Invasive Birds in Japan

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Abstract

Introduction of exotic birds has been increasing throughout the world. We conducted inquiry investigations in order to reveal the spread of introduced bird species in Japan. Forty-three exotic species are known to breed or are regarded as breeding. The Feral Pigeon (*Columba livia*), Chinese Bamboo Partridge (*Bambusicola thoracica*), Red-billed Leiothrix (*Leiothrix lutea*), Budgerigar (*Melopsittacus undulatus*) and Red Avadavat (*Amandava amandava*) are widely distributed. The Rose-ringed Parakeet (*Psittacula krameri manillensis*), Common Peafowl (*Pavo cristatus*), Light-vented Bulbul (*Pycnonotus sinensis*), Melodious Laughing Thrush (*Garrulax canorus*), Crested Myna (*Acridotheres cristatellus*) and Black-billed Magpie (*Pica pica sericea*) are locally common, though they have limited ranges. Sightings of other species were sporadic and regionally limited. Most of introduced birds in Japan are escaped caged birds. About two thirds are from tropical regions, and show diet generalism as omnivores, granivores and frugivorous or herbivorous insectivores. Light-vented Bulbuls, Feral Pigeons and Common Peafowl exerted crop damage. Although adverse effects by exotic birds are not prominent yet, there is great concern that exotic birds will become a threat to native biota and habitats in future.

Key words: introduced birds, invasive birds, naturalization

1. Introduction

Disturbance of native biota and devastation of local ecosystems are major adverse effects caused by invasive species. Nile perches prey upon native fishes and have changed the fauna of Lake Victoria remarkably (Hunter, 1996). Goats, cattle and pigs destroy the habitat of islands (Pimentel *et al.*, 2000). Introduced carnivorous mammals, including rats, cats, mongooses and weasels, have caused local extinction of native bird species in the Hawaiian Islands, New Zealand and other oceanic islands (Pimentel *et al.*, 2000; Levy, 2003). Many birds have also been transported by humans to new areas of the world. The ecological effects of introduced birds on native biota and ecosystems are known, and some are harmful (Long, 1981; Lever, 1987).

The introduction process is composed of three phases; arrival or entry, establishment and integration (Vermeij, 1996). A population may increase after establishment and proceed into the integration phase, where the species interacts with the recipient community and, eventually, is integrated into it. Increasing population of exotic species can exert a negative influence on local ecosystems and native biota. In general, biotic invasion causes disturbance or destruction of habitats and ecosystems, and decline or extinction of native biota (Long, 1981; Lever, 1987).

Risk assessment is a tool for decision-making on whether or not a species is to be introduced or imported and for establishing eradication or regulation programmes against established invasive species (Wittenberg & Cock, 2001). It assesses the rate of entry, likelihood of successful introduction, rate and extent of spread, and economic and ecological impact. However, because no effective methods of assessing the risk of entry, establishment and spread of potential invasive species have been established yet, risk assessment schemes have been practiced in only a limited number of countries (Wittenberg & Cock, 2003). In Japan, the Ministry of Environment formulated a new Biodiversity Strategy in 2002 (http://www.biodic.go.jp/nbsap.html). It includes three main actions in response to biotic introduction:

prevention, assessment and management (regulation and eradication). Because most of the introduced birds and mammals in Japan are escaped captive animals, a legal barrier against the import of animals is necessary. Legal regulation depends on accurate evaluation of invasibility and risk of ecological and economic damage to the recipient ecosystem and biotic community.

In Japan, there have been no nationwide studies on the spread of invasive birds so far. In this study, we carried out a nationwide inquiry of the distribution of exotic birds and their impact on native biota and habitats. Most of these species were recently introduced and the number of founders is small, probably less than ten pairs. The aim of this study is to attempt to assess the likelihood of spread and risk of impact of these invasive species on native biota and ecosystems. Diamond and Case (1986) listed mechanisms of effects on biota by avian invasion: habitat destruction, predation, brood parasitism, disease, hybridization and competition. In this paper, we will review various influences of introduced birds on native biota and ecosystems overseas in regions where they have been introduced first and then discuss possible risks of impact in Japan.

2. Methods

Exotic species known to breed in Japan are listed in Table 1. Species are nominated based on information collected by inquiries, supplemented with data from literature; Iozawa (2000), Ornithological Society of Japan (2000) and Tanaka & Takehara (2003). Investigations were conducted in 2000 and 2001 by e-mail to members of mailing lists ("jeconet" and "banding") and the Ornithological Society of Japan, inquiries on journals of the Wild Bird Society of Japan (WBSJ), Sapporo Branch of WBSJ, Kitakyushu Branch of WBSJ and Japanese Society for Preservation of Birds, and personal inquiries.

Inquiry data were collected from 34 of Japan's 47 prefectures. Information is not evenly distributed throughout the country, but biased to the western half (Fig. 1a). In total, information on 564 cases was collected. While a lot of information, 351 cases, was collected from three prefectures, Nagano, Hyogo and Saitama, there were fewer than ten cases each from 22 prefectures. Therefore, we could show only rough tendencies. The extent of spread for each species was shown at the prefectural level (Table 1, Fig. 1). We included only species in which breeding is suspected according to evidence such as breeding behavior, existence of juveniles, or grouping of adult males and females during the putative breeding season. Only species in which sightings at more than one place were obtained were included. We omitted the data if an individual was suspected to be a vagrant.

