‘Survival ecology’: an urgent ecological study of birds imperilled by the cage-bird trade across Java and Bali, Indonesia

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‘Survival ecology’: an urgent ecological study of birds imperilled by the cage-bird trade across Java and Bali, Indonesia

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General abstract

Throughout Southeast Asia, the trade in wild-caught songbirds—prized for their vocal ability, plumage, rarity and cultural significance—is having a massive effect on wild populations, such that an Asian Songbird Crisis was declared in 2017. Indonesia, particularly its most populous island of Java, is widely regarded as the epicentre of Southeast Asia's cage-bird trade, with millions of birds sold annually at markets irrespective of their legal status and astonishing levels of bird ownership that have led to estimates that there may actually be more songbirds kept in cages across Java than there are in the wild. The aim of this thesis was to understand the ecology and management needs of some of the passerines most threatened by the cage-bird trade across Java and Bali, Indonesia, to guide in situ conservation actions. First, I implemented a citizen science event in Java and Bali to gather bird occurrence data and examine the potential for citizen science as a conservation tool in Indonesia. I then used these data to model the distributions of 23 of Java's lowland birds in order to assess the convergence between the current distribution and previous distribution maps and the network of protected areas. Following this, I studied the ecology and conservation management of two of Java and Bali's most endangered sturnids, the Black-winged Myna (*Acridotheres melanopterus*) and Bali Myna (*Leucopsar rothschildi*).

There are large existing gaps in biological data coverage that hinder efforts to generate robust baseline information on the distribution and abundance of birds across Java and Bali. I attempted to address this by designing and implementing a month-long citizen science event, 'BigMonth2020', which had the dual aim of engaging Indonesian society in citizen science and generating a large bird occurrence dataset. The event was publicised through social media and incentivised with grants and competitions. A huge number of bird records (n = 102,887) were submitted to the 'Burungnesia' phone app during the event, resulting in a massive increase (147%) in spatial coverage of data, so that now 79.3% of grid squares contain at least some data. Three quarters of Java and Bali's bird species (n = 353) were recorded and this included 27 globally threatened species, many of which were recorded in new areas. The event was more inclusive in terms of female participation (23.4% of participants were female) than other bird-related pastimes in Indonesia, such as bird-keeping and songbird contests, and the vast majority (71.8%) of participants were under 30-years old. The project cost less than

US$10,000 to run, and serves as a model for rapidly establishing a distributional baseline for monitoring biodiversity trajectories.

The current distributions of many of Java's lowland passerines remain poorly understood, and this lack of baseline data precludes efforts to monitor distribution changes in threatened species. Data generated from BigMonth2020 were combined with other citizen science bird datasets available for Java (eBird, Burungnesia and the Indonesian Bird Atlas) to assess the current distributions of 23 of Java's lowland birds. Most species exhibited relatively patchy distributions that were often significantly smaller than existing estimates for their extent of occurrence. Among the environmental variables used in modelling, land-cover-based predictors were ultimately the most important in the models for the majority of species (20/23), with landscape-scale habitat diversity, the proportion of forest, and the proportion of cultivated land most commonly the most important predictor. The lack of convergence between the current distribution of the modelled species with Java's formally protected areas suggests that future conservation for these and other lowland birds, which are likely to come under increasing anthropogenic pressure, will need to occur alongside people and involve other effective area-based conservation measures (OECMs). My findings highlight the considerable value of continued citizen science efforts across Java, and indeed elsewhere in data-poor yet biodiverse regions.

Sturnids are popular cage-birds owing to their vocal ability and the bold colours and striking patterns of their plumage and in some cases bare facial skin. As a result, there have been disastrous population declines in some of Java's sturnids. I documented the plight of the Black-winged Myna, a Java and Bali endemic that has been trapped to near extinction. I estimated the current range and population size of the species at Baluran National Park, which supports Java's last known population, and used species distribution modelling to evaluate the suitability of currently unoccupied areas across the park to identify priorities for management intervention. I estimated that the Black-winged Myna population numbers 179 individuals (95% CI: 111–288) and that its current range is restricted to a small area (12.3 km^2) of savanna and dry deciduous woodland, while my model indicated that a considerable extra portion (72.1 km^2) has potentially suitable habitat. I inferred that the main cause for the disparity between its current and potential range is trapping, compounded by savanna loss and degradation due to overgrazing by cattle and the spread of invasive thorny acacia (*Vachellia nilotica*). The recent partial clearance of acacia appears to have assisted a modest population recovery by the myna, but its further population growth depends on effective management of illegal poaching, further clearance of acacia, and easing grazing pressure on areas of savanna, particularly through engagement with human communities living inside the park.

Continuing with the theme of studying endangered sturnids, I measured the viability of the Bali Myna population at Bali Barat National Park (BBNP). Despite decades of conservation efforts, in the 2000s it was reported that there were probably no Bali Mynas left in the wild, and it is unlikely these reports would be false considering how well-known the location of the last individuals' was and that mynas can be readily detected by call. Since then, reintroductions of captive-bred birds and other management interventions have led to population growth. To plan for the next decade of conservation management, I modelled the Bali Myna population at BBNP to explore the effects of (1) changes to population supplementation and (2) an increase in trapping intensity. A baseline model was validated using population census and captive-bred release data from the last ten years and the model was projected ten years into the future. The population was predicted to increase under current levels of supplementation, while stopping supplementation in five years had only a small effect. I modelled the differential effects of two trapping methods used by poachers and three trapping volumes. The population was resilient to low levels of trapping with and without population supplementation but declined under high levels of trapping. On current trajectory, I estimated that the population will approach self-sustainability in the next 5–10 years. The supplementation programme at BBNP could then either be scaled back or repurposed as a translocation project to expand the myna's range, and nest-boxes could be used to support population growth.

There is much more work needed to address issues related to the Asian Songbird Crisis, and I conclude by providing some recommendations for future work that are related to the topics covered in this thesis. Among these are the need to continue to grow citizen science efforts across Indonesia, a recommendation for urgent fieldwork to understand the status of at least three of Sulawesi's six island endemic sturnids (and indeed other Indonesian sturnids that are poorly known), a call for community-based conservation projects, and further ecological fieldwork to support reintroductions and conservation management of threatened species.

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Contents

Chapter 1

1 Introduction

1.1 The global biodiversity crisis

Over the last 550 million years there have been five mass extinction events—relatively short periods of geological time during which 76% of living species were lost (Jablonski, 1991). All of these extinction events were triggered by natural catastrophes and separated by tens of millions of years (Schulte et al., 2010; Penn et al., 2018). Occurring at just over half the average inter-extinction interval, there are clear signs that we are now entering Earth's sixth mass extinction event, but unlike those that preceded it, this extinction event is being driven by a single element of biodiversity—it is human-induced (Ceballos et al., 2015b). The enormous expansion of humans since the last ice age and the effects we have had on ecosystems through destructive technologies and the staggering growth in populations of domesticated animals has had such a profound impact on the planet that we are said to have entered a new, human-dominated geological epoch dubbed the Anthropocene (Crutzen, 2002; Steffen et al., 2008).

It is estimated that the current rates of species extinction are about a thousand times greater than the estimated background extinction rate (Pimm et al., 2014). Human activities are the primary cause of the current global decline in biodiversity (Purvis et al., 2000), and the principal drivers are habitat destruction and fragmentation (Malhi et al., 2008; Gibson et al., 2011), the spread of invasive species (Gurevitch and Padilla, 2004), climate change (Malcolm et al., 2006), pollution (Gibbon et al., 2000), and overexploitation—the harvesting of wildlife at rates that cannot be compensated for with reproduction or regrowth (Sodhi et al., 2004; Sadovy de Mitcheson et al., 2013; Maxwell et al., 2016). However, none of these drivers act on ecosystems in isolation: their overall impact on biodiversity is the product of the drivers' interactions, which are complex and include synergies and feedbacks (Brook et al., 2008; Dirzo et al., 2022). As an indication of the scale of the crisis, it is estimated that at the end of the last ice age the total vertebrate mass was 300 million tonnes, with less than 1% accounted for by the human population (Smil, 2015). By 2015, this total had reached 1850 million tonnes, with domesticated animals accounting for 76% of the total, humans 23%, and wildlife just 1%, although some taxonomic groups (e.g. birds and amphibians) were not included in the study (Smil, 2015). There have been massive population depletions of both vertebrates and invertebrates (Dirzo et al., 2014), and the pervasive loss of biodiversity has serious negative consequences for ecosystem functioning (Pires et al., 2018). Finally, although just under 700 vertebrate extinctions have been documented to date (Ceballos et al., 2015a), many more have gone extinct without our knowledge (Tedesco et al., 2014).

1.2 Global wildlife trade

The global wildlife trade is a growing lucrative industry that provides livelihoods to many people but is also a leading driver of the global biodiversity crisis (Challender and MacMillan, 2014; Maxwell et al., 2016). Wildlife trade includes all sales or exchanges of wild animal and plant resources, involving live animals and plants, and a diverse array of derived products such as skins, medicinal ingredients, fuel, timber, food, and religious or tourism-related items (TRAFFIC, 2008). The demand for global wildlife and wildlife products is estimated to be worth billions of dollars per year (Wyler and Sheikh, 2008; Barber-Meyer, 2010), involving upwards of 100 million animals annually (Harfoot et al., 2018). The major sources for demand of wildlife and wildlife products are for use as pets (Bush et al., 2014), food (Veríssimo and Wan, 2019), medicine (Hughes, 2021), and ornamental purposes (Phelps and Webb, 2015). The regulation of international wildlife trade is orchestrated through the Convention in International Trade in Endangered Species of Wild Flora and Fauna (CITES), which aims to prevent extinctions of traded species (CITES, 2018) and has had varying levels of success in achieving this (Martin, 2000). In addition to the legal wildlife trade, there is a substantial illegal wildlife trade (Symes et al., 2018b), which is estimated to be worth US$5–20 billion per year, placing it among the world’s most lucrative illicit businesses (Wyler and Sheikh, 2008).

It is estimated that almost a fifth of vertebrates are traded (Scheffers et al., 2019) along with many plant species (Liu et al., 2019). Given that there is a significant research and reporting bias that has resulted in a major focus on vertebrates, many taxa, such as fungi and invertebrates, are ignored by the IUCN and CITES reporting mechanisms and are, therefore, not included in the global reporting of taxa affected by trade (Fukushima et al., 2020). While the sustainability of trade in many species is poorly understood and sometimes mischaracterised in the scientific literature (Challender et al., 2022), it is clearly one of the most pervasive threats to biodiversity (Cardoso et al., 2021). Birds are one of the most heavily traded taxonomic groups, with nearly a third of extant species identified as threatened by trade (Butchart, 2008), and parrots (Psittaciformes), songbirds (Passeriformes), and falcons (Falconiformes) are the most commonly traded orders (Bush et al., 2014).

1.3 Wildlife trade in Southeast Asia

Linked to its complex geological history, the exceptional species richness and levels of endemism mean that most of Southeast Asia is considered a biodiversity hotspot (Myers et al., 2000; Sodhi et al., 2004). However, the region also harbours the highest proportion of threatened species across all taxonomic groups considered (reptiles, mammals, birds, and vascular plants) except amphibians (Sodhi, Posa, et al., 2010). The principal threats to biodiversity are deforestation and the overexploitation of wildlife for food and trade (Nijman, 2010; Wilcove et al., 2013; Hughes, 2017; Symes et al., 2018a). The market for wildlife trade is both international and domestic, and in some countries of Southeast Asia, such as Indonesia, it is likely that the domestic trade dwarfs international trade (Sodhi et al., 2004; Nijman, 2010; Symes et al., 2018b). The drivers of Southeast Asia's wildlife trade are comparable to those reported at the global level, that is, for use as food (Hughes, 2017), pets (Harris et al., 2017), medicine (Harrison et al., 2016b), and ornamental purposes (Collar, 2015). In recent decades, some of the trade has moved online, making tracking and reporting the volume and effects of trade more complex (Harrison et al., 2016a; Rowley et al., 2016). Many taxa are threatened due to unsustainable levels of trade including pangolins (Shepherd, 2009), turtles (Chen et al., 2009), plants (Phelps and Webb, 2015), amphibians (Rowley et al., 2016), and birds (Nash, 1993; Sykes, 2017).

1.4 The Asian Songbird Crisis

The effect of wildlife trade on some taxa has been noted for several decades, and the effect on the bird community in general has been a conservation concern for at least three decades (Nash, 1993; Jepson, 2016). Throughout Southeast Asia, the trade in wild-caught songbirds—prized for their vocal ability, plumage, rarity and cultural significance—is having a massive effect on wild populations (Nijman, 2010; Lee et al., 2016; Symes et al., 2018a; Indraswari et al., 2020), and more than 1,000 bird species have been identified as traded (Nijman, 2010; Chng et al., 2015; Harris et al., 2017; Marshall et al., 2020b). The resulting 'Asian Songbird Crisis' has left many species facing extinction, while for many others the damage trade has wrought on their populations is still poorly understood due to insufficient monitoring (Eaton et al., 2015; Shepherd and Cassey, 2017; Sykes, 2017; Bergin et al., 2018; Marshall et al., 2020b). While negative impacts of trade have been reported from all parts of Southeast Asia (Yohe et al., 2014; Shepherd and Cassey, 2017), Indonesia is undoubtedly at the epicentre of the cage-bird trade owing to the deep-rooted songbird-keeping culture and the rising popularity of songbird competitions (Sodhi et al., 2004; Collar et al., 2012; Eaton et al., 2015; Collar and Wirth, 2022). Indeed, such are the levels of bird ownership across Java that, unbelievably, it has been estimated that there may actually be more birds in captivity than there are in the wild (Marshall et al., 2020b). There is a high risk of imminent trade-driven extinctions in Indonesia (Eaton et al., 2015), and half of Indonesia's 64 globally threatened songbirds (order Passeriformes), most of which occur on Java, are threatened primarily by trade (IUCN, 2022).

1.5 Approaches to tackling the songbird crisis

In recognition of the plight of songbirds across Southeast Asia, Wildlife Reserves Singapore and TRAFFIC convened the first Asian Songbird Trade Crisis Summit in 2015, which was followed by a second summit in 2017 (Sykes, 2017). The stated objective of the first meeting was to 'reduce the threat from trade to songbird taxa in the Greater Sunda region and work towards the survival of these species' (Sykes, 2017). The first task undertaken at these meetings was to identify the songbird taxa in Southeast Asia whose populations are seriously affected by trade, and this resulted in a two-tier priority list being compiled, with Tier 1 including the most threatened taxa requiring immediate conservation action, and Tier 2 comprising taxa of high conservation concern but requiring further research (Lee et al., 2016). During the 2017 summit, it was proposed to form an IUCN Species Survival Commission group to represent the group working on the Asian songbird trade, and the IUCN approved the formation of this 'Asian Songbird Trade Specialist Group' in May 2017 (Sykes, 2017).

There are five main themes to the work being carried out by members of the ASTSG in order to tackle the songbird crisis: field research, genetic research, conservation breeding and reintroduction, trade and legislation, and education and community engagement (Lee et al., 2016). In the following sections, I will briefly address some of the work that has been undertaken so far in each of these categories.

1.5.1 Field research

Field research is needed to identify and map the distributions of taxa, and to monitor population sizes and trends in order to improve our understanding of the status of taxa and guide conservation actions. This information is also used to inform national and international protection lists, including the IUCN Red List (IUCN Standards and Petitions Working Group, 2022). The general lack of baseline data for most species seriously hinders and, in many cases, precludes efforts to monitor birds across Southeast Asia (Lee et al., 2016; Collar and Wirth, 2022). Furthermore, reintroduction work also relies on ecological knowledge to identify suitable release sites for taxa and to enable conservation management following releases.

An example of the field research underway for birds affected by trade to date is a project led by Burung Indonesia that has been assessing the avifauna of West Java's mountains, which are home to some heavily traded songbirds that have reportedly suffered serious population declines such as the Javan Green Magpie (*Cissa thasalinna*) and Rufous-fronted Laughingthrush (*Garrulax rufifrons*) (Eaton et al., 2015). A review of the state of West Java's mountains was undertaken, including a forest cover loss analysis, which identified 20 key areas that needed urgent biological and socio-economic surveys (Higginbottom et al., 2019). Subsequent biological survey work has revealed the richness of some of these areas for upland birds (Devenish et al., 2022), and socio-economic research is ongoing to determine patterns of forest use and bird trapping around these areas. The results of this and other similar fieldwork can be used to inform future reintroduction work and could lead to changes in the protected area network to safeguard important sites.

Other field research has attempted to determine the direct impact of trapping on bird communities (Harris et al., 2017), worryingly finding on Sumatra that trappers travel at least 5 km into forests to obtain birds, and that most remaining forests are within 5 km of roads across the island, meaning remoteness is becoming less of a barrier to the pervasive effects of trapping. Another study on Sumatra found that species in mixed flocks affected by trade had significantly declined over a period of 20 years when compared to non-traded species, suggesting wider impacts on other non-traded species that rely on mixed-species flocking (Marthy and Farine, 2018).

1.5.2 Genetic research

There are three main purposes for genetic research under the auspices of the Asian Songbird Crisis: to reliably identify taxa deserving of protection as separate conservation units, guide breeding programmes to avoid inbreeding between closely related individuals and identify pure individuals for use in conservation breeding programmes. Research to date has helped resolve taxonomic confusion in the White-rumped Shama (*Copsychus malabaricus*) complex (Wu and Rheindt, 2022) and hill mynas (*Gracula* spp.) (Ng et al., 2021), and contributed to conservation breeding efforts (Baveja et al., 2021).

1.5.3 Conservation breeding and reintroductions

Ex situ interventions in the Asian Songbird Crisis are easier to implement than the challenging in situ interventions that will be needed to restore habitat and promote population growth. Conservation breeding programmes for individual taxa that are most threatened by trade have been established to buy time for the wild populations. There are several examples of projects being undertaken at various institutions, many of which are profiled by Collar and Wirth (2022). As an example, conservation breeding efforts for the Javan Pied Starling (*Gracupica jalla*)—a species probably now extinct in the wild but with a captive population of possibly one million birds on Java alone (Marshall et al., 2020b)—have been initiated at the Prigen Conservation Breeding Ark in spite of complex issues related to genetic mixing with the former conspecific Asian Pied Starling (*G. contra*) (Collar and Wirth, 2022).

1.5.4 Education and community engagement

To date, some resources for education and examples of community engagement have been made available on the ASTSG web page, which includes seminars on the songbird crisis (ASTSG, 2022a). There have also been art projects to promote awareness of the illegal trade in Asian songbirds (Kunzova et al., 2022), and conservation NGOs, such as Begawan Foundation, include the topic in their educational work, as well as housing a captive-breeding and release programme of Bali Mynas (Begawan, 2022). Community engagement programmes also play an important role in preparations for reintroduction work of species affected by trade. This work was instigated for a small pilot release of Black-winged Mynas in West Java (Owen et al., 2014) and is underway in the areas surrounding Bali Barat National Park in support of future reintroductions to expand the range of the Bali Myna (Squires et al., in review).

1.5.5 Trade and legislation

Research on the trade in songbirds tends to focus on either the supply or demand side. Most research to date on the supply side has involved market surveys to assess the volume and turnover of birds (Chng et al., 2015, 2018; Harris et al., 2015; Chng and Eaton, 2018), and to identify overexploited species (Harris et al., 2015). Species-specific supply-side research has also been carried out (Shepherd et al., 2016b; Nijman et al., 2018, 2021). Meanwhile, on the demand side there have been household surveys to measure bird ownership levels (Jepson, 2010; Jepson et al., 2011; Marshall et al., 2020b), efforts have been made to characterise different bird-keeping user groups to support demand reduction campaigns in the future (Marshall et al., 2020a), and the dynamics of songbird ownership have been assessed (Chiok et al., 2022).

Important issues related to the legal protection of certain taxa and better enforcement of the existing wildlife laws have been raised by members of the ASTSG. Species-specific studies related to illegal trade and species protection have been carried out for some of the most threatened birds including the Straw-headed Bulbul (Shepherd et al., 2013; Bergin et al., 2018), *Garrulax* laughingthrushes (Shepherd et al., 2016a), and Black-winged Myna (Shepherd et al., 2016b). Efforts to improve reporting of the status of species and the need to better protect them culminated in the Indonesian Government adding several taxa to the protected species list, although in a stark reminder of what conservationists are up against, just weeks later several of these were removed again in response to fierce lobbying by the country's bird traders (Leupen and Shepherd, 2018; Collar and Wirth, 2022).

Table 1.1 The Asian Songbird Trade Specialist Group's priority taxa list (ASTSG, 2022b). Only taxa whose range includes Java or Bali are included here.

Common name	**Scientific name**	**Red List**	**CITES**	**Population status**
TIER 1 - Urgent conservation priority				
Black-winged Myna	*Acridotheres m. melanopterus*	EN		possibly extinct in the wild[1]
Grey-backed Myna	*Acridotheres m. tricolor*	EN		c.180[2], few known populations (Chapter 4)
Grey-rumped Myna	*Acridotheres m. tertius*	EN		35–100[1] at possibly only two sites
Brown-cheeked Bulbul	*Alophoixus bres*	EN		size unknown; rapid decline[1]
Javan Leafbird	*Chloropsis cochinchinensis*	EN		size unknown; rapid decline[1]
Greater Green Leafbird	*Chloropsis sonnerati*	EN		size unknown; rapid decline[1]
Javan Green Magpie	*Cissa thalassina*	CR		50–249; rapid decline[1]
Oriental Magpie-robin (East Java/Bali)	*Copsychus saularis amoenus*	LC		subspecies in critical status[1]
Javan Jungle-flycatcher	*Cyornis banyumas*	LC		rare[3]; slow decline[1]
Rufous-fronted Laughingthrush (west Java)	*Garrulax r. rufifrons*	CR		50–249; very rapid decline[1]
Rufous-fronted Laughingthrush (Mt. Slamet)	*Garrulax rufifrons slametensis*	CR		unrecorded since 1925[1]
Orange-headed Thrush (Java/Bali)	*Geokichla citrina rubecula*	LC		size unknown; drastic decline of this subspecies[1]
Common Hill Myna	*Gracula religiosa*	LC	II	common; declining[1]
Javan Pied Starling	*Gracupica jalla*	CR		1–49; no known populations[1]
White-rumped Shama (Central Java)	*Copsychus malabaricus javanus*	LC		very rare[4] and declining[1]
White-rumped Shama (East Java)	*Copsychus m. omissus*	LC		very rare[4] and declining[1]
Bali Myna	*Leucopsar rothschildi*	CR	I	reintroduced population >100 individuals (Chapter 5)
Horsfield's Bushlark (Java)	*Mirafra j. javanica*	LC		unknown
Crested Jay (Java)	*Platylophus g. galericulatus*	NT		almost extinct on Java[4]; declining[1]
Javan Scimitar-babbler	*Pomatorhinus montanus*	LC		unknown
Straw-headed Bulbul	*Pycnonotus zeylanicus*	CR	II	probably extinct in Java; extreme decline[1]
Javan White-eye	*Zosterops flavus*	EN		size unknown; rapid decline[1]
Sangkar White-eye	*Zosterops melanurus*	VU		apparently common; rapid decline[1]

Common name	**Scientific name**	**Red List**	**CITES**	**Population status**
TIER 2 - 'watch list'; taxa present in trade but impact on wild populations unclear; more research needed.				
Chestnut-capped Thrush	*Geokichla interpres*	EN		size unknown; rapid decline[1]
Javan Heleia	*Heleia javanica*	LC		size unknown; suspected decline[1]
Bar-winged Prinia	*Prinia familiaris*	NT		size unknown; very rapid decline in Java[1]
Plain Prinia	*Prinia inornata*	LC		common; stable population[1]
Orange-spotted Bulbul	*Pycnonotus bimaculatus*	NT		locally common; slow to moderate decline[1]
Ruby-throated Bulbul	*Pycnonotus dispar*	VU		size unknown; suspected rapid decline[1]
White-bellied Fantail	*Rhipidura euryura*	LC		size unknown; possible decline[1]
White-bibbed Babbler	*Stachyris thoracica*	LC		size unknown; suspected decline[1]
Lemon-bellied White-eye	*Zosterops chloris*	LC		size unknown; suspected stable[1]

IUCN Red List status: CR critically endangered, EN endangered, VU vulnerable, NT near threatened, LC least concern

[1] (BirdLife International, 2022); [2] (Squires et al., 2022); [3] (Eaton et al., 2021); [4] (Billerman et al., 2022)

1.6 Study area: Java and Bali, Indonesia

Java is the fifth largest island of the Indonesian archipelago, covering an area of 127,000 km^2 (Figure 1.1), and the most densely populated island in the world, with more than 141 million people living at a density of over 1,000 people km^{-2}, accounting for 55% of Indonesia's 255 million population (Statistics Indonesia, 2018). Population growth in the last 300 years has transformed the landscape of Java, and by the middle of the 20th century almost all cultivable land had been cleared of its natural vegetation and replaced with agricultural crops, primarily rice (Padmanaba et al., 2017). The fertile volcanic soils across the islands support some of the most intensive agriculture in the world (Whitten et al., 1996). Forest loss has been so severe that Java's lowlands have less than 3% cover of highly fragmented primary forest, increasing to 17% and 45% in the less accessible upland and montane regions respectively (Margono et al., 2014). Indonesia's rapid economic growth since the 1970s was concentrated in Java and has led to an increasing proportion of the population moving from agricultural labour to employment in labour-intensive manufacturing industries and services, causing Java to become a more urbanised society (Verburg et al., 1999). The large human population has placed immense pressure on Java's natural resources, and most of the richest remaining areas for biodiversity are isolated and in areas difficult to cultivate, primarily the chain of volcanoes that runs west to east across the island, which coincide with many of Java's protected areas (Sodhi et al., 2005).

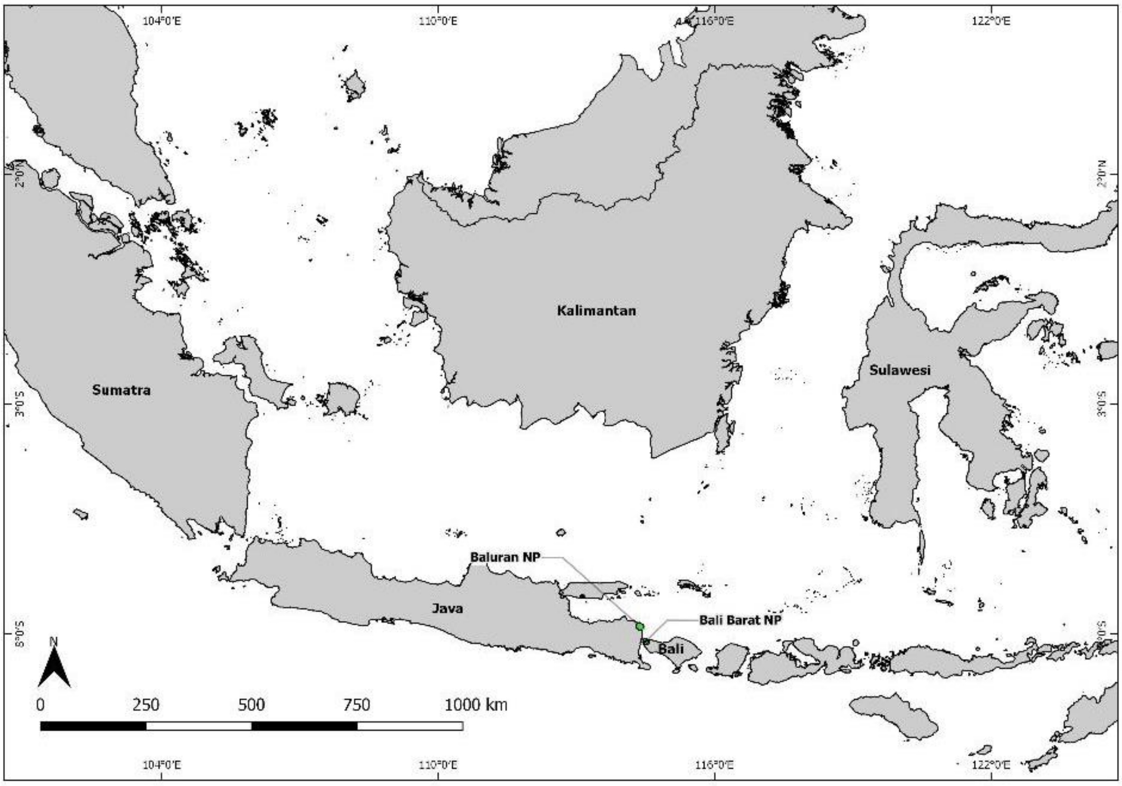

Figure 1.1 The location of Java and Bali in relation to the other major islands in the western part of Indonesia.

1.7 Focal study species

1.7.1 Black-winged Myna

The Black-winged Myna is endemic to Java and Bali and now survives exclusively in protected areas, but was once widespread in the lowlands, predominantly savannas and cultivated areas up to 1,200 m in West Java and reportedly 2,400 m in East Java (Feare & Craig 1998; Collar et al. 2001). Three subspecies are recognised: the nominate *melanopterus* was distributed across Java from the west up to a small hybridisation zone just east of Surabaya, where mixing with the East Javan *tricolor* subspecies may have occurred, and the *tertius* subspecies occurs on Bali (BirdLife International, 2022). The species has been present in both domestic and international trade for decades, despite its protection under Indonesian law since 1979 (Minister of Agriculture, Decree no. 757/Kpts/Um/12/1979). It is, however, domestic trade that is largely responsible for the precipitous decline in the wild population, which began in the 1960s but was most pronounced in the 1990s, and the sharp decline in numbers traded in the 2000s gave a clear indication that wild populations were vanishing because of trapping (Collar et al., 2001, 2012; Eaton et al., 2015; Shepherd et al., 2016b; Nijman et al., 2018). Genetic integrity in the subspecies has been lost due to mixing between individuals following escape from captivity, and there are concerns that birds are being purposefully interbred in captivity to achieve a whiter plumage, which can fetch a higher price in the trade (Collar et al., 2012). Due to its affinity with open agricultural areas, it is suspected that pesticide applications, which can reach high levels in Java (Yuda, 2008), are also affecting the species, although this requires clarification (BirdLife International, 2018).

Although small numbers may persist in recently unsurveyed areas including some nature reserves, the only known wild population of Black-winged Myna left on Java occurs in Baluran National Park, East Java (Winnasis et al., 2020; eBird, 2021), while a small number persist at two sites in Bali, with 35 birds at Bali Barat National Park (Brillianti et al., 2019) and 12 at another unspecified site (Eaton et al., 2015). The small number of birds known from two sites near Jakarta, Java (Eaton et al., 2015) are unlikely to persist (pers. obs.), and the latest evidence suggests subspecies *melanopterus* is extinct in the wild (Eaton et al., 2021). At Baluran, the population has been extremely low over the past decade: the largest flocks observed in 2009 and 2010 numbered 25 and 12 individuals, respectively (Winnasis et al., 2011; Eaton et al., 2015), although in 2016 a flock of 37 was recorded (BirdLife International, 2021). Accordingly, the global population size of wild Black-winged Mynas is considered to be below 100 individuals, probably around 85. Given the extremely small wild population and the current volume of trade in birds across Java (estimated to number 10,000–15,000 birds year^{-1}), the vast majority of birds sold must be captive bred (Nijman et al., 2018), although trapping of wild-caught birds does certainly still occur (Bruslund et al., 2021).

Conservation actions for the species to date have included the establishment of a conservation breeding programme for subspecies *melanopterus*, which has had considerable success despite a major robbery at the complex (Eaton et al., 2015; Collar and Wirth, 2022). This programme has been used to stage an unsuccessful experimental reintroduction in West Java, which might have succeeded had it not been for trapping, given that successful breeding was recorded (Owen et al., 2014). There has, however, now been a successful release of *melanopterus* at the large and well protected Taman Safari in Bogor, with around 60 individuals reported in 2020 (Collar and Wirth, 2022). A small population of *tricolor* has been established at Batu Secret Zoo in East Java, and there are plans to increase this programme. No conservation breeding programme exists for *tertius* so far.

1.7.2 Bali Myna

The Bali Myna is a Critically Endangered species confined to Bali Barat National Park (BBNP) in north-west Bali, Indonesia, where it became so rare through the conversion of its open monsoon woodland habitat and, especially in the past fifty years, intensive trapping for trade that it now survives in the wild only through the release of captive-bred birds (Collar et al., 2001; Sutedi, 2012; Jepson, 2016; Hernowo, 2017; Yuni et al., 2022). This programme of supplementation (and indeed likely complete replacement) of the wild population by birds of captive origin has not been documented in great detail; there are no data on the ages, sex or relatedness of releasees. Moreover, although birds have responded favourably to the provision of nestboxes and food in the vicinity of release sites, there has been no systematic post-release monitoring of birds in terms of survival, dispersal, habitat use or, apart from a single recent study (Yuni et al., 2022).

In addition to the BBNP population, which is the only one inside the recognised historic native range for the species, there have been a number of releases of Bali Mynas elsewhere in an attempt to establish free-flying populations. A benign introduction has been undertaken on Nusa Penida, a small island south-east of Bali, by the Friends of the National Parks Foundation (Dijkman, 2007), and while it seemed as though the population had all but disappeared after nestboxes fell into disrepair and monitoring became sporadic (Mattison, 2016), a reinvigorated community engagement and monitoring programme helped discover a few remaining individuals, which have since successfully bred in well-maintained nestboxes and now there are at least 25 occupied nestboxes and flocks of juvenile birds reported (D. Donato, pers. comm.). Begawan Foundation also continues to attempt to establish a population around its headquarters in southern Bali (Nijman et al., 2017).

1.8 Thesis aim and objectives

In this thesis, I aimed to understand the ecology and management needs of some of the passerines most threatened by the cage-bird trade across Java and Bali, Indonesia, to guide in situ conservation actions. To achieve this aim, the remaining chapters of this thesis have the following objectives:

Chapter 2:	Implement a citizen science event in order to 1) gather occurrence data for under-recorded birds across Java and Bali, and 2) engage with Indonesian society and examine levels of participation in citizen science and the barriers to increasing them further.
Chapter 3:	Assess the current distributions of 23 of Java's lowland bird species using appropriate environmental predictors and examine the role of habitat, trapping and protected areas in relation to these.
Chapter 4:	Estimate the population size and distribution of Java's last remaining population of the Endangered Black-winged Myna (Acridotheres melanopterus) at Baluran National Park, and identify potentially suitable but unoccupied areas in the park and the possible barriers preventing population expansion to these.
Chapter 5:	Assess the near-future population viability of Bali's largest remaining population of the Critically Endangered Bali Myna (Leucopsar rothschildii).
Chapter 6:	Discuss the findings and conservation implications of Chapters 2–5 and outline priorities for the conservation of birds threatened by the cage-bird trade across Indonesia.

1.9 Thesis structure

Chapter 2: Citizen science rapidly delivers extensive distribution data for birds in a key tropical biodiversity area

Citizen science is a practical way to obtain broad-scale ecological data to evaluate species distributions and their responses to environmental change that would be beyond the reach of professional researchers due to the limited resources available for data collection. Despite recent rapid growth, citizen science projects remain rare in biodiverse yet data-poor countries, contributing to a shortfall in generating data for biodiversity monitoring and promoting public stewardship of nature. In this chapter I document and analyse BigMonth2020, a month-long birdwatching event held in Java and Bali, during which a huge number of bird records were gathered, resulting in a massive increase in spatial coverage of data. I analyse the participant demographics and show that the event had higher female participation than other bird-related activities in Indonesia, and that the vast majority of participants were relatively young and members of bird clubs or studying at university. My findings highlight the potential for relatively low cost citizen science projects to help close extensive remaining gaps in biological data across Indonesia to improve the monitoring of bird species, and to deliver positive social outcomes. BigMonth2020 serves as a model for rapidly establishing a distributional baseline for monitoring biodiversity trajectories.

Chapter 3: Modelling the current distributions of Java's lowland birds: the role of trade, habitat and protected areas

Scientific knowledge of the forms and functions of Earth's biodiversity remains extremely limited, at a time when biodiversity faces unprecedented threats from human activities. Species conservation measures will need to be prioritised as long as resources for conservation are limited, a process typically achieved using species-specific range- and population-based metrics which require distribution and abundance data to be calculated. Across Java, Indonesia, the principal threats to wild bird populations are trapping to supply the cage-bird trade and habitat loss. In this chapter I use citizen science datasets to generate the first species distribution models for a suite of Java's lowland birds that face varying levels of these threats. My findings show that the current distribution for most species are relatively patchy and often significantly smaller than the extent of occurrence. Among the environmental variables used in modelling, land-cover-based predictors are the most important in the models for most species. The lack of convergence between the current distribution of the modelled species with Java's formally protected areas suggests that future conservation for these and other lowland birds, which are likely to come under increasing anthropogenic pressure, will need to involve other effective community-based conservation measures. My findings highlight the urgent need to promote continued citizen science efforts across Java, and indeed elsewhere in data-poor yet biodiverse regions, and to commission further survey work to discover areas of high biodiversity value.

Chapter 4: Controlling trapping, overgrazing and invasive vegetation is key to saving Java's last population of the Black-winged Myna

The Black-winged Myna has been trapped to near extinction across its native range of Java and Bali and the population is estimated to number fewer than 100 individuals. In this chapter I estimate the current range and population size of the species at Baluran National Park, the location of its last known population, and use species distribution modelling to evaluate the potential suitability of unoccupied areas across the park. I estimate that the Black-winged Myna population numbers 179 individuals and that its current range is restricted to a very small area of savanna and dry deciduous woodland, while my model shows there is a considerable extra portion of potentially suitable habitat. I infer from the findings that the principal cause for the disparity between its current and potential range is trapping, compounded by savanna loss and degradation due to illegal domestic cattle grazing and the spread of invasive vegetation. The further population growth and range expansion of Black-winged Mynas in Baluran depends on effective management of illegal poaching, further clearance of acacia, and easing domestic cattle-grazing pressure on areas of savanna, particularly through engagement with human communities living inside the park.

Chapter 5: Assessing the near-future population viability of the Critically Endangered Bali Myna Leucopsar rothschildi

The iconic Bali Myna (*Leucopsar rothschildi*) is Critically Endangered after suffering decades of overexploitation for the cage-bird trade. Despite decades of conservation efforts, in the 2000s it is likely there were no birds left in the wild. Since then, however, various management interventions at Bali Barat National Park (BBNP), including consistent annual releases of a greater number of captive-bred birds and a revamped artificial nestbox scheme, have led to population growth. To plan for the next decade of conservation management, in this chapter I model the free-living Bali Myna population at BBNP to explore the effects of (1) changes to population supplementation and (2) an increase in the level of illegal trapping (still the main threat). I validate a baseline matrix population model using data covering the last ten-year period (2012–21) and project the model ten years into the future (to 2032). The findings show that the population is resilient to low levels of trapping irrespective of the method used to trap birds, but at high levels of trapping the population is projected to steeply decline under both trapping scenarios. On current trajectory I estimate that the population will reach self-sustainability, and at this time the supplementation programme could either be scaled back or repurposed as a translocation project to repopulate vacant parts of the myna's range, and nest-boxes could be used as a potential tool to support population growth. I discuss the limitations associated with the population model and the need to enlist the support of local communities to generate new data to improve the model.

Chapter 6: General conclusions and recommendations for future work

In this chapter I discuss the key findings from Chapters 2–5 and use them to produce some general recommendations for future conservation work related to the topics broadly covered by this thesis. I begin by discussing the plight of Indonesia's starlings and mynas (family Sturnidae) and make some specific recommendations for fieldwork to address some important knowledge gaps for species that are being increasingly targeted by trappers for the cage-bird trade. I go on to briefly discuss issues and recommendations related to biodiversity monitoring across Indonesia using citizen science, ecological fieldwork for rare and threatened species, the development of community-based conservation projects to initiate conservation work outside protected areas, and the improvement of reintroduction methods across the region.

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Chapter 2

2 Citizen science rapidly delivers extensive distribution data for birds in a key tropical biodiversity area

Abstract

Citizen science projects remain rare in biodiverse yet data-poor countries, contributing to a shortfall in generating data for biodiversity monitoring and promoting public stewardship of nature. We document and analyse BigMonth2020, a month-long birdwatching event across Java and Bali, publicised through social media and incentivised with grants and competitions. Over 20,000 lists containing 100,000 bird records were submitted to the 'Burungnesia' phone app. Spatial coverage extended to 71% of the islands' 3,408 atlas grid squares (6.9 × 6.9 km), including 1,613 previously undocumented squares, with 353 bird species recorded, representing 74% of Java and Bali's avifauna excluding vagrants; 27 threatened species were recorded, with new records for 204 grid squares. Almost 25% of contributors were female, 72% were under 30 years old, and most were graduates and members of birdwatching clubs. The project cost less than US$10,000 to run and serves as a model for rapidly establishing a distributional baseline for monitoring biodiversity trajectories.

2.1 Introduction

Obtaining broad-scale ecological data to evaluate species distributions and their responses to environmental change requires resources unavailable to most researchers (Dickinson et al., 2010). Citizen science is a practical way to bridge the resource gap, with projects typically mobilising volunteers to gather and/or classify data following a protocol developed by experts (Dickinson et al., 2012). Ecology-related citizen science projects vary widely, ranging from online exercises (Shamir et al., 2014; Swanson et al., 2015; Rosenthal et al., 2018) to field surveys (Preston, 2013; Gillings et al., 2019). Scientists benefit from citizen science by obtaining large datasets with higher coverage, the volunteers experience direct involvement in science and enhance their skills (Dickinson et al., 2010), and wider society benefits, as volunteers often share their knowledge, increasing levels of scientific literacy and environmental advocacy among peers (Johnson et al., 2014).

Citizen science has recently proliferated in developed countries but remains rare in developing countries (Chandler et al., 2017). This is problematic for conservation, since biodiversity hotspots predominantly coincide with these data-poor, highly threatened areas (Brooks et al., 2006; Fisher and Christopher, 2007). Barriers to citizen science in developing countries include low awareness of opportunities (for both participants and institutions) (Pocock et al., 2019), low appreciation of its environmental and societal value (Chandler et al., 2017), and low levels of expertise, time, money, and perceived personal benefits (Requier et al., 2020). By way of counterbalance, the global rise in smartphone ownership and internet coverage in many developing countries gives citizen science both practicality and appeal (August et al., 2015; Taylor and Silver, 2019).

Indonesia is one of the most biodiverse nations on earth, but habitat loss through land-use change is a major threat to wildlife and habitats, while illegal trapping of wild birds has triggered an ‘Asian Songbird Crisis’ (Margono et al., 2014; Lee et al., 2016; Hughes, 2017). This trade affects at least 32 threatened species in Indonesia and many common species (Eaton et al., 2015; BirdLife International, 2021), with households in Java, Indonesia’s most populous island, keeping some 74 million cage-birds (Marshall et al., 2020). Baseline distribution data for widespread Javan species are now urgently required to identify future changes. To date, such data have been gathered by Indonesian birdwatchers for the first Indonesian Bird Atlas (‘Atlas Burung Indonesia’; Taufiqurrahman et al., 2016), and through eBird (Sullivan et al., 2014). However, these data are predominantly gathered in urban centres, ecotourism hotspots and protected areas, leaving large intervening spaces. To develop baseline distribution models for common birds, data need to cover the range of habitats and land-use types within the study area (Phillips et al., 2006).

To this end, we developed ‘BigMonth2020’, a citizen science project held in Java and Bali during January 2020 which aimed to engage Indonesian society, expand the coverage of bird distribution data, and incidentally contribute to the Indonesian Bird Atlas. Here we outline the scope and design of BigMonth2020, the data collection and validation protocols followed, and the promotional campaign and incentive scheme intended to attract participation. We then assess the bird data collected for their novelty, composition, and quality, and examine the demographic of those who contributed to BigMonth2020. Finally, we review the project’s outcomes and the benefits and pitfalls of a citizen science event, providing lessons learnt for similar initiatives and the continuation of the work on Java and Bali.

2.2 Methods

2.2.1 Inception

We developed an outline plan for an incentive-driven inclusive birdwatching event, to be promoted primarily through social media, and enlisted the involvement of two Indonesian partner organisations: the Indonesian Ornithologists' Union (IdOU), whose members are predominantly academics or work for conservation NGOs, and Birdpacker, a grassroots birdwatching community whose citizen science phone app 'Burungnesia' (*burung*, Indonesian 'bird'; *-nesia* from Indonesia) was released in 2016. This app enables birdwatchers to submit georeferenced bird lists in support of efforts to produce *Atlas Burung Indonesia*, the country's first national bird atlas.

2.2.2 BigMonth2020

The event's scope was limited to Java and Bali to ensure its logistics were manageable and it lasted a full month to maximise data accumulation within the constraint of limited administrative resources. This gave contributors ample opportunity to log data yet was short enough to maintain social media interest. We timed the event to coincide with university and national holidays, when participants had more free time and were dispersed from large urban centres.

A competition was promoted via social media. We purchased ornithological equipment (binoculars, telescope, field guides, etc.) as prizes for various categories, including the overall top-ten contributors of bird lists, the best social media influencer, and the best photograph. We also established a small grant scheme, administered online with simple bank transfers, to cover transport and subsistence for trips to under-recorded areas. After the third week, we identified the five largest remaining unrecorded areas and offered grants to people to visit them. A total of IDR 27M (US$1,850) was divided among 51 applicants, in grants ranging from US$6.80 for one person on a day trip to US$200 for a seven-day trip by eleven students. Overall operational costs, including a small team assembled by Birdpacker to administer the various aspects of the event (i.e., social media, data handling and expert validation), were covered by US$7,000 from the Oriental Bird Club (OBC) and US$400 by Idea Wild.

Other indirect project-related costs included the incidental funding of TMS and SJM, as well as the in-kind cost of developing and running the data-logging app.

2.2.3 Promotion

In November 2019 partners from Manchester Metropolitan University (MMU) and Birdpacker presented BigMonth2020 at the annual Indonesian birdwatchers' conference. Thereafter, promotion was carried out on social media. We posted Indonesian-language promotions via Facebook and Instagram, and English-language promotions via Twitter. We directly contacted 33 naturalist clubs (22 of them university societies), eleven Indonesian NGOs, two Indonesian zoos, and the European (EAZA) and North American (AZA) zoo associations via email and social media. Thirty-four organisations became official supporters and their logos featured on promotional material.

Social media promotions began with a digital project poster (Figure 2.1) on Facebook, Instagram, and Twitter two weeks before the event, followed by information about its aims and objectives, the data collection protocol, and the competition rules. A BigMonth2020 webpage provided tutorials for data collection and input, and an identification guide for lookalike species. Once BigMonth2020 commenced, social media posts were made almost daily on Instagram and Facebook, providing updates on the prize competition, data accumulation, unusual findings, and priority grid squares. Many participants shared our promotions or created their own content, increasing the project's reach (Appendix 2.1). The MMU partners visited the Birdpacker team in Malang, East Java, in mid-January and collected data alongside students, NGO staff and government officials, while the OBC chairperson recorded data with members of the Birdpacker team in East Java.

birdpacker Bagi yang nonton live dadakan kita pagi ini pasti paham kalau kita baru saja kedatangan paket khusus! Ini dia penampakan... more

Figure 2.1 From top left clockwise: BigMonth2020 promotional poster; promotion of competition prizes on Instagram; prizes being awarded at the end of BigMonth2020; young participants wearing their BigMonth2020 T-shirts and holding the field guides they won as prizes for their contribution.

2.2.4 Data collection

Participants were asked to focus their efforts on low-elevation land (<800 m altitude) outside protected areas, because the largest gaps in data occur in these relatively accessible areas, and to use the Burungnesia phone app to submit their data; we did promote the use of eBird but no participants chose to use this during BigMonth2020. Participants compiled lists of bird species recorded at a unique location as either presence-only or count data (sample data and full protocol in Appendix 2.2). Participants were encouraged to search for birds around the start location for at least one hour and to begin a new list if they travelled 3 km away from the start point. As Burungnesia currently lacks the functionality to record extensive list metadata, we could not obtain data on survey effort (distance travelled, survey duration) or list completeness (i.e., whether all species encountered were recorded).

We increased the resolution of the Indonesian Bird Atlas grid system (0.25° × 0.25°, WGS 84) by dividing each square into 16 cells (0.0625° × 0.0625°; 6.9 × 6.9 km), resulting in 3,408 grid squares. Using existing data from eBird and the Indonesian Bird Atlas we categorised squares as unvisited (no bird lists) or visited at two levels (1–5 bird lists, >5 bird lists). Data for the Indonesian Bird Atlas were collected manually until the Burungnesia app was released in 2016; here, both datasets are combined and referred to as 'Indonesian Bird Atlas data'. BigMonth2020 participants could download the map as a *.kmz* file. Trips to unvisited squares were incentivised using a weighted point-scoring system for the competition, with extra points awarded if five bird lists were submitted from an unvisited grid square. The map and grid square status were updated every three days and a new download made available.

2.2.5 Data validation

Six experts manually validated all submitted data throughout and after the event. A bird list was flagged for further review if (1) a location description did not match the GPS coordinates; (2) the habitat description did not match the habitat depicted on Google Earth; or (3) a species record was deemed unusual in terms of location, time of year or habitat. For flagged records, the observer was asked for supporting evidence, and depending on the response the record was either retained in or removed from the database. Records without coordinates were omitted. All records were adjusted so that taxonomy followed HBW & BirdLife International (2022).

2.2.6 Participant questionnaire

An Indonesian-language questionnaire (Appendix 2.3) was posted online to learn more about the event's participants, with a free BigMonth2020 T-shirt (Figure 2.1) offered to all respondents. Participants provided demographic data (age, education level, employment status) and information on their birdwatching expertise, motivations and perceptions of conservation issues.

2.2.7 Ethical statement

The questionnaire was administered by Universitas Atma Jaya Yogyakarta, followed their research guidelines, and conformed to standards in BSA (2017). It explained its objectives at the start and participants provided informed consent by answering the questions. Questionnaire data were accepted from adults only (>18 years) and anonymised before analysis.

2.2.8 Data analysis

The species recorded were classified into six functional groups to examine differences in data recording. Birds were categorised as either raptors, aerial feeders or waterbirds based on taxonomy and feeding strategy, while all species outside those categories were grouped by preferred habitat (woodland, open/agriculture, or scrub/savanna; see Appendix 2.4) using BirdLife Data Zone information (BirdLife International, 2020). We calculated Shannon's evenness (E_H) (Peet, 1974) for the six classes to measure the within-group relative abundance of records for each species, with values ranging from 0 (group dominated by few taxa) to 1 (records evenly distributed among taxa).

To identify participant attributes associated with high survey effort and data composition, we fitted two generalised linear models (GLMs) using the dataset of 134 participants' questionnaire responses combined with the bird data they submitted to Burungnesia. The survey effort model used the number of grid squares visited as the dependent variable, while the data composition model used a 'rarity recording' metric, calculated following August et al. (2020): every species was ranked according to the number of times it was recorded and assigned a rarity value from 1 = most common to 100 = most rare; we then subtracted the median rarity value across all observations in the dataset from the median rarity value across all records for the participant, so that negative values of the metric indicate that the participant recorded common species more frequently than expected and positive values show that the participant recorded rare species more frequently than expected. The predictors used in both models were age in years, gender, occupation (formal employment, freelance-type work, student), birdwatching experience in years, and bird club membership; the number of grid squares visited was included as a predictor in the data composition model. All statistical analyses were conducted in R (4.0.2, R Core Team 2020) using package 'MASS' (Venables and Ripley, 2002).

2.3 Results

2.3.1 Data accumulation

During BigMonth2020 a total of 22,055 bird lists were submitted across Java and Bali comprising 102,887 bird records (Table 2.1). The daily number of bird lists submitted grew throughout the event, punctuated by peaks in submissions at weekends (average 55% increase vs. preceding weekdays) (Figure 2.2). The difference in data accumulation between the first (2,564 lists, 11.6%) and last (8,470 lists, 38.4%) seven-day period was particularly sharp.

Table 2.1 Summary statistics for two existing citizen science bird distribution datasets over Java and Bali and the BigMonth2020 dataset.

Characteristic	**eBird**	**Indonesian Bird Atlas**	**BigMonth2020**
Years covered	1970–2020	2003–2020	2020
Number of records	180,975	39,011	102,887
Number of bird lists	11,666	4,130	22,055
Species recorded (threatened)	517 (39)	469 (38)	353 (27)
Median species recorded per bird list (IQR)	9 (3–19)	6 (2–13)	4 (3–6)
Grid squares (exclusive to dataset)	594 (67)	827 (135)	2,417 (1,613)
Contributors	1,241	483	218*
Median number of lists per contributor	4 (1–11)	2 (1–6)	8.5 (3–27)

* unique Burungnesia users who submitted data. Some people recorded in groups and the total number of participants is estimated at 373. IQR: interquartile range.

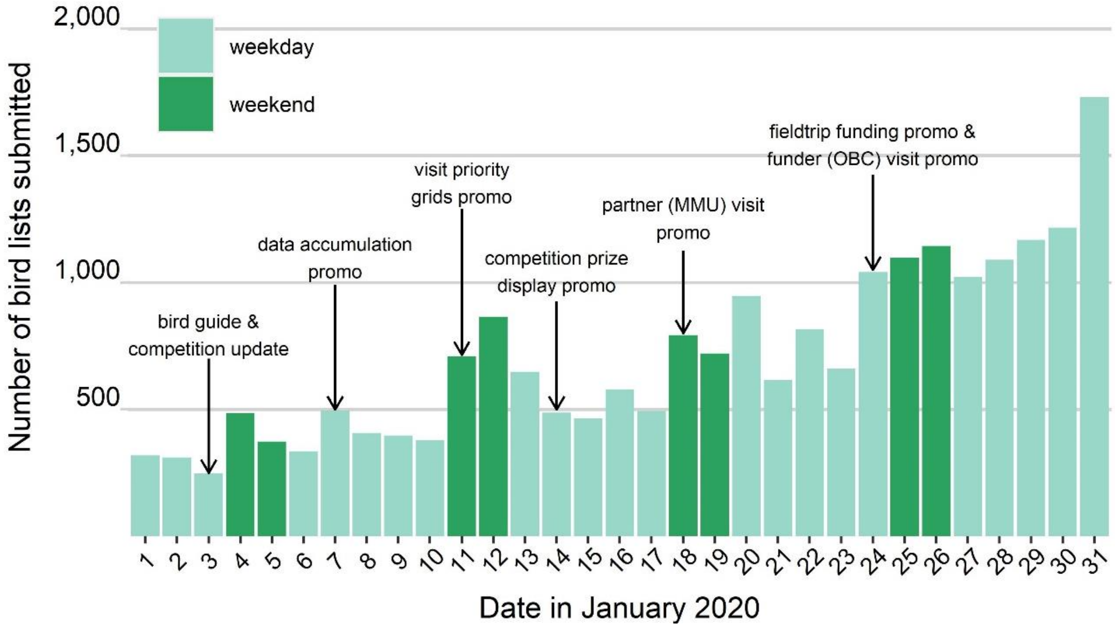

Figure 2.2 Daily number of bird lists uploaded to the Burungnesia phone app during BigMonth2020. Annotations show the timing and content of popular social media posts by Birdpacker. Submission peaks on 13 and 20 January are probably data-reporting lags from weekends, and the peak on the final day is probably contributors entering data before the competition cut-off time.

Over the course of the event 218 unique users submitted data to Burungnesia, although the total number of participants was an estimated 373 because some participants worked in groups with only one member submitting data. Ten contributors collected 72% of all bird lists (16,090), 25 contributors submitted over 100 bird lists, and 99 submitted at least ten. A median of four species (IQR 3–6 species) per bird list was slightly lower than the eBird and Indonesian Bird Atlas datasets (Table 2.1).

2.3.2 Data coverage

At least one bird list was recorded in 2,417 (70.9%) of the 3,408 grid squares across Java and Bali (Figure 2.3). Data were initially concentrated around major cities, but coverage steadily expanded to remoter areas (Appendix 2.5). Many low-elevation agricultural areas were surveyed for the first time. Coverage was greatest in Central Java and least in remote parts of West and East Java, with limited road access and a higher proportion of forested uplands.

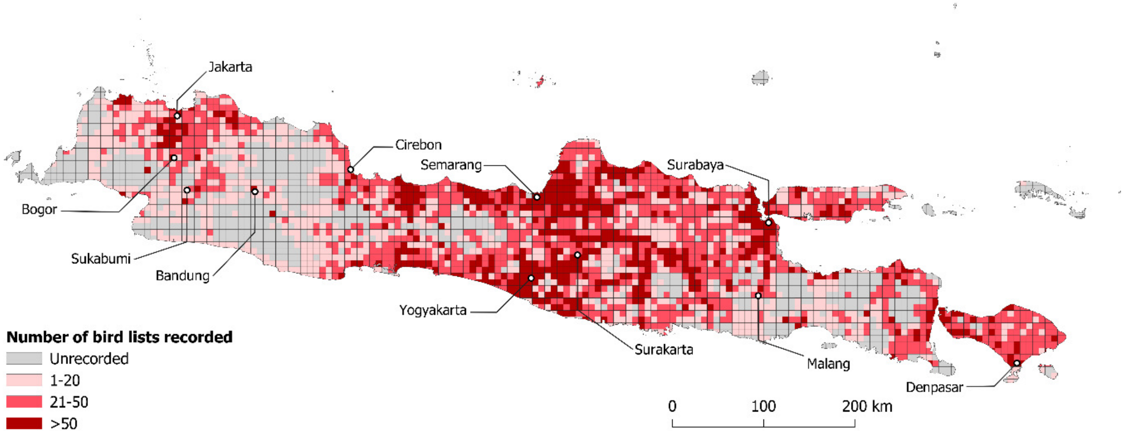

Figure 2.3 Data coverage for BigMonth2020. Grid squares (6.9 × 6.9 km) are coloured according to the number of bird lists recorded within them. Major cities in Java and Bali are shown.

Prior to BigMonth2020, bird occurrence data were available for 1,092 (32% overall; eBird 17.4%; Indonesian Bird Atlas 24.2%) of the grid squares across Java and Bali (Table 2.1). BigMonth2020 extended bird distribution data to a further 1,613 (47.2%) atlas grid squares, representing a 147% increase in coverage. Combined with eBird and the Indonesian Bird Atlas data, total coverage is now 79.3% of grid squares (Figure 2.4). Coverage increased by over 50% for 72 species and over 100% for 37 (Appendix 2.4).

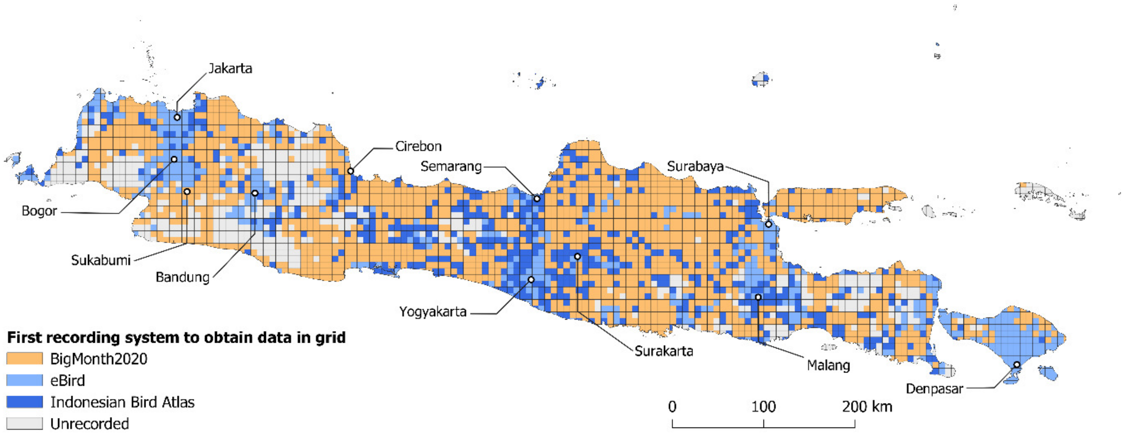

Figure 2.4 Bird distribution data coverage for Java and Bali. Grid squares (6.9 × 6.9 km) are coloured according to which recording system was first to obtain data there: BigMonth2020 (n = 1,613; 47.3%); Indonesian Bird Atlas (n = 575; 16.9%); eBird (n = 514; 15.1%); unrecorded (n = 706; 20.7%).

2.3.3 Data composition

There were 353 bird species recorded during BigMonth2020, representing 74% of those known from Java and Bali excluding vagrants (Lepage, 2020). Cave Swiftlet (*Collocalia linchi*), Eurasian Tree Sparrow (*Passer montanus*) and Sooty-headed Bulbul (*Pycnonotus* *aurigaster*) made up 16.2%, 10.2% and 6.0% of the 102,887 records, respectively. Twenty-seven species on the IUCN Red List (14 Vulnerable, 9 Endangered, 4 Critically Endangered) were recorded, ten of which are significantly affected by the cage-bird trade (BirdLife International 2020b). Six threatened species were recorded on >20 lists: Javan Myna (*Acridotheres javanicus*) (142 lists), Javan Coucal (*Centropus nigrorufus*) (101), Sangkar White-eye (*Zosterops melanurus*) (66), Ruby-throated Bulbul (*Rubigula dispar*) (62), Milky Stork (*Mycteria cinerea*) (33) and Java Sparrow (*Lonchura oryzivora*) (23). Threatened species were recorded for the first time in 204 grid squares, and seven species were recorded for the first time in at least ten squares, with grid square coverage for these increasing by 15.5–69.8% (Appendix 2.4).

Species of open country and farmland were most frequently observed (Table 2.2), with Eurasian Tree Sparrow, Scaly-breasted Munia (*Lonchura punctulata*) and Javan Munia (*L. leucogastroides*) comprising 70.8% of these records. Nearly half the species inhabit woodland but accounted for only a quarter of observations. Aerial feeders were over-represented in the dataset (4.3% of all species recorded accounting for 24.1% of all observations), as were scrub and savanna birds; waterbirds and raptors were under-represented.

Table 2.2 Summary of bird data recorded during BigMonth2020, with species grouped based on taxonomy and feeding strategy (raptors, waterbirds and aerial feeders) or preferred habitat (woodland birds, birds of open country/agricultural areas, and scrub/savanna birds).

Group	Species	% of all species	% of records	Threatened species	Evenness (EH)
Open country/agriculture	31	8.8	31.4	3	0.57
Woodland	172	49.0	26.7	15	0.50
Aerial feeders	15	4.3	24.1	0	0.42
Scrub/savanna	21	6.0	9.3	2	0.56
Waterbirds	91	25.9	7.9	6	0.66
Raptors	21	6.0	0.6	1	0.70

2.3.4 Data quality

During data validation, 845 bird lists (3.8%) were flagged for review. Data from 494 (58.5%) lists were retained in the database following verification, 253 (29.9%) were retained with updated location or species data, and 98 (11.6%) were removed for lack of supporting evidence. Some easily misidentified species commonly required review, notably tailorbirds (*Orthotomus* spp.): 19 of 60 records of Ashy Tailorbird (*O. ruficeps*) were reviewed, of which 13 were accepted with evidence, five re-identified as Olive-backed Tailorbird (*O. sepium*) and one as Common Tailorbird (*O. sutorius*).

2.3.5 Participant characteristics

Of the estimated 373 participants, 188 (50.4%) answered the questionnaire, all of whom were Indonesian. Of these, 23.4% were female and 71.8% were under 30 years old. Most respondents lived in East Java (28.2%), Yogyakarta (21.8%) and Central Java (14.9%), with fewer in West Java (12.8%), Jakarta (5.3%), Banten (2.1%) and Bali (2.1%), and the remainder (12.8%) lived elsewhere in Indonesia. Most were members of a bird club (67%) and discovered BigMonth2020 through their club (39%) or social media (20% Instagram; 9% Facebook); 36.2% owned a camera but not binoculars, 30.3% had both a camera and binoculars, 16.5% used binoculars alone, and 17% had no equipment. Top-ranking motives for their participation in BigMonth2020 were ‘contributing to conservation’ (74% of respondents) followed by ‘seeing new bird species’ (64%) (Appendix 2.6). The cage-bird trade and habitat loss were considered equally important threats to birds in Java, followed by climate change (Appendix 2.6). The number of grid squares visited by participants (sampling effort) was significantly higher for participants with more birdwatching experience ($z = 2.79 \pm 0.03$, $p < 0.01$) and who were male ($z = 2.32 \pm 0.28$, $p = 0.02$). In terms of rarity recording, participants who visited more grids during BigMonth2020 tended to record common birds more frequently than expected ($t = -2.44 \pm 0.006$, $p < 0.01$). Full GLM parameters are provided in Appendix 2.7.

2.4 Discussion

BigMonth2020 demonstrates the viability of citizen science in Indonesia and could be replicated in other countries where citizen science projects are scarce and biodiversity seriously under-recorded (Meyer et al., 2015). Over 300 Indonesians (Appendix 2.8) generated a dataset comprising over 100,000 bird records, half of which were collected in previously unsurveyed areas (see https://bigmonth2020.shinyapps.io/shiny_app/).

2.4.1 Data coverage and composition

BigMonth2020 has more than doubled bird distribution data coverage on Java and Bali, extending to almost 80% of grid squares. Sampling biases related to contributor distribution are a common and expected feature of citizen science data (Dennis and Thomas, 2000; Romo et al., 2006), and the spatial distribution of data here broadly reflects the accessibility of squares and the distribution of contributors, the most prolific of whom mainly lived in Central Java, Yogyakarta, and East Java. Consequently, only one in ten bird lists were submitted in western Java (Banten, Jakarta and West Java provinces) despite half Java's population residing there (Badan Pusat Statistik, 2016). While inaccessible upland areas in western Java account for the largest remaining gaps in data coverage, some accessible areas close to urban centres were unvisited. In part this is because Central Java and Yogyakarta possess more bird clubs, which are associated with the region's cluster of biology-focused universities. It may also reflect cultural differences in interest in birds between the Sundanese in western Java and the Javanese in central and eastern Java (Jepson and Ladle, 2005). Moreover, it could be linked to the rapid urbanisation of western Java (Firman, 2017), producing a human–nature disconnect and reduction in pro-environmental feeling (Cleary et al., 2020).

The considerable increase in data coverage for many commoner species will enable us to develop robust distribution models to establish a distributional baseline against which to monitor the stability of the environment, as changes in the distribution of common species representative of major habitat types can reveal patterns of wider ecosystem health (Caro and O'Doherty, 1999). Distribution models for common species, which contribute most to patterns of overall species richness (Vázquez and Gaston, 2004), could be used to identify areas of relatively high biodiversity value in under-recorded regions of Java. Estimating the distribution of rare and threatened species is another important aspect of biodiversity monitoring (BirdLife International, 2021), and BigMonth2020 delivered valuable data for 27 threatened species, for nine of which we obtained at least the minimum number of records needed to build accurate distribution models (Proosdij et al., 2016). Some of Java's Critically Endangered species, such as Black-winged Myna (*A. melanopterus*) and Javan Pied Starling (*Gracupica jalla*), were conspicuous absentees from the dataset, highlighting the disastrous declines of some species due to bird trapping in the region.

2.4.2 Participation and demographic

BigMonth2020 engaged with over 300 Indonesian citizens (Appendix 2.8), a level of participation comparable to similar schemes in Africa and Taiwan (Ko et al., 2014; APLORI, 2020). BigMonth2020 had more participants under 30 years old than projects in countries where birdwatching has a longer tradition with a wider spectrum of cohorts

(Wright et al., 2015; MacPhail and Colla, 2020). It also attracted people who were not already birdwatchers, suggesting that such events can promote engagement with nature and conservation issues. Although this demographic may present challenges relating to capacity to participate (e.g., less disposable income, limited transport) and data quality (i.e., less birdwatching experience, limited access to equipment), it indicates a growing community of nature enthusiasts who could rapidly become a significant body of conservation advocates. Retaining participants is, however, critical if BigMonth2020's baseline is to serve its purpose, because participant expertise can be expected to increase over time, especially if project goals and data use are effectively communicated (Forrester et al., 2017). This is best achieved by continuing to appeal to peoples' varied initial motivations for participating (Clary and Snyder, 1999; Bruyere and Rappe, 2007).

Birds have a deep cultural significance in Indonesia (Jepson and Ladle, 2005), but bird-keeping and songbird competitions are almost exclusively male-dominated activities (Marshall et al. 2020); encouragingly, however, a quarter of questionnaire respondents for BigMonth2020 were female. Nevertheless, female participants visited fewer grid squares than average, suggesting that gender-specific barriers to participation still exist and initiatives to encourage female participation are warranted. Even so, we speculate that birdwatching could develop as an inclusive pursuit in Indonesia, irrespective of sex, age or social class, and events like BigMonth2020 are ideally placed to promote this. The distribution of contributors to BigMonth2020 mirrors the prevalence of bird-keeping across Java (Jepson and Ladle, 2009; Marshall et al., 2020), suggesting that people from bird-keeping households could be attracted to birdwatching and conservation as an alternative means to enjoy birds, thereby helping to reduce the threat from the cage-bird trade.

2.4.3 Project design and data collection

Some adjustments to the sampling strategy we used for BigMonth2020 could help address the spatial bias and remaining gaps in data coverage. Besides the bias we introduced by asking volunteers to visit low-elevation unprotected areas, survey bias was linked to human population density and accessibility, a common problem when ad hoc sampling is used, reflecting the trade-off between protocol complexity (data quality) and ease of participation (Bird et al., 2014; Geldmann et al., 2016). While it should be minimised, spatial bias does not preclude the accurate estimation of species distributions (Johnston et al., 2020). Moreover, the uptake of our small incentives to explore under-recorded areas suggests that further such incentives to visit grid squares remote from major roads would help reduce the current spatial bias.

Some issues related to the data collection protocol can be addressed by modifications to the data-logging app. First, the number of taxa recorded per bird list for BigMonth2020 was low relative to other reporting systems for the same area, suggesting that either sampling effort per list (not recorded) or bird detection frequency was lower. The design of the competition, kept simple to promote engagement, probably contributed to this by awarding points for every bird list submitted, thereby encouraging low sampling effort; this is corroborated by our finding that participants who submitted most data tended to record commoner birds more frequently than expected. Requiring a minimum sampling effort for every bird list could resolve this issue, and highlights the need to design incentives carefully. Second, contributors may not have reported all species they encountered—possibly ignoring common species or those posing identification challenges (Snäll et al., 2011; Tulloch et al., 2013)—so inferring species absence was not possible. If absences and sampling effort are known, biases can be accounted for statistically, so these metadata should be required by future versions of the app (Fink et al., 2020). Finally, we manually validated photographic evidence requested from users after bird records were flagged. To expedite this process in future, users should be able to attach photographic evidence to their records during data submission, and the proportion of correctly identified photographs could also be tracked as a metric of data quality (Vantieghem et al., 2017).

2.4.4 Biodiversity monitoring in Java

BigMonth2020 delivered high geographic coverage of the study region and valuable distribution data for most of Java's bird species. The immediate aim following the event is to widen the network of citizen science birdwatchers and improve the utility of the data collected, in order to establish distributional baselines for birds across Indonesia. Extending survey effort beyond the populous islands of Java and Bali poses a logistical challenge given Indonesia's geography, but, beyond simply replicating the efforts described here, in more remote regions organisers could seek to engage with local stakeholders and integrate forms of traditional and indigenous knowledge into the project (Leach and Fairhead, 2002). It would also be desirable to extend the monitoring protocol to better enable the calculation of population trends for common birds from the dataset, which as previously discussed is not possible with the data collected in BigMonth2020. To achieve this, a repeated samples protocol is needed consisting of a random selection of fixed sites, stratified by habitat type, to be surveyed at regular intervals. Meanwhile, the ad hoc sampling adopted for Bigmonth2020 would be retained because it is inclusive, offers training opportunities for less experienced volunteers, and helps recruit, retain and involve more casual participants (Higby et al., 2012). Finally, spatiotemporal data coverage can be extended and duplication of effort avoided by establishing closer connections with existing initiatives including the Asian Waterbird Census (International Waterbird Census, 2020), eBird and Raptor Watch (Yuda, 2017).

2.4.5 Conclusion

We have demonstrated the potential of citizen science to address gaps in biodiversity distribution data coverage that are unlikely to be filled by traditional fieldwork, as well as its ability to engage with a young demographic, not all of whom were seasoned birdwatchers. Our approach was based on a tailored incentive scheme and targeted social media promotion campaign and stimulated a data collection approach built on existing local efforts. We have identified key aspects of the incentive scheme and data collection protocol that can be adapted to improve data quality, and what would be required to monitor population trends as well as distributions. Considering the popularity of citizen science among funders (Gura, 2013) and the benefits that can be derived from it (McKinley et al., 2017), we hope that the findings and processes reported here will prove a basis, guide and stimulus to similar endeavours across the tropics.

Supplementary Information

Pre- and post-BigMonth2020 data coverage for species recorded (excluding ‘sensitive species’) can be viewed at https://bigmonth2020.shinyapps.io/shiny_app/).

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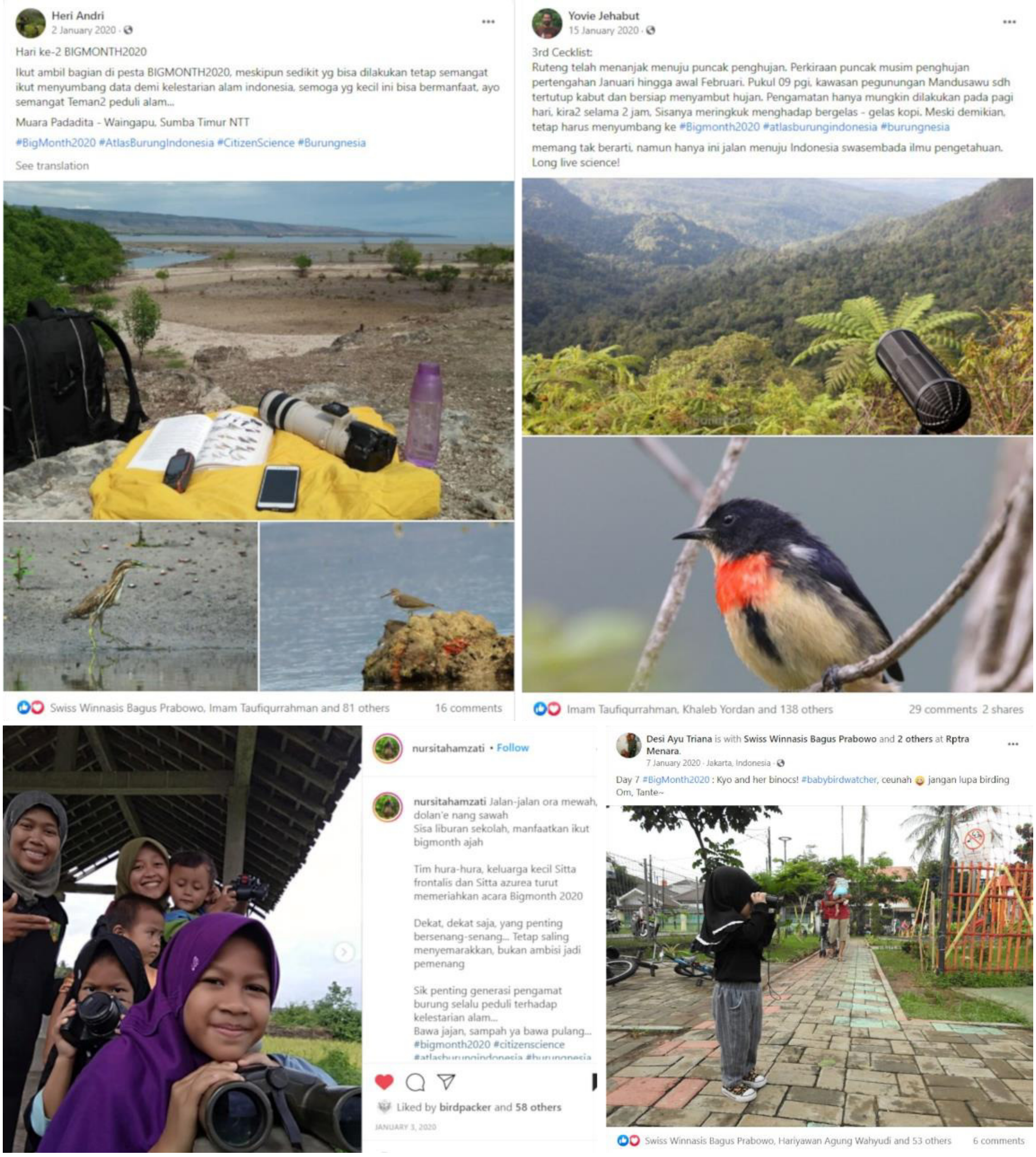
Appendices

Appendix 2.1 Social media posts during BigMonth2020. From top left clockwise: a participant visiting a coastal area to record birds; another participant using audio equipment to record birds; a young participant was recording birds with her mother at an urban park; and young citizen scientists took part in BigMonth2020 with the support of their teacher.



Heri Andri
2 January 2020

Hari ke-2 BIGMONTH2020

Ikut ambil bagian di pesta BIGMONTH2020, meskipun sedikit yg bisa dilakukan tetap semangat ikut menyumbang data demi kelestarian alam indonesia, semoga yg kecil ini bisa bermanfaat, ayo semangat Teman2 peduli alam...

Muara Padadita - Waingapu, Sumba Timur NTT

#BigMonth2020 #AtlasBurungIndonesia #CitizenScience #Burungnesia

See translation

Swiss Winnasis Bagus Prabowo, Imam Taufiqurrahman and 81 others 16 comments

Yovie Jehabut
15 January 2020

3rd Cecklist:
Ruteng telah menanjak menuju puncak penghujan. Perkiraan puncak musim penghujan pertengahan Januari hingga awal Februari. Pukul 09 pgi, kawasan pegunungan Mandusawu sdh tertutup kabut dan bersiap menyambut hujan. Pengamatan hanya mungkin dilakukan pada pagi hari, kira2 selama 2 jam. Sisanya meringkuk menghadap bergelas - gelas kopi. Meski demikian, tetap harus menyumbang ke #Bigmonth2020 #atlasburungindonesia #burungnesia

memang tak berarti, namun hanya ini jalan menuju Indonesia swasembada ilmu pengetahuan. Long live science!

Imam Taufiqurrahman, Khaleb Yordan and 138 others 29 comments 2 shares

nursitahamzati • Follow

nursitahamzati Jalan-jalan ora mewah, dolan'e nang sawah
Sisa liburan sekolah, manfaatkan ikut bigmonth ajah

Tim hura-hura, keluarga kecil Sitta frontalis dan Sitta azurea turut memeriahkan acara Bigmonth 2020

Dekat, dekat saja, yang penting bersenang-senang... Tetap saling menyemarakkan, bukan ambisi jadi pemenang

Sik penting generasi pengamat burung selalu peduli terhadap kelestarian alam...
Bawa jajan, sampah ya bawa pulang...
#bigmonth2020 #citizenscience
#atlasburungindonesia #burungnesia

Liked by **birdpacker** and **58 others**
JANUARY 3, 2020

Desi Ayu Triana is with **Swiss Winnasis Bagus Prabowo** and **2 others** at **Rptra Menara**.
7 January 2020 · Jakarta, Indonesia

Day 7 #BigMonth2020 : Kyo and her binocs! #babybirdwatcher, ceunah 😀 jangan lupa birding Om, Tante~

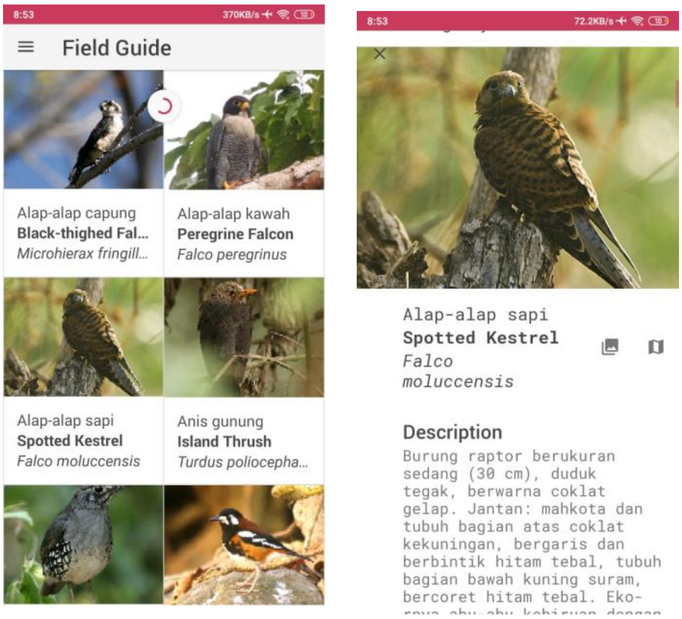
Swiss Winnasis Bagus Prabowo, Hariyawan Agung Wahyudi and 53 others 6 comments

Appendix 2.2 Data recording protocol using the Burungnesia app

The process for recording data in Burungnesia is as follows:
1. The user visits any location to record birds.
2. A new bird list is initiated within the app.
3. The following list data are recorded: location name, habitat (free text description), observer name, latitude, and longitude (automatically taken from the device's GPS, or input manually by selecting position on a map).
4. Every new species encountered and positively identified is added to the list and a count of individuals can be included (optional). The app includes a field guide to aid identification (this can be downloaded for offline use).

Additional notes on the recording protocol

The time and date of the bird list are taken by the app from the time of upload to the central database. The app allows species to be added to an existing list for visits at later dates. However, for BigMonth2020, we requested that a new bird list be recorded if the same location was visited on a different date.
Contributors were asked to spend at least one hour in the general vicinity of the point at which they began the list, and to begin a new list if they changed major habitat type (e.g., from rice paddy to woodland) or moved a significant distance (>3 km) from the start point.



Alap-alap capung
Black-thighed Fal...
Microhierax fringill...

Alap-alap kawah
Peregrine Falcon
Falco peregrinus

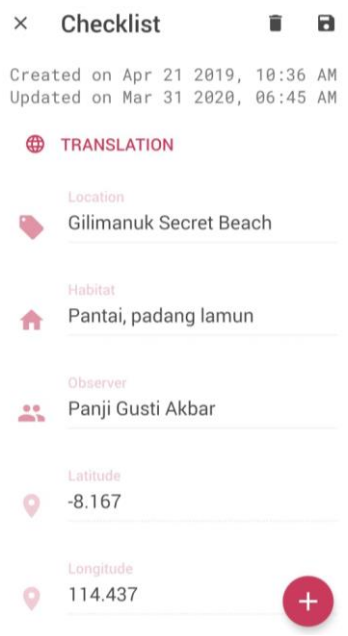
Alap-alap sapi
Spotted Kestrel
Falco moluccensis

Anis gunung
Island Thrush
Turdus poliocepha...

Alap-alap sapi
Spotted Kestrel
Falco moluccensis

Description
Burung raptor berukuran sedang (30 cm), duduk tegak, berwarna coklat gelap. Jantan: mahkota dan tubuh bagian atas coklat kekuningan, bergaris dan berbintik hitam tebal, tubuh bagian bawah kuning suram, bercoret hitam tebal. Eko-

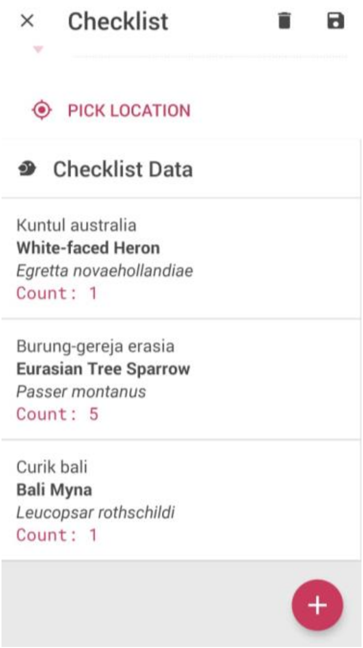
Field guide available within the Burungnesia app. This can be downloaded for use offline.



Created on Apr 21 2019, 10:36 AM
Updated on Mar 31 2020, 06:45 AM

Location: Gilimanuk Secret Beach
Habitat: Pantai, padang lamun
Observer: Panji Gusti Akbar
Latitude: -8.167
Longitude: 114.437

List data screen in the Burungnesia app.



Kuntul australia
White-faced Heron
Egretta novaehollandiae
Count: 1

Burung-gereja erasia
Eurasian Tree Sparrow
Passer montanus
Count: 5

Curik bali
Bali Myna
Leucopsar rothschildi
Count: 1

Species recording screen. Counts of individuals can be entered optionally.

Appendix 2.3 BigMonth2020 Feedback Questionnaire

This questionnaire was administered by Universitas Atma Jaya, Yogyakarta, in Bahasa Indonesia; this is a translation to English language.

We would like to invite you to take part in a research study which aims to gather feedback for BigMonth2020. This will take you around 20 minutes to complete. Your feedback will help us ensure that next year's Big Month is even better. Your participation is completely voluntary, and you are free to quit the survey at any time without submitting the response. To have your response included in the survey, please press the “submit” button at the end of the survey. The information you give us will be anonymised, so the data will be untraceable.
Please consider your answers and answer as completely as possible.
Do you agree to take part in this research?
Yes/No
Birdwatching
These questions are related to your birdwatching experience. Think about the birdwatching you did before BigMonth2020 when answering these questions.
How many years have you been birdwatching for? *Answer in years*

<u>Optional</u>: please tell us a short (100 words) story about how you became interested in birding
(responses may be used for promotional material in the future)

How did you become interested in birdwatching? *unlimited selection*
friend group / birdwatching club / family / developed personal interest / had pet birds / nature documentary / social media / photography / other
if other, please specify:

Are you affiliated with a birdwatching club?
Yes / No

If you are a member of a birdwatching club, please specify which club
name of club

When did you join the club? d*ate*

Not including BigMonth2020, how often do you go birdwatching?
Never / I have, but not routinely / once in more than 6 months / once every 6 months / once every 3 months / monthly / at least twice a month / weekly

In 2019, which habitats and places did you visit to go birdwatching?
mountain / agricultural area / coast / wetland / park / urban area / national park / nature reserve / forest / islands apart from Java/Bali / other

In 2019, what was the furthest distance you travelled to go birdwatching? *answer in km*

Including you, what is your normal group size when you go birdwatching?

What limits the amount of birdwatching you do?
time / money / work / having nobody to go with / transport / equipment / no free time / family / other

What equipment do you have?
binoculars / telescope / camera / telephoto lens / audio recorder / bird ringing equipment / other

Which statement best describes your level of bird identification skill?
I can identify most birds aurally
I can identify about half of birds I encounter aurally
I mostly use visual features to identify birds, but some common species I know by sound
I almost always use visual features
I can identify common species that I often see
I usually have to check the identification guide
I am new to bird watching

Which bird field guides do you own?
MacKinnon (A Field Guide to the Birds of Borneo, Sumatra, Java and Bali)
Coates and Bishop (A Guide to the Birds of Wallacea: Sulawesi, the Moluccas, and Lesser Sunda Islands, Indonesia)
Pratt and Beehler (Birds of New Guinea)
King, Woodcock, and Dickinson (Birds of South-East Asia)
Robson (Birds of South-East Asia)
Hayman, Marchant, and Prater (Shorebirds)
Eaton, van Balen, Brickle and Rheindt (Birds of the Indonesian Archipelago)
Sundev and Yamazaki (Raptors of Asia)
Bhushan et al. (A Field Guide to the Waterbirds of Asia)
I do not have any field guides

How many species of bird have you seen in your life?
<50 / 50–100 / 100–200 / >200 / >500

Do you have any other hobbies?
free text answer

BigMonth2020

This section is specific to the impact of BigMonth2020 and what was good/bad about it, and how it could be improved in the future.

How did you hear about BigMonth2020?
friend / birdwatching club / Facebook / Instagram / Twitter / university or school / other

Do you feel proud about BigMonth2020 taking place in Indonesia? If so, please explain why (50 words)

What stopped you from collecting more data for BigMonth2020? (select all that apply)
money / transport / time / work / nobody to go with / study / family commitment / other

Has BigMonth changed your bird watching habits, and if so, how? (50 words)

How could we attract more participants for BigMonth in the future?
Please rank the importance to you of the following motivations to participate in BigMonth (1 = not important, 10 = very important)

chance to win prizes
personal goal
being part of a citizen science project
represent your region
contribution to conservation
seeing new birds
visiting new places
socialising

How important were the following aspects of BigMonth2020 to you? (1 = not important, 10 = very important)

Contribution to international conservation
Contribution towards the first Indonesian Bird Atlas
To try and encourage others to become involved in birdwatching
To show how much Indonesians love birdwatching
To encourage the wider community to find out about decreasing numbers of birds

Status of Java and Bali's birds
These questions are related to your overall impressions of birdwatching in Java and Bali.

During your participation with BigMonth, the number of birds you saw was…

Much more than I expected
More than I expected
Less than I expected
Much less than I expected
About what I expected
I did not have an expectation

What species were you most surprised to see during BigMonth2020, and why?

What was your favourite species that you saw during BigMonth, and why?

Please describe your impression of the status of Javanese and Balinese birds based on your experiences during the BigMonth2020

Based on your knowledge, what are the most important threats to birds in Java and Bali?
hunting for food / cagebird trade / climate change / pollution / urbanisation habitat loss / invasive species

Personal Information
Name:
Age:
Gender:
Home province: (*options for all provinces of Java/Bali and other Indonesian islands*)
Education level: *SLTP/SLTA/Strata 1/ Strata 2 / Strata 3/*
Profession: *student / lecturer / teacher / researcher / conservation consultant / NGO / Ministry of Environment / entrepreneur / farm worker / house husband or wife / retired / freelance / other*
Email address:

Original questionnaire administered in Indonesian
Kami ingin mengundang Anda untuk berpartisipasi dalam penelitian yang bertujuan untuk mengumpulkan umpan balik dari Big Month 2020. Survei ini memakan waktu sekitar 20 menit. Umpan balik Anda akan membantu kami memastikan Big Month selanjutnya bisa menjadi lebih baik. Partisipasi Anda bersifat sukarela, dan Anda bebas berhenti mengisi survei ini kapan saja tanpa mengirim tanggapan. Untuk mengikutsertakan tanggapan Anda di survei ini, mohon untuk menekan tombol submit di akhir wawancara. Informasi yang Anda berikan bersifat anonim, sehingga data Anda tidak dapat kami lacak.
Mohon mempertimbangkan jawaban Anda dan menyelesaikan seluruh jawaban selengkap mungkin.

Apakah Anda bersedia mengikuti penelitian ini?
Ya / Tidak

Pengamatan Burung
Pertanyaan ini berhubungan dengan pengalaman pengamatan burung Anda. Silahkan mengingat kegiatan pengamatan burung yang telah Anda lakukan sebelum Big Month 2020 ketika menjawab pertanyaan ini.

Berapa tahun Anda telah melaksanakan kegiatan pengamatan burung? *Mohon jawab dalam satuan tahun*

Opsional: mohon beritahu kami cerita pendek (100 kata) mengenai bagaimana Anda mulai tertarik dengan kegiatan pengamatan burung (*tanggapan ini mungkin akan digunakan untuk materi promosi di masa depan)*

Bagaimana Anda tertarik dengan pengamatan burung? *Bisa pilih lebih dari satu*
a. teman / kelompok pengamat burung / keluarga / diri sendiri / pernah memelihara burung / film dokumenter alam / media sosial / fotografi / lainnya
b. jika pilih lainnya, mohon dipesifikasikan:

Apakah Anda berafiliasi dengan kelompok pengamata burung?
Ya / Tidak

Jika Anda merupakan anggota klub pengamat burung, mohon dispesifikasikan nama klubnya
nama klub pengamatan burung
Kapan Anda bergabung dengan klub tersebut? *tanggal*

Tidak termasuk kegiatan BigMonth 2020, seberapa sering Anda melakukan kegiatan pengamatan burung?
Tidak pernah / Pernah namun tidak rutin / Sekali setiap lebih dari 6 bulan / Sekali setiap 6 bulan/ Sekali setiap 3 bulan / Setiap bulan / Setidaknya dua kali sebulan / Setiap minggu

Di tahun 2019, di habitat mana saja Anda pernah melakukan kegiatan pengamatan burung?
Gunung / perkebunan / pantai / lahan basah / taman / kawasan urban / taman nasional / cagar alam / hutan / pulau di luar Jawa-Bali / lainnya

Di tahun 2019, seberapa jauh jarak yang Anda tempuh untuk melakukan kegiatan pengamatan burung? *jawab dalam satuan km*

Termasuk dii Anda sendiri, berapa orang yang biasanya terlibat ketika Anda mengamati burung?

Apa yang membatasi Anda dalam melakukan kegiatan pengamatan burung?
waktu / uang / pekerjaan / tidak ada teman / transportasi / peralatan / tidak ada waktu luang / keluarga / lainnya

Peralatan apa yang Anda miliki?
binokuler / teleskop / kamera / lensa telefoto / perekam suara / peralatan pencincinan burung / lainnya

Dari pernyataan berikut, mana yang paling bisa mendeskripsikan kemampuan identifikasi burung Anda?

Saya bisa mengidentifikasi semua burung melalui suara
Saya bisa mengidentifikasi sekitar setengah spesies burung yang saya jumpai melalui suara
Saya lebih sering mengidentifikasi secara visual, namun saya juga tahu beberapa suara burung
Saya hampir selalu menggunakan fitur visual untuk mengidentifikasi bruung
Saya bisa mengidentifikasi spesies-spesies umum yang sering saya lihat
Saya biasanya harus melihat panduan identifikasi untuk mengidentifikasi burung
Saya sangat baru terhadap kegiatan pengamatan burung

Buku panduan lapangan pengamatan burung mana yang Anda miliki?

MacKinnon (A Field Guide to the Birds of Borneo, Sumatra, Java, and Bali)
Coates and Bishop (A Guide to the Birds of Wallacea: Sulawesi, the Moluccas, and Lesser Sunda Islands, Indonesia)
Pratt and Beehler (Birds of New Guinea)
King, Woodcock, and Dickinson (Birds of South-East Asia)
Robson (Birds of South-East Asia)
Hayman, Marchant, and Prater (Shorebirds)
Eaton, van Balen, Brickle and Rheindt (Birds of the Indonesian Archipelago)
Sundev and Yamazaki (Raptors of Asia)
Bhushan et al. (A Field Guide to the Waterbirds of Asia)
Saya tidak memiliki buku panduan lapangan pengamatan burung

Berapa banyak spesies burung yang telah Anda lihat seumur hidup?
<50 / 50–100 / 100–200 / >200 / >500
Apakah Anda memiliki hobby lain?
jawaban bebas

BigMonth2020

Bagian ini bersifat spesifik pada dampak BigMonth2020 dan hal-hal yang baik/buruk mengenai kegiatan tersebut, serta bagaimana kegiatan ini dapat diperbaiki di masa depan.

Bagaimana Anda mengetahui kegiatam BigMonth 2020
teman / klub pengamatan burung / Facebook / Instagram / Twitter / sekolah atau kampus / lainnya

Apakah Anda merasa bangga dengan terlaksananya BigMonth 2020 di Indonesia? Jika iya, mohon jelaskan mengapa (50 kata)

Apa yang membuat Anda berhenti mengumpulkan data untuk BigMonth2020 (pilih semua yang sesuai)
uang / transportasi / waktu / pekerjaan / tidak ada teman / kuliah atau sekolah / komitmen keluarga / lainnya

Apakah BigMonth mengubah kebiasaan pengamatan burung Anda? dan jika iya, apa yang berubah? (50 kata)

Bagaimana kami bisa menarik lebih banyak peserta BigMonth di masa depan?

Tolong berikan peringkat seberapa penting motivasi berikut yang mempengaruhi Anda berpartisipasi dalam BigMonth (1 = tidak penting, 10 = sangat penting)

kesempatan memenangkan hadiah
tujuan personal
ingin menjadi bagian dari proyek citizen science
ewakili daerah Anda
berkontribusi dalam kegiatan konservasi
melihat spesies burung balu
mengunjungi lokasi baru
bersosialisasi

Bagi Anda, seberapa penting aspek-aspek dari BigMonth2020 berikut? (1 = tidak penting, 10 = sangat penting)

Kontribusi terhadap kegiatan konservasi international
Kontribusi terhadap Atlas Burung Indonesia
Menarik perhatian orang lain untuk terlibat kegiatan pengamatan burung
Menunjukan kecintaan warga Indonesia terhadap kegiatan pengamatan burung
Menarik perhatian masyarakat untuk mengetahui berkurangnya populasi burung

Status burung-burung di Jawa dan Bali

Pertanyaan ini berhubungan dengan impresi keseluruhan Anda mengenai pengamatan burung di Jawa dan Bali.

Selama berpartisipasi dalam BigMonth, jumlah burung yang Anda lihat...
Jauh lebih banyak dari yang saya duga / Lebih banyak dari yang saya duga
Lebih sedikit dari yang saya duga / Jauh lebih sedikit dari yang saya duga
Sekitar sebanyak yang syaa duga / Saya tidak punya ekspetasi apa-apa

Spesies apa yang paling membuat Anda terkejut ketika Anda jumpai di BigMonth 2020? Mengapa?

Spesies apa yang berhasil Anda jumpai dan menjadi favorit Anda selama BigMonth? Mengapa?

Tolong jelaskan impresi Anda terhadap status burung-burung di Jawa dan Bali berdasarkan pengalaman Anda selama BigMonth2020

Berdasarkan pengetahuan Anda, apa yang menjadi ancaman terbesar bagi burung-burung di Jawa dan Bali?
Perburuan untuk dimakan / Perdagangan burung peliharaan / Perubahan iklim polusi / urbanisasi / kehilangan habitat / spesies invasif

Informasi Pribadi
Nama:
Usia:
Jenis kelamin:
Provinsi asal: (pilihan untuk seluruh provinsi di Jawa/Bali dan pulau-pulau lainnya)
Tingkatan edukasi: *SLTP/SLTA/Strata 1/Strata 2/ Strata 3*
Profesi: *pelajar / dosen / guru / peneliti / konsultan konservasi / LSM / Kementrian Lingkungan Hidup dna Kehutanan / pengusaha / peternak / rumah tangga / pensiunan / freelance / lainnya*
Alamat email:

Appendix 2.4 Summarised data for all species recorded during BigMonth2020. Data were recorded in 2,417 of the 3,408 grid squares across Java and Bali. Functional groups: AF—aerial feeders; OP—birds of open/agricultural areas; SC—scrub/savanna birds; WL—woodland birds; WA—waterbirds; RA—raptors.

Common name	Scientific name	Functional group	Red List status	Records	Surveyed squares present (%)	Increase in coverage (%)
Cave Swiftlet	*Collocalia linchi*	AF	LC	16,702	95.4	236.2
Eurasian Tree Sparrow	*Passer montanus*	OP	LC	10,596	80.8	276.7
Sooty-headed Bulbul	*Pycnonotus aurigaster*	WL	LC	7,193	75.8	228.3
Scaly-breasted Munia	*Lonchura punctulata*	OP	LC	6,334	68.6	282.8
Javan Munia	*Lonchura leucogastroides*	OP	LC	5,964	62.5	201.9
Scarlet-headed Flowerpecker	*Dicaeum trochileum*	WL	LC	5,548	59.8	279.4
House Swallow	*Hirundo javanica*	AF	LC	3,978	55.6	543.4
Olive-backed Sunbird	*Cinnyris jugularis*	WL	LC	3,918	55.9	198.2
Eastern Spotted Dove	*Spilopelia chinensis*	SC	LC	3,789	56.2	182.4
Zitting Cisticola	*Cisticola juncidis*	OP	LC	2,770	41.3	322.4
Zebra Dove	*Geopelia striata*	SC	LC	2,313	33.9	273.8
Collared Kingfisher	*Todiramphus chloris*	OP	LC	2,120	45.5	147.2
Javan Pond-heron	*Ardeola speciosa*	WA	LC	1,857	30.9	154.9
Barn Swallow	*Hirundo rustica*	AF	LC	1,794	29.4	216
Yellow-vented Bulbul	*Pycnonotus goiavier*	WL	LC	1,737	31.9	136.8
Common Tailorbird	*Orthotomus sutorius*	SC	LC	1,555	33.9	260.9
Javan Kingfisher	*Halcyon cyanoventris*	OP	LC	1,396	31.9	141.1
Little Egret	*Egretta garzetta*	WA	LC	1,319	20.6	156.7
Common Iora	*Aegithina tiphia*	WL	LC	1,213	27.4	156.7
White-headed Munia	*Lonchura maja*	OP	LC	1,091	20.3	185.3
Brown-throated Sunbird	*Anthreptes malacensis*	WL	LC	931	19.3	126.1
Small Minivet	*Pericrocotus cinnamomeus*	WL	LC	921	22.5	180.8
Cattle Egret	*Bubulcus ibis*	WA	LC	855	16	103
Brush Cuckoo	*Cacomantis variolosus*	WL	LC	845	22.6	215.8
Plaintive Cuckoo	*Cacomantis merulinus*	WL	LC	814	21.7	163.9
Red-rumped Swallow	*Cecropis daurica*	AF	LC	770	18	160.2
Plain Prinia	*Prinia inornata*	OP	LC	711	17.5	178.7
White-breasted Woodswallow	*Artamus leucoryn*	AF	LC	700	18.1	117.7
Olive-backed Tailorbird	*Orthotomus sepium*	SC	LC	606	16.3	104.4
Cerulean Kingfisher	*Alcedo coerulescens*	WA	LC	478	10.9	93.2
Golden-bellied Gerygone	*Gerygone sulphurea*	WL	LC	473	9.5	83.5
Common Sandpiper	*Actitis hypoleucos*	WA	LC	449	9.4	73.5
Sunda Pygmy Woodpecker	*Picoides moluccensis*	WL	LC	428	12.5	138.5
Long-tailed Shrike	*Lanius schach*	SC	LC	352	11	74.2
House Swift	*Apus nipalensis*	AF	LC	314	9.4	75.3
White-breasted Waterhen	*Amaurornis phoenicurus*	WA	LC	298	8.8	108.8
Green-backed Heron	*Butorides striata*	WA	LC	293	5.8	51.7
Rock Dove	*Columba livia*	OP	LC	284	6.5	111.9
Lesser Coucal	*Centropus bengalensis*	SC	LC	277	8.9	92.5
Sunda Pied Fantail	*Rhipidura javanica*	WL	LC	273	7.3	63.4

Common name	Scientific name	Functional group	Red List status	Records	Surveyed squares present (%)	Increase in coverage (%)
Barred Buttonquail	*Turnix suscitator*	SC	LC	268	8.2	92.3
Freckle-breasted Woodpecker	*Dendrocopos analis*	WL	LC	266	9	65.2
Crested Serpent-eagle	*Spilornis cheela*	RA	LC	250	8.2	53.2
Coppersmith Barbet	*Psilopogon haemacephalus*	WL	LC	209	6.4	70.6
Pied Triller	*Lalage nigra*	WL	LC	209	5.7	89.4
Savanna Nightjar	*Caprimulgus affinis*	OP	LC	206	5.6	62.9
Edible-nest / Black-nest Swiftlet	*Aerodramus fuciphagus/maximus*	AF	LC	188	5.6	0
Blue-eared Kingfisher	*Alcedo meninting*	WA	LC	179	5	56.2
Great White Egret	*Ardea alba*	WA	LC	178	2.9	35.9
Intermediate Egret	*Ardea intermedia*	WA	LC	175	4.2	67.2
Horsfield's Babbler	*Malacocincla sepiaria*	WL	LC	167	4.8	48.7
Javan Plover	*Charadrius javanicus*	WA	NT	164	4.2	56.8
White-shouldered Triller	*Lalage sueurii*	WL	LC	156	4.3	150
Yellow Bittern	*Ixobrychus sinensis*	WA	LC	145	4	75
White-capped Munia	*Lonchura ferruginosa*	OP	LC	143	2.9	116.7
Javan Myna	*Acridotheres javanicus*	OP	VU	142	4.5	48.8
Black-crowned Night-heron	*Nycticorax nycticorax*	WA	LC	141	3.2	46.1
Orange-bellied Flowerpecker	*Dicaeum trigonostigma*	WL	LC	140	3.4	70.9
Purple Heron	*Ardea purpurea*	WA	LC	138	3.2	36.4
Blue-tailed Bee-eater	*Merops philippinus*	AF	LC	133	3.6	52.3
Pink-necked Green-pigeon	*Treron vernans*	WL	LC	133	3.4	36.5
Bar-winged Prinia	*Prinia familiaris*	OP	NT	131	3.6	48.4
Grey Heron	*Ardea cinerea*	WA	LC	123	2.7	34.7
Square-tailed Drongo-cuckoo	*Surniculus lugubris*	WL	LC	115	3.3	74
Cinnamon Bittern	*Ixobrychus cinnamomeus*	WA	LC	106	3.4	65.4
Javan Coucal	*Centropus nigrorufus*	SC	VU	101	2.8	69.8
Grey-rumped Treeswift	*Hemiprocne longipennis*	AF	LC	95	3.2	47.1
Sunda Collared-dove	*Streptopelia bitorquata*	SC	LC	95	3	43.5
Wood Sandpiper	*Tringa glareola*	WA	LC	92	2.8	52.4
White-winged Tern	*Chlidonias leucopterus*	WA	LC	90	1.5	60.4
Whimbrel	*Numenius phaeopus*	WA	LC	88	1.7	24.3
Little Black Cormorant	*Phalacrocorax sulcirostris*	WA	LC	76	1.5	35.5
Banded Bay Cuckoo	*Cacomantis sonneratii*	WL	LC	74	2.8	88.7
Black-winged Flycatcher-shrike	*Hemipus hirundinaceus*	WL	LC	74	2.6	33.1
Green Junglefowl	*Gallus varius*	WL	LC	74	2.7	34.2
Little Tern	*Sternula albifrons*	WA	LC	74	1.5	44.8
Little Spiderhunter	*Arachnothera longirostra*	WL	LC	72	2.3	33.3
Common Barn-owl	*Tyto alba*	RA	LC	68	2.1	61.2
Sangkar White-eye	*Zosterops melanurus*	WL	VU	66	2.4	27.6
Yellow-eared Barbet	*Psilopogon australis*	WL	LC	66	2.2	35.2
Ruby-throated Bulbul	*Rubigula dispar*	WL	VU	62	2	52
Whiskered Tern	*Chlidonias hybrida*	WA	LC	62	1.7	72.5
Golden-headed Cisticola	*Cisticola exilis*	OP	LC	61	2	62.3

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Greater Coucal	*Centropus sinensis*	SC	LC	61	2.2	41
Yellow-bellied Prinia	*Prinia flaviventris*	OP	LC	61	2.2	70.1
Black Eagle	*Ictinaetus malaiensis*	RA	LC	57	1.9	22
Ashy Tailorbird	*Orthotomus ruficeps*	SC	LC	56	1.9	19.4
Black-naped Oriole	*Oriolus chinensis*	WL	LC	56	1.8	22.7
Black Drongo	*Dicrurus macrocercus*	OP	LC	55	1.9	33.1
Chestnut-breasted Malkoha	*Phaenicophaeus curvirostris*	WL	LC	53	1.9	23.8
Eastern Yellow Wagtail	*Motacilla tschutschensis*	OP	LC	53	1.3	91.4
Oriental Honey-buzzard	*Pernis ptilorhynchus*	RA	LC	52	1.9	20.1
White-browed Crake	*Amaurornis cinerea*	WA	LC	52	1.2	10.7
Pacific Swift	*Apus pacificus*	AF	LC	50	1.9	30.8
Javan Banded Pitta	*Hydrornis guajanus*	WL	LC	47	1.6	33.3
Striated Grassbird	*Megalurus palustris*	OP	LC	46	1.4	20.4
Australasian Reed-warbler	*Acrocephalus australis*	OP	LC	45	1.6	194.4
Common Moorhen	*Gallinula chloropus*	WA	LC	42	1.2	27.9
Spotted Kestrel	*Falco moluccensis*	RA	LC	41	1.4	26.8
Sunda Teal	*Anas gibberifrons*	WA	NT	41	1.1	22.9
Sunda Scops-owl	*Otus lempiji*	RA	LC	40	1.5	15.8
Rufous-browed Babbler	*Pellorneum capistratum*	WL	LC	39	1.2	25.3
Ashy Drongo	*Dicrurus leucophaeus*	WL	LC	38	1.4	15.2
Black-winged Stilt	*Himantopus himantopus*	WA	LC	38	0.9	39.5
Scarlet Minivet	*Pericrocotus flammeus*	WL	LC	38	1.3	18
Black-naped Monarch	*Hypothymis azurea*	WL	LC	34	1.2	22.8
Great Tit	*Parus major*	WL	LC	34	1.3	21.1
Glossy Ibis	*Plegadis falcinellus*	WA	LC	33	0.3	19.2
Grey-capped Emerald Dove	*Chalcophaps indica*	WL	LC	33	1.2	19.8
Milky Stork	*Mycteria cinerea*	WA	EN	33	0.7	25
Pacific Golden Plover	*Pluvialis fulva*	WA	LC	29	0.9	17.8
Pacific Reef-egret	*Egretta sacra*	WA	LC	29	1	12
Common Redshank	*Tringa totanus*	WA	LC	28	0.7	30.6
Chestnut-headed Bee-eater	*Merops leschenaulti*	AF	LC	27	1	22.3
Purple-backed Starling	*Agropsar sturninus*	SC	LC	26	0.8	51.9
Little Pied Flycatcher	*Ficedula westermanni*	WL	LC	25	0.8	17
Sacred Kingfisher	*Todiramphus sanctus*	WA	LC	25	1	18.5
Oriental Darter	*Anhinga melanogaster*	WA	NT	24	0.6	15.8
Oriental Reed-warbler	*Acrocephalus orientalis*	OP	LC	24	0.4	23.5
Sunda Forktail	*Enicurus velatus*	WL	LC	24	0.8	25
Greater Crested Tern	*Thalasseus bergii*	WA	LC	23	0.8	13.7
Grey-cheeked Green-pigeon	*Treron griseicauda*	WL	LC	23	0.8	7.2
Java Sparrow	*Lonchura oryzivora*	OP	EN	23	0.7	15.5
Olive-winged Bulbul	*Pycnonotus plumosus*	WL	LC	23	0.7	30.4
Red-breasted Parakeet	*Psittacula alexandri*	WL	NT	23	0.7	7.8
Brown Prinia	*Prinia polychroa*	SC	LC	21	0.7	29.1
Common Tern	*Sterna hirundo*	WA	LC	21	0.5	21.3

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Ruddy Cuckoo-dove	*Macropygia emiliana*	WL	LC	21	0.7	13.8
Black-naped Tern	*Sterna sumatrana*	WA	LC	20	0.7	18.7
Black Bittern	*Ixobrychus flavicollis*	WA	LC	20	0.5	37.9
Great-billed Heron	*Ardea sumatrana*	WA	LC	20	0.3	15.4
Asian Woollyneck	*Ciconia episcopus*	WA	VU	19	0.6	34.2
Changeable Hawk-eagle	*Nisaetus cirrhatus*	RA	LC	19	0.8	17.2
Common Greenshank	*Tringa nebularia*	WA	LC	19	0.6	16.7
Flame-fronted Barbet	*Psilopogon armillaris*	WL	LC	19	0.7	10.4
Lesser Whistling-duck	*Dendrocygna javanica*	WA	LC	18	0.5	10.5
Asian House Martin	*Delichon dasypus*	AF	LC	17	0.6	23.5
Common Gull-billed Tern	*Gelochelidon nilotica*	WA	LC	17	0.6	33.3
Javan Hawk-eagle	*Nisaetus bartelsi*	RA	EN	17	0.5	8.7
Javan Whistling-thrush	*Myophonus glaucinus*	WL	LC	17	0.6	19.6
Black-banded Barbet	*Psilopogon javensis*	WL	NT	16	0.5	10.3
Black-naped Fruit-dove	*Ptilinopus melanospilus*	WL	LC	16	0.4	7.4
Common Flameback	*Dinopium javanense*	WL	LC	16	0.6	16.7
Javan Sunbird	*Aethopyga mystacalis*	WA	LC	16	0.5	10.4
Oriental Pratincole	*Glareola maldivarum*	WL	LC	16	0.6	31.1
Slender-billed Crow	*Corvus enca*	WL	LC	16	0.6	11.5
Asian Brown Flycatcher	*Muscicapa dauurica*	WL	LC	15	0.5	11.8
Bali Myna	*Leucopsar rothschildi*	WA	CR	15	0.3	0
Bar-tailed Godwit	*Limosa lapponica*	WL	NT	15	0.4	29.6
Racquet-tailed Treepie	*Crypsirina temia*	WL	LC	15	0.5	10
Lesser Shortwing	*Brachypteryx leucophris*	WL	LC	14	0.5	6.8
Mangrove Whistler	*Pachycephala cinerea*	WL	LC	14	0.5	11.9
Velvet-fronted Nuthatch	*Sitta frontalis*	WL	LC	14	0.5	12.5
White-bellied Sea-eagle	*Haliaeetus leucogaster*	RA	LC	14	0.5	5
Cream-vented Bulbul	*Pycnonotus simplex*	WL	LC	13	0.5	20
Lesser Sandplover	*Charadrius mongolus*	WA	LC	13	0.2	15.6
Short-tailed Starling	*Aplonis minor*	WL	LC	13	0.5	22.6
Yellow-throated Hanging-parrot	*Loriculus pusillus*	WL	NT	13	0.4	4.1
Brown-cheeked Bulbul	*Alophoixus bres*	WL	NT	12	0.3	2.6
Marsh Sandpiper	*Tringa stagnatilis*	WA	LC	12	0.3	27.3
Oriental Cuckoo	*Cuculus saturatus*	WL	LC	12	0.5	27.8
Oriental Magpie-robin	*Copsychus saularis*	WL	LC	12	0.4	5.1
Peregrine Falcon	*Falco peregrinus*	RA	LC	12	0.5	14.7
Red-billed Malkoha	*Zanclostomus javanicus*	WL	LC	12	0.5	22.5
Asian Glossy Starling	*Aplonis panayensis*	WL	LC	11	0.4	7.8
Common Buttonquail	*Turnix sylvaticus*	SC	LC	11	0.5	32.3
Greater Sandplover	*Charadrius leschenaultii*	WA	LC	11	0.3	17.1
Little Cormorant	*Microcarbo niger*	WA	LC	11	0.3	14.3
Orange-spotted Bulbul	*Pycnonotus bimaculatus*	WL	NT	11	0.3	18.2
Pygmy Cupwing	*Pnoepyga pusilla*	WL	LC	11	0.4	5.9
Red Junglefowl	*Gallus gallus*	WL	LC	11	0.5	11.2
Black-headed Bulbul	*Brachypodius atriceps*	WL	LC	10	0.4	12.9

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Brahminy Kite	*Haliastur indus*	RA	LC	10	0.4	11.1
Grey Wagtail	*Motacilla cinerea*	OP	LC	10	0.3	12.5
Large-tailed Nightjar	*Caprimulgus macrurus*	WL	LC	10	0.3	9
Mountain Warbler	*Phylloscopus trivirgatus*	WL	LC	10	0.4	8.1
Ruddy-breasted Crake	*Zapornia fusca*	WA	LC	10	0.4	15.6
Streaky-breasted Spiderhunter	*Arachnothera affinis*	WL	LC	10	0.3	6.7
Terek Sandpiper	*Xenus cinereus*	WA	LC	10	0.3	36.8
Blue Nuthatch	*Sitta azurea*	WL	LC	9	0.3	5
Crescent-chested Babbler	*Cyanoderma melanothorax*	WL	LC	9	0.3	2.7
Eurasian Curlew	*Numenius arquata*	WA	NT	9	0.2	26.1
Javan Warbler	*Phylloscopus grammiceps*	WL	LC	9	0.3	7.4
Oriental Dwarf-kingfisher	*Ceyx erithaca*	WA	LC	9	0.3	10.6
Paddyfield Pipit	*Anthus rufulus*	OP	LC	9	0.3	12.7
Pin-tailed Parrotfinch	*Erythrura prasina*	SC	LC	9	0.2	27.8
Purple Swamphen	*Porphyrio porphyrio*	WA	LC	9	0.1	6.9
Schrenck's Bittern	*Ixobrychus eurhythmus*	WA	LC	9	0.3	53.8
Sooty Tern	*Onychoprion fuscatus*	WA	LC	9	0.1	100
Streaked Weaver	*Ploceus manyar*	OP	LC	9	0.4	11.8
Sunda Cuckooshrike	*Coracina larvata*	WL	LC	9	0.2	6.1
Arctic Warbler	*Phylloscopus borealis*	WL	LC	8	0.2	3.8
Blue Whistling-thrush	*Myophonus caeruleus*	WL	LC	8	0.3	7.3
Common Myna	*Acridotheres tristis*	OP	LC	8	0.2	3.8
Indigo Flycatcher	*Eumyias indigo*	WL	LC	8	0.2	4.9
Javan Broadbill	*Eurylaimus javanicus*	WL	NT	8	0.2	9.8
Lineated Barbet	*Psilopogon lineatus*	WL	LC	8	0.3	7.7
Little Cuckoo-dove	*Macropygia ruficeps*	WL	LC	8	0.2	0
Mountain Tailorbird	*Phyllergates cucullatus*	WL	LC	8	0.3	7.8
Oriental Pied Hornbill	*Anthracoceros albirostris*	WL	LC	8	0.2	4.2
Pheasant-tailed Jacana	*Hydrophasianus chirurgus*	WA	LC	8	0.2	14.3
Red-necked Stint	*Calidris ruficollis*	WA	NT	8	0.3	14.6
Silver-rumped Spinetail	*Rhaphidura leucopygialis*	AF	LC	8	0.3	46.7
Australasian Grebe	*Tachybaptus novaehollandiae*	WA	LC	7	0.2	22.7
Blood-breasted Flowerpecker	*Dicaeum sanguinolentum*	WL	LC	7	0.3	5.7
Chinese Sparrowhawk	*Accipiter soloensis*	RA	LC	7	0.3	5.9
Crimson Sunbird	*Aethopyga siparaja*	WL	LC	7	0.3	46.7
Lemon-bellied White-eye	*Zosterops chloris*	WL	LC	7	0.3	7
Long-billed Spiderhunter	*Arachnothera robusta*	WL	LC	7	0.2	19.2
Osprey	*Pandion haliaetus*	RA	LC	7	0.2	42.9
Yellow-rumped Flycatcher	*Ficedula zanthopygia*	WL	LC	7	0.2	23.1
Asian Palm-swift	*Cypsiurus balasiensis*	AF	LC	6	0.2	2.4
Banded Kingfisher	*Lacedo pulchella*	WL	LC	6	0.2	19.2
Black-winged Kite	*Elanus caeruleus*	RA	LC	6	0.2	27.3
Greater Racquet-tailed Drongo	*Dicrurus paradiseus*	WL	LC	6	0.2	4.2

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Grey-headed Canary-flycatcher	*Culicicapa ceylonensis*	WL	LC	6	0.2	4.4
Javan Bulbul	*Ixos virescens*	WL	LC	6	0.2	1.9
Javan Flameback	*Chrysocolaptes strictus*	WL	VU	6	0.1	8.8
Lesser Adjutant	*Leptoptilos javanicus*	WA	VU	6	0.2	2.2
Little Pied Cormorant	*Microcarbo melanoleucos*	WA	LC	6	0.2	9.1
Little Ringed Plover	*Charadrius dubius*	WA	LC	6	0.2	10.3
Pintail / Swinhoe's Snipe	*Gallinago stenura/megala*	WA	LC	6	0.2	0
Black-thighed Falconet	*Microhierax fringillarius*	RA	LC	5	0.2	8.9
Chestnut-backed Scimitar-babbler	*Pomatorhinus montanus*	WL	LC	5	0.2	5.8
Chestnut-winged Cuckoo	*Clamator coromandus*	WL	LC	5	0.2	57.1
Christmas Frigatebird	*Fregata andrewsi*	WA	CR	5	0.1	2
Copper-throated Sunbird	*Leptocoma calcostetha*	WL	LC	5	0.2	17.6
Crimson-winged Woodpecker	*Picus puniceus*	WL	LC	5	0.2	10.3
Greater Painted-snipe	*Rostratula benghalensis*	WA	LC	5	0.2	15.8
Grey Plover	*Pluvialis squatarola*	WA	LC	5	0.1	7.4
Javan Fulvetta	*Alcippe pyrrhoptera*	WL	LC	5	0.2	4.3
Lesser Frigatebird	*Fregata ariel*	WA	LC	5	0.2	4.2
Ruby-cheeked Sunbird	*Chalcoparia singalensis*	WL	LC	5	0.2	8.1
Snowy-browed Flycatcher	*Ficedula hyperythra*	WL	LC	5	0.2	6.2
Sunda Minivet	*Pericrocotus miniatus*	WL	LC	5	0.1	1.4
Wandering Whistling-duck	*Dendrocygna arcuata*	WA	LC	5	0.2	11.8
Wreathed Hornbill	*Rhyticeros undulatus*	WL	VU	5	0.2	2.9
Asian Blue Quail	*Synoicus chinensis*	SC	LC	4	0.1	11.8
Black-headed Ibis	*Threskiornis melanocephalus*	WA	NT	4	0.1	0
Brown-throated Barbet	*Psilopogon corvinus*	WL	LC	4	0.2	14.3
Chestnut-fronted Shrike-babbler	*Pteruthius aenobarbus*	WL	LC	4	0.1	6.2
Common Snipe	*Gallinago gallinago*	WA	LC	4	0.2	200
Crested Goshawk	*Accipiter trivirgatus*	RA	LC	4	0.1	2.6
Dark-backed Imperial-pigeon	*Ducula lacernulata*	WL	LC	4	0.2	2
Javan Grey-throated White-eye	*Heleia javanica*	WL	LC	4	0.2	1.3
Javan Leafbird	*Chloropsis cochinchinensis*	WL	EN	4	0.1	2.7
Javan White-eye	*Zosterops flavus*	WL	EN	4	0.1	3.6
Kentish Plover	*Charadrius alexandrinus*	WA	LC	4	0.2	13.8
Laced Woodpecker	*Picus vittatus*	WL	LC	4	0.2	14.3
Large Cuckooshrike	*Coracina javensis*	WL	LC	4	0.2	3.3
Long-tailed Parakeet	*Psittacula longicauda*	WL	VU	4	0.1	33.3
Long-toed Stint	*Calidris subminuta*	WA	LC	4	0.2	10
Orange-breasted Trogon	*Harpactes oreskios*	WL	LC	4	0.2	3
Pied Imperial-pigeon	*Ducula bicolor*	WL	LC	4	0.2	9.4
Pygmy Tit	*Psaltria exilis*	WL	LC	4	0.2	6.2
Rufous Piculet	*Sasia abnormis*	WL	LC	4	0.1	8.3
White-crowned Forktail	*Enicurus leschenaulti*	WL	LC	4	0.2	4.5

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Aberrant Bush-warbler	*Horornis flavolivaceus*	WL	LC	3	0.1	2.9
Barred Cuckoo-dove	*Macropygia unchall*	WL	LC	3	0.1	1.8
Barred Dove	*Geopelia maugeus*	SC	LC	3	0.1	37.5
Baya Weaver	*Ploceus philippinus*	OP	LC	3	0	4.3
Blue-and-white Flycatcher	*Cyanoptila cyanomelana*	WL	LC	3	0.1	0
Chestnut-capped Babbler	*Timalia pileata*	WL	LC	3	0.1	13
Far Eastern Curlew	*Numenius madagascariensis*	WA	EN	3	0.1	21.4
Green Imperial-pigeon	*Ducula aenea*	WL	LC	3	0.1	1.9
Green Peafowl	*Pavo muticus*	OP	EN	3	0.1	4
Grey-cheeked Tit-babbler	*Mixornis flavicollis*	WL	LC	3	0.1	0
Grey-tailed Tattler	*Tringa brevipes*	WA	NT	3	0.1	0
Hill Blue-flycatcher	*Cyornis banyumas*	WL	LC	3	0.1	3.6
Javan Blue-banded Kingfisher	*Alcedo euryzona*	WL	CR	3	0	0
Javan Yellownape	*Chrysophlegma mentale*	WL	NT	3	0.1	0
Large-billed Crow	*Corvus macrorhynchos*	WL	LC	3	0.1	2.6
Pied Bushchat	*Saxicola caprata*	SC	LC	3	0.1	5.3
Pied Shrike-babbler	*Pteruthius flaviscapis*	WL	LC	3	0.1	1.7
Pink-headed Imperial-pigeon	*Ducula rosacea*	WL	NT	3	0.1	0
Sanderling	*Calidris alba*	WA	LC	3	0.1	7.7
Slaty-breasted Rail	*Lewinia striata*	WA	LC	3	0.1	8.8
White-bibbed Babbler	*Stachyris thoracica*	WL	LC	3	0.1	0
White-breasted Kingfisher	*Halcyon smyrnensis*	WA	LC	3	0	8.3
Asian Golden Weaver	*Ploceus hypoxanthus*	OP	NT	2	0.1	7.1
Black-browed Reed-warbler	*Acrocephalus bistrigiceps*	OP	LC	2	0.1	0
Buffy Fish-owl	*Ketupa ketupu*	RA	LC	2	0.1	7.7
Chestnut-bellied Partridge	*Arborophila javanica*	WL	LC	2	0.1	2.9
Crimson-breasted Flowerpecker	*Prionochilus percussus*	WL	LC	2	0.1	3.1
Fulvous-chested Jungle-flycatcher	*Cyornis olivaceus*	WL	LC	2	0.1	2.7
Great Slaty Woodpecker	*Mulleripicus pulverulentus*	WL	VU	2	0	16.7
Grey-rumped Myna	*Acridotheres tertius*	SC	CR	2	0	33.3
Horsfield's Frogmouth	*Batrachostomus javensis*	WL	LC	2	0.1	2.8
Japanese Sparrowhawk	*Accipiter gularis*	RA	LC	2	0.1	1.3
Javan Tesia	*Tesia superciliaris*	WL	LC	2	0	2.5
Javan Trogon	*Apalharpactes reinwardtii*	WL	VU	2	0.1	0
Lesser Crested Tern	*Thalasseus bengalensis*	WA	LC	2	0.1	5.3
Mountain White-eye	*Zosterops japonicus*	WL	LC	2	0.1	1.4
Mugimaki Flycatcher	*Ficedula mugimaki*	WL	LC	2	0.1	7.1
Orange-breasted Green-pigeon	*Treron bicinctus*	WL	LC	2	0.1	2.4
Orange-headed Thrush	*Geokichla citrina*	WL	LC	2	0.1	18.2
Oriental Dollarbird	*Eurystomus orientalis*	WL	LC	2	0.1	2.3
Oriental Paradise-flycatcher	*Terpsiphone affinis*	WL	LC	2	0	0

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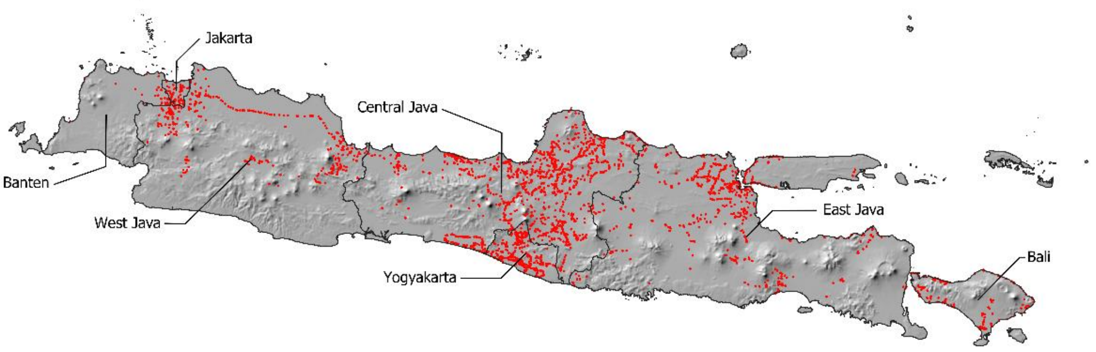
Pink-headed Fruit-dove	*Ptilinopus porphyreus*	WL	LC	2	0.1	3.3
Plain Flowerpecker	*Dicaeum minullum*	WL	LC	2	0.1	2.6
Rhinoceros Hornbill	*Buceros rhinoceros*	WL	VU	2	0.1	9.1
Rufous Woodpecker	*Micropternus brachyurus*	WL	LC	2	0.1	22.2
Spotted Crocias	*Laniellus albonotatus*	WL	NT	2	0	0
Stork-billed Kingfisher	*Pelargopsis capensis*	WA	LC	2	0.1	6.9
Sunda Cuckoo	*Cuculus lepidus*	WL	LC	2	0.1	2
Sunda Grasshopper-warbler	*Locustella montis*	SC	LC	2	0.1	0
Thick-billed Flowerpecker	*Dicaeum agile*	WL	LC	2	0.1	40
Tiger Shrike	*Lanius tigrinus*	SC	LC	2	0.1	11.8
Western Koel	*Eudynamys scolopaceus*	WL	LC	2	0.1	2.4
White-bellied Fantail	*Rhipidura euryura*	WL	LC	2	0.1	3.4
White-flanked Sunbird	*Aethopyga eximia*	WL	LC	2	0.1	1.9
Asian Dowitcher	*Limnodromus semipalmatus*	WA	NT	1	0	33.3
Baillon's Crake	*Zapornia pusilla*	WA	LC	1	0	25
Barred Eagle-owl	*Bubo sumatranus*	RA	LC	1	0	4.5
Black-faced Munia	*Lonchura molucca*	OP	LC	1	0	0
Black-tailed Godwit	*Limosa limosa*	WA	NT	1	0	0
Brown Boobook	*Ninox scutulata*	RA	LC	1	0	9.1
Brown Booby	*Sula leucogaster*	WA	LC	1	0	2.6
Chestnut-capped Thrush	*Geokichla interpres*	WL	NT	1	0	0
Crow-billed Drongo	*Dicrurus annectens*	WL	LC	1	0	10
Curlew Sandpiper	*Calidris ferruginea*	WA	NT	1	0	5
Dark-sided Flycatcher	*Muscicapa sibirica*	WL	LC	1	0	5
Great Knot	*Calidris tenuirostris*	WA	EN	1	0	9.1
Greater Green Leafbird	*Chloropsis sonnerati*	WL	EN	1	0	4.5
Hair-crested Drongo	*Dicrurus hottentottus*	WL	LC	1	0	0
Horsfield's Bronze-cuckoo	*Chalcites basalis*	WL	LC	1	0	2.6
Indian Cuckoo	*Cuculus micropterus*	WL	LC	1	0	20
Island Thrush	*Turdus poliocephalus*	WL	LC	1	0	4.2
Javan Cochoa	*Cochoa azurea*	WL	VU	1	0	5.6
Javan Owlet	*Glaucidium castanopterum*	RA	LC	1	0	2.9
Large Wren-babbler	*Turdinus macrodactylus*	WL	NT	1	0	0
Lesser Cuckooshrike	*Lalage fimbriata*	WL	LC	1	0	3.2
Little Curlew	*Numenius minutus*	WA	LC	1	0	20
Malay Night-heron	*Gorsachius melanolophus*	WA	LC	1	0	25
Malay Plover	*Charadrius peronii*	WA	NT	1	0	0
Mangrove Blue-flycatcher	*Cyornis rufigastra*	WL	LC	1	0	0
Narcissus Flycatcher	*Ficedula narcissina*	WL	LC	1	0	20
Pale Blue-flycatcher	*Cyornis unicolor*	WL	LC	1	0	0
Red-legged Crake	*Rallina fasciata*	WA	LC	1	0	8.3
Red Knot	*Calidris canutus*	WA	NT	1	0	12.5
Salvadori's Nightjar	*Caprimulgus pulchellus*	WL	NT	1	0	9.1
Sharp-tailed Sandpiper	*Calidris acuminata*	WA	LC	1	0	9.1

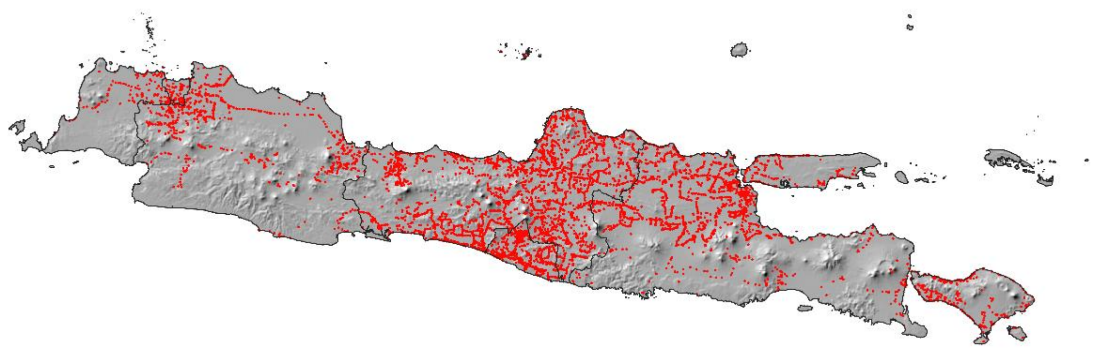
Common name	**Scientific name**	**Functional group**	**Red List status**	**Records**	**Surveyed squares present (%)**	**Increase in coverage (%)**
Siberian Blue Robin	*Larvivora cyane*	WL	LC	1	0	0
Siberian Thrush	*Geokichla sibirica*	WL	LC	1	0	5.6
Tawny-breasted Parrotfinch	*Erythrura hyperythra*	OP	LC	1	0	9.1
Temminck's Babbler	*Trichastoma pyrrogenys*	WL	LC	1	0	0
White's Thrush	*Zoothera aurea*	WL	LC	1	0	7.7
White-bellied Woodpecker	*Dryocopus javensis*	WL	LC	1	0	0
White-faced Partridge	*Arborophila orientalis*	WL	VU	1	0	0
White-tailed Tropicbird	*Phaethon lepturus*	WA	LC	1	0	0
White-throated Needletail	*Hirundapus caudacutus*	AF	LC	1	0	7.7

Appendix 2.5 Weekly accumulation of data during BigMonth2020. Each point represents the location of a bird list submitted to Burungnesia during January 2020.

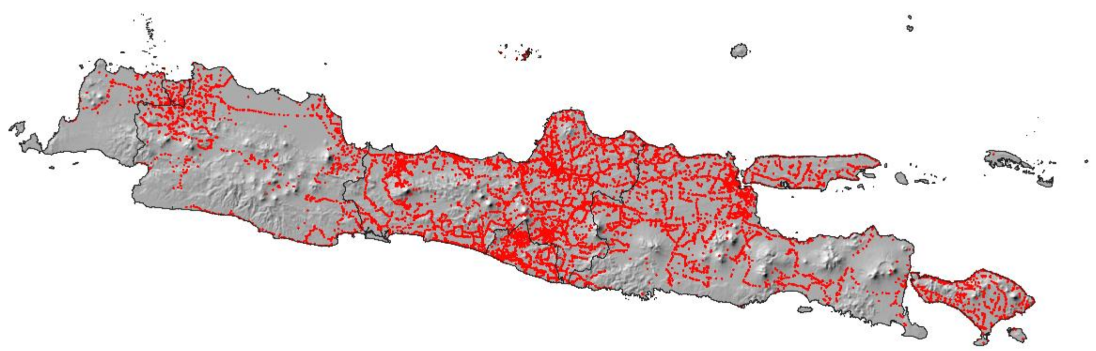
Week 1. 1-8 January 2020



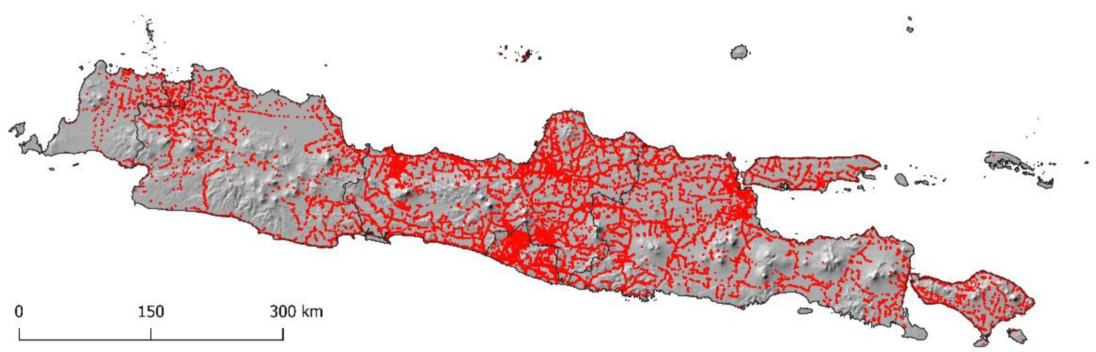
Week 2. 9-16 January 2020



Week 3. 17-24 January 2020

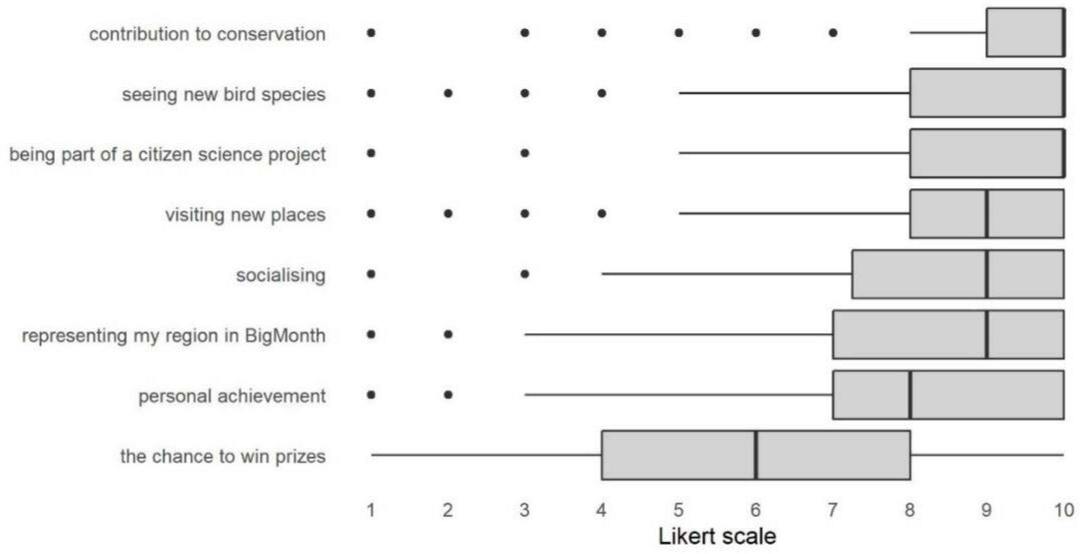


Week 4. 25-31 January 2020

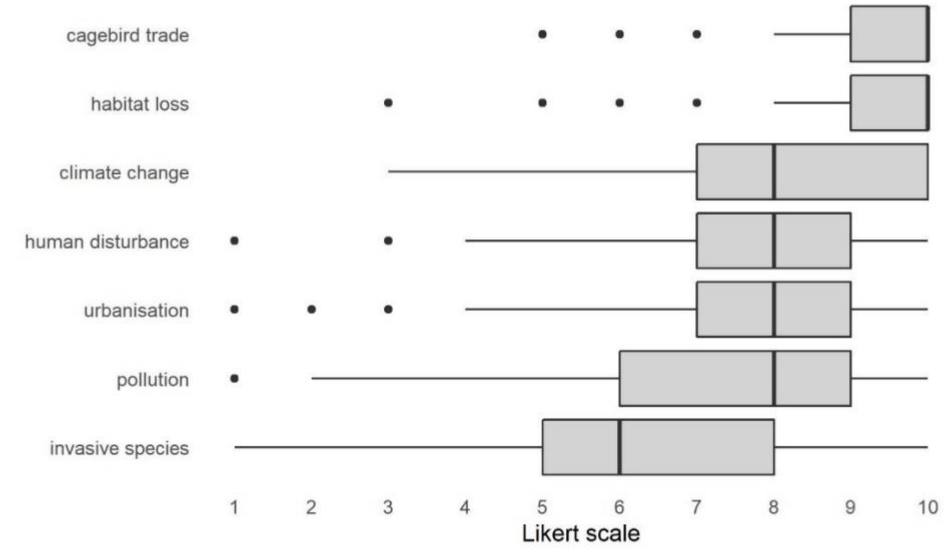


Appendix 2.6 Questionnaire responses from BigMonth2020 participants on a Likert scale of importance from 1 = not important to 10 = very important. These two questions were asked to understand why participants took part in BigMonth2020 and what they consider the most important threats facing Javan birds are. The box represents the interquartile range, the line in the box is the median, the whiskers show the minimum and maximum values, and outliers are shown as points.

(a) How important were the following factors to your participation in BigMonth2020?



(b) How important do you consider the following threats to be to Javan birds?



Appendix 2.7 Participant analysis GLM parameters

Modelling sampling effort (grid squares visited) as a function of participant characteristics

The number of grid squares visited and bird lists recorded were significantly correlated ($r_s = 0.97$, $p < 0.001$), so the number of grid squares visited alone was used as the dependent variable representing sampling effort for BigMonth2020. The dataset consisted of 134 observations. A negative binomial GLM was fitted using package 'MASS' (Venables & Ripley 2002) in R (R Core Team 2020). The independent variables included in the model were age, birdwatching experience, gender, bird club membership, and occupation with three levels (freelance, student, formal contract).

Appendix Table 2.7.1 Negative binomial generalised linear model parameters.

	Coefficient	SE	z value	Pr (>|z|)
Intercept	2.538	0.397	6.394	<0.01 **
Age	0.235	0.174	1.352	0.18
Birdwatching experience	0.305	0.126	2.421	0.02 *
Gender: male	0.699	0.283	2.471	0.01 *
Bird club membership: true	0.305	0.290	1.053	0.29
Occupation reference: formal employment				
Occupation: freelance work	–0.111	0.313	–0.355	0.73
Occupation: student	0.168	0.412	0.407	0.68

* $p < 0.05$, ** $p < 0.01$

Modelling rarity recording as a function of participants' demographic attributes and survey effort

To model rarity recording, we excluded 35 observations from the global dataset of 134 observations because those participants submitted fewer than 30 records to Burungnesia during BigMonth2020, which would have introduced extreme rarity recording metric values into the dataset. We fitted a Gaussian generalised linear model with the rarity recording metric as the dependent variable (August et al. 2020), and the independent variables included were age, birdwatching experience, gender, bird club membership, occupation (with three levels), and number of grid squares visited during BigMonth2020. The analysis was conducted in R (R Core Team 2020).

Appendix Table 2.7.2 Rarity recording generalised linear model parameters.

	Coefficient	SE	t value	Pr (>|t|)
Intercept	2.945	1.290	2.283	0.02 *
Age	1.014	0.566	1.792	0.08
Birdwatching experience	–0.077	0.414	–0.187	0.85
Gender: male	0.535	0.923	0.579	0.56
Bird club membership: true)	–0.441	0.943	–0.467	0.64
Occupation reference: formal employment				
Occupation: freelance work	0.184	1.023	0.180	0.86
Occupation: student	0.053	1.342	0.040	0.97
Grid squares visited	–0.019	0.007	–2.842	<0.01 **

* $p < 0.05$, ** $p < 0.01$

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Venables, W. N. and Ripley, B. D. (2002) *Modern Applied Statistics with S*. 4th ed., New York, USA: Springer.

Appendix 2.8 Contributors to BigMonth2020 data collection

We are grateful to the following people for their contribution to BigMonth2020:
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Chapter 3

3 Modelling the current distributions of Java's lowland birds: the role of trade, habitat, and protected areas

Abstract

Scientific knowledge of the forms and functions of Earth's biodiversity remains extremely limited, at a time when biodiversity faces unprecedented threats from human activities. Species conservation measures will need to be prioritised as long as resources for conservation are limited, a process typically achieved using species-specific range- and population-based metrics which require distribution and abundance data to be calculated. Across Java, Indonesia, the principal threats to wild bird populations are trapping to supply the cage-bird trade and habitat loss. I used citizen science datasets for the region to generate the first species distribution models for a suite of Java's lowland birds that face varying levels of these threats. The current distribution for most species was patchy and often significantly smaller than the extent of occurrence. Among the environmental variables used in modelling, land-cover-based predictors were the most important in the models for most species. The lack of convergence between the current distribution of the modelled species with Java's formally protected areas suggests that future conservation for these and other lowland birds, which are likely to come under increasing anthropogenic pressure, will need to involve other effective community-based conservation measures. My findings highlight the urgent need to promote continued citizen science efforts across Java, and indeed elsewhere in data-poor yet biodiverse regions, and to commission further survey work to discover areas of high biodiversity value.

3.1 Introduction

Scientific knowledge of the forms and functions of Earth's biodiversity remains extremely limited (Whittaker et al., 2005; Mora et al., 2008; Joppa et al., 2011; Scheffers et al., 2012), especially in the tropics (Collen et al., 2008). Even for birds (class Aves), the most well-studied taxa, there are many challenges facing conservation biology, particularly in biodiverse yet data-poor regions (Fisher and Christopher, 2007; Richardson and Whittaker, 2010). In the face of severe and urgent threats to biodiversity

(Pimm et al., 2014), it is widely judged that conservation measures need to be prioritised due to the limited resources available (Wilson et al., 2007), although the extent of such limitations is disputed (Wiedenfeld et al., 2021). The IUCN Red List is often used by concerned managers as a guide to the prioritisation of species conservation, as it evaluates overall extinction risk based on species-specific metrics including the extent of occurrence, area of occupancy, and both size and speed of population trajectory (Mace et al., 2008; IUCN Standards and Petitions Working Group, 2022). For a large number of species, the general lack of data precludes our ability to estimate these metrics reliably (Dickinson et al., 2010). However, the proliferation of citizen science over recent decades has helped to begin to close the gaps in biodiversity data (Fontaine et al., 2021). By 2021, the eBird database (Sullivan et al., 2014) had accrued one billion bird records, while the iNaturalist database (2022) now comprises more than 100 million records covering all taxonomic groups. These large and growing datasets are available for use in species distribution modelling (SDM), a suite of methods developed to enable the use of distribution data to predict the probability of occurrence in non-sampled areas by relating recorded presences to abiotic factors thought to drive the species' distribution (Elith and Leathwick, 2009).

Indonesia is one of the most biodiverse countries on Earth (Myers et al., 2000) but also has among the highest number of threatened species (Duckworth et al., 2012; IUCN, 2022) and the highest number of globally threatened birds (BirdLife International, 2022). Among the main human activities threatening biodiversity across the archipelago are habitat loss (Fitzherbert et al., 2008; Hughes, 2017, 2018) and the direct exploitation of wildlife for trade (Nash, 1993; Jepson and Ladle, 2005; Harrison et al., 2016; Symes et al., 2018b) and food (Harris et al., 2017). The scale of illegal trapping of wild birds to supply the cage-bird trade has led to an 'Asian Songbird Crisis' (Sykes, 2017), which Indonesia—its most populous island Java in particular—plays by far the greatest part in driving, for various socio-economic and cultural reasons (Jepson, 2010; Sykes, 2017). Millions of birds are trapped in the wild annually across the country to be sold mostly in domestic markets (Chng et al., 2015, 2016, 2018; Nijman et al., 2017) to supply a seemingly insatiable demand for cage-birds (Marshall et al., 2020). This pervasive threat facing Java's avifauna has already led to the possible extinction in the wild of the Javan Pied Starling (*Gracupica jalla*), a formerly widespread species (van Balen and Collar, 2021), and has resulted in serious declines in other species (Eaton et al., 2015).

Despite Java being heavily populated, surprisingly little remains known about the distribution of many of its bird species. Data hotspots including popular birdwatching sites and areas well-visited by tourists and bird tours (Squires et al., 2021) tend to be in the remaining good-quality forests at high altitudes, where there is high diversity and most endemic species are found. Most of Java's lowlands, which include large open agricultural areas, plantation woodlands, and peri-urban areas, receive little attention in comparison. It is important that our knowledge of these areas improves, as formerly common lowland birds across Java are disappearing from their natural habitat, presumably in large part due to the cage-bird trade (Collar et al., 2012; Bergin et al., 2018). Recent citizen science efforts on Java, particularly the BigMonth2020 event (Squires et al., 2021), have augmented the existing datasets available from eBird and Burungnesia (Winnasis et al., 2018). Here, I combine these citizen science datasets to generate SDMs for a suite of Java's lowland species suspected to be impacted by trade and habitat loss to varying degrees. First, I assess whether it is possible to produce robust models for the study species with the currently available citizen science data. I go on to examine the potential impact of bird ownership on distribution patterns, utilising bird ownership data from a recent household survey by Marshall et al. (2020). Finally, I measure the convergence of predicted current distributions with previous distribution maps and the network of formally protected areas, and by combining the modelled ranges of all study species I identify potential priority areas where predicted modelled species richness is high and compare this area with protected area coverage.

3.2 Methods

3.2.1 Study area

The modelled area included the islands of Java and Madura, Indonesia, matching the study area adopted by Marshall et al. (2020) to measure levels of bird ownership. Several small groups of satellite islands that fall under Java's administration (e.g. Karimunjawa and Pulau Ribu) were excluded from the modelling area because they are not connected biogeographically with the main island and different small island ecological processes are likely to be driving species distributions on them (Whitten et al., 1996).

3.2.2 Study species

Twenty-three species were selected for inclusion in this study (Table 3.1) because they predominantly inhabit the lowlands of Java (elevation 0–1000 m following Margono et al., 2014); were recorded during the BigMonth2020 citizen science event (Squires et al., 2021); represent a broad spectrum of the bird families affected by trade in Java; and are impacted to varying degrees by the cage-bird trade, from species traded in low volumes with little apparent effect on wild populations (e.g. Black-winged Flycatcher-shrike *Hemipus hirundinaceus*), to those traded in high volumes or undergoing a serious population decline apparently owing to trapping (e.g. Brown-cheeked Bulbul *Alophoixus bres*) (BirdLife International, 2022). Here, a cage-bird is considered a bird kept or sold as a pet or for use in songbird competitions in either households or markets (Chng et al., 2015). As the cage-bird trade on Java primarily involves songbirds (Passeriformes), all species selected were passerines except for Coppersmith Barbet (*Psilopogon haemacephalus*) and Yellow-eared Barbet (*P. australis*), which are members of the Megalaimadae (Piciformes) but have been considered part of the cage-bird trade by other studies in the region (Marshall et al., 2020) and are trapped, sold and kept in the same way as songbirds (Chng et al., 2015). Species were ranked according to the threat that trapping poses to their wild populations based on (1) the volume of trade in the species, the latest market survey data for Java and Bali from Chng et al. (2015, 2016, 2018) and Chng and Eaton (2016); (2) their presence on the Asian Songbird Trade Specialist Group’s (ASTSG) priority taxa list (ASTSG, 2022), although note that this list excludes taxa threatened by trade and considered globally threatened by the IUCN but which have introduced populations outside their native range (e.g. Java Sparrow *Lonchura oryzivora* and Javan Myna *Acridotheres javanicus*); (3) the impact trapping is having on the wild population as estimated by experts (BirdLife International, 2022); and (4) the level of ownership of the species across Java based on a recent estimates (Marshall et al., 2020). Although this ranking is somewhat subjective in nature, it should be noted that it was only intended for use as a guide to order the presentation of results and as a tool with which to observe possible patterns in the data.

Table 3.1 The current conservation status and habitat preference of the study species. Species were assigned a trapping threat rank according to whether they are heavily traded on Java and Bali using data from Chng et al. (2015, 2016, 2018) and Chng and Eaton (2016); their listing on the ASTSG's latest priority species list (ASTSG, 2022); the estimated impact of trapping on the species (BirdLife International, 2022); and levels of ownership reported by Marshall et al. (2020).

Species	**Red List status**	**ASTSG Priority Status**	**Heavily traded**	**Trapping threat[2]**	**Trapping threat rank**	**Habitat preference (species are labelled as forest (F) or non-forest (NF) species for grouping in results)**
Brown-cheeked Bulbul *Alophoixus bres*	EN	Tier 1	yes	high: 9	1	F—various forest types including degraded forest[1]
Sangkar White-eye *Zosterops melanurus*	VU	Tier 1	yes	med: 7	2	F—various forest and woodland[1]; parks and gardens[2]
Javan Myna *Acridotheres javanicus*	VU	-	yes	high: 8	3	NF—cultivated land, grassy areas, scrub and urban areas[2]
Oriental Magpie-robin *Copsychus saularis*	LC	Tier 1	yes	low: 4	4	NF—urban and rural areas, cultivation, disturbed forest[1]
Ruby-throated Bulbul *Rubigula dispar*	VU	Tier 2	yes	med: 7	5	F—open woodland, scrub, degraded areas, town edge[1]
Bar-winged Prinia *Prinia familiaris*	NT	Tier 2	yes	low: 5	6	NF—shrubby and wooded habitats[3]; disturbed areas[2]
Black-naped Oriole *Oriolus chinensis*	LC	-	yes	non-trivial	7	F—forest, edge and wooded cultivation[1]
Pied Triller *Lalage nigra*	LC	-	no	non-trivial	8	NF—open habitats, mangroves, coastal scrub[1]
Scarlet Minivet *Pericrocotus flammeus*	LC	-	no	non-trivial	9	F—various forest types including disturbed forest[3]
Yellow-eared Barbet *Psilopogon australis*	LC	-	no	non-trivial	10	F—forest, woodland, parkland and cultivation[1]
Sooty-headed Bulbul *Pycnonotus aurigaster*	LC	-	yes	n.r.	11	NF—open habitats including villages, cultivation and scrub[1]
Scaly-breasted Munia *Lonchura punctulata*	LC	-	yes	n.r.	12	NF—cultivation, gardens, open areas, scrub[1]
Long-tailed Shrike *Lanius schach*	LC	-	yes	n.r.	13	NF—edge, cultivated and open habitats[1]
Javan Banded Pitta *Hydrornis guajanus*	LC	-	no	n.r.	14	F—various forest types including plantations[3]
Common Iora *Aegithina tiphia*	LC	-	yes	n.r.	15	NF—open woodland, edge, scrub, mangroves, parkland[1]
Sunda Pied Fantail *Rhipidura javanica*	LC	-	no	n.r.	16	NF—open habitats, mangroves, scrub, gardens[1]
Yellow-vented Bulbul *Pycnonotus goiavier*	LC	-	yes	n.r.	17	NF—open habitats: parks, villages, scrub, forest clearings[1]
Plain Prinia *Prinia inornata*	LC	Tier 2	no	n.r.	18	NF—scrubby grasslands, cultivated areas, gardens[3]
Common Tailorbird *Orthotomus sutorius*	LC	-	no	n.r.	19	NF—shrubby areas, gardens, parks, edge, plantations[3]
Black-naped Monarch *Hypothymis azurea*	LC	-	no	n.r.	20	F—forest, edge, secondary growth and scrub[1]
Coppersmith Barbet *Psilopogon haemacephalus*	LC	-	no	n.r.	21	NF—open woodland, parks, plantations and urban areas[1]
Horsfield's Babbler *Malacocincla sepiaria*	LC	-	no	n.r.	22	F—broadleaf evergreen, logged and secondary forest[1]
Black-winged Flycatcher-shrike *Hemipus hirundinaceus*	LC	-	no	n.r.	23	F—forest, edge[1], plantations and wooded gardens[3]

n.r., none reported; [1] Eaton et al. (2021), [2] BirdLife International (2022), [3] Billerman et al. (2022); ASTSG Tier 1 species are urgent conservation priorities, while Tier 2 species are on a 'watch list' because they are present in trade but the impacts on wild populations are unclear and require more research.

3.2.3 Species occurrence data

Occurrence records for each study species (Table 3.2) were gathered from four sources: (a) BigMonth2020, a citizen science database generated during a month-long birdwatching event in Java and Bali (Squires et al., 2021); (b) Burungnesia, an Indonesian citizen science records database (Winnasis et al., 2018); (c) Atlas Burung Indonesia, a database of Indonesian citizen science records contributed by amateur Indonesian birdwatchers and used to create Indonesia’s first bird atlas (Taufiqurrahman et al., 2016; Winnasis et al., 2020); and (d) eBird, a global online database of citizen science records contributed by amateur birdwatchers (Sullivan et al., 2014). All records were adjusted so that taxonomy followed the Handbook of Birds of the World (HBW) and BirdLife International (2022a).

Occurrences from BigMonth2020 data were validated following the protocol described in the *Methods* section of Chapter 2 and Squires et al. (2021). Briefly, records were flagged for review by a team of six experts if they were deemed unusual in terms of location, time of year or habitat. Records were also manually verified if the GPS coordinates did not match the location description or if the habitat description did not match the habitat depicted on Google Earth. In total, 845 bird lists (3.8%) were flagged for review. Data from 494 (58.5%) lists were retained in the database following verification, 253 (29.9%) were retained with updated location or species data, and 98 (11.6%) were removed for lack of supporting evidence. Records included in the Burungnesia and Atlas Burung Indonesia databases undergo a similar validation process overseen by the same group of experts (Taufiqurrahman et al., 2016; Winnasis et al., 2018).

Records in the eBird database are assigned to a specific location irrespective of the distance covered or duration of the bird recording session. To avoid the inclusion of spatially inaccurate data, records from a trip with either a transect length greater than 10 km or duration greater than 6 h were excluded from the final dataset. When recording data using eBird, the user decides whether to create a custom location for their recording session or use a pre-existing eBird ‘hotspot’—a permanently marked location on the eBird map established for frequently visited birdwatching locations. Using hotspots makes recording easier and faster on eBird and allows users to compare their efforts at a particular site with those of other users. While the hotspots system is effective for small sites in Java, the low density of hotspots means that for some large areas, especially national parks, there is only one hotspot, which can lead to records being attributed to points far from the actual position of observation. The location of all eBird hotspots across Java was manually verified using Google Earth, and records attached to hotspots covering a particularly large area were removed (e.g. all records from the Bromo Tengger Semeru National Park hotspot, which is arbitrarily located at the centre of the park, which is the bare volcano summit).

Occurrences were validated by passing them through species-specific elevation and native extent of occurrence filters. The elevational range for each species was taken from BirdLife International (2022) or Eaton et al. (2021). The digitised range maps compiled by HBW and BirdLife International were used as the range filter (requested and downloaded from BirdLife International, 2022). Records that were obviously erroneous or most likely of captive origin (e.g., forest species recorded in an urban area) were removed from the dataset. All records of Brown-cheeked Bulbul, Oriental Magpie-robin (*Copsychus saularis*) and Ruby-throated Bulbul (*Rubigula dispar*) from urban areas were excluded because these species are unlikely to occur naturally in these areas and the number of individuals kept in captivity is high, meaning there is a strong chance that these records are from birds of captive origin. It is possible that the final modelled range for these species is therefore an underestimate owing to the possible exclusion of genuine urban occurrences. Following these filtering steps, the occurrences from all datasets were combined and spatial duplicates were removed, so that each 1-km cell across the study area contained only a single occurrence.

As the aim of this study was to model the current distribution of species, only recent occurrence records were included, although this was balanced with the need for sufficient data to produce robust models. SDM performance differs depending on the ecological range of the species (Papeş and Gaubert, 2007), and the minimum number of records needed to achieve satisfactory performance varies accordingly. Studies have reported that for sample sizes under 14–25 records performance declines drastically (Stockwell and Peterson, 2002; Papeş and Gaubert, 2007; Proosdij et al., 2016). Adopting a cautious approach, here all records in and since 2016 up to February 2022 were included, but for any species with fewer than 30 occurrences in that period, data from successive earlier years were admitted until 30 records were obtained (Brown-cheeked Bulbul), and otherwise all records going back to 2012 inclusive were admitted and the species was still included (Oriental Magpie-robin).

Table 3.2 Number of occurrence records used to model the distribution of the study species. The elevation range for each species was used during the record validation process. The final number of records for each species is lower than the sum of the 'After filtering' steps for each dataset.

Species	Elevation range (m)	Earliest record used	BigMonth2020		Burungnesia		Atlas Burung Indonesia		eBird		Final no. of records[*]
			Raw no. records	After filtering	Raw no. records	After filtering	Raw no. records	After filtering	Raw no. records	After filtering	
Sooty-headed Bulbul	0–1800[a]	2016	6,970	6,512	1,226	1,104	34	27	3,522	742	1,671
Scaly-breasted Munia	0–1800[a]	2016	6,109	5,823	471	451	22	19	1,592	389	1,430
Common Tailorbird	0–1500[a]	2016	1,534	1,453	402	373	14	11	595	189	838
Yellow-vented Bulbul	0–1900[a]	2016	1,271	1,158	479	441	21	17	1,679	331	720
Common Iora	0–1000[a]	2016	981	909	432	393	15	11	1,325	216	656
Plain Prinia	0–1500[a]	2016	703	658	228	218	5	4	430	118	497
Long-tailed Shrike	0–1600[a]	2016	308	303	284	260	14	13	757	141	345
Horsfield's Babbler	0–1700[b]	2016	164	146	313	273	7	5	1,009	131	237
Pied Triller	0–1000[a]	2016	200	190	101	96	1	-	312	90	213
Sunda Pied Fantail	0–1500[a]	2016	204	179	211	181	6	4	1,005	111	191
Coppersmith Barbet	0–2250[b]	2016	140	130	185	167	7	6	972	165	171
Javan Myna	-	2016	119	114	178	163	8	7	834	151	160
Black-winged Flycatcher-shrike	0–1500[a]	2016	74	66	189	161	4	2	1,088	111	151
Bar-winged Prinia	0–1500[a]	2016	127	124	59	54	2	1	817	45	129
Sangkar White-eye	100–2500[b]	2016	61	61	134	115	8	5	984	85	129
Yellow-eared Barbet	0–2000[b]	2016	31	29	135	126	1	1	251	68	102
Scarlet Minivet	0–1500[a]	2016	33	28	145	130	3	3	262	52	96
Javan Banded Pitta	0–1300[b]	2016	44	41	116	100	-	-	438	72	88
Ruby-throated Bulbul	0–1000[b]	2016	52	42	121	94	4	3	276	29	74
Black-naped Oriole	0–1200[a]	2016	35	31	94	82	7	4	611	53	61
Black-naped Monarch	0–1500[a]	2016	30	28	40	34	4	3	151	33	58
Brown-cheeked Bulbul	0–1500[b]	2014	9	3	69	33	13	2	455	18	31
Oriental Magpie-robin	0–1900[b]	2012	6	2	13	11	-	-	74	9	16

[a] Eaton et al. (2021), [b] BirdLife International (2022); [*] the final number of records for each species is lower than the sum of the 'After filtering' steps for each dataset because spatial duplicates were excluded after combining the occurrences from all datasets.

3.2.4 Predictors included in the model

The predictors used in the final models were based on land-cover data, digital elevation model (DEM) data and normalised difference vegetation index (NDVI) data. Land-cover data for Java were obtained from Gong et al. (2019) as single land-cover type rasters produced using 30-m resolution Landsat images from 2017. The land-cover rasters were resampled to a resolution of 1 km and all remaining land-cover classes (bare ground was lost during resampling due to its scarcity) were combined to give a categorical land-cover raster with forest, cultivation, grass and shrubland (grassland and shrubland rasters grouped), wetlands (wetland and inland water land covers grouped), and urban land-cover types. This land-cover raster was first used to produce a landscape-scale habitat diversity predictor by calculating the neighbourhood joint entropy within a circular window (2-km radius) around each raster cell using the ‘landscapemetrics’ package (Hesselbarth et al., 2019) in R (R Core Team, 2022). The land-cover rasters were then further processed to calculate the proportion of each land-cover type within a moving window (2-km radius) around the focal cell to produce final land-cover predictors. The urban land-cover class was not included in models as a predictor because it was strongly correlated with the bird ownership layer (see Section 3.2.6). An elevation predictor was calculated from the Shuttle Radar Topography Mission (SRTM) digital elevation model with 30-m resolution (van Zyl, 2001), resampled to 1-km resolution. NDVI predictors at 1-km resolution were obtained from the Dynamic Habitat Indices which were derived from 2015 MODIS data (Hobi et al., 2017). The final predictors used were the minimum NDVI and variance in NDVI over 23 time periods in 2015 (one image collected every 16 days).

3.2.5 Modelling methods

Occurrence data were spatially thinned to a distance of 5 km to reduce the potential spatial bias that could result from repeated sampling at certain sites and clustered records (Fourcade et al., 2014). Background data were generated by randomly sampling 5 sets of 10,000 pseudoabsences from a random sample of 50,000 points across the whole modelling area but excluding cells containing a record of occurrence (Barbet-Massin et al., 2012).

MaxEnt, a widely favoured SDM method that has been shown to have high performance (Valavi et al., 2022), was used to build species-specific distribution models in R (R Core Team, 2022). Model tuning was undertaken using seven combinations of features consisting of linear, product, quadratic and threshold transformations of predictors across six values of the regularisation parameter (from 1–6, in steps of 1) in the ‘maxnet’ package (Phillips et al., 2017). Model performance was evaluated with the ‘ENMeval’ package (Muscarella et al., 2014) using fourfold cross-validation in a double checkerboard pattern at scales of 5 and 10 km to prevent spatial autocorrelation inflating evaluation metrics. Four model performance metrics were calculated for each model: the area under the receiver operating characteristic curve (AUC), Tjur’s R^2 (Tjur, 2009), and the biserial correlation coefficient were calculated using the ‘dismo’ package (Hijmans et al., 2017), and Boyce’s index values were calculated using the ‘ecospat’ package (Di Cola et al., 2017). The best model for each pseudoabsence dataset was selected based on the highest AUC value, and this was then projected onto the environmental predictors to estimate the probability of occurrence for the species across the modelling area. A final predicted suitability raster was produced by averaging across the five models resulting from the five pseudoabsence runs and these were then converted to species-specific binary predictions of presence–absence by assuming the species was present in cells with suitability values greater than the 10th percentile of known presences (Pearson et al., 2007). Using this threshold gives a conservative range prediction for a species that is more likely to reflect the current area of occupancy. All modelling was conducted in R, and other packages used but not yet specified include ‘sf’ (Pebesma, 2018), ‘raster’ (Hijmans, 2020) and ‘terra’ (Hijmans, 2022).

3.2.6 Modelling bird ownership

Bird ownership data for households across Java were obtained from Marshall et al. (2020). The sampling unit in this dataset is the total number of birds kept in households within communities (*n* = 89) in rural and urban areas across Java’s six provinces. After removing non-native species from the dataset, all native species were grouped, and the mean number of individuals owned per household in each community was calculated. It was not possible to calculate species-specific ownership levels for the study species because some are missing from the ownership dataset, the sample sizes in general are small, and some taxa with similar plumages are grouped in the dataset to reduce the likelihood of mis-reporting which species are owned.

Three predictors were used to model bird ownership across Java, all of which were created from regency-level data from the 2010 Indonesia census conducted by the national Indonesian statistics authority (Badan Pusat Statistik, BPS) and downloaded from the Minnesota Population Center (2020). As keeping cage-birds is deeply rooted in Javanese culture (Jepson, 2010), a predictor of the density of people identifying as Javanese was created by multiplying the regency-level proportion of Javanese residents with 1-km resolution population density data downloaded from WorldPop (2020), with the assumption that Javanese people are evenly distributed within regencies. As differing levels of educational attainment and the varying distributions of people between urban or rural areas may drive patterns of bird ownership (Jepson and Ladle, 2009), these two predictors were also created from the census data. BPS differentiates urban and rural areas using a composite score across factors including population density, infrastructure availability and the number of households working in agriculture (Badan Pusat Statistik, 2010).

Modelling of bird ownership was undertaken in R. First, correlations among predictors were checked by calculating the variance inflation factor (VIF) using the 'usdm' package (Naimi et al., 2014). No predictors had a VIF greater than 5 and all were retained in the model. A quasi-Poisson generalised linear model was constructed with the number of birds per household included as the response variable and the density of Javanese people, proportion of people educated to secondary school level, and proportion of people living in urban areas included as the predictors. The model was projected onto the predictors over the modelling area and the rate of birds owned per household was converted to a density by multiplying it with the 1-km resolution population density raster to give a final density of birds owned per km^2 (Figure 3.1). From this, a binary raster of high–low bird ownership was generated, with high levels of bird ownership considered equal to or greater than the value at the 90th percentile. The distance from every cell in the model area to the nearest area with high bird ownership was then calculated (Figure 3.2).

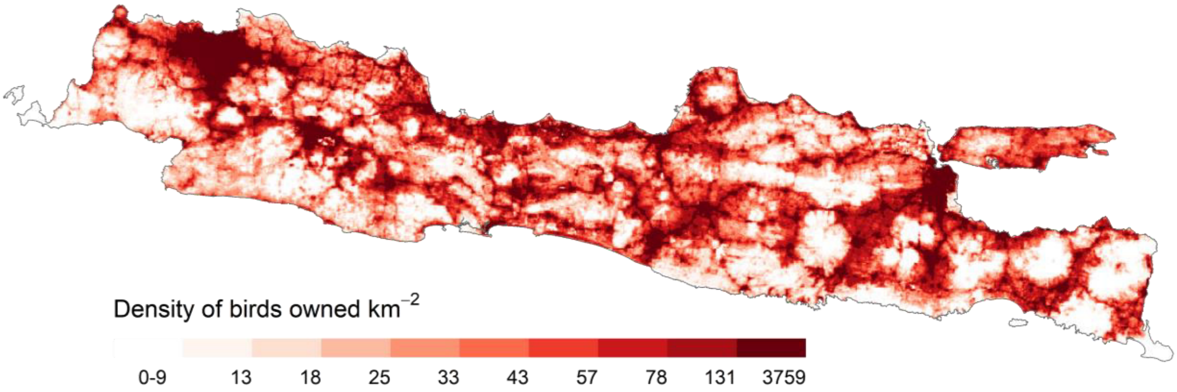


Figure 3.1 Density of bird ownership (1-km resolution) across Java. Colours on the plot are assigned on a quantile scale (*n* = 10) to account for the small volume of data with high values of bird ownership, which were primarily distributed in the urban centres of Jakarta and Surabaya in the west and east of Java, respectively.

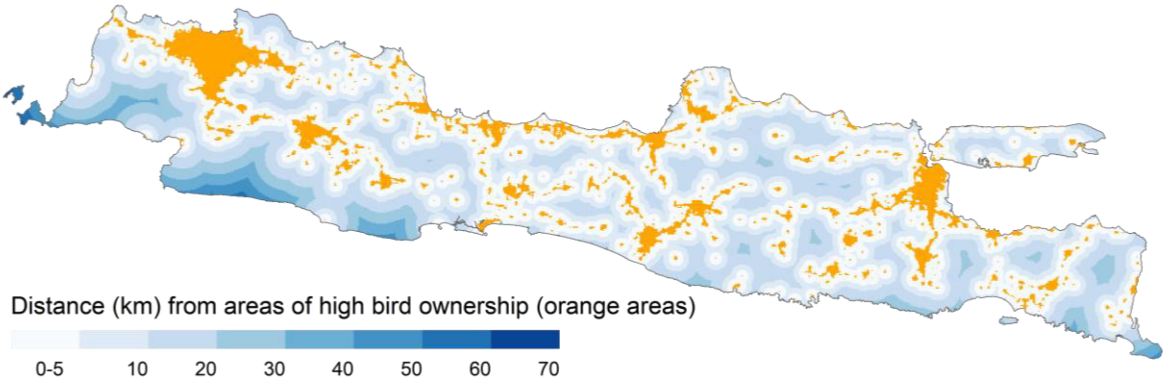


Figure 3.2 The distance from each 1-km cell across Java to the nearest area of high bird ownership (orange areas).

To assess whether any pattern of bird ownership impact could be detected in the results, a random sample of 1,000 points across the modelling areas was taken for each species, with 500 from locations within the distribution as defined by HBW and BirdLife International (best described as the extent of occurrence for the species) but outside the modelled range (most similar to the area of occupancy for the species), and 500 where the HBW and BirdLife International and modelled ranges overlapped. Grouped boxplots are presented to show whether there are any differences between these two samples in terms of their proximity to areas of high bird ownership, and therefore presumed higher trapping pressure.

3.2.7 Priority areas, range comparisons and protected area coverage

The modelled range for each species was compared with the distribution maps produced by HBW and BirdLife International (2022b), which are based on locality data, broad habitat preferences and elevational range of occurrence, and the protected area network across Java (UNEP-WCMC and IUCN, 2022). All terrestrial protected areas in the dataset were included (IUCN categories I–VI). To search for areas of potentially high importance, a map of binary high–low 'modelled species richness' (the number of species predicted present in each 1 km^2 cell of the modelled area) was generated by assuming high richness in cells with 70% of modelled species present (17 of the 23 modelled species). Large contiguous areas with high species richness outside the current protected area network were visually identified from the resulting map as potential priority areas.

3.3 Results

3.3.1 Model performance

The model performance for the species under study ranged from good (mean AUC = 0.86 for Brown-cheeked Bulbul) to relatively poor (mean AUC = 0.61 for Long-tailed Shrike) (Table 3.3). Of the four evaluation metrics used, AUC and Tjur's r^2 values exhibited similar results ($r = 0.97$, df = 21, $p < 0.001$), as did values of biserial correlation and the Boyce index ($r = 0.7$, df = 21, $p < 0.001$), while all other combinations of metrics were not closely related ($r < 0.50$).

The models for species with smaller predicted ranges tended to perform better (Figure 3.3). Two variables likely to be related to model performance—the predicted range size and the number of occurrence records used for modelling—were strongly correlated ($r = 0.75$, df = 21, $p < 0.001$). Univariate GAMs revealed a linear negative association between the percentage of the modelled area occupied and the mean AUC, while the number of occurrences used to produce the model had a weaker negative non-linear association with mean AUC (Figure 3.3).

Table 3.3 Performance of the top-ranked model (selected using highest AUC) for each species measured by the results of the four evaluation metrics being averaged over the four cross-validation folds. The tuning parameters used in each model (features used as transformations of predictors and the regularisation multiplier—Reg) are also presented.

Species	mean AUC (sd)	mean biserial correlation (sd)	mean Tjur's R2 (sd)	mean Boyce index (sd)	Features	Reg
Brown-cheeked Bulbul	0.86 (0.09)	0.07 (0.02)	0.33 (0.11)	0.55 (0.15)	qpt	2
Scarlet Minivet	0.81 (0.03)	0.12 (0.02)	0.35 (0.03)	0.80 (0.07)	l	5
Oriental Magpie-robin	0.80 (0.19)	0.04 (0.02)	0.24 (0.13)	0.81 (0.15)	l	1
Sunda Pied Fantail	0.79 (0.02)	0.19 (0.01)	0.29 (0.01)	0.84 (0.07)	qpt	2
Yellow-eared Barbet	0.79 (0.03)	0.10 (0.02)	0.30 (0.03)	0.76 (0.13)	lq	2
Sangkar White-eye	0.76 (0.01)	0.13 (0.03)	0.25 (0.05)	0.75 (0.10)	lqp	1
Javan Banded Pitta	0.75 (0.03)	0.08 (0.01)	0.24 (0.02)	0.77 (0.11)	qpt	1
Black-winged Flycatcher-shrike	0.74 (0.03)	0.10 (0.01)	0.22 (0.03)	0.84 (0.08)	lqp	1
Black-naped Oriole	0.73 (0.07)	0.07 (0.02)	0.19 (0.05)	0.68 (0.14)	l	4
Coppersmith Barbet	0.73 (0.03)	0.11 (0.02)	0.19 (0.03)	0.84 (0.07)	qpt	1
Horsfield's Babbler	0.72 (0.02)	0.11 (0.01)	0.18 (0.02)	0.92 (0.03)	lqp	6
Ruby-throated Bulbul	0.72 (0.03)	0.07 (0.01)	0.18 (0.02)	0.63 (0.10)	lqp	2
Javan Myna	0.71 (0.07)	0.11 (0.04)	0.18 (0.06)	0.75 (0.05)	l	3
Pied Triller	0.71 (0.01)	0.11 (0.02)	0.17 (0.02)	0.90 (0.03)	l	2
Scaly-breasted Munia	0.70 (0.01)	0.22 (0.01)	0.15 (0.01)	0.98 (0.01)	lqp	1
Plain Prinia	0.69 (0.02)	0.14 (0.02)	0.14 (0.01)	0.95 (0.04)	lq	1
Bar-winged Prinia	0.68 (0.02)	0.08 (0.01)	0.16 (0.01)	0.66 (0.09)	lqpt	1
Black-naped Monarch	0.68 (0.06)	0.05 (0.01)	0.15 (0.05)	0.44 (0.16)	lqpt	1
Yellow-vented Bulbul	0.67 (0.01)	0.15 (0.01)	0.11 (0.01)	0.90 (0.06)	lqp	1
Sooty-headed Bulbul	0.65 (0.01)	0.18 (0.02)	0.08 (0.01)	0.94 (0.05)	lqp	1
Common Iora	0.64 (0.02)	0.12 (0.01)	0.09 (0.01)	0.95 (0.02)	lqp	1
Common Tailorbird	0.64 (0.01)	0.12 (0.01)	0.08 (0.01)	0.88 (0.08)	lqpt	1
Long-tailed Shrike	0.61 (0.02)	0.07 (0.02)	0.05 (0.01)	0.79 (0.13)	lqp	5

Features used in the models as predictor transformations: (l) linear, (p) product, (q) quadratic, and (t) threshold

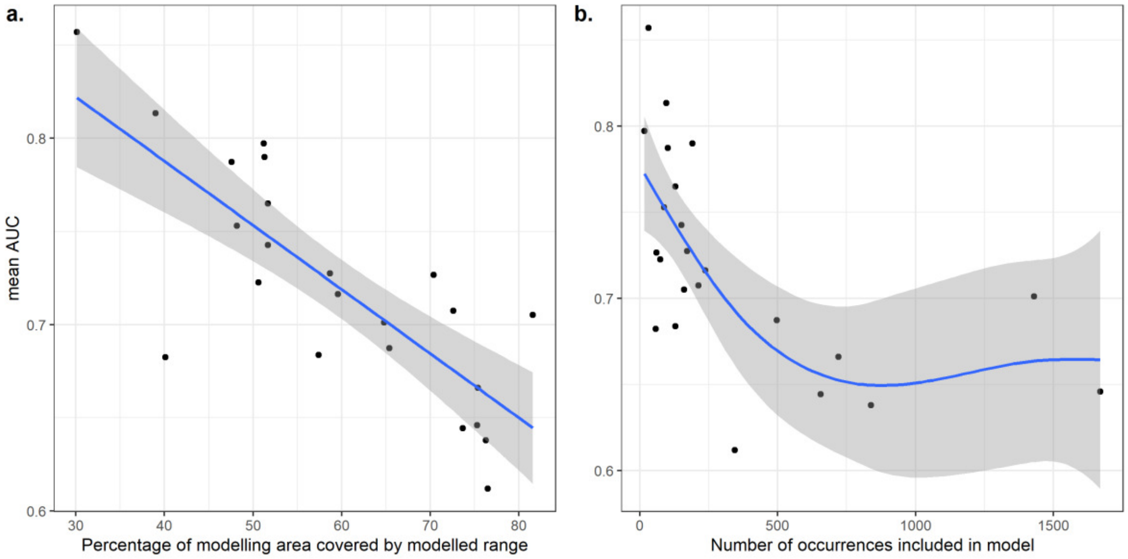


Figure 3.3 Model performance (mean AUC) in relation to **(a)** the area of Java predicted to be suitable for each species and **(b)** the number of occurrences included in the model.

The most important predictor for the forest species modelled was always the percentage of either cultivated land or forest (themselves highly correlated), and all other predictors had relatively low importance in the models (Figure 3.4). There was a greater diversity of responses among non-forest species to the predictors, but landscape-scale habitat diversity was most important for six (Common Iora *Aegithina tiphia*, Common Tailorbird *Orthotomus sutorius*, Javan Myna, Scaly-breasted Munia *Lonchura punctulata*, Sooty-headed Bulbul *Pycnonotus aurigaster* and Yellow-vented Bulbul *Pycnonotus goiavier*), with increasing values of habitat diversity leading to higher suitability values for all. Elevation- and NDVI-based predictors were relatively more important for non-forest birds than forest birds. The wetland and grass/shrubland land-cover predictors had generally low importance.

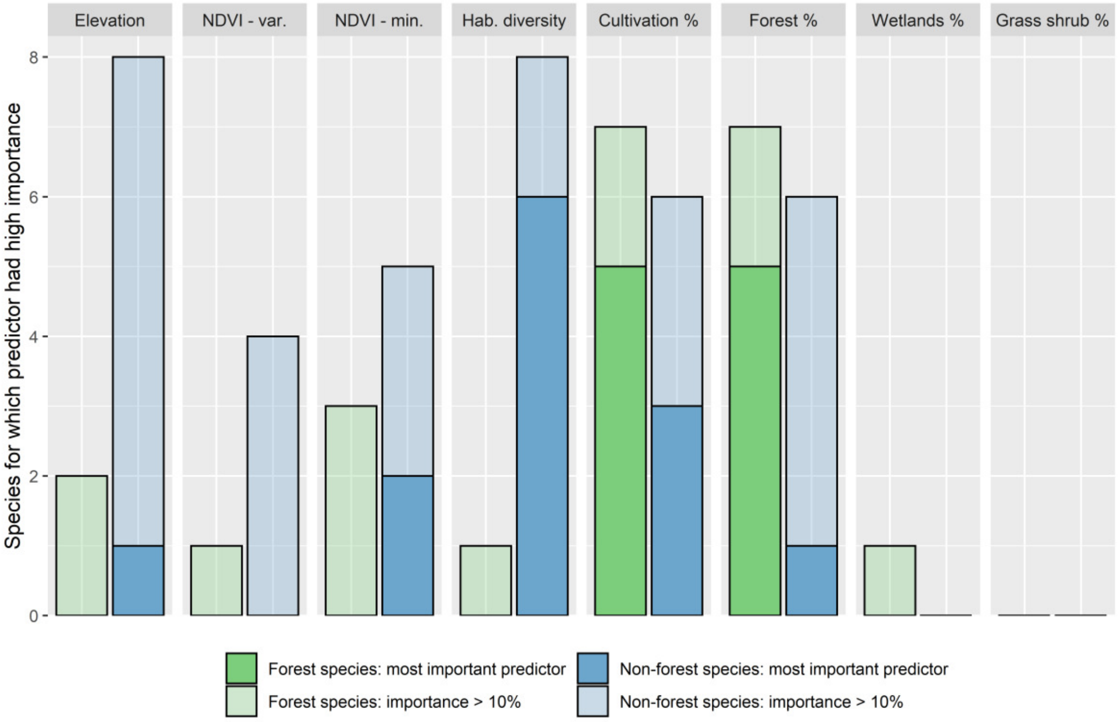


Figure 3.4 Variable importance for the predictors included in the final SDMs. The bars—grouped according to whether a species was considered a forest (green) or non-forest (blue) bird—present the number of species for which each predictor was 'important' (average contribution of >10% to the final model), and the number for which it was the most important predictor in the final model.

3.3.2 Modelled species distributions

In general, the areas of highest suitability for most forest birds were concentrated in the southern part of Java, particularly around the chain of volcanoes where most remaining forest occurs (Figure 3.6). The suitability of urban areas was high for two of the forest species (Sangkar White-eye *Zosterops melanurus* and Black-naped Oriole *Oriolus chinensis*). For most forest species there were strong differences between suitable and unsuitable areas, the latter predominantly being the large areas of cultivated land found across northern Java, urban areas, and high altitude zones.

The non-forest species exhibited a wider range of patterns of suitability across Java. Widespread generalists, such as Sooty-headed Bulbul, Common Iora and Common Tailorbird, had relatively high suitability values with little variation across the island. Species of open country, such as Scaly-breasted Munia, Pied Triller and Plain Prinia, inverted the suitability pattern of forest species, with higher values in the large cultivated areas of the north. Some species, particularly Javan Myna, Sunda Pied Fantail and Coppersmith Barbet, showed a preference for urban areas, which tend to be complex environments with high habitat diversity.

Compared to the range maps published by HBW and BirdLife International, the modelled ranges tended to be patchier and smaller than the BirdLife ranges, and this pattern is clearