding author

Fenner School of Environment and Society, The Australian National University, Canberra, ACT,
0200, Australia Fenner School of Environment and Society

+ Invasive Animals Cooperative Research Centre, University of Canberra, ACT, 2617, Australia.

kate.grarock@anu.edu.au

T: +84 12 6606 2714

F: +61 2 6125 0746

²Christopher R. Tidemann

Fenner School of Environment and Society, The Australian National University, Canberra, ACT,
0200, Australia Fenner School of Environment and Society

chris.tidemann@anu.edu.au

³Jeffrey T. Wood

Fenner School of Environment and Society, The Australian National University, Canberra, ACT,
0200, Australia Fenner School of Environment and Society

jeff.wood@anu.edu.au

⁴David B. Lindenmayer

Fenner School of Environment and Society, The Australian National University, Canberra, ACT,
0200, Australia Fenner School of Environment and Society

+Australian Research Council Centre of Excellence for Environmental Decisions

+ National Environmental Research Program, Canberra, ACT, Australia.

david.lindenmayer@anu.edu.au

Abstract Habitat modification and invasive species are significant drivers of biodiversity decline. However, distinguishing between the impacts of these two drivers on native species can be difficult. For example, habitat modification may reduce native species abundance, while an invasive species may take advantage of the new environment. This scenario has been described as the driver-passenger model, with ‘passengers’ taking advantage of habitat modification and ‘drivers’ causing native species decline. Therefore, research must incorporate both habitat modification and invasive species impact to successfully investigate native species decline. In this paper, we used the common myna (*Acridotheres tristis*) as a case study to investigate the driver-passenger model. We investigated changes in bird abundance, over two and a half years, in relation to different habitat types and common myna abundance. We hypothesised that the common myna is both a passenger of habitat change and a driver of some bird species decline. Our results indicated that the abundance of many native species is greater in high tree density nature reserves, while the common myna was uncommon in these areas. Common myna abundance was almost three times higher in urban areas than nature reserves and declined rapidly as tree density in nature reserves increased. Our findings indicated that the common myna is primarily a passenger of habitat change. However, we also observed negative associations between common myna abundance and some bird species. We stress the importance of simultaneously investigating both invasive species impact and habitat modification. We suggest habitat restoration could be a useful tool for both native species recovery and invasive species control. Understanding the drivers of native species decline will help inform impact mitigation and direct further research.

Keywords: Introduced species, pest management, Indian myna, *Sturnus tristis*, driver-passenger model.

INTRODUCTION

Both habitat modification and the introduction of invasive species pose major threats to biodiversity across the globe (Clavero and García-Berthou 2005; Davis 2003; Pimentel *et al.* 2005; Sala *et al.* 2000; Westphal *et al.* 2008). Invasive species can often benefit from habitat modification (Chace and Walsh 2006; Didham *et al.* 2005b; Farnsworth 2004; MacDougall and Turkington 2005). However, the impact on native species can be a direct result of the invading species, or it can be compounded by habitat modification (Didham *et al.* 2005b; Gurevitch and Padilla 2004; MacDougall and Turkington 2005). As a result of this complexity, it can be difficult to identify the true cause of native species decline (Didham *et al.* 2005b; MacDougall and Turkington 2005; Shine 2010).

Research often investigates the impact of human-induced habitat modification *or* the effects of invasive species on native species abundance (Didham *et al.* 2005a; Fazey *et al.* 2005). Studies that rely on correlations between native and invasive species abundance allow only weak inferences of impact to be made (Gurevitch and Padilla 2004). In fact, studies reliant on these correlations may mistakenly identify an invasive species as a driver of change (Didham *et al.* 2005b; Gurevitch and Padilla 2004). For example, in New England, USA, native plant species decline was primarily driven by the same habitat variable that promoted invasive species, however, correlation did not imply causation (Farnsworth 2004). In this scenario, the invasive species were ‘passengers’ of habitat modification, not ‘drivers’ of native species decline (Didham *et al.* 2005b; MacDougall and Turkington 2005). Therefore, to successfully investigate native species decline, research must incorporate both habitat modification and invasive species impact (Bradshaw *et al.* 2007; Davis *et al.* 2011; Didham *et al.* 2005b; MacDougall and Turkington 2005; Parker *et al.* 1999).

In this paper, we used the introduced common myna (*Acridotheres tristis*) as an Australian case study to investigate the driver-passenger model. Specifically, we asked whether the common myna is a passenger of habitat modification or a driver of native species decline.

The common myna is listed as one of the world’s worst invasive species (Lowe *et al.* 2000). There is concern that the common myna displaces native species through competitive domination of

cavity-nest sites and territories (Dhami and Nagle 2009; Harper *et al.* 2005; Lowe *et al.* 2011; Pell and Tidemann 1997b). Despite this concern, there is only limited scientific evidence that demonstrates the impact of the common myna (Lowe *et al.* 2011). Some studies have found no evidence of common myna impact on native bird species (Lowe *et al.* 2011; Parsons *et al.* 2006; Wood 1996), while other research postulates that the common myna could drive native species decline (Dhami and Nagle 2009; Grarock *et al.* 2012; Pell and Tidemann 1997b). The species is described as being abundant in modified urban landscapes (Lowe *et al.* 2011; Parsons *et al.* 2003; Peacock *et al.* 2007; Pell and Tidemann 1997a; van Rensburg *et al.* 2009; White *et al.* 2005; Wood 1996), but tends to avoid high density native woodland areas (Crisp and Lill 2006; Lowe *et al.* 2011; Pell and Tidemann 1997a; Tracey *et al.* 2007; White *et al.* 2005).