In Table 1, we also included the mode of introduction, habitat type in Japan, clutch size, diet, habitat type and climate type of the original region, impact of introduced populations on native biota and habitats in and outside of Japan. Information on Japan was based on inquiry data, the Yamashina Institute for Ornithology (1979), Kuroda & Komiya (1987), Kinjo *et al.* (1987), Kinjo (1998), Eguchi & Takeishi (1997) and Tanaka & Takehara (2003). Information on original habitats and regions of introduction overseas were from Long (1981), Lever (1987), del Hoyo *et al.* (1992, 1997), Robson (2000) and Zheng & Zhang (2002).

3. Invasive Species in Japan

In total, 43 exotic species are known to breed in Japan (Table 1). The Muscovy Duck *Cairina moschata* and Feral Pigeon (*Columba livia*) are feral escapees of domesticated races introduced as food or racing pigeons. The Chinese Bamboo Partridge (*Bambusicola thoracica*) and Ring-necked Pheasant (*Phasianus colchicus karpowi*) were introduced and released for hunting more than 70 years ago (Kiyosu, 1978; Kuroda & Komiya, 1987). The Black-billed Magpie (*Pica pica sericea*) was introduced and released in northern Kyushu in the late 16th century, and it is very common there now (Eguchi & Kubo, 1992). All other species are escaped caged birds imported as pets. Therefore, Psittidae, Timaliidae, Estrilidae, Ploceidae and Sturnidae accounted for more than half of the species listed in Table 1.

Feral Pigeons have spread throughout the country because of a long history of introduction, having been introduced at least 600 years ago (Yamashina Institute for Ornithology, 1979). For Chinese Bamboo Partridges, we obtained sight records from 17 prefectures. However, a nationwide census conducted by the Agency of Environment (1981) showed that this species was distributed in about two thirds of the country, mainly in the western half of Japan. Besides these two species, the Budgerigar (*Melopsittacus undulatus*), Red-billed Leiothrix (*Leiothrix lutea*) and Red Avadavat (*Amandava amandava*) have spread in more than 10 prefectures (Table 1, Fig. 1b – 1d). Budgerigars and Red Avadavats are concentrated in the Kanto region and scattered in southwestern Japan (Fig. 1b, 1c). Distribution of the Red-billed Leiothrix are concentrated around the Kanto region, Kinki region and Kyushu (Fig. 1d), in particular they inhabit all of Kyushu in high abundance (Eguchi & Amano, 2000).

Most introduced land birds inhabit habitats disturbed by humans such as croplands, riparian grasslands and town areas, which correspond to their original habitats (Table 1). Exceptions include four species of Timaliidae, the White-browed Laughing Thrush (*Garrulax sannio*), Spectacled Laughing Thrush (*G. perspicullatus*), Melodious Laughing Thrush (*G. canorus*) and Red-billed Leiothrix, which inhabit forests. Particularly, latter two species have been

Table 1 Exotic birds breeding in Japan.

Species	In Japan				Original populations[2]				Overseas introduced populations[2]
	Mode of entry	No. of cases/ prefectures[*]	Habitats	Impact reported[1]	Habitat type	Climate type	Diet	Clutch size	Impact reported
Anseriformes									
Cygnus olor	E	16/7(2)	waters	n/k	waters	temp	omni	3-7	crop/comp
Aix sponsa	E	4/2	waters	n/k	waters	temp	omni	6-15	n/k
Cairina moschata	F	7/5	waters	n/k	waters	wet trop	herb	8-15	---
Ciconiformes									
Mycteria leucocephala	E	3/1	waters	n/k	waters	wet trop	omni	3-4	---
Galliformes									
Bambusicola thoracica	I	27/17	for	n/k	for	temp	herb/ins	3-7	n/k
Phasianus colchicus	I	11/3	for/crop	hybrid	for/crop	temp	herb/ins	8-14	crop
Colinus virginianus	E	6/3	for/rip	n/k	for/sav/crop	temp	herb/ins	12-18	n/k
Pavo cristatus[3]	E[3]	---/---(1)[3]	for/crop[3]	crop/pred[3]	sav/crop	wet trop	omni	3-8	n/k
Columbiformes									
Columba livia	F	---/coutrywide	town	crop/dis	rocky	temp/trop	herb	2	crop/comp
Psittiformes									
Melopsittacus undulatus	E	24/9(3)	town/rip	n/k	sav	arid temp	gran/ins	4-8	crop
Myiopsitta monachus	E	2/2	town	n/k	sav	dry trop	gran	1-11	crop
Psittacula eupatria[4]	E[4]	---/---(1)[4]	town[4]	n/k	for	wet trop	frug	3-4	crop/dis/comp
Psittacula krameri	E	17/6	town	n/k	for/crop	wet trop	frug/gran	3-4	crop
Psittacula alexandri	E	3/1(1)	town	n/k	for	wet trop	herb	3-4	crop
Passeriformes									
Pycnonotus jocosus	E	2/2	town	n/k	sav/for	wet trop	frug/ins	2-4	crop
Pycnonotus sinensis[5]	E[5]	---/---(1)[5]	town/crop[5]	crop[5]	for/crop	temp	frug/ins	3-4	---
Garrulax perspicillatus	E	6/3	for	n/k	for/crop	temp/trop	frug/ins	2-5	---
Garrulax sannio	E	2/2	for	n/k	for	temp	frug/ins	3-4	n/k
Garrulax canorus	E	36/6(3)	for	n/k	for	temp	frug/ins	3-4	comp
Leiothrix lutea	E	78/21(1)	for	n/k	for	temp	frug/ins	3-4	crop/dis/comp
Paroaria coronata	E	4/3(1)	rip/crop	n/k	for	temp	herb/ins	2-4	n/k
Estrilda troglodytes	E	3/2(1)	rip/crop/town	n/k	sav/rip	dry trop	gran/ins	3-6	n/k
Estrilda melpoda[4]	E[4]	---/---(1)[4]	rip/crop[4]	n/k	sav/crop	dry trop	gran	5-6	n/k
Lonchura punctulata	E	8/3	rip	n/k	sav/crop	dry trop	gran	5-7	crop
Lonchura malacca	E	10/4(1)	rip	n/k	sav/crop/rip	wet trop	gran	5-7	crop
Lonchura atricapilla	E	8/6(3)	rip/crop	n/k	sav/crop	temp	gran	3-6	---
Lonchura striata[6]	E[6]	---/---(1)[6]	crop[6]	n/k	sav/crop	wet trop	gran	5-6	crop
Lonchura maja	E	13/5(1)	rip	n/k	sav/crop	wet trop	gran	4-5	---
Padda oryzivora	E	9/2(2)	rip/crop	n/k	crop	wet trop	gran	4-5	crop/comp
Amandava amandava	E	63/12(2)	rip	n/k	sav/crop/rip	wet trop	gran /ins	4-7	crop
Vidua paradisaea	E	2/2(1)	town	n/k	sav	dry trop	gran	parasite	n/k
Vidua macroura	E	4/3	rip	n/k	sav/crop	dry trop	gran	parasite	n/k
Euplectes afer[6]	E[6]	---/---(1)[6]	rip/crop[6]	n/k	sav	dry trop	gran	3-4	n/k
Euplectes orix	E	4/4	rip/crop	n/k	sav	dry trop	gran	2-5	crop
Ploceus manyar	E	2/2	rip	n/k	rip/crop	wet trop	gran/ins	2-5	---
Ploceus intermedius[4]	E[4]	---/---(1)[4]	rip/crop[4]	n/k	sav	dry trop	gran/ins	---	---
Sturnus contra	E	3/1(1)	town	n/k	crop/town	wet trop	omni[7]	4-6	---
Acridotheres tristis	E	4/1(2)	town	n/k	sav/crop	wet trop	omni	3-6	crop/dis/pred/ comp
Acridotheres ginginianus	E	2/1(1)	town	n/k	sav/crop	wet trop	omni[7]	---	---
Acridotheres fuscus	E	3/1(2)	town	n/k	sav/crop	wet trop	omni	4-6	crop/hybrid
Acridotheres cristatellus	E	51/5(2)	town/rip	n/k	crop	temp/trop	omni	4-6	crop/comp
Pica pica	I	16/5(4)	town/crop	crop	for/sav/crop	temp	omni	5-9	n/k
Urocissa caerulea	E	3/2	town	n/k	for	temp	omni	3-8	---

