

Threats from introduced birds to native birds

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Abstract. Invasion by introduced species is a global threat to the natural environment, with major consequences for biodiversity, economies and societies. This paper reviews the literature documenting interactions between introduced and native bird species. Surprisingly, we found only ten cases of an introduced bird being involved in a process that threatened a population of a native bird with extinction and conclude there is little evidence that introduced birds are a major threat to avian diversity globally. The conservation priorities for managing interactions between introduced and native birds are: (1) strong, precautionary biosecurity policies and practices to discourage future introductions; (2) gaining knowledge about inter-specific interactions; (3) managing threats where they affect the persistence or recovery of threatened and endemic island bird species; (4) managing the threat of hybridisation and (5) best practice, cost-effective management that defines threatening processes to avian diversity and uses adaptive management for threat abatement. Our review highlights the lack of evidence for detrimental effects of introduced birds on native avifauna and highlights the need for future work in this area.

Additional keywords: alien, brood parasitism, disease, hybridisation, interspecific competition, invasive birds, predation.

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Introduction

Birds move – they fly and walk over land, and fly over and swim in water – and for millennia, these natural movements have been supplemented by people. An introduced species is one that has had human-assisted transport and free release outside its historically known range (Long 1981; IUCN 1987). Some introduced species form self-sustaining populations and become naturalised in their non-native environment. *Introduced Birds of the World* (Long 1981) lists 426 species that have been moved by people within or between 89 regions of the world and documents thousands of instances of avian introductions. In the past, people moved birds to provide food from domestic and wild stock, for recreational hunting and for aesthetics. From the mid-1800s, birds such as the House Sparrow (*Passer domesticus*) were introduced into agricultural areas with the expectation that they would control insect pests (Long 1981). In addition to these deliberate introductions, accidental introductions have occurred through avicultural escapees and birds hitching rides on boats and planes. More than half of the introductions tabulated by Long (1981) did not result in the establishment of a naturalised population. Nevertheless, there are many hundreds of naturalised populations of introduced bird species throughout the world.

Invasive alien species are considered to be a principal cause of animal extinctions globally and a principal threat to food security, human and animal health and biodiversity (IUCN 2012). Over the past few decades, knowledge has improved about the drivers of invasion and there has been a growing discussion of theoretical frameworks (e.g. Richardson *et al.* 2000; Blackburn *et al.* 2011). Lowe *et al.* (2000) constructed a

list of the 100 worst invasive species using two criteria: (1) seriousness of impacts on biological diversity or human activities and (2) their illustration of important issues surrounding biological invasion. The list contains three bird species: Common Myna (*Acridotheres tristis*), Red-vented Bulbul (*Pycnonotus cafer*) and Common Starling (*Sturnus vulgaris*), although Lowe *et al.* (2000) cautioned that there are many other invasive alien species not included on the list. Some introduced bird species have significant economic impacts, particularly on agricultural production (Bomford and Sinclair 2002; Pimental 2011; IUCN 2012) and human health (IUCN 2012; WHO 2012). Leaving these aside, there is a growing interest in defining and managing the threats that introduced birds may have on biodiversity, particularly native birds (Kumschick and Nentwig 2010; Strubbe *et al.* 2011).

In this paper, we review the world literature on negative interactions between introduced birds and native birds in the context of threatening processes and discuss the implications for the conservation of native avian diversity. For several decades, biodiversity conservation policies (e.g. United Nations 1992) and strategies (e.g. Commonwealth of Australia 1996) have typically striven to conserve ecosystems and their component species, as well as the genetic diversity within and among populations. It is implicit in such statements that introduced species are not part of natural ecosystems and, thus, are unwelcome and potentially pose threats to biodiversity. The scientific literature deals with the nomenclature, classification and management of threats (e.g. Salafsky *et al.* 2008; Balmford *et al.* 2009; Auld and Keith 2009) and some jurisdictions, for instance Australia, have

threatened species legislation that includes provision to list and manage key threatening processes, which may include introduced species. However, in many jurisdictions, threatening processes are not required to be assessed or listed explicitly by biodiversity conservation legislation or policy, although the threats from introduced species may be managed directly through on-ground actions specified in management strategies (e.g. in Europe and New Zealand) and action plans (e.g. in Canada) or indirectly through protection and reservation of habitat for native species (e.g. in Europe, South Africa and United States of America).

The establishment of an introduced species will change the species composition of a community and the interactions within it. Although preservation of natural communities is an ideal for conservation, funds are limited and a high priority for biodiversity conservation is to deal with the processes that threaten species with extinction (Bradstock *et al.* 1995). Thus, this review focusses on interactions where an introduced bird species poses a threat to the survival of a population of a native bird. There is overwhelming evidence that introduced mammals can be disastrous for native vertebrate biodiversity (Dickman 1996). For example, predation by the Red Fox (*Vulpes vulpes*) is a threatening process in Australia for which threat abatement plans are being implemented (e.g. Mahon 2009). Furthermore, there is a growing interest in invasive mammal eradications (Phillips 2010). However, an assessment of global avian extinctions at the species and subspecies level did not cite birds as a driver of extinctions (Szabo *et al.* 2012). So what is the evidence for introduced birds interacting with native birds? Are there cases where introduced birds pose a threat to native birds? And what are the priorities for cost-effective conservation management of the threats of introduced birds to native avian diversity?