We investigated changes in bird abundance, over two and a half years, in relation to different habitat types and common myna abundance. Our study was not as robust as controlled removal and long-term before and after experiments (Blossey 1999; Lindenmayer *et al.* 2005; Newton 1998; Ricciardi *et al.* 2000). However, we used present day variations in habitat and invasive species abundance to infer the effects of long-term changes on native species abundance. By including environmental variables in our analysis, we aimed to identify whether the common myna was a driver of native species decline or a passenger of habitat modification.

To investigate the passenger-driver model for the common myna, we developed and tested three hypotheses:

- 1) the common myna would be more abundant in urban habitats than in nature reserves;
- 2) common myna abundance would increase as tree density declined;
- 3) common myna abundance would have a negative impact on the abundance of some cavity-nesting species and small bird species.

Understanding the drivers of native species decline will help direct further research and better inform managers to conduct impact mitigation (Didham *et al.* 2005b; MacDougall and

Turkington 2005). For example, invasive species management may not assist threatened species recovery if habitat quality is the major cause of species decline (Didham *et al.* 2005b).

METHODS

Survey Sites

To investigate our hypotheses, we selected survey sites in the city of Canberra, Australia (Figure 1). Each site was located in a suburban subdivision adjacent to a nature reserve. Sites were restricted to subdivisions greater than 20 years old and were situated at least two kilometers apart. These criteria limited the potential study sites to 23. We estimated tree density in the 23 nature reserves by walking a one kilometer transect scoring the vegetation cover at 20 m intervals. We allocated one point for tree cover overhead or zero points for no tree cover.

In order to select 15 survey sites, we chose 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) (Figure 2a). 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) (Figure 2b). Using a Geographic Information System (GIS) layer of native street trees in urban areas, we observed that tree density fell into the same high, medium and low categories as the adjoining nature reserves. We confirmed this using an ANOVA (Figure 2c). As a result of this finding, we broadly defined each site (consisting of both urban and nature reserve area) as having either high, medium or low tree density.

Bird Surveys

At each site we established two transect surveys. The first transect ran through the nature reserve and the second, at a right-angle to the first, through the urban area (suburb) (Figure 1). Transects were one kilometer in length, 100 m wide in the nature reserves, and 60 m wide in the urban areas. Within each site, we expected that the habitat surrounding one transect may influence the abundance and diversity of birds in the other transect due to their close proximity (1 km²) (Chapman and Reich 2007; Palomino and Carrascal 2007). Therefore, paired transects (in urban and nature reserves) were treated as a single site, with 15 sites in total.

We surveyed our 15 sites every second month (January, March, May, July, September, November) from September 2008 to March 2011. Experienced bird observers identified birds by both sight and call, using line transect surveys. We assigned each observer to a group of three sites (six transects) to survey. Observers walked transects for 20 minutes, within 3 hours of sunrise. We ensured surveys were undertaken in good weather conditions with little or no rain or wind. Fifteen observers completed a total of 1071 transect surveys. Each transect was walked two or three times per survey month.

Preliminary data analysis

Earlier research indicated that the common myna may affect cavity-nesting species and small bird species, but not large bird species (Dhami and Nagle 2009; Grarock *et al.* 2012; Harper *et al.* 2005; Pell and Tidemann 1997b; Tindall *et al.* 2007). We defined cavity-nesting species to be those that primarily nested in cavities and had a mean body weight greater than 3.03 g. Species lighter than 3.03 g (2 g less than the common myna) could potentially occupy cavities that were too small for the common myna. Small species were defined as having a mean body weight of 0.5 g less than the common myna. To conduct the analysis, we selected the 40 most abundant bird species within the following categories: ten cavity-nesting species (>3.03 g), 20 small bird species (≤4.53g) and ten large bird species (>4.53 g) (Appendix S1 and S2). Three of the small bird species can use cavities

for nesting, however their small size (<2.85 g) may limit cavity-nesting competition (Appendix S1). The ten large bird species were not cavity-nesting species. We provide a list of the 40 target species analysed in Appendix S2. We also selected the following variables for analysis: total species abundance and richness, native cavity-nester bird abundance and richness, small native bird abundance and richness, and large native bird abundance and richness. The common myna was not included in any of the above species groupings.

For each transect, we calculated the average number of birds observed per km² (for every species) over the replicated surveys in a month. This reduced the large number of zero values in the data. We then applied a log(x+1) transformation to ensure data were approximately normally distributed.

To check that bird detection rates were similar for both reserve and suburb transects, we assumed that the transect surveys for a season (breeding or non-breeding season) could be treated as replicates. We estimated detection rates for each transect type, for selected species, from binomial mixtures models (Kery, Royle and Schmid, 2005) using the R package unmarked (Fiske and Chandler, 2011). Unfortunately, the low abundance of many species prevented us from performing meaningful detectability analysis. Therefore, we performed detectability analysis on the six most abundant species. These species were the common myna, magpie lark (*Grallina cyanoleuca*), crimson rosella (*Platycercus elegans*), spotted pardalote (*Pardalotus punctatus*), Australian magpie (*Gymnorhina tibicen*) and brown thornbill (*Acanthiza pusilla*).

