Global variation in the availability of data on the environmental impacts of alien birds

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Abstract

Alien birds are widely distributed across the globe, but information on their environmental impacts is available for less than a quarter of the regions in which they are located. We test a series of hypotheses better to understand why impact data are available for some regions but not others. Information on factors hypothesised to influence spatial variation in the availability of impact were collated for 60 regions with actual, recorded alien bird impacts, and 187 regions without. These data were analysed using mixed effects models. The characteristics of alien bird invasions most strongly influence the availability of impact data, which are more likely to be available for regions with higher alien bird species richness and longer alien bird residence times. There are many regions of the world that lack impact data but are characterised by high alien bird species richness and long alien bird residence times: it is likely that the impacts of alien birds are going unnoticed within them. To a lesser extent, impact data are also more likely to be available for regions characterised by higher economic development. Improving the capacity for research amongst less developed regions may therefore be a key strategy to improve our understanding of the impacts of alien birds. Impact data availability was not found to be associated with impact severity, and therefore we cannot conclude that regions lacking impact data do so because the impacts sustained within them are less severe.

Keywords

Alien birds; Biological invasions; Data deficient; Impact data; Alien species richness; Human development

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Introduction

Alien species are broadly distributed across the globe (Turbelin, Malamud and Francis, 2017) and their numbers continue to rise (Seebens *et al.*, 2017). One-sixth of the global land surface is highly vulnerable to biological invasions (Early *et al.*, 2016), and while the impacts of some alien species are relatively benign (Goodenough, 2010), others can be particularly damaging to native biodiversity. For example, alien species threaten the existence of one-quarter of all bird and amphibian species listed as threatened on the IUCN Red List (Bellard, Genovesi and Jeschke, 2016), and have been associated with the extinction of more species since 1500CE than any other cause (Blackburn, Bellard and Ricciardi, 2019).

In recent years, progress has been made to improve our understanding of the types of alien species that cause the most damage. The development of protocols to quantify and categorise the impacts of alien species, such as the Environmental Impact Classification for Alien Taxa (EICAT: Blackburn et al. 2014), have enabled useful comparisons regarding the severity and type of impacts generated by alien species within and across different taxonomic groups and geographic regions (e.g. Kumschick & Nentwig, 2010; Kumschick et al. 2015a; Measey et al. 2016). This may assist in directing management interventions to the most damaging species and most severely affected locations. However, our understanding of alien species impacts, and hence our ability to manage them, is compromised by the fact that for many regions of the world we have no data on their environmental impacts (Kumschick et al. 2015b). Alien birds are no exception: they are present in 247 regions of the world but data on their environmental impacts is only available for 60 (24%) of these regions (Evans et al. Submitted).

From the data that are available, we know that the severity of impacts generated by alien birds varies substantially across regions: impacts in some regions are negligible, whilst in others they are severe, and include native species extirpations and extinctions (Evans et al. 2016; Evans et al. 2018a). For example, predation by the great horned owl (*Bubo virginianus*) has contributed to the extinction of the red-moustached fruit-dove (*Ptilinopus mercierii*) on Hiva Oa (Marquesas Islands: (Shine, Reaser and Guiterrez, 2003), while competition with the Japanese white eye (*Zosterops japonicus*) has caused a collapse in the population of the Hawaii akepa (*Loxops coccineus*) on Mauna Kea (Freed, Cann and Bodner, 2008). The number and type of alien bird species that have been introduced to regions for which we have no impact data varies substantially. Therefore, whilst it is possible that all 187 of these regions lack impact data because alien birds have no impacts within them, it is more likely that impact severity also varies across these regions: impacts are likely to be negligible in some, but severe in others. Spatial variation in the availability of impact data is thus a significant knowledge gap for conservation science. If we could identify the factors that influence spatial variation in the availability of impact data, it may help us to understand why alien birds remain unstudied in certain regions, and may also reveal where the impacts of alien birds are likely to be going unnoticed.