Methods

We searched the primary literature in Web of Science (Thompson Reuters, New York, NY, USA; see <http://thomsonreuters.com>, accessed April–August 2012) from 1965 to the present initially using three keywords: bird* and behav* (for behaviour or behaviours) and interact* (for interaction, interactions or interacting). This yielded 2024 results that were refined using terms for introduced species (e.g. non-native, non-indigenous, exotic, alien, invade) and interactions (e.g. competition, interspecific, aggression, displacement, exclusion) to identify relevant articles. In addition, we used the taxonomic and common names of the 31 invasive bird species in the Global Invasive Species Database (GISD; <http://www.issg.org/database/welcome/>, accessed April–August 2012) to search on the Web of Science for articles on interactions with native birds. There was a large number of articles identified and the search was refined using the terms listed above, in addition to the ecological impacts recognised by the GISD. Using Google Scholar (<http://scholar.google.com.au/>, accessed October 2013), we searched with phrases like ‘invasive birds and their aggressive behaviour towards natives’ and ‘impact of exotic birds on the native bird community’ resulting in over 1700 references for which the titles were searched to identify relevant articles. Additional articles were found through references cited in articles resulting from our searches.

The review is based on reports (from all continents except Antarctica, and islands in the Atlantic, Indian, Pacific and South-

ern Oceans) of interspecific interactions between 33 introduced species and more than 150 native species taken from 94 articles published between 1945 and 2012 (Table S1 of the Supplementary material). The literature provided evidence of interactions between introduced and native birds, which we separated into seven categories based on ecological processes: competition for nesting sites, competition for food, interference competition, predation, brood parasitism, hybridisation and disease.

Evidence of interactions was divided into four levels: (1) theoretical, meaning suppositions and speculations that use observation or ecological theory to suggest negative interactions; (2) anecdotal, which includes isolated observations, qualitative methods and quantitative observations with small sample sizes; and (3) quantitative, which we subdivided into (a) correlative evidence, whereby data are presented but cannot demonstrate causal relationships and (b) causal evidence, whereby an effect is measured directly (e.g. rate of predation) or experimental data are presented that might demonstrate cause and effect. Generally, the relative strength of the evidence increases from theoretical to causal. For each interaction given in Table S1 of the Supplementary material, we generally excluded articles with only theoretical evidence if there were articles with anecdotal or quantitative evidence. We also excluded articles about birds that had formed naturalised populations without human assistance.

Evidence for interactions in ecological processes

Competition for nesting sites

The abundance and quality of nesting sites is of particular importance because sites may be a limiting resource for native birds (Löhms and Remm 2005; Orchan *et al.* 2013). Competition may be observed directly or demonstrated indirectly, through correlative evidence for overlap of nesting habitats or reduced reproductive success. Thirty-three articles reported competition for nesting sites (Table S1) of which 27 involved introduced and native birds competing for nest-hollows and cavities (hereafter hollows). The 12 introduced species were: Crimson Rosella (*Platycercus elegans*; 1 article), Common Myna (5), Common Starling (11), Eurasian Collared Dove (*Streptopelia decaocto*; 1), Great Kiskadee (*Pitangus sulphuratus*; 1), Green-backed Firecrown (*Sephanoides sephanoides*; 1), House Finch (*Carpodacus mexicanus*; 1), House Sparrow (3), Little Corella (*Cacatua sanguinea*; 1), Long-billed Corella (*Cacatua tenuirostris*; 1), Rainbow Lorikeet (*Trichoglossus haematodus*; 1) and Rose-ringed Parakeet (*Psittacula krameri*; 5) (Table S1). There was some strong evidence for a lack of competition in interactions with the Common Starling (see ‘Evidence for threats involving by introduced species’, below), Common Myna (Lowe *et al.* 2011), Green-backed Firecrown and House Sparrow (Hahn *et al.* 2011). There were just two cases where competition for nesting sites might be considered a threatening process: weak evidence for a threat from the Crimson Rosella on Norfolk Island and strong evidence for a threat from the Rose-ringed Parakeet in Europe (Table 1; see ‘Evidence for threats involving introduced species’, below).

Competition for food

Food can be a limiting resource for birds. Competition can be observed directly through monopolisation of food by an

Table 1. Introduced bird species that interact with native birds showing the process of interaction and level of evidence in articles where a population-level threat was demonstrated

See Table S1 of the Supplementary material for more detail on these articles and a further 72 articles where no population-level threat was shown