Autoregressive model analysis

We fitted mixed models in GenStat 14® (VSN International 2011) using restricted maximum likelihood to investigate the factors that influenced: (1) common myna abundance (2) the abundance of the 40 target species, (3) the abundance and richness of particular species groupings. We included the following explanatory variables in the model: area (urban or nature reserve), tree

density (high, medium and low), urban housing density per km², season (breeding: November to March and non-breeding: May to September) and year (1, 2 and 3). We also included common myna abundance as an explanatory variable for all analyses except for those exploring abundance of the common myna itself. We treated site as a random effect and used Wald tests for dropping individual terms from the full fixed model, until only significant (<0.05) explanatory variables remained.

We used the table of effects from the GenStat 14® (VSN International 2011) output, to observe the *influence* (either positive >0.00 or negative <0.00) each significant variable had on species abundance or species richness. We also used the table of predicted means, from the model output, to estimate (reverse log(x+1) transformed) species abundance across different areas.

The breeding season for many species in Canberra starts later than in other areas of Australia due to the altitude (605 m above sea level) and colder weather (Gibbs et al. 2011; Lenz 1979). Therefore, we defined the breeding season as beginning in October and ending in March. We sourced data on house density from the Australian Bureau of Statistics (ABS 2006).

RESULTS

Detection rates

The estimated detection rates in the nature reserve transect (100 m wide) and the urban transect (60 m wide) were not significantly different for four of the six species we analysed (common myna, magpie lark, spotted pardalote, and brown thornbill). Detectability bias may occur for the crimson rosella, Australian magpie and other species not analysed. Therefore, results for these species should be treated with some caution. However, we believe bird survey estimates represent a constant proportion of the actual population present. Our study did not attempt to quantify the exact

number of individuals present at each site, rather we examined the general influence of habitat and the common myna on bird abundance.

Common myna response to habitat variation

We found significant relationships between the abundance of the common myna and urban area, tree density, season and year (Table 1). The reverse $\log(x+1)$ transformed predicted means (derived from the model in Table 1), indicated that common myna abundance in nature reserves was 13.6 birds per km^2 , while in urban areas it was almost three times higher at 41.1 birds per km^2 . This finding supported our hypothesis that the common myna would be more abundant in urban habitats than in nature reserves.

We also identified a significant negative relationship between common myna abundance and tree density in nature reserves ($F_{2,14}=23.4$, $P<0.001$). The predicted means, derived from the model in Table 1, indicated that common myna abundance in low tree density nature reserves was 44.2 birds per km^2 , 18.5 birds per km^2 for medium tree density and 2.5 birds per km^2 for high tree density nature reserves. This finding supported our hypothesis that common myna abundance would increase as tree density declines.

Tree density in urban areas also influenced common myna abundance. The predicted means, derived from the model in Table 1, indicated that high tree density urban areas had lower common myna numbers (27.8 birds per km^2) than in medium (55.5 birds per km^2) and low (44.7 birds per km^2) tree density areas. Therefore, the common myna was strongly influenced by habitat and appeared to be a passenger of habitat change.

Species abundance and richness

Total species abundance was positively associated with high tree density areas ($F_{2,19}=39.5$, $P<0.001$), the non-breeding season ($F_{1,130}=15.9$, $P<0.001$), later years ($F_{2,129}=9.7$, $P<0.001$) and

abundance of the common myna ($F_{1,140}=8.7$, $P=0.004$). We found no significant relationship between common myna abundance and total species richness (Table 1).

Cavity-nesting species

We did not identify any significant relationships between common myna abundance and native cavity-nesting species abundance or richness (Table 1). Native cavity-nester abundance was positively associated with urban areas ($F_{1,133}=21.1$, $P<0.001$) and the non-breeding season ($F_{1,133}=29.5$, $P<0.001$). Native cavity-nester richness was positively associated with nature reserve areas ($F_{1,131}=22.9$, $P<0.001$), high tree density areas ($F_{2,14}=5.8$, $P=0.037$) and the non-breeding season ($F_{1,131}=7.4$, $P=0.007$) (Table 1).

We found a negative relationship between the abundance of the common myna and the abundance of the gang-gang cockatoo (*Callocephalon fimbriatum*) ($F_{1,145}=11.8$, $P<0.001$) and crimson rosella ($F_{1,145}=4.5$, $P=0.036$) (Appendix S2). These findings supported our hypothesis that the common myna would negatively affect the abundance of some cavity-nesting species and, therefore, potentially drive the decline of these species.

We observed significant positive relationships between common myna abundance and three cavity-nesting species: the eastern rosella (*Platycercus eximius*) ($F_{1,646}=10.5$, $P=0.001$), red-rumped parrot (*Psephotus haematonotus*) ($F_{1,108}=7.6$, $P=0.007$) and common starling (*Sturnus vulgaris*) ($F_{1,122}=19.1$, $P<0.001$). No significant relationships were observed between common myna abundance and little corolla (*Cacatua sanguinea*), sulphur-crested cockatoo (*Cacatua galerita*), Australian king-parrot (*Alisterus scapularis*) or laughing kookaburra (*Dacelo novaeguineae*) abundance (Appendix S2).