[*]: figures in parentheses are additional data based on literature.

[References] [1]: Yamashina Institute for Ornithology (1979), Kinjo *et al*. (1987), Kuroda & Komiya (1987), Eguchi & Takeishi (1997), Kinjo (1998), Tanaka & Takehara (2003), [2]: Long (1981), Lever (1987), del Hoyo *et al.* (1992, 1997), Roberts (1992), Robson (2000), Zheng & Zhang (2002), [3]: Tanaka and Takehara (2003), [4]: Ornithological Society of Japan (2000), [5]: Kinjo *et al*. (1987), Kinjo (1998), [6]: Iozawa (2000), [7]: Roberts (1992).

[Abbreviation] Mode of entry: E=escapees; F=feral individuals, I=intentional releases, Habitats/Habitat type: waters= rivers, lakes, ponds and estuaries; for= indigenous forests, woodlots and bamboo thickets; rip= swamps, reed beds, riparian grasslands and grasslands; crop= paddy fields, croplands, and orchards; town= parks, town areas, and human habitations; sav= savannas, forest edges and woodlands; rocky=coastal escarpments, Climate type: temp=temperate; trop=tropic, Diet: omni=omnivore; herb=herbivore; gran=granivore; frug=frugivore; ins=insectivore; Impact reported: crop=crop damage; hybrid=hybridization; comp=competition with native species; dis=disease; pred=predation; n/k=not known; ---=no available data.

increasing rapidly in indigenous forests (Eguchi & Amano, 2000; Sato, 2000; Kawakami & Yamaguchi, 2004).

Thus, while the Feral Pigeon and Chinese Bamboo Partridge have reached the integration phase and penetrated into the community or urban ecosystem, the Melodious Laughing Thrush and Red-billed Leiothrix may be in proceeding into the integration phase. Budgerigars are widely distributed (Fig. 1b), but each population is quite small and is not increasing. Red Avadavats are also widely distributed (Fig. 1c). Although this species sometimes becomes locally common (Lever, 1987), it does not maintain a self-sustaining population for a long time probably due to a great year-to-year population fluctuation (Shigeta, 1997). Crested Mynas (*Acridotheres cristatellus*) are currently limited to a small region (Table 1), but are increasing rapidly in urban areas in Hyogo