Introduced bird	Native bird	Process	Level of evidence	Threat	Reference
Chukar (<i>Alectoris chukar</i>)	Red-legged Partridge (<i>Alectoris rufa</i>), Rock Partridge (<i>Alectoris graeca</i>)	Hybridisation	Correlative	Potentially	Baratti <i>et al.</i> (2005), Barbanera <i>et al.</i> (2005), Barilani <i>et al.</i> (2007), Tejedor <i>et al.</i> (2007), Blanco-Aguiar <i>et al.</i> (2008)
Common Myna (<i>Acridotheres tristis</i>)	Tahiti Monarch (<i>Pomarea nigra</i>)	Competition – interference	Correlative	Yes	Blanvillain <i>et al.</i> (2003)
	Seychelles Magpie-Robin (<i>Copsychus sechellarum</i>)	Competition – interference	Correlative	Yes	Komdeur (1996)
Crimson Rosella (<i>Platycercus elegans</i>)	Tasman Parakeet (<i>Cyanoramphus cookii cookii</i>), Norfolk Island Southern Boobook (<i>Ninox novaeseelandiae undulata</i>)	Competition – nesting sites	Anecdotal	Yes	Garnett <i>et al.</i> (2011)
Japanese White-eye (<i>Zosterops japonica</i>)	Hawaii Akepa (<i>Loxops coccineus</i>)	Competition – food	Causal	Yes	Freed and Cann (2009)
Northern Mallard (<i>Anas platyrhynchos</i>)	Hawaiian Duck (Koloa) (<i>Anas wyvilliana</i>)	Hybridisation	Theoretical	Potentially	BirdLife International (2012)
	Hawaiian Duck (Koloa)	Hybridisation	Causal	Potentially	Fowler <i>et al.</i> (2009)
	Pacific Black Duck (<i>Anas superciliosa</i>)	Hybridisation	Correlative	Yes	Gillespie (1985)
	<i>Anas</i> spp.	Hybridisation	Theoretical	Potentially	Global Invasive Species Database (2010)
	Pacific Black Duck	Hybridisation	Theoretical (review)	Potentially	Guay and Tracey (2009)
	Pacific Black Duck	Hybridisation	Causal	Yes	Rhymer <i>et al.</i> (1994)
	Pacific Black Duck	Hybridisation	Correlative	Yes	Tracey <i>et al.</i> (2008)
	Hawaiian Duck (Koloa)	Hybridisation	Anecdotal	Yes	US Fish and Wildlife Service (2011)
Red-vented Bulbul (<i>Pycnonotus cafer</i>)	Tahiti Monarch	Competition – interference	Correlative	Yes	Blanvillain <i>et al.</i> (2003)
Rock Dove (<i>Columba livia</i>)	Galapagos Dove (<i>Zenaida galapagoensis</i>)	Disease	Anecdotal	Potentially	Wikelski <i>et al.</i> (2004)
Rose-ringed Parakeet (<i>Psittacula krameri</i>)	Eurasian Nuthatch (<i>Sitta europaea</i>) and other hollow-nesting species	Competition – nesting sites	Correlative	Potentially	Strubbe and Matthysen (2007)
	Eurasian Nuthatch	Competition – nesting sites	Causal	Potentially	Strubbe and Matthysen (2009)
Ruddy Duck (<i>Oxyura jamaicensis</i>)	White-headed Duck (<i>Oxyura leucocephala</i>)	Hybridisation	Causal	Potentially	Muñoz-Fuentes <i>et al.</i> (2007)
Shiny Cowbird (<i>Molothrus bonariensis</i>)	Puerto Rican Vireo (<i>Vireo latimeri</i>)	Brood parasitism	Causal	Potentially	Woodworth (1997)

introduced species or indirectly through behaviour or health of native species. Twelve articles reported competition for food between native birds and six introduced species: Common Myna (4 articles), Common Starling (2), Eurasian Collared Dove (2), Japanese White-eye (*Zosterops japonica*; 2), Great Kiskadee (1) and Rainbow Lorikeet (1) (Table S1). There was strong evidence for a lack of competition in interactions with the Eurasian Collared Dove (Poling and Hayslette 2006). There was strong evidence that competition with the Japanese White-eye for food is a threat to small native birds in Hawaii but not in the Bonin Islands (Table 1; see ‘Evidence for threats involving introduced species’, below).

Interference competition

Aggression, particularly territorial aggression, within and between bird species is common. Behaviours include vocalisation, harassment, chasing, pecking, supplanting and swooping. Twenty-four articles reported interference competition that was not clearly related to nesting sites or food resources from ten introduced species: Australian Magpie (*Cracticus tibicen*; 6 articles), Canada Goose (*Branta canadensis*; 1), Common Myna (8), Common Starling (2), House Sparrow (2), Northern Mallard (*Anas platyrhynchos*; 1), Mute Swan (*Cygnus olor*; 1), Purple Swamphen (*Porphyrio porphyrio*; 1), Red-vented Bulbul (1) and Red-whiskered Bulbul (*Pycnonotus jocosus*; 1)

(Table S1). There was some strong evidence for a lack of competition in interactions with the Common Myna (Parsons *et al.* 2006; Lowe *et al.* 2011; Borowske *et al.* 2012), Common Starling (Williamson and Gray 1975; Vierling 1998) and Mute Swan (Conover and Kania 1994). There was correlative evidence that the Australian Magpie reduced the abundance of native species in New Zealand (Morgan *et al.* 2006). There was correlative evidence that the Common Myna is a threat to the Seychelles Magpie-Robin (*Copsychus sechellarum*) and that the Common Myna and Red-vented Bulbul are a threat to the Tahiti Monarch (*Pomarea nigra*) (Table 1; see 'Evidence for threats involving introduced species', below).

A correlative study in West Mexico that focussed on avian community structure (MacGregor-Fors *et al.* 2010) showed that areas with the introduced House Sparrow had reduced species richness of native birds, which was attributed to interspecific aggression. House Sparrows were observed to attack native species such as the Lesser Goldfinch (*Carduelis psaltria*), White-collared Seedeater (*Sporophila torqueola*) and Golden-fronted Woodpecker (*Melanerpes aurifrons*) at feeding sites. MacGregor-Fors *et al.* (2010) suggested that the presence of the House Sparrow is the cause of the loss of native species. However, because House Sparrows had only established in urban and agricultural areas, the change to the avian community could also have been driven by the loss or degradation of native bird habitats associated with the creation of House Sparrow habitat.