There are at least three broad reasons why impact data may be available for some regions but not others. First, alien bird species may be more readily available for study in some regions. Here, availability may relate to the characteristics of invasions sustained by a region, and to the characteristics of the invading bird species. For example, with regard to the characteristics of invasions, more impact data may be available for regions that have been subject to invasions for longer periods of time, as there will have been greater opportunity to study alien birds in these regions. Regions with greater alien bird species richness are also less likely to lack impact data, as the opportunity to study alien birds in these regions will be increased. With regard to species characteristics, more impact data may be available for regions supporting widespread alien birds, as species occupying a broad area are more likely to be noticed and studied. Regions supporting generalist alien bird species (both in terms of their diet and habitat preferences) may also be less likely to lack impact data, as these species tend to use and occupy a broader range of habitats (Carrascal *et al.*, 2008) and are therefore more likely to be observed. More impact data may be available for regions supporting large-brained alien birds, as this trait (which is an indicator of ecological flexibility) has been shown to facilitate colonisation of variable habitats by birds (Fristoe, Iwaniuk and Botero, 2017), to be associated with bird species exposed to greater environmental variation (Sayol *et al.*, 2016), to be linked to increased abundance in native birds (Shultz *et al.*, 2005) and to higher survival rates for alien birds and mammals (Sol *et al.*, 2007, 2008). Large brained birds are also more tolerant of urban environments (Maklakov *et al.*, 2011; Sol *et al.*, 2014) which brings them into direct contact with humans where their impacts may be noticed, and they often possess charismatic traits that attract research (e.g. Auersperg et al. 2011). More impact data may also be available for regions supporting greater numbers of conspicuous alien bird species (e.g. those that are large, brightly coloured, or have loud calls), because these species are more likely to be noticed and recorded (*sensu* McCallum 2005).

Second, the severity of alien bird impacts sustained by a region may also influence the availability of impact data, because alien bird species with damaging impacts tend to be more frequently studied than those with minor impacts (Martin-Albarracin et al. 2015; Evans et al. 2016; Evans et al. 2018b). Indeed, amongst alien species more generally, those with documented impacts are more frequently studied than species with no documented impacts (Pyšek *et al.*, 2008). A number of factors have been linked to the severity of impacts generated by alien birds. Widespread alien bird species have more severe impacts (Evans et al. 2018a), and so do generalist alien birds (Shirley & Kark, 2009; Kumschick et al. 2013; Evans et al. 2014; Evans et al. 2018a). Alien bird impacts have been found to be more severe on oceanic islands in comparison to continents (Evans, Kumschick and Blackburn, 2016), and on small islands (<100km^2) in comparison to larger islands and mainland locations (Evans et al. Submitted). Regions with low native species richness have also been found to sustain more severe alien bird impacts (Evans et al. Submitted). In the above cases, regions that lack impact data would be those less likely to sustain severe impacts.

Third, the characteristics of a region may influence the availability of impact data. Uninhabited or sparsely populated regions are more likely to lack impact data, as the impacts of alien species may go unnoticed in these regions. Less impact data may be available for regions with lower levels of economic development where capacity to undertake and publish research on the impacts of alien species is limited (for example due to a lack of political will or resources). Indeed, invasive species research has generally been found to focus on regions in the developed world (Pyšek et al. 2008; Bellard & Jeschke 2015). In the above cases, data availability would be unrelated to the severity of alien bird impacts sustained by a region.

Here, we test a series of hypotheses (H) to better understand why data on the environmental impacts of alien birds is available for some regions but not for others. Based on the factors discussed above, we expect to find proportionately more regions lacking impact data amongst those: (H1) supporting alien birds with shorter residence times, (H2) with lower levels of alien bird species richness, (H3) supporting alien birds with smaller alien ranges, (H4) supporting specialist alien birds, (H5) supporting alien birds with smaller relative brain sizes, (H6) supporting less conspicuous alien birds, (H7) which are continents or large islands (in comparison to islands <100km^2), (H8) with higher levels of native bird species richness, (H9) with lower human population densities, and / or (H10) with lower economic development.