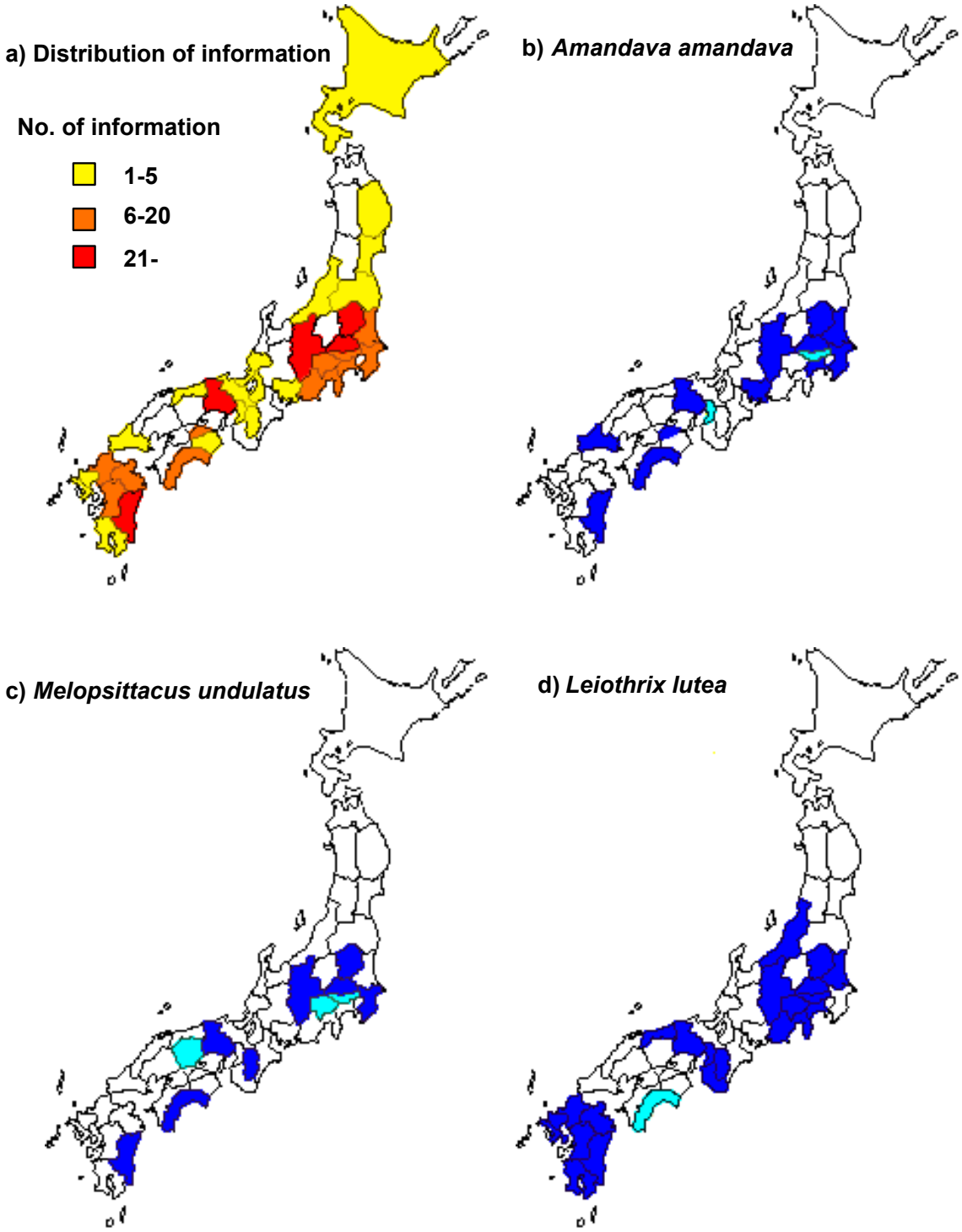

Fig. 1 Distribution of information and major exotic species. (a) Number of items of information. Data (two items) from Okinawa Prefecture are not shown in the figure, (b) Red Avadavat, (c) Budgerigar, (d) Red-billed Leiothrix. In Figs. 1b, 1c and 1d, data for prefectures marked with dark blue and light blue are based on our inquiries and information from the Ornithological Society of Japan (2000), respectively.

Prefecture (H. Kuroda, pers. comm.). For other species, because sightings are sporadic, they have hardly been established yet or, if they have, are at a very low density.

4. Impacts of Exotic Birds on Local Ecosystem and Native Biota

4.1 Influence on agriculture and ecosystem

4.1.1 Crop damage and other economic damage

Economic damage to agriculture and other human activities are one of the largest costs of biotic invasion (Pimentel *et al.*, 2000). Because such damage is easily recognized, many reports of damage caused by introduced species are available (cf. Long, 1981). About a half of introduced birds in Japan are agricultural pests overseas where they have been introduced (Table 1). Particularly, parrots damage cereals and fruits; waxbills and weavers, cereals; and mynas, cereals and fruits. In the USA, Feral Pigeons and European Starlings (*Sturnus vulgaris*) are serious crop pests, estimated to cause 1.9 billion dollars in damage per year (Pimentel *et al.*, 2000).

In Japan, Feral Pigeons are a serious crop pest, particularly, on soybeans (Yamashina Institute for Ornithology, 1979). The Light-vented Bulbul (*Pycnonotus sinensis*) causes damage to fruits in Okinawa Prefecture (Kinjo *et al.*, 1987; Kinjo, 1998). The high density of the Common Peafowl (*Pavo cristatus*) is a possible cause of crop damages on islets of the Ryukyu Islands, though no damage to crops has been claimed because there is no commercial agriculture there (Tanaka & Takehara, 2003). Many Black-billed Magpies in high density locally build nests on power poles (Eguchi & Takeishi, 1997), which causes an increase in electrical accidents. Rose-ringed Parakeets (*Psittacula krameri manillensis*) also are locally common around Tokyo, but no economic damage has been reported yet. Because of very low densities and restricted ranges, no economic damage has been reported for other exotic species.