Small bird species

Our research identified significant negative relationships between common myna abundance and small native species abundance ($F_{1,141}=4.5$, $P=0.035$) and small native bird species richness ($F_{1,143}=21.3$, $P<0.001$) (Table 1). This supported our hypothesis that the common myna would negatively affect the abundance of some small bird species. We observed a significant positive relationship between nature reserve areas and small native species richness ($F_{1,134}=30.4$, $P<0.001$) (Table 1). Urban areas with high tree density supported a greater abundance of small native species than urban areas with medium or low tree density (Table 1).

We identified significant negative relationships between common myna abundance and the abundance of eight of the 20 small bird species we examined (Appendix S2). These eight species were: white-throated treecreeper (*Cormobates leucophaea*), spotted pardalote, speckled warbler (*Pyrrholaemus saggitatus*), brown thornbill, buff-rumped thornbill (*Acanthiza reguloides*), eastern spinebill (*Acanthorhynchus tenuirostris*), grey fantail (*Rhipidura fuliginosa*) and grey butcher bird (*Cracticus torquatus*). The brown thornbill, buff-rumped thornbill and grey fantail were significantly more likely to be found in nature reserves than in urban areas (Appendix S2). These findings supported our hypothesis that the common myna would negatively affect the abundance of some small bird species and therefore, potentially drive the decline of these species.

Large bird species

We found a significant positive relationship between large native bird abundance and common myna abundance ($F_{1,144}=11.6$, $P<0.001$). We did not identify a significant relationship between common myna abundance and large native bird species richness (Table 1). Large native bird abundance was significantly higher in urban areas ($F_{1,135}=195.0$, $P<0.001$), while large native bird species richness was significantly lower in these areas ($F_{1,131}=40.6$, $P<0.001$) (Table 1).

No negative relationships between common myna abundance and the abundance of any of the ten large bird species were observed (Appendix S2).

DISCUSSION

Our investigation of the common myna in Canberra provided a valuable case study to test the driver-passenger model. We used linear models to investigate changes in bird abundance and make inferences about the different drivers of native species decline. The key findings of our study were:

1. The common myna appeared to be primarily a passenger of habitat change.
2. In combination with habitat change, the common myna also appeared to be a driver of some bird species decline.
3. It is essential to simultaneously investigate invasive species impact and habitat modification to assist with differentiating between the impacts on native species.

The common myna was strongly influenced by tree density and showed a preference for urban areas. We did not find negative associations between common myna abundance and total species abundance and richness, native cavity-nester abundance and richness or large native bird abundance and richness. These results are consistent with the conclusions of Lowe *et al.* (2011) and Parsons *et al.* (2003) that the common myna is a passenger of habitat modification. However, unlike these earlier studies, we did find that small native species abundance and richness and some individual cavity-nesting species and small bird species, were negatively associated with the abundance of the common myna. This indicates that the common myna may also drive the decline of *some* bird species. This finding is supported by our previous research (Grarock *et al.* 2012) that investigated the long-term abundance of bird species, pre- and post common myna invasion of Canberra, Australia.

Clearly not all causal variables have been included in our model with some unexpected results observed. For example, common myna abundance had a positive association with eastern rosella abundance. There may be a complex relationship between reductions in other cavity-nesting

species (for example the crimson rosella) and increases in eastern rosella abundance. We also did not attempt to include the influence of other species such as the noisy miner (*Manorina melanocephala*) or the domestic cat (*Felis catus*) on native bird species abundance. The patterns observed in our study (limited to 15 sites in Canberra) may differ from those in other bioregions and in areas experiencing different stages of common myna invasion.

We suggest tree density is the key difference between our study and other investigations that found no negative correlations between common myna abundance and native species (Lowe *et al.* 2011; Parsons *et al.* 2003). Our study encompassed low, medium and high tree density sites, while Lowe *et al.* (2011) described their study site as having high tree density.

Low tree density in nature reserves, or fragmentation of native vegetation, may enhance habitat quality for the common myna enabling the species to spread into new areas and compete for resources with native species (Lowe *et al.* 2011; Tracey *et al.* 2007). Low tree density sites are probably similar to the natural environment of the common myna in India and central and southern Asia (Feare and Craig 1998; Pell and Tidemann 1997a).

We observed that the abundance of many native species was greater in nature reserves than in urban areas, and that abundance increased further with higher tree density. We also observed that the common myna was uncommon in high tree density nature reserves. Common myna abundance was almost three times higher in urban areas than nature reserves and declined rapidly as tree density in nature reserves increased. This observation is supported by studies that documented high densities of the common myna in disturbed and urban habitats (Crisp and Lill 2006; Dhami and Nagle 2009; Feare and Craig 1998; Lowe *et al.* 2011; Parsons *et al.* 2003; Pell and Tidemann 1997a; Perry *et al.* 2009; van Rensburg *et al.* 2009; White *et al.* 2005). Other studies have also reported increased native species abundance in response to greater native tree density (Blair 2001; Clergeau *et al.* 1998; Crooks *et al.* 2004; Gardali and Holmes 2011; Munro *et al.* 2009; Sewell and Catteral 1998).

The results of our study suggest that the effects of habitat modification and invasive species are interrelated. Many species are strongly influenced by habitat, with greater abundance in high quality habitat than in low quality habitat (Kajzer *et al.* 2011). However, high quality habitat for one species may not constitute high quality habitat for another species (Didham *et al.* 2005b; Farnsworth 2004; MacDougall and Turkington 2005). Our results indicated that both human habitat modification and the common myna had negative impacts on some bird species. Therefore, in combination, habitat modification and the introduction of invasive species could have a substantial impact on native taxa (Gibbons and Lindenmayer 2002; Lindenmayer *et al.* 2009).

Given the influence of habitat on species abundance, we suggest that habitat restoration and tree planting may be useful tools to both control common myna abundance and aid native bird species recovery. Other Australian studies have suggested vegetation regeneration and dense planting of vegetation in reserves could be used as a method to control bird species such as the noisy miner and the bell miner (*Manorina melanophrys*) (Clarke and Schedvin 1999; Kemmerer *et al.* 2008; Lindenmayer *et al.* 2010; Montague-Drake *et al.* 2011; Taylor *et al.* 2008).

ACKNOWLEDGEMENTS

We thank volunteer bird observers, including: Barbara Allen, Heather Allsopp, Judith Bourne, John Brannan, Malcolm Fyfe, Bill Handke, Owen Holton, Anne I'Ons, Daryl King, Sue Lashko, Barbara Levings, Bruce Lindenmayer, Chris Marsh and Peter Ormay. We also thank Georgia Davis, Hamish Dalley and six anonymous reviewers. This work was undertaken in accordance with animal ethics approval Protocol No. C.RE.51.08 and is supported in part by Stuart Leslie Bird Research Award, BirdLife Australia. The views expressed herein do not necessarily reflect the views of this organisation.

REFERENCES

- ABS. (2006) 'Census of Population and Housing, Australian Capital Territory (1996, 2001, 2006). Australian Bureau of Statistics, Canberra.
- Blair R. B. (2001) *Creating a homogeneous avifauna*. Kluwer Academic Publishers, Boston.
- Blossey B. (1999) Before, during and after: The need for long-term monitoring in invasive plant species management. *Biological invasions* **1**, 301-11.
- Bradshaw C. J. A., Field I. C., Bowman D. M. J. S., Haynes C. & Brook B. W. (2007) Current and future threats from non-indigenous animal species in northern Australia: A spotlight on World Heritage Area Kakadu National Park. *Wildlife Research* **34**, 419-36.
- Chace J. F. & Walsh J. J. (2006) Urban effects on native avifauna: A review. *Landscape and Urban Planning* **74**, 46-69.
- Chapman K. A. & Reich P. B. (2007) Land use and habitat gradients determine bird community diversity and abundance in suburban, rural and reserve landscapes of Minnesota, USA. *Biological Conservation* **135**, 527-41.
- Clarke M. F. & Schedvin N. (1999) Removal of bell miners *Manorina melanophrys* from Eucalyptus radiata forest and its effect on avian diversity, psyllids and tree health. *Biological Conservation* **88**, 111-20.
- Clavero M. & García-Berthou E. (2005) Invasive species are a leading cause of animal extinctions. *Trends in Ecology and Evolution* **20**, 110.
- Clergeau P., Savard J. P. L., Mennechez G. & Falardeau G. (1998) Bird abundance and diversity along an urban-rural gradient: A comparative study between two cities on different continents. *Condor* **100**, 413-25.
- Crisp H. & Lill A. (2006) City slickers: Habitat use and foraging in urban Common Mynas *Acridotheres tristis*. *Corella* **30**, 9-15.