Predation

Predation can be observed directly or demonstrated indirectly through reduced reproductive success and observations of destroyed nests. There were 14 articles reporting predation by ten introduced species: Austral Thrush (*Turdus falcklandii*; 1 article), Australian Magpie (2), Cattle Egret (*Bubulcus ibis*) (1), Common Myna (4), House Crow (*Corvus splendens*; 1), Red-vented Bulbul (1), Red-whiskered Bulbul (1), Sacred Ibis (*Threskiornis aethiopicus*; 1), Swamp Harrier (*Circus approximans*; 1) and Tasmanian Masked Owl (*Tyto novaehollandiae castanops*; 1) (Table S1). There was strong evidence for predation (Hughes *et al.* 2008) and against predation (Thibault *et al.* 2002) by the Common Myna and against predation by the Red-vented Bulbul (Thibault *et al.* 2002). There was no case demonstrating that predation by an introduced bird species was a threat to a native bird species.

Two cases of predation by raptors introduced to islands were confounded by the co-occurrence of the introduced Black Rat (*Rattus rattus*). Predation by the Swamp Harrier, introduced to Tahiti in 1885, was considered to be a possible cause of the extinction of the Polynesian Imperial-Pigeon of Tahiti (*Ducula aurorae wilkesii*) and the Blue Lorikeet (*Vini peruviana*) and to have seriously affected other native bird populations (Meyer 2003). However, Black Rats may have driven the endemic land birds to extinction, especially the lorikeets (*Vini* spp.), monarch flycatchers (*Pomarea* spp.) and the ground-doves (*Gallicolumba* spp.) (Meyer 2003). The Tasmanian Masked Owl, introduced to Lord Howe Island in the 1920s (DECC 2007), is known to prey upon the Lord Howe Woodhen (*Gallirallus sylvestris*, listed as vulnerable), White Tern (*Gygis alba*), Black-winged Petrel (*Pterodroma nigripennis*) and Providence Petrel (*P. solandri*) (Hutton

1991). However, predation by the Black Rat, which eats birds' eggs, is listed as a key threatening process on Lord Howe Island (NSW Scientific Committee 2011), and there is no evidence of a population-level threat by the Tasmanian Masked Owl.

Brood parasitism

We found only one article (Woodworth 1997) that described brood parasitism by an introduced species. The Shiny Cowbird (*Molothrus bonariensis*) was introduced to the island of Puerto Rico in c. 1955 and was recorded parasitising 17 native species, including a single-island endemic, the Puerto Rican Vireo (*Vireo latimeri*) (Woodworth 1997, and references cited therein) (Table 1; see 'Evidence for threats involving introduced species', below).

Hybridisation

Hybridisation is common in birds (Grant and Grant 1992) and introduced species can change the genetic trajectory of native taxa, as evidenced in phenotypic (morphological) characterisation and molecular genotyping. Hybridisation can contribute to the decline of the genetic integrity of native species in two ways: (1) by causing infertility through a reduction in fecundity or survivorship and (2) by contamination of the native gene pool through genetic introgression and swamping. Twenty-one articles, involving six introduced species, reported hybridisation: Chukar Partridge (*Alectoris chukar*; 5 articles), Little Corella (1), Long-billed Corella (1), Northern Mallard (10), Rock Partridge (*Alectoris graeca*; 2) and Ruddy Duck (*Oxyura jamaicensis*; 2) (Table S1). There was strong evidence that hybridisation by the Northern Mallard with other species of *Anas* and by the Ruddy Duck with the White-headed Duck (*Oxyura leucocephala*) is a threatening process (see 'Evidence for threats involving introduced species', below). There was also strong evidence for a threat of hybridisation among species of partridge, mainly through introgression from the Chukar (Table 1; see 'Evidence for threats involving introduced species', below).

Disease

Birds can be vectors or reservoirs for a variety of diseases, including parasites, and the potential for introduced birds to transfer disease to the native avian community is strong (Ishtiaq *et al.* 2006; Deem *et al.* 2008). Seventeen articles, involving nine introduced species, reported diseases: Common Myna (4 articles), House Finch (3), Japanese White-eye (1), Little Corella (1), Long-billed Corella (1), Nutmeg Mannikin (*Lonchura punctulata*) (1), Rainbow Lorikeet (1), Red Junglefowl (i.e. domestic poultry, *Gallus gallus*; 3) and Rock Dove (*Columba livia*; 2). There was strong evidence that introduced birds carry introduced strains of diseases (Farmer *et al.* 2005; Ishtiaq *et al.* 2006). One article (Soos *et al.* 2008) provided correlative evidence that diseases carried by Red Junglefowl in the Galápagos Islands did not occur in native birds. There was weak but mounting evidence that the Rock Dove is spreading *Trichomonas gallinae*, which may be a threat to the Galapagos Dove (*Zenaida galapagoensis*) (Table 1; see 'Evidence for threats involving introduced species', below).

Circumstantial evidence suggested that avian malaria (*Plasmodium* spp.) was important in the decline and extinction of some

endemic Hawaiian drepaniids (honeycreepers) (Warner 1968; Lowe *et al.* 2000). Although avian malaria is reported anecdotally in the introduced Japanese White-eye in parts of Hawaii (Warner 1968), there is no evidence of disease transmission and the disease probably occurred in birds in Hawaii before human settlement, through migratory ducks and waders (Warner 1968). It most likely spread to Hawaiian endemics with the introduction of a mosquito (*Culex* sp.) in 1826 (Warner 1968). Drepaniids exposed to the disease do not show immunogenic capacity and they die (Atkinson *et al.* 1995). In addition, avian pox viruses (*Avipoxvirus* spp.), which were surmised to have been introduced to Hawaii on domestic poultry (Red Junglefowl) and which was common in the introduced House Finch and other introduced species, caused severe infestation in native bird species because of their lack of immunity (Warner 1968). In captive trials, Nutmeg Mannikins and a native honeycreeper, the Iiwi (*Vestiaria coccinea*), were inoculated with avian malaria (Atkinson *et al.* 1995). The introduced species was not infected but the native species was, thus making it a more likely carrier of the disease (Atkinson *et al.* 1995).