4.1.2 Influence on ecosystem and habitat

Cases of habitat destruction by introduced birds have been rarely reported. However, if large birds are introduced into small islands and increase in number, adverse effects on the environment may be possible. The expansion of the introduced Common Peafowl on small islets of the Ryukyu Islands, has probably had an adverse effect on both local biota and habitats. This species is omnivorous. On islets where peafowl exist in high density, local vegetations have been modified and some animals including the endemic lizard *Eumeces kishinouyei* have decreased (Tanaka &Takehara, 2003).

In the Hawaiian Islands, the Common Myna (*Acridotheres tristes*) is known as a major seed disperser of the introduced harmful weed, *Lantana camara* (Pimentel *et al.*, 2000). Thus, mynas have caused a change in habitats indirectly by seed dispersal of an exotic weed. In Japan, some introduced frugivorous species like mynas and Red-whiskered Bulbuls inhabit town areas and suburbs where exotic plants are common. If these exotic birds increase, they may disperse seeds of weeds and exotic trees to adjacent indigenous habitats. Of course, one cannot necessarily condemn only exotic birds for seed dispersal of weeds and exotic plants; native frugivores like the Brown-eared Bulbul (*Hypsipetes amaurotis*) do that in the same way.

4.2 Influence on native avifauna

4.2.1 Predation and interference

The Common Myna and Red-vented Bulbul (*Pycnonotus cafer*) act aggressively sometimes towards native bird species in Mascarenes and Tahiti (Jones, 1996; Thibault *et al.*, 2002; Blanvillain *et al.*, 2003). In Tahiti, native endangered Tahiti Flycatchers (*Pomarea nigra*) have been decreasing mainly due to predation by rats. In the process of carrying on a recovery program, Blanvillain *et al.* (2003) assessed the impact of Common Mynas and Red-vented Bulbuls on the reproductive success of flycatchers. Nest failure and death of newly fledged young occurred often in territories where aggressive interactions by introduced birds were observed frequently. The fact that a higher number of interactions with Common Mynas was observed around nests containing eggs and nestlings suggests nest predation by this species (Blanvillain *et al.*, 2003). Common Mynas have been introduced into many islands for controlling insect pests (Long, 1981). However, this species is reputed to be a nest thief (Blanvillain *et al.*, 2003).

On the other hand, Red-vented Bulbuls affect flycatchers through aggressive competitive interaction for food, not direct nest predation (Blanvillain *et al.*, 2003). Aggressive interaction by this species is presumed to have a possible negative impact also on other monarch (*Pomarea*) populations in the Polynesian Islands (Thibault *et al.*, 2002). It is likely that endangered species on a small remote oceanic island are susceptible to decline of numbers through interference by introduced birds.

If a brood parasite species is introduced, new host species incapable of egg rejection will suffer serious reproductive losses, *e.g.*, impact of the Brown-headed Cowbird (*Molothrus ater*) (Rothstein *et al.*, 1980). So far, no cases have been reported of predation or brood parasitism by introduced species in Japan. In Japan, however, two species of *Vidua* are known to breed (Table 1). *Vidua* species are brood parasites in their original habitats (Davies, 2000). Although host species have not been observed yet, these species may influence the breeding of native species.

4.2.2 Disease

A decline in native bird species due to disease derived from the invasion of introduced birds has been well documented in the Hawaiian Islands (van Riper *et al.*, 1986). In the Hawaiian Islands, avian malaria

and its vector have accompanied the invasion of introduced birds and reduced the number of native species (van Riper *et al.*, 1986; Dobson & May, 1991). In the Mascarene Islands, avian pox and parrot diseases derived from introduced birds have prevailed (Jones, 1996). Species on small and remote islands are particularly susceptible to new pathogens and parasites brought in by introduced birds.

Furthermore, exotic species living near human beings and livestock, such as European Starlings, House Sparrows (*Passer domesticus*), Common Mynas, Feral Pigeons and Rose-ringed Parakeets are reservoirs and vectors of human and livestock diseases, including parrot fever and ornithosis (Lever, 1987; Pimentel *et al.*, 2000).

In Japan, no epidemics arising from introduced exotic birds have prevailed yet. However, there have been very few studies conducted on diseases and parasites of Japanese native birds. Several studies have shown that the prevalence of blood parasites is low in grassland dwelling warblers and buntings in Japan (Sodhi *et al.*, 1996, 1999; Nagata & Sodhi, 2003). Introduced birds could act as a large reservoir of various parasites and pass them on to native parasite-free birds.

Further, the risk of infection with diseases which also infect humans and livestock like parrot fever or West Nile fever is high because many caged birds are imported every year (Agency of Environment, 1986; Nash, 1993).

4.2.3 Hybridization and Introgression

Introduced species can hybridize with closely related native species. This may cause the extinction of endangered native species. Sterility in hybrids results in wasted reproductive efforts, which may cause a population decline. If hybrids are fertile, introgressive hybridization results in a loss of original morphological traits in either species (Rhymer & Simberloff, 1996; Allendorf *et al.*, 2001). Where an introduced species outnumbers a native species, original traits of the native species disappear.

In Anatidae, interspecific hybridization occurs often and, in many cases, hybrids are fertile (Parkin, 1996). In particular, introduced Mallards (*A. plathyrhynchos*) or domesticated races have introgressively hybridized with endemic species, causing a decline in the latter (Long, 1981; Hunter, 1996; Rhymer & Simberloff, 1996). The Hawaiian Duck (*Anas wyvilliana*) in Hawaii (Browne *et al.*, 1993), Florida Mottled Duck (*A. f. fulvigula*) in south Florida (Rhymer & Simberloff, 1996; Wittenberg & Cock, 2001), Australian Black Duck (*A. superciliosa rogersi*) in Australia (Long, 1981; Lever, 1987) and Grey Duck (*A. superciliosa*) in New Zealand (Haddon, 1984; Gillespie, 1985; Rhymer *et al.*, 1994) have been decreasing due to introgressive hybridization with introduced Mallards. The Ruddy Duck (*Oxyura jamaicensis*) introduced from North America also threatens the native White-headed Duck (*O. leucocephala*) with extinction through hybridization in Britain (Hughes, 1996).