- 394 Crooks K. R., Suarez A. V. & Bolger D. T. (2004) Avian assemblages along a gradient of
395 urbanization in a highly fragmented landscape. *Biological Conservation* **115**, 451-62.
- 396 Davis M. A. (2003) Biotic globalization: Does competition from introduced species threaten
397 biodiversity? *BioScience* **53**, 481-9.
- 398 Davis M. A., Chew M. K., Hobbs R. J., Lugo A. E., Ewel J. J., Vermeij G. J., Brown J. H.,
399 Rosenzweig M. L., Gardener M. R., Carroll S. P., Thompson K., Pickett S. T. A., Stromberg
400 J. C., Tredici P. D., Suding K. N., Ehrenfeld J. G., Philip Grime J., Mascaro J. & Briggs J.
401 C. (2011) Don't judge species on their origins. *Nature* **474**, 153-4.
- 402 Dhami M. K. & Nagle B. (2009) Review of the biology and ecology of the common myna
403 (*Acridotheres tristis*) and some implications for management of this invasive species
404 [Internet document]. Report undertaken for the Pacific Invasives Initiative Auckland, New
405 Zealand. [Cited 30 Jan 2013.] Available from URL:
406 [http://www.issg.org/cii/Electronic%20references/pii/references/pii_review_of_the_biology_](http://www.issg.org/cii/Electronic%20references/pii/references/pii_review_of_the_biology_and_ecology_of_the_common_myna_and_some_implications_for_management.pdf)
407 [and_ecology_of_the_common_myna_and_some_implications_for_management.pdf](http://www.issg.org/cii/Electronic%20references/pii/references/pii_review_of_the_biology_and_ecology_of_the_common_myna_and_some_implications_for_management.pdf).
- 408 Didham R. K., Ewers R. M. & Gemmell N. J. (2005a) Comment on "Avian extinction and
409 mammalian introductions on oceanic islands". *Science* **307**, 1412.
- 410 Didham R. K., Tylianakis J. M., Hutchison M. A., Ewers R. M. & Gemmell N. J. (2005b) Are
411 invasive species the drivers of ecological change? *Trends in Ecology & Evolution* **20**, 470-4.
- 412 Farnsworth E. J. (2004) Patterns of plant invasions at sites with rare plant species throughout New
413 England. *Rhodora* **106**, 97-117.
- 414 Fazey I., Fischer J. & Lindenmayer D. B. (2005) What do conservation biologists publish?
415 *Biological Conservation* **124**, 63-73.
- 416 Feare C. & Craig A. (1998) *Starlings and Mynas*. Helm, London.
- 417 Fiske I. & Chandler, R. B. (2011) Unmarked: An R package for fitting hierarchical models of
418 wildlife occurrence and abundance. *Journal of Statistical Software* **43**(10) 1-23.
- 419 Gardali T. & Holmes A. L. (2011) Maximizing benefits from riparian revegetation efforts: Local-
420 and landscape-level determinants of avian response. *Environmental Management* **48**, 28-37.

- 421 Gibbons P. & Lindenmayer D. B. (2002) *Tree Hollows and Wildlife Conservation in Australia*.
422 CSIRO Publishing, Collingwood.
- 423 Gibbs H. M., Chambers L. E. & Bennett A. F. (2011) Temporal and spatial variability of breeding
424 in Australian birds and the potential implications of climate change. *Emu* **111**, 283-91.
- 425 Garrock K., Tidemann C. R., Wood J. & Lindenmayer D. B. (2012) Is it benign or is it a pariah?
426 Empirical evidence for the impact of the common myna (*Acridotheres tristis*) on Australian
427 birds. PLoS ONE 7(7): e40622. doi:10.1371/journal.pone.0040622.
- 428 Gurevitch J. & Padilla D. K. (2004) Are invasive species a major cause of extinctions? *Trends in*
429 *Ecology and Evolution* **19**, 470-4.
- 430 Harper M. J., McCarthy M. A. & van der Ree R. (2005) The use of nest boxes in urban natural
431 vegetation remnants by vertebrate fauna. *Wildlife Research* **32**, 509-16.
- 432 Kajzer J., Lenda M., Kośmicki A., Bobrek R., Kowalczyk T., Martyka R. & Skórka P. (2011) Patch
433 occupancy and abundance of local populations in landscapes differing in degree of habitat
434 fragmentation: A case study of the colonial black-headed gull, *Chroicocephalus ridibundus*.
435 *Journal of biogeography* **39**, 371-81.
- 436 Kemmerer E. P., Shields J. M. & Tidemann C. R. (2008) High densities of bell miners *Manorina*
437 *melanophrys* associated with reduced diversity of other birds in wet eucalypt forest:
438 Potential for adaptive management. *Forest Ecology and Management* **255**, 2094-102.
- 439 Kery M., Royle J. A. & Schmid H. (2005) Modeling avian abundance from replicated counts using
440 binomial mixture models. *Ecological Applications* **15**, 1450-1461.
- 441 Lenz M. (1979) Bird recording in the A.C.T. - the need for improvement and a change in policy:
442 breeding birds census. *Canberra Bird Notes* **4**, 2-7.
- 443 Lindenmayer D. B., Cunningham R. B. & Peakall R. O. D. (2005) The recovery of populations of
444 bush rat *Rattus fuscipes* in forest fragments following major population reduction. *Journal*
445 *of Applied Ecology* **42**, 649-58.

- 446 Lindenmayer D. B., Knight E. J., Crane M. J., Montague-Drake R., Michael D. R. & MacGregor C.
447 I. (2010) What makes an effective restoration planting for woodland birds? *Biological*
448 *Conservation* **143**, 289-301.
- 449 Lindenmayer D. B., Welsh A., Donnelly C., Crane M., Michael D., Macgregor C., McBurney L.,
450 Montague-Drake R. & Gibbons P. (2009) Are nest boxes a viable alternative source of
451 cavities for hollow-dependent animals? Long-term monitoring of nest box occupancy, pest
452 use and attrition. *Biological Conservation* **142**, 33-42.
- 453 Lowe K., Taylor C. & Major R. (2011) Do Common Mynas significantly compete with native birds
454 in urban environments? *Journal of Ornithology* **152**, 909-21.
- 455 Lowe S., Browne M., Boudjelas S. & Poorter M. D. (2000) 100 of the World's Worst Invasive
456 Alien Species A selection from the Global Invasive Species Database. In: *First published as*
457 *special lift-out in Aliens 12, December 2000. Updated and reprinted version: November*
458 *2004* p. 7. The Invasive Species Specialist Group (ISSG) a specialist group of the Species
459 Survival Commission (SSC) of the World Conservation Union (IUCN).
- 460 MacDougall A. S. & Turkington R. (2005) Are invasive species the drivers or passengers of change
461 in degraded ecosystems? *Ecology* **86**, 42-55.
- 462 Montague-Drake R. M., Lindenmayer D. B., Cunningham R. B. & Stein J. A. (2011) A reverse
463 keystone species affects the landscape distribution of woodland avifauna: A case study
464 using the Noisy Miner (*Manorina melanocephala*) and other Australian birds. *Landscape*
465 *Ecology* **26**, 1383-94.
- 466 Munro N. T., Fischer J., Wood J. & Lindenmayer D. B. (2009) Revegetation in agricultural areas:
467 The development of structural complexity and floristic diversity. *Ecological applications*
468 **19**, 1197-210.
- 469 Newton I. (1998) *Population limitation in birds*. Academic Press, London.
- 470 Palomino D. & Carrascal L. M. (2007) Threshold distances to nearby cities and roads influence the
471 bird community of a mosaic landscape. *Biological Conservation* **140**, 100-9.