Evidence for threats involving introduced species

Chukar

Populations of the Red-legged Partridge (*Alectoris rufa*) and Rock Partridge are declining in parts of their native range owing to overhunting and restocking with the Chukar and Chukar hybrids, causing further hybridisation with both species (Barbanera *et al.* 2005; Barilani *et al.* 2007). Correlative studies using molecular genotyping indicated that hybridisation with the Chukar is widespread across the natural range of the Red-legged Partridge (Baratti *et al.* 2005; Barbanera *et al.* 2005; Barilani *et al.* 2007; Tejedor *et al.* 2007; Blanco-Aguilar *et al.* 2008) and present in the native distribution of the Rock Partridge, leading to introgressive hybridisation (Barilani *et al.* 2007). Hybridisation between Red-legged and Rock Partridges occurs naturally in hybrid zones in the southern French Alps (Barilani *et al.* 2007) but in Spain, the Rock Partridge has been deliberately introduced to breed with the Red-legged Partridge for hunting, which may be threatening the genetic integrity of native Red-legged Partridge populations (Arruga *et al.* 1996; Negro *et al.* 2001). Although there is some regulation of the stock that is released for hunting, hybridisation remains a potential threat, particularly hybridisation with the Chukar (Barbanera *et al.* 2005; Baratti *et al.* 2005; Barilani *et al.* 2007).

Common Myna

In the Seychelles Archipelago, the critically endangered Seychelles Magpie-Robin is threatened by the clearing of all of the natural vegetation and numerous introduced species (Komdeur 1996). Its nesting success was reduced to zero in the presence of the Common Myna because the native species abandoned its nesting sites when the introduced species co-occupied its nesting trees (Komdeur 1996). Komdeur (1996) emphasised that the only self-sustaining Seychelles Magpie-Robin population was on an island never colonised by rats but colonised by the Common Myna. Nevertheless, interference competition from the Common Myna is a threat that needs to be addressed in the recovery of the species.

A 3-year study in Tahiti (Blanvillain *et al.* 2003) correlated a significant decline in the nesting success of the critically endangered endemic Tahiti Monarch, with encounters and interactions with the Common Myna and Red-vented Bulbul, both of which utilised the forest habitat of the Tahiti Monarch. Blanvillain *et al.* (2003) suggested that the aggressive interactions were indicative of predation by the Common Myna. However, using artificial nests monitored by cameras, Thibault *et al.* (2002) found no evidence for predation by the Common Myna. Nevertheless, the threat of interference competition from the Common Myna needs to be addressed in the recovery of the Tahiti Monarch (Blanvillain *et al.* 2003).

Crimson Rosella

The Crimson Rosella was introduced to Norfolk Island in the 1830s (Higgins 1999). It is recognised as a competitor for nesting hollows with two critically endangered endemics: the Southern Boobook (*Ninox novaeseelandiae undulata*) and Tasman Parakeet (*Cyanoramphus cookii cookii*) (Garnett *et al.* 2011). Evidence for this threat is only anecdotal and further information is needed to assist in the recovery of these two species (Garnett *et al.* 2011).

Japanese White-eye

The Japanese White-eye was introduced to both the Hawaiian and Bonin Islands in the 1920s. Separate studies examined competition for food resources through changes in morphometric characteristics over time. In the Hawaiian Islands, after an increase in the population of the introduced species within old-growth forest, the juveniles of eight native species had shorter bills, shorter tarsi, a lower body mass and less furcular fat, which is indicative of reduced foraging efficiency leading to poor nutrition and lower survival (Freed and Cann 2009). The Japanese White-eye was considered to be responsible for the decline of the native species because at nearby sites with fewer Japanese White-eyes, juveniles of native species had normal measurements (Freed and Cann 2009). One of the native species, the Hawaii Akepa (*Loxops coccineus coccineus*), is endangered and appears to be threatened by the competition for food with the Japanese White-eye (Freed and Cann 2009).

In contrast, a high degree of dietary overlap was also found between the Japanese White-eye and the endemic Bonin Island White-eye (*Apalopteron familiare*) but the native White-eye was not affected by interspecific competition for food resources (Kawakami and Higuchi 2003). The fledgling weight of the native species, an indicator of good health and nutrition availability, was similar in populations allopatric and sympatric with the introduced species. Moreover, the native species adjusted its foraging height to accommodate the presence of the introduced species. These results may be attributed to high availability of food, perhaps where the introduced species has not reached peak density (Kawakami and Higuchi 2003).