Methods

Data

A list of 119 alien bird species with documented impacts to biodiversity was taken from Evans et al. (2016). A further 296 alien bird species with self-sustaining populations globally have no recorded data on impacts (these species are categorised as Data Deficient (DD) under EICAT). These species were not included in this study, as to include them would be to assume that they actually have impacts to native biodiversity, when whether or not they do is unknown.

A list of 247 regions of the world known to support one or more of the 119 alien birds, comprising 60 regions with recorded alien bird impacts and 187 regions without, was taken from Evans et al. (Submitted). Regions were delineated following the Natural Earth mapping dataset (1:10 million, map subunits: http://www.naturalearthdata.com/downloads/10m-cultural-vectors, downloaded 13 February 2019), which identifies regions that are not contiguous but part of the same country, including islands. Thus, for example, mainland Australia, Tasmania and Macquarie Island represent three separate regions.

Data on the following variables (V) were assembled to test the 12 hypotheses listed in the Introduction.

V1 (residence time): Regional alien bird residence time scores (the average residence time (in years) for all alien bird species present in a region).

V2 (alien bird species richness): Regional alien bird species richness scores (the number of alien bird species present in a region).

V3 (widespread alien birds): Regional alien range size scores (the geometric mean of the alien range sizes of all alien bird species present in a region).

V4 – V6 (habitat and diet specialist alien birds): Regional habitat and diet breadth scores (the average habitat breath score and diet breadth score for all alien bird species present in a region), calculated following Evans et al. (Submitted). Regions supporting generalist alien bird species have higher average scores. As native range size is also a measure of generalism, regional native range size scores (the geometric mean of the native range sizes of all alien bird species present in a region) were also used.

V7 (relative brain size): Regional relative brain size scores (the average relative brain size for all alien bird species present in a region), calculated using data from Sol et al. (2012).

V8 (conspicuousness): The proportion of species present in a region considered to be inconspicuous and conspicuous. Species were divided by family into two categories (inconspicuous and conspicuous) based on their size and colour. This approach follows Evans et al. (2018b).

V9 (islands): Regions were divided into two categories: islands (<100km^2) and remaining regions.

V10 (native bird species richness): Regional native bird species richness scores (the number of native bird species present in a region).

V11 (population density): Regional population density scores (the number of people / km^2).

V12 (economic development): quantified using Regional Human Development Index (HDI) scores. HDI scores were unavailable for some regions: in these cases, following Evans et al. (2018a), HDI scores were taken from associated nations (e.g. Galapagos Islands, HDI = Ecuador).

For V1 – V6 and V9 – V12, variables were calculated using data taken from Evans et al. (Submitted).

Analysis

The presence or absence of data on the impacts of alien birds for each of the 247 regions was analysed as a binary response variable (0 = no impact data; 1 = impact data). We compared regions with no available impact data with regions that support at least one alien bird species for which impact data are available. For a complete list of regions see Appendix A.

The relationship between each predictor variable and the presence or absence of impact data across regions was assessed using generalized linear mixed effects models using the lme4 package (Bates *et al.*, 2015). A random effect for continent was included to account for potential autocorrelation among regions within continents. Independent analysis of each predictor variable was followed by multivariate analysis for all predictor variables using the dredge function in the MuMIn package (Bartoń, 2018). Relative importance values (the sum of the Akaike weights over all models for each predictor variable) were obtained using the Importance function (MuMIn). Delta AIC values were calculated by obtaining the AIC for the best model with each variable removed, and subtracting this value from the AIC for the best multivariate model. *P* values were obtained using the lmerTest function (Kuznetsova, Brockhoff and Christensen, 2017).

Data for alien bird residence time, native and alien bird species richness, native and alien range size and human population density were $\log_{10}$ transformed for analysis. HDI data were not normally distributed and could not be log transformed. Here the data were divided into four categories: low (0 – 0.549), medium (0.550 – 0.699), high (0.700 – 0.799) and very high (0.800 and above). These are the four formal HDI categories adopted by the United Nations Development Programme (http://hdr.undp.org/en/composite/HDI).

The car package (Fox & Weisberg 2011) was used to calculate variance inflation factors for all variables, to check for the potential effects of multicollinearity (see Table S1 in the Supporting Information). All statistical analyses were undertaken using RStudio version 1.1.383 (R Core Team 2017).

Mapping

A map showing global variation in the availability of alien bird impact data was produced in R using the Natural Earth mapping dataset (1:10m cultural vectors: http://www.naturalearthdata.com/downloads/10m-cultural-vectors), and the following packages: sp (Bivand, Pebesma and Gomez-Rubio, 2013), rgeos (Bivand & Rundel 2017), rgdal (Bivand, Keitt and Rowlingson, 2017), raster (Hijmans 2016) and maptools (Bivand & Lewin-Koh 2017).

Results

Data on environmental impacts of alien birds is available for 60 regions of the world. However, alien bird species are more widely distributed than this, being present in a further 187 regions where no data is available on their environmental impacts (Fig. 1). Univariate analysis revealed positive relationships between impact data availability and alien bird residence time, alien bird species richness and economic development. A negative relationship was revealed for diet breadth (Table 1, Fig. 2), such that areas are more likely to have recorded impacts if they house alien bird species with narrower diets.

In multivariate analysis, it is the characteristics of alien bird invasions that most strongly influence the availability of impact data across regions. Impact data are more likely to be available for regions with higher alien bird species richness and for regions supporting alien birds for longer periods of time. These variables had by far the highest relative importance values of 1 and 0.97, respectively, while removing these predictors from the most likely multivariate model led to large (>8) increases in model AIC (Table 2).

To a lesser extent, characteristics of regions were also found to influence the availability of impact data across regions: impact data are more likely to be available for regions with higher economic development, as measured by HDI (Table 2). The characteristics of alien bird species were also found to influence the availability of impact data, which is more likely to be available for regions supporting diet specialist birds, those with larger relative brain sizes and those with smaller native ranges (Table

2). However, in all cases, removing these trait variables from the most likely multivariate model led to relatively small increases in AIC (<3 in all cases). Aside from a negative association with diet breadth, we found no relationships between variables associated with the severity of impacts generated by alien birds and the availability of impact data across regions (Table 2).

Discussion

Data on the environmental impacts of alien birds are available for 60 regions of the world. These regions tend to be economically developed and relatively wealthy, such as those within Western Europe, North America and Australasia. However, alien birds with recorded impacts are more widely distributed than this, being present in over three times as many regions where their environmental impacts are unknown. These regions are broadly distributed across the globe, and are often within less developed areas such as Africa, India, Central and South America, Eastern Europe and Southeast Asia (Fig. 1). The availability of impact data across regions is primarily influenced by the characteristics of alien bird invasions (both the number of alien bird species being introduced to a region and the length of time alien birds have been present in a region). It is also influenced by the level of economic development in a region, as Fig. 1 illustrates. Importantly, aside from a negative association with diet breadth, we found no relationships between variables previously shown to be associated with the severity of impacts generated by alien birds and the availability of impact data across regions. This means that impact data availability is not necessarily associated with impact severity. As such we cannot conclude that regions lacking impact data do so because the impacts sustained within them are less severe.

Impact data are more likely to be available for regions supporting greater numbers of alien bird species. This is likely to be because there will be greater opportunity to study these species, and because of the increased likelihood that one or more of those species will have an impact that is observed. The average number of species within regions with available impact data is 11 (median = 8); for regions with no impact data it is 3 (median = 2). Nevertheless, some regions that lack impact data support many alien bird species. For example, the United Arab Emirates (UAE) supports 18 species; Reunion supports 15; Oman, Malaysia, the Dominican Republic and Bahrain support 11. Furthermore, some of these regions support alien bird species that harm native biodiversity in other locations. For example, the UAE, Oman, Malaysia and Bahrain all support introduced populations of the Indian house crow (*Corvus splendens*), which through predation (nest raiding) has caused declines in native bird species in Mombasa, Kenya (Ryall 1992). Reunion supports an introduced population of the common myna (*Acridotheres tristis*) which through competition, has been shown to cause declines in populations of two native cavity nesting birds in Australia (Grarock *et al.*, 2013). It is likely that the impacts of alien birds are going unnoticed in these regions.