The case of the Grey Duck is particularly serious. Gillespie (1985) demonstrated the decrease of pure Grey Duck populations, based on an analysis of the morphometric variations, due to introgressive hybridization with introduced Mallards in Otago, New Zealand. In New Zealand, Mallards were introduced for the first time during the 1860s. Mallards easily hybridized with native Grey Duck and the hybrids were fertile. Since the first introduction of Mallards, the number of pure Grey Ducks has been decreasing. In 1981-1982, the proportions of pure Grey Ducks, hybrids and Mallards in samples collected during the hunting seasons were 4.5%, 51.4% and 44.1%, respectively (Gillespie, 1985). By the end of the 1950s, the Mallards already outnumbered Grey Ducks and hybrids have increased rapidly since then (Gillespie, 1985). Because pure Grey Duck individuals occupy a very small proportion in the whole Mallard-Grey Duck collective, hybridization is inevitable. Therefore, local disappearance of pure Grey Ducks is feared. Rhymer *et al.* (1994) also demonstrated, based on the mtDNA analysis, the introgressive hybridization occurring between these two species throughout the country.

Thus, introgressive hybridization between Mallards and Grey Ducks has expanded drastically to an extent that some authors have stated that nearly all Grey Duck populations have become genetic admixtures (Allendorf *et al.*, 2001). In Japan, hybridization between domestic races of the Mallard and the Spot-billed Duck (*A. poecilorhyncha*) has also been reported (Nakamura, 1994), but the impact on native species is not prominent yet.

In a polytypic species, introduction beyond a range of one particular subspecies causes a loss of original traits of each of the other subspecies adapted to local environments. Therefore, the release of game birds beyond the borders of individual locally adapted populations is harmful.

In Japan, few studies have been conducted on hybridization among birds in natural habitats. The Ring-necked Pheasant was introduced onto three main islands, Honshu, Shikoku and Kyushu, around 1920, and hybridization with the native Green Pheasant *Phasianus versicolor* occurred (Kuroda & Komiya, 1987; Lever, 1987). Thereafter, introduction was ceased for fear that hybridization would spread in these islands. On the other hand, introduction was started around 1930 and continued in Hokkaido where no closely related species lived.

The Green Pheasant comprises four subspecies (Ornithological Society of Japan, 2000). Captive breeding and release for hunting have been conducted. However, because release was conducted beyond the range of each subspecies, the uniqueness of each subspecies has been lost due to introgressive hybridization among them (Ornithological Society of Japan,

2000). Reintroduction of endangered species should be carefully planned so as to reduce the likelihood of hybridization.

4.2.4 Interspecific competition for nest sites

Hole-nesting introduced birds such as the European Starling, House Sparrow, Common Myna and Rose-ringed Parakeet can compete with native species for nesting holes (Long, 1981; Lever, 1987; Feare, 1996; Lindell, 1996, Pimentel *et al.*, 2000). Competition for nesting sites is more severe among hole-nesters than open-nesters because natural nest holes are often in short supply (Lindell, 1996).

In North America, European Starlings harass various cavity excavating species like woodpeckers with aggressive interaction and usurp freshly excavated nest cavities. In Mississippi, 52% of nest cavities of the Red-bellied Woodpecker (*Melanerpes carolinus*) were usurped by starlings (Ingold, 1989). In Ohio, 39%, 15% and 14% of nest cavities excavated by Red-bellied Woodpeckers, Red-headed Woodpeckers (*M. erythrocephalus*) and Northern Flickers (*Colaptes auratus*), were usurped by starlings, respectively (Ingold, 1994). Because starlings and Red-bellied Woodpeckers start breeding early in highly overlapping periods, the proportion of usurped nest cavities was highest.

Nest usurpation did not prevent the host species from breeding at all, but forced them to delay breeding within the season. Usurped Red-bellied Woodpeckers started replacement nests in the vicinity of their previous nests and eventually fledged young (Ingold, 1994). However, the delayed breeding caused a decline in fecundity more or less, such as seasonal declines in clutch size and fledging success due to a decreased food supply. Therefore, Ingold (1998) concluded that nest usurpation by starlings had an adverse effect on the breeding success of host species.

Nest usurpation by European Starlings is also prominent in other species: Gila Woodpeckers, *Melanerpes uropygialis* (Kerpez & Smith, 1990) and Acorn Woodpeckers, *M. formicivorus* (Troetschler, 1976). In all reports so far, the adverse effect of nest usurpation is unambiguous. However, the demographic data collected in larger areas showed no significant long-term declines in native cavity nesting species. Koenig (2003) compared mean densities of cavity-nesting species native to North America before and after the invasion of European Starlings using Breeding Bird Surveys and Christmas Bird Counts. Of 27 native species, he could detect a significant decline in population density after the invasion only in sapsucker species (*Sphyrapicus* spp.). There are some possible reasons which account for the lack of long-term declines despite the unambiguous adverse effect of nest usurpation. Starlings usurping nest cavities do not always drive host pairs away from their nesting areas. Therefore, individual woodpecker pairs whose nests have been usurped excavate a new nest in the vicinity of their initial nests and some of them eventually fledge young (Ingold, 1989, 1994; 1998). A delay in breeding may lower their fecundity. However, the decline in fecundity due to a delay in breeding may be small or nonexistent. For example, in the Northern Flicker, a significant decline occurred only in clutch size and not in the number of fledglings of the usurped pairs (Ingold, 1998). Therefore, a large year-to-year fluctuation in reproductive output due to factors other than nest usurpation may mask the decrease in fecundity due to nest usurpation. This demonstrates the difficulty of predicting the effect of invasive species.

In Japan, three species of *Acridotheres* and the Rose-ringed Parakeet have been increasing. Yet no usurpation has been reported of nest cavities of native species by these species, probably because the distribution ranges are restricted to urban areas so far.

4.2.5 Interspecific competition for food

In the Mascarene Islands, there are some cases of interspecific competition, which have caused a decline in native species, *e.g.*, the Common Myna and Red-whiskered Bulbul affecting the Mauritius Bulbul (*Hypsipetes olivaceus*) and Mauritius Cuckoo-shrike (*Coracina typica*), the Madagascar Fody (*Foudia madagascariensis*) affecting the Mauritius Fody (*F. rubra*), and the Rose-ringed Parakeet affecting Mauritius Parakeet (*Psittacula eques*) (Jones, 1996).

In Hawaiian native forests, generalist species such as the Japanese White-eye (*Zosterops japonicus*), Melodious Laughing Thrush and Red-billed Leiothrix outnumber native species specializing in a particular food resource (Mountainspring & Scott, 1985). Although authors have attributed the decline to competition for food, no evidence for the mechanism of competition was shown.

In Japan, the Red-billed Leiothrix has also invaded broadleaved deciduous forests. This species is dominant over the avifauna of such habitats in Kyushu (Eguchi & Masuda, 1994; Eguchi & Amano, 2000). Among the avifauna of broadleaved deciduous forests, major species occupying a similar niche to this species are the Japanese Bush-warbler (*Cettia diphone*), Great Tit (*Parus major*), Varied Tit (*P. varius*), Coal Tit (*P. ater*), Willow Tit (*P. montanus*) and Long-tailed Tit (*Aegithalos caudatus*). Although there is no evidence to confirm the presence of interspecific competition between introduced and native species for food resources or nest sites, predation on Japanese Bush-warblers nesting in the same nesting habitat as the Red-billed Leiothrix has been found to be higher than in other regions where the latter species has not yet invaded (Amano & Eguchi, 2002a, 2002b). Even if there is no direct competition, a high density of the Red-billed Leiothrix may cause a decline in reproductive success in sympatric species through “apparent competition” (Martin & Martin, 2001) mediated by attraction of predators.

Kawakami and Higuchi (2003) could not detect

any negative interspecific relationships between the endemic *Apalopteron familiare* and introduced Japanese White-eye in spite of similar food habits in the Bonin Islands.

Thus, it is even more difficult to demonstrate the existence of direct interspecific competition for food than for nest-sites. So far, in some cases, native bird species are presumed to have decreased in number due to interspecific competition with introduced species. However, no demonstration with quantitative evidence has been done in either case. Introduced species are common in disturbed habitats like croplands and town areas where native species have already disappeared (Green, 1984; Blair, 1996; Case, 1996; this study). Therefore, they may compete with other introduced species (Moulton & Pimm, 1986). In indigenous forests, introduced species interact with native species. Because the habitat structure is complicated and food resources are diverse there, we might be unable to find a negative interspecific relationship between native species and introduced species successfully invading such a habitat. If so, interspecific competition for food is usually not severe between closely related species or between species belonging to the same guild, contrary to the competition for nest holes. In most instances where an introduced species has exterminated a native species, the species interactions were between trophic levels (Davis, 2003). Probably, a top-down effect caused by introduced species causes more serious results for native species. For example, the Common Peafowl in the Ryukyu Islands modified the vegetation and caused a decrease in some prey animals, which may have caused a decline in sympatric birds such as White-eyes (Tanaka & Takehara, 2003).

5. Invasiveness of Introduced Birds in Japan

In total, 79 exotic species were sighted out of captivity in our inquiry investigation. Forty-three of them were likely to breed. However, there have been few species establishing self-sustaining populations so far. Most exotic species arrived not so long ago. It took only a couple of decades for the Red-billed Leiothrix to spread throughout Kyushu (Amano & Eguchi, 2002a). Hence, it is likely that the number of species establishing a self-sustaining population will increase during the next decade.

There is no doubt that the success of introduction is determined primarily by the effort put into introducing a species, i.e., the size and number of releases (Veltman *et al.*, 1996; Duncan, 1997; Green, 1997). However, if efforts to introduce them are controlled, there are significant relationships between some ecological traits of both the introduced species and the recipient community and the success of introduction; higher success rates are seen in species which are more sedentary (Veltman *et al.*, 1996; Cassey, 2002), have a larger body mass (Green, 1997; Duncan *et al.*, 1999), small clutch size (Green, 1997), higher level of behavioral flexibility (larger relative brain size, higher frequency of foraging innovation and higher habitat generalism; Sol & Lefevre, 2000; Sol *et al.*, 2002; Cassey, 2002) and originate from latitudes closer to those of the areas into which they are being introduced (Blackburn & Duncan, 2001). Species with smaller clutches may have a lower reproductive rate but survive longer, and it is not clear whether reproduction or survival is more important (Green, 1997). Smaller differences in latitude may mean that the species is being introduced into areas with the same climate as the original habitats and can establish itself successfully (Blackburn & Duncan, 2001). However, ecological traits are not always a reliable predictor in assessing invasiveness of each species (Willenberg & Cocks, 2001). Outcomes of introduction are often different between related species. Simberloff and Boecklen (1991) stated that the most reliable predictor is whether or not the species was successful at establishing itself in other areas.

Systematic and large-scale introductions, such as those by acclimatization societies, is no longer done. A typical pattern of establishment is that escaped caged birds gather and breed by chance. However, ecological traits of both the introduced species and the recipient community and abiotic factors of habitat may determine whether their number increases or not. Most exotic birds breeding in Japan are from the tropics (Table 1). In general, birds in the tropics have a long breeding season and lay several clutches within a single season. Indeed, the Red Avadavat in Japan breeds out of the ordinary season when most native birds breed (Shigeta, 1997). The Red-billed Leiothrix has a longer breeding season than native species (Eguchi & Amano, unpublished data). If global warming progresses, it may become more likely that exotic species from the tropics will establish themselves and increase in Japan.

The hole-nesting Rose-ringed Parakeet and mynas currently inhabit town areas. However, they may begin to compete with native hole-nesting species if they increase and extend their ranges towards rural areas.

Many species closely related or identical to Japanese races have been imported as pets mainly from the Eurasian continent (Nash, 1993). It is feared that these birds may hybridize with Japanese races and that the original traits of the Japanese races will disappear through introgressive hybridization.

Currently, there are only a few exotic species with high density, and serious adverse effects are prominent for only several species: the Feral Pigeon, Light-vented Bulbul and Common Peafowl. However, closely related or identical species have increased in after being introduced into areas other than Japan and have damaged native biota and crops there (cf. Long, 1981; Lever, 1987). Even if only a few pairs breed currently, the risk that these species will increase and

become a threat to native biota and habitats in the near future is high. Most exotic birds in Japan are omnivores, granivores, or frugivorous insectivores. These species may easily become crop pests if they occur at high densities.

Finally, the effects of introduced birds on ecosystems, biota and human society are not obvious. Most introduced birds inhabit disturbed habitats (*e.g.*, Blair, 1996; Case, 1996). Therefore, introduced birds replace native species that have disappeared due to human disturbance, and hence play a role in maintaining ecosystems and mitigating environments for human beings (Case, 1996). For example, introduced game birds occupy a similar niche held by now-extinct or rare native species and facilitate seed dispersal and germination of native plants in disturbed habitats in Hawaii (Cole *et al.*, 1995). Therefore, people believe that because introduced birds are harmless compared to other animals, they should be allowed to exist in disturbed habitats until they start to exhibit negative impacts on the ecosystem and humans. However, proliferation of introduced species in local regions means a loss of unique regional avifauna and facilitates homogenization of fauna on a global level. Rather, we should try to regenerate disturbed habitats and create room for native birds to return there.

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Appendix 1 English names and Scientific names of invasive birds in Japan.

Order	Family	English name	Scientific name
Anseriformes	Anatidae	Mute Swan	*Cygnus olor*
		American Wood Duck	*Aix sponsa*
		Muscovy Duck	*Cairina moschata*
Ciconiiformes	Ciconiidae	Painted Stork	*Mycteria leucocephala*
Galliformes	Phasianidae	Chinese Bamboo Partridge	*Bambusicola thoracica*
		Ring-necked Pheasant	*Phasianus colchicus karpowi*
		Northern Bobwhite	*Colinus virginianus*
		Common Peafowl	*Pavo cristatus*
Columbiformes	Columbidae	Feral Pigeon	*Columba livia*
Psittiformes	Psittacidae	Budgerigar	*Melopsittacus undulatus*
		Monk Parakeet	*Myiopsitta monachus*
		Alexandrine Parakeet	*Psittacula eupatria*
		Rose-ringed Parakeet	*Psittacula krameri manillensis*
		Moustached Parakeet	*Psittacula alexandri*
Passeriformes	Pycnonotidae	Red-whiskered Bulbul	*Pycnonotus jocosus*
		Linght-vented Bulbul	*Pycnonotus sinensis*
	Timaliidae	Spectacled Laughing Thrush	*Garrulax perspicillatus*
		White-browed Laughing Thrush	*Garrulax sannio*
		Melodious Laughing Thrush	*Garrulax canorus*
		Red-billed Leiothrix	*Leiothrix lutea*
	Emberizidae	Red-crested Cardinal	*Paroaria coronata*
	Estrildidae	Black-rumped Waxbill	*Estrilda troglodytes*
		Orange-cheeked Waxbill	*Estrilda melpoda*
		Nutmeg Mannikin	*Lonchura punctulata*
		Tricolored Mannikin	*Lonchura malacca*
		Chestnut Mannikin	*Lonchura atricapilla*
		White-rumped Munia	*Lonchura striata*
		White-headed Mannikin	*Lonchura maja*
		Java Sparrow	*Padda oryzivora*
		Red Avadavat	*Amandava amandava*
	Ploceidae	Paradise Whydah	*Vidua paradisaea*
		Pin-tailed Whydah	*Vidua macroura*
		Yellow-crowned Bishop	*Euplectes afer*
		Red Bishop	*Euplectes orix*
		Streaked Weaver	*Ploceus manyar*
		Lesser Masked Weaver	*Ploceus intermedius*
	Sturnidae	Asian Pied Starling	*Sturnus contra*
		Common Myna	*Acridotheres tristis*
		Bank Myna	*Acridotheres ginginianus*
		Indian Jungle Myna	*Acridotheres fuscus*
		Crested Myna	*Acridotheres cristatellus*
	Corvidae	Black-billed Magpie	*Pica pica*
		Formosan Blue Magpie	*Urocissa caerulea*

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