Northern Mallard

The Northern Mallard has hybridised with congeneric, endemic duck populations, some of which are extinct or threatened with extinction (Global Invasive Species Database 2010). The extinction of some populations of the Pacific Black Duck (*Anas super-*

ciliosa) has occurred in New Zealand and on Lord Howe Island (Tracey *et al.* 2008; Guay and Tracey 2009) and there is evidence that introgression has occurred on Macquarie Island (Norman 1990). In New Zealand in the 1980s, the frequency of Mallard-like hybrids was >60%, that of Pacific Black Duck-like hybrids 11.7%, and the frequency of pure Pacific Black Duck was 4.5%, a level that was considered possibly too low to ensure its survival (Gillespie 1985). Mitochondrial DNA analysis corroborates the decline of the New Zealand native species through introgressive hybridisation (Rhymer *et al.* 1994). In South Australia, a phenotypic study demonstrated that hybridisation between the Northern Mallard and the Pacific Black Duck is increasing (Paton *et al.* 1992) but the extent and threat of hybridisation in mainland Australia is still not known. Northern Mallard hybridisation is reducing the number of pure Hawaiian Ducks (*Anas wyvilliana*) and is recognised as a primary threat to their recovery (US Fish and Wildlife Service 2011; BirdLife International 2012). The distribution and abundance of hybrids in Hawaii is not clear because it is difficult to distinguish between hybrids by phenotypic characters (US Fish and Wildlife Service 2011). Although introgression was highly apparent in Hawaiian Ducks, molecular genotyping distinguished between hybrid and pure Hawaiian Ducks and showed that there were varying hybrid frequencies between the islands and that recovery of this species may be possible using island-specific management (Fowler *et al.* 2009). Hybridisation by the Northern Mallard is a threatening process of global significance.

Red-vented Bulbul

Encounters and aggressive interactions between the Red-vented Bulbul (together with the Common Myna) and Tahiti Monarch are correlated with a decline in nesting success of the endemic Monarch (Thibault *et al.* 2002; Blanvillain *et al.* 2003). The threat of interference competition from the Red-vented Bulbul needs to be addressed in the recovery of the Tahiti Monarch along with predation by the Black Rat (Thibault *et al.* 2002; Blanvillain *et al.* 2003).

Rock Dove

The Rock Dove, introduced to the Galápagos Islands in the early 1970s, is a carrier of *Trichomonas gallinae*, a protozoan parasite that is transmitted through drinking water and causes a disease commonly called 'canker' (Harmon *et al.* 1987). The parasite was detected in three of nine endemic Galapagos Doves on islands where the Rock Dove occurred but none of the 18 Galapagos Doves from islands thought to be free of the Rock Dove (Harmon *et al.* 1987). More recently, Wikelski *et al.* (2004) stated that populations of Galapagos Doves had declined rapidly on islands inhabited by Rock Doves. This suggests that the transmission of canker from the introduced Dove could be threatening the island endemic with extinction.

Rose-ringed Parakeet

The Rose-ringed Parakeet was introduced to Europe and United Kingdom in c. 1970. In Belgium, Strubbe and Matthysen (2007) found a negative correlation between the abundance of the Rose-ringed Parakeet and Eurasian Nuthatch (*Sitta europaea*) but not with other native hollow-nesters. In a related study, Strubbe and

Matthysen (2009) found that blocking breeding hollows of Rose-ringed Parakeets resulted in a significant decline in numbers of Eurasian Nuthatch, attributed to Rose-ringed Parakeets displacing Nuthatches from their hollows. Strubbe *et al.* (2010) then used distribution modelling to predict that competition between these two species across Flanders might affect up to one-third of the Nuthatch population but considered this did not justify a Rose-ringed Parakeet eradication program.

In the United Kingdom, Butler (2003) observed no competition between the Rose-ringed Parakeet sharing nesting trees with a variety of native species over 3 years. Newson *et al.* (2011) modelled survey data gathered during 1994–2008 from 180 sites in south-eastern England and found that, although there was a weak but significant negative relationship between the abundance of the Rose-ringed Parakeet and Eurasian Nuthatch, this relationship did not persist when the degree of urbanisation was considered. Newson *et al.* (2011) concluded that the Rose-ringed Parakeet did not have a negative effect on the Nuthatch or seven other native hollow-nesting species.

The population of Rose-ringed Parakeets is growing and expanding in the United Kingdom and Europe (Butler 2003). The population and the potential threat of competition with native species for nesting hollows should be monitored into the future (Butler 2003; Newson *et al.* 2011).

Ruddy Duck

The Ruddy Duck, introduced to Europe after 1948, is recognised as a threat to the endangered White-headed Duck through hybridisation (Hughes 1996; Muñoz-Fuentes *et al.* 2007). Although evidence for hybridisation is apparent, no extensive introgression has been found in the native populations, which may be a result of an effective control program that began in the 1990s and included 11 countries by the mid-2000s (Hughes *et al.* 2006; Muñoz-Fuentes *et al.* 2007).

Shiny Cowbird

The Shiny Cowbird was introduced to Puerto Rico in 1955 (Woodworth 1997). It parasitised up to 83% of the nests of the Puerto Rican Vireo, which reduced the fledging rate by 82% and potentially threatened to reduce the range of the Vireo (Woodworth 1997). That the Shiny Cowbird is able to parasitise many native species on the island while affecting such a high proportion of a single native species, suggests that the Puerto Rican Vireo could eventually be threatened with extinction by this process. However, despite a lack of information on population parameters for the species, it is currently evaluated as Least Concern (BirdLife International 2013).

Summary of evidence

Surprisingly, there is little evidence supporting a general and primary role for introduced species in extinctions (Gurevitch and Padilla 2004), including extinction of birds (Strubbe *et al.* 2011). In this review, we found very little evidence that introduced birds are a major threat to avian biodiversity globally. Obviously, the addition of introduced birds will change the species composition and dynamics of an avian community. However, of the many hundreds of naturalised populations of introduced birds, we found only ten cases of an introduced bird being involved in a process

that threatened a population of a native bird with extinction. This lack of evidence for threats is not a result of a lack of interest in or a paucity of studies about interactions between introduced and native birds. There are also many quantitative studies showing a lack of effect when introduced and native birds interact.

Level and strength of evidence

In describing interspecific interactions we have used articles with theoretical (19 articles), anecdotal (26), correlative (33) and causal (25) evidence. Theoretical and anecdotal reports need to be treated with caution (Strubbe *et al.* 2011) but, even though the evidence they provide might be only weak, it might be important. For instance, a negative interaction discussed in a theoretical article may trigger quantitative studies. If the native species being studied is reduced to one population, particularly if it is small or already threatened with extinction, it will be difficult to conduct experiments with replicates and control treatments to provide strong evidence. In such cases, anecdotal evidence (e.g. competition for nests from the Crimson Rosella on Norfolk Island) or correlative studies with small datasets (e.g. interference competition from the Common Myna on the Seychelles Islands) may be the best level of evidence obtainable.

Sound quantitative evidence of interactions may be provided by a study but quantitative evidence of a process or threat may not be measured. For instance, on Moturoa Island, New Zealand, after the Common Myna was culled, there was a significant increase in the abundance of some of the native and other introduced bird species (Tindall *et al.* 2007). Although release from interference competition was inferred by the study, this process was not measured in the study.

Quantitative studies are typically limited in size, time and space. So, multiple studies can help to establish general patterns of interactions between an introduced species and a range of native species. For instance, competition for nesting sites from Common Starlings has been well studied in North America and seems not to be a threatening process. Troetschler (1976) found that the Acorn Woodpecker (*Melanerpes formicivorus*) out-competed Common Starlings for hollows, readily excavated new hollows, or successfully nested or re-nested later in the season and maintained local populations, which was taken to indicate that the Common Starling was not reducing the fecundity of the native species. Vierling (1998) observed 59 pairs of the Lewis's Woodpecker (*Melanerpes lewis*) at their nest-hollows and only one hollow was lost to Common Starlings. A series of studies (Ingold 1989, 1994, 1996, 1998) showed that in urban and rural areas, the Common Starling competed successfully with native woodpeckers. In one case, competition with the Red-bellied Woodpecker (*Melanerpes carolinus*) was intense; 52% of hollows were lost, leading to a significant reduction in fecundity (Ingold 1989). However, for this species and the Red-headed Woodpecker (*Melanerpes erythrocephalus*) and Northern Flicker (*Colaptes auratus*), re-nesting and moving into more heavily forested areas where the Common Starling was sparsely distributed were successful strategies to reduce competition with the Common Starling. There was no evidence for reduced fecundity of the Mountain Bluebird (*Sialia currucoides*) or Tree Swallow (*Tachycineta bicolor*) in the presence of the Common Starling, although the Tree Swallow did shift its nesting niche (Koch *et al.* 2012). Notably, analysis of

long-term (30+ years) bird distribution datasets for 27 native, hollow-nesting species (including the Red-bellied Woodpecker) showed that only one common species, the Yellow-bellied Sapsucker (*Sphyrapicus varius*), had a population decline plausibly owing to competition with the Common Starling (Koenig 2003). Overall, there is strong corroborative evidence for a lack of population-level effect of competition for nesting sites from the Common Starling on native birds.

Multiple studies of an introduced bird may lead to differing conclusions about potential threats. This was the case with studies of competition from the Common Myna with native birds in south-eastern Australia. The evidence we reviewed indicates that the Common Myna is not implicated in any threatening process in Australia. In cases like this, further rigorous study is needed to give a better understanding of the patterns of interactions and the potential for threats to native avian diversity.

Confounding factors

The greatest threat to biodiversity is habitat destruction or modification (IUCN 2012), which disadvantages most native birds and can provide suitable habitat for introduced bird species (e.g. Yom-Tov *et al.* 2012). Generally, small populations are vulnerable to multiple threats and competition alone is rarely the cause of extinction (Davis 2003). As demonstrated by many of the articles we reviewed, distinguishing among the effects of introduced birds, habitat loss and predation by rats is difficult (Gurevitch and Padilla 2004). For instance, the Seychelles Magpie-Robin had to contend with nest-trees being cleared for farming land, feeding habitat trampled by stock and nest predation from introduced rats (Komdeur 1996); interference competition with the Common Myna was in addition to these threats. Predation by introduced rats is also an issue for native birds in Tahiti (Meyer 2003), Lord Howe Island (NSW Scientific Committee 2011), Ascension Island (Hughes *et al.* 2008), Norfolk Island (Garnett *et al.* 2011), and the Hawaiian and other islands (Drake and Hunt 2009).

Introduced mammals other than rats can also confound the effect of introduced birds. On Ascension Island, ten seasons of monitoring nests of Sooty Terns (*Onychoprion fuscatus*) implicated egg predation by the Common Myna in the failure of ~10% of all nests, although the effect of this was uncertain as the population began to increase after the eradication of cats from the island in the seventh year of monitoring (Hughes *et al.* 2008).

Temporal factors may also confound the evidence of threats posed by introduced species. For instance, introduced species go through an establishment phase before they become a potential management issue (Blackburn *et al.* 2011). Currently, some introduced birds may be in this lag phase and their potential to become involved in a threatening process may not be obvious. Furthermore, 'extinction debt', the loss of species long after the fragmentation of their habitat (Tilman *et al.* 1994), may be exacerbated by interactions with introduced species.