- 472 Parker I. M., Simberloff D., Lonsdale W. M., Goodell K., Wonham M., Kareiva P. M., Williamson
473 M. H., Von Holle B., Moyle P. B., Byers J. E. & Goldwasser L. (1999) Impact: Toward a
474 framework for understanding the ecological effects of invaders. *Biological invasions* **1**, 3-
475 19.
- 476 Parsons H., French K. & Major R. E. (2003) The influence of remnant bushland on the composition
477 of suburban bird assemblages in Australia. *Landscape and Urban Planning* **66**, 43-56.
- 478 Parsons H., Major R. E. & French K. (2006) Species interactions and habitat associations of birds
479 inhabiting urban areas of Sydney, Australia. *Austral Ecology* **31**, 217-27.
- 480 Peacock D. S., Van Rensburg B. J. & Robertson M. P. (2007) The distribution and spread of the
481 invasive alien Common Myna, *Acridotheres tristis* L. (Aves: *Sturnidae*), in southern Africa.
482 *South African Journal of Science* **103**, 465-73.
- 483 Pell A. S. & Tiedemann C. R. (1997a) The ecology of the Common Myna in urban nature reserves in
484 the Australian Capital Territory. *Emu* **97**, 141-9.
- 485 Pell A. S. & Tiedemann C. R. (1997b) The impact of two exotic hollow-nesting birds on two native
486 parrots in savannah and woodland in eastern Australia. *Biological Conservation* **79**, 145-53.
- 487 Perry L. G., Cronin S. A. & Paschke M. W. (2009) Native cover crops suppress exotic annuals and
488 favor native perennials in a greenhouse competition experiment. *Plant Ecology* **204**, 247-59.
- 489 Pimentel D., Zuniga R. & Morrison D. (2005) Update on the environmental and economic costs
490 associated with alien-invasive species in the United States. *Ecological Economics* **52**, 273-
491 88.
- 492 Ricciardi A., Steiner W. W. M., Mack R. N. & Simberloff D. (2000) Toward a global information
493 system for invasive species. *BioScience* **50**, 239-44.
- 494 Sala O. E., Chapin III F. S., Armesto J. J., Berlow E., Bloomfield J., Dirzo R., Huber-Sanwald E.,
495 Huenneke L. F., Jackson R. B., Kinzig A., Leemans R., Lodge D. M., Mooney H. A.,
496 Oesterheld M., Poff N. L., Sykes M. T., Walker B. H., Walker M. & Wall D. H. (2000)
497 Global biodiversity scenarios for the year 2100. *Science* **287**, 1770-4.

- Sewell S. R. & Catterall C. P. (1998) Bushland modification and styles of urban development: Their effects on birds in south-east Queensland. *Wildlife Research* **25**, 41-63.
- Shine R. (2010) The Ecological Impact of Invasive Cane Toads (*Bufo Marinus*) in Australia. *The Quarterly Review of Biology* **85**, 253-91.
- Taylor R. S., Oldland J. M. & Clarke M. F. (2008) Edge geometry influences patch-level habitat use by an edge specialist in south-eastern Australia. *Landscape Ecology* **23**, 377-89.
- Tindall S. D., Ralph C. J. & Clout M. N. (2007) Changes in bird abundance following Common Myna control on a New Zealand island. *Pacific Conservation Biology* **13**, 202-12.
- Tracey J., Bomford M., Hart Q., Saunders G. & Sinclair R. (2007) *Managing Bird Damage to Fruit and Other Horticultural Crops*. Bureau of Rural Sciences, Canberra.
- van Rensburg B. J., Peacock D. S. & Robertson M. P. (2009) Biotic homogenization and alien bird species along an urban gradient in South Africa. *Landscape and Urban Planning* **92**, 233-41.
- VSN International. (2011) GenStat 14. VSN International, Hemel Hempstead, UK.
- Westphal M. I., Browne M., MacKinnon K. & Noble I. (2008) The link between international trade and the global distribution of invasive alien species. *Biological invasions* **10**, 391-8.
- White J. G., Antos M. J., Fitzsimons J. A. & Palmer G. C. (2005) Non-uniform bird assemblages in urban environments: The influence of streetscape vegetation. *Landscape and Urban Planning* **71**, 123-35.
- Wood K. A. (1996) Bird assemblages in a small public reserve and adjacent residential area at Wollongong, New South Wales. *Wildlife Research* **23**, 605-20.

Table 1. Significant explanatory variables for restricted maximum likelihood analysis of common myna (*Acridotheres tristis*) abundance and species groupings in Canberra, South East Australia.

Species/ Species	Area	Tree density	Human	Season	Year	common

Group						Housing Density (km ²)				myna abundance
	nature to urban	nature		urban			breeding to non- breeding	year 1 to year 2	year 1 to year 3	
		high to med	high to low	high to med	high to low					
common myna (<i>Acridotheres tristis</i>)	0.91 ±0.10 p<0.001	0.74 ±0.24 p<0.001	1.11 ±0.24 p<0.001	0.29 ±0.24 p<0.001	0.20 ±0.24 p<0.001		-0.24 ±0.07 p=0.002	-0.15 ±0.08 p=0.013	-0.23 ±0.08 p=0.013	N/A
Total Species abundance		-0.05 ±0.06 p<0.001	-0.12 ±0.06 p<0.001	-0.06 ±0.06 p<0.001	-0.16 ±0.06 p<0.001		0.08 ±0.02 p<0.001	0.07 ±0.02 p<0.001	0.09 ±0.02 p<0.001	0.07 ±0.02 p=0.004
Total Species richness	0.01 ±0.02 p=0.003	0.06 ±0.03 p=0.018	-0.03 ±0.03 p=0.018	0.01 ±0.03 p=0.018	-0.07 ±0.03 p=0.018	-0.0003 ±0.0001 p=0.020	-0.02 ±0.01 p=0.002	0.03 ±0.01 p=0.003	0.03 ±0.01 p=0.003	
Native cavity-nester bird abundance	0.12 ±0.03 p<0.001						0.15 ±0.03 p<0.001			
Native cavity-nester bird richness	-0.04 ±0.02 p<0.001	-0.01 ±0.04 p=0.037	-0.13 ±0.04 p=0.037	-0.03 ±0.04 p=0.037	-0.13 ±0.04 p=0.037		0.03 ±0.01 p=0.007			
*Small native bird abundance		0.09 ±0.01 p=0.002	-0.08 ±0.01 p=0.002	-0.04 ±0.01 p=0.002	-0.23 ±0.01 p=0.002	-0.001 ±0.0004 p=0.029				-0.08 ±0.04 p=0.035
*Small natives richness	-0.04 ±0.02 p<0.001						-0.03 ±0.02 p=0.049	0.05 ±0.02 p=0.003	0.05 ±0.02 p=0.003	-0.06 ±0.02 p<0.001
Large native bird abundance	0.20 ±0.02 p<0.001						0.11 ±0.02 p<0.001	0.07 ±0.02 p<0.001	0.08 ±0.02 p<0.001	0.09 ±0.02 p<0.001
Large native bird richness	-0.05 ±0.01 p<0.001						-0.03 ±0.01 p=0.002	-0.01 ±0.01 p=0.028	-0.03 ±0.01 p=0.028	

524

525

526

527

528

529

Analysis initially included all explanatory variables in the model and then we sequentially removed non-significant variables until only significant (<0.05) variables remained. Values in the table indicate the *effect* (either positive >0.00 or negative <0.00) each variable had on log(x+1) transformed species abundance or species richness (±SE). For example, the positive value of 0.91 indicates that common myna abundance was greater in urban areas than in nature reserves. Nature reserve is given a zero value and the effect of urban areas relative to nature reserve is reported.

Likewise, tree density is reported as a deviation from high tree density to either medium or low tree density, season is reported as a deviation from breeding season to non-breeding season and year is reported as a deviation from year 1 to either year 2 or year 3. The values reported for human housing density and the common myna give the coefficients of these variables. An asterisk (*) indicates species groupings that were negatively associated with common myna abundance.

Figure 1. Study area and location of the 15 survey sites in Canberra, South East Australia. Sites were located in a suburb greater than 20 years old and adjacent to a nature reserve. At each site we set up two, one km, line transect surveys. Transects were one kilometer in length, 100 m wide in the nature reserves and 60 m wide in the urban areas.

Figure 2. Tree density of the 15 sites across Canberra, South East Australia. (a) Examples of the

high, medium and low tree densities categories in nature reserves. (b) Graph of transect mean tree density score ($\pm$ SE) of nature reserve across the three tree density categories. (c) Graph of the mean tree density per km² ($\pm$ SE) of urban areas across the three tree density categories. Tree density in suburbs ($F_{2,12}=22.0$, $P<0.001$) and nature reserves ($F_{2,12}=12.5$, $P=0.001$) was significantly different between the high, medium and low tree density sites.

SUPPORTING INFORMATION

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

Appendix S1: List of species observed throughout the survey period on transect surveys and their relevant species groupings.

Appendix S2: Significant explanatory variables for restricted maximum likelihood analysis of 10 cavity-nesting species, 20 small bird species and 10 large bird species in Canberra, South East Australia.

Figure 1

Figure 2