Impact data are also more likely to be available for regions supporting alien birds for longer periods of time. Longer residence time may both increase the opportunity to study these species, and give them more time to spread and generate impacts. The median residence time for regions with available impact data is 88 years, versus 44 years for regions with no impact data. However, some regions lacking impact data have supported alien birds for a long time: the Nicobar Islands (India), Egypt and

Madagascar have average residence times of 236, 133 and 130 years, respectively. Some of these regions also support alien birds that have significant impacts to biodiversity elsewhere. For example, Egypt is reported to support an introduced population of the rose-ringed parakeet (*Psittacula krameri*), which through competition has been found to cause declines in a native cavity-nesting bird species in Belgium (Strubbe & Matthysen 2009). Impacts may also be being overlooked in regions like these.

Impact data are more likely to be available for regions characterised by higher economic development. This is most likely because these regions have greater capacity to undertake and publish research on the impacts of alien species. This confirms the findings of previous studies which indicate that invasive species research is primarily being undertaken in the developed world (Early *et al.*, 2016; Wilson *et al.*, 2016). This is of concern as there are many less developed countries supporting a broad range of alien bird species which have been found to cause damage to native wildlife in other locations. For example, Nancowry Island (Nicobar Islands, India) supports an introduced population of the red-whiskered bulbul (*Pycnonotus jocosus*). On Mauritius, this species is believed to have caused declines in native birds including endemic white-eyes (*Zosterops* spp.), and the eradication of spiders of the genus *Neophilia* from the island (Diamond 2009; Linnebjerg et al. 2010).

Alien bird impact data are also more likely to be available for regions supporting diet specialist birds, those with larger relative brain sizes and those with smaller native ranges. These results are likely to be associated with the presence of alien parrots in our dataset. Parrots tend to have relatively specialist diets, large brains and small native ranges (accounting for half of the 30 species with the smallest native ranges in our dataset). They represent over 20% of the alien birds for which we have impact data worldwide, and are present in 60% of regions with impact data but only 29% of those without. Indeed, 20 alien parrot species in our dataset are present on mainland USA (a region with impact data), whilst only 12 species are present across all regions with no impact data. However, parrots tend to have relatively minor impacts as aliens (Evans, Kumschick and Blackburn, 2016), and the effects of species traits on the availability of impact data across regions in our models are small.

Conclusions

This study demonstrates that we likely have much to learn about the impacts of alien birds globally, which may well be going unnoticed in many regions. A key issue here is a lack of data on the current status of many alien bird populations: results from the Global Avian Invasions Atlas (GAVIA), which represents the most comprehensive source of information of the global distribution of alien birds, indicate that over 80% of alien bird species have at least one alien population for which the status is unknown (Dyer, Redding and Blackburn, 2017). Studies should be carried out to confirm the presence of these alien bird populations, with priority given to alien bird species with potential for significant impacts to biodiversity (such as habitat generalist alien birds (Evans et al. 2018a)).

As impact data availability is not necessarily associated with impact severity, we cannot conclude that regions lacking impact data do so because the impacts sustained within them are less severe. However, impact data availability is associated with human development, and improving the capacity for research amongst less developed regions may therefore be a key strategy to improve our understanding of the impacts of alien birds (and alien species in general). To extend the global reach of invasion science, collaborations between invasion scientists in developed and developing regions should be undertaken in order to develop national alien species registers, and to undertake research to identity the impacts of alien birds. Where impacts are identified, EICAT workshops should be undertaken in order to formally quantify and categorise the impacts of alien birds. EICAT will shortly be adopted by the IUCN, and any such EICAT assessments will contribute to a global, freely accessible resource on the impacts of alien species.

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