Conservation priorities for managing introduced birds that threaten avian diversity

Our findings relate only to interspecific interactions among birds. The seriousness, complexity and intractability of threats to other aspects of biodiversity and the economic and social costs asso-

ciated with invasive species are well articulated in the scientific literature (e.g. Bradstock *et al.* 1995; Mack *et al.* 2000) and policy statements (e.g. IUCN 2012). For biodiversity conservation, management is about conserving populations and ecological communities (May 1994) as well as natural genetic variation (e.g. Roberts *et al.* 2011); it is not about maximising the numbers of each native species or minimising numbers of each introduced species. Furthermore, funding for biodiversity conservation is grossly inadequate (Joseph *et al.* 2009) and a relatively miniscule consideration in national budgets compared with economic and social security. It is imperative not to squander precious biodiversity conservation resources (Odling-Smee 2005; Wilson *et al.* 2006).

Biosecurity

As a high priority, deliberate introductions should be strongly discouraged by precautionary national and international biosecurity policies and practices. 'Prevention is better than cure', and proposed new introductions need to be thoroughly assessed (Mack *et al.* 2000; Jeschke and Strayer 2005; Keller *et al.* 2007). New accidental introductions should be evaluated for potential effects and the feasibility of eradication, although detection may be problematic and assessing natural range extensions of birds and naturally occurring vagrants may also need to be considered.

Increasingly, translocation (supplementation, reintroduction or introduction) is being used as a conservation technique for threatened species (Griffith *et al.* 1989; Armstrong and McLean 1995; Fischer and Lindenmayer 2000) and, given habitat loss and the effects of climate change, in some situations, introduction may be the only viable option. If the options of translocating birds to supplement an existing population or to reintroduce a population within the historical range of the species are not viable, then the benefits of the introduction of a bird must be balanced against the risks, especially to the host environment.

Weight of evidence

Increasing our knowledge of interspecies interactions with high-quality studies is a priority (Gurevitch and Padilla 2004; Didham *et al.* 2005). Causal studies, where possible, are likely to provide the strongest evidence of threats and the best chance to secure funds for conservation management (Gurevitch and Padilla 2004). Studies are needed that separate the effects of introduced

species from confounding factors. Long-term monitoring may provide correlative evidence for emerging threats.

Islands

Islands have high rates of endemism, high susceptibility to threats, and many threatened species and extinctions. Hence, Phillips (2010) argued for eradications of invasive mammals from islands. Seven of the introduced species we have listed are involved in a threat to native bird populations on islands. Managing these threats where they affect the persistence or recovery of threatened and endemic island bird species is a priority. Eradication of vertebrates is rarely possible (Bomford and O'Brien 1995) but may be achievable and ultimately cost effective on small islands, although integrated invasive species management is needed to avoid one threat being replaced by another (Orchan *et al.* 2013).

Hybridisation

Hybridisation is a widespread concern, particularly for the conservation of threatened or rare native species (Rhymer and Simberloff 1996; Fowler *et al.* 2009; Guay and Tracey 2009). Hybridisation with the Northern Mallard has already caused the extinction of two populations of the Pacific Black Duck. The recovery of the Hawaiian Duck from hybridisation with the Northern Mallard is a priority, as is further study of the potentially global threat of hybridisation by the Northern Mallard. Continuation of the recovery actions already underway to address the threat of hybridisation of White-headed Duck by the Ruddy Duck is a priority. Identifying and protecting pure populations of the Red-legged and Rock Partridges from hybridisation, particularly with the Chukar and Chukar hybrids, is a priority. Programs of restocking for partridge hunting need to be better regulated to protect wild populations from the threat of hybridisation.

Best practice

Best-practice wildlife management needs to be adaptive (Holling 1978; Burbidge *et al.* 2011) and, for introduced species, not simply the *ad hoc* culling of birds. However, community-based programs may have more political and public support if they involve culling introduced birds (e.g. Canberra Indian Myna Action Group, see <http://www.indianmynaaction.org.au/>, accessed 2 April 2012), than if they involve protection and restoration of native bird habitat, even though the latter is likely

Box 1. Cost-effective management of threats involving introduced birds (*sensu* Lunney *et al.* 2007)

- (1) *A planning approach.* Plans such as threat abatement or threatened species recovery plans must be evidenced-based using the best available science together with community participation in their preparation and implementation. They must conform to local regulations and manage threats in the context of the environment, both natural and anthropogenic, and with consideration to ethical issues such as the humane treatment of all birds.
- (2) *Define the threatening process.* Introduced birds are not a threat *per se*. They may threaten native birds by one or more of the processes described in this paper. It is essential that the threatening process be defined and that an acceptable threshold of effect be identified.
- (3) *Adaptive management is best-practice in biodiversity conservation management.* This allows for plans to define threats and actions. Importantly, it allows for testing research hypotheses, monitoring of the success of actions and the progress towards the goals of the plan whether they are to abate, ameliorate or eliminate that threat.

to have greater biodiversity benefits (Tindall *et al.* 2007). Nevertheless, community support is often an essential component of control programs (Lunney *et al.* 2007; Strubbe *et al.* 2011) and there are instances of programs being rendered ineffective because of public resistance (Genovesi 2005). Above all, programs that purport to be dealing with introduced birds for the sake of biodiversity conservation must define the threat that is being managed, assess the efficacy of the methods being used, and define, monitor and evaluate threat abatement (Field *et al.* 2007; Davis *et al.* 2011; Lowe *et al.* 2011; see Box 1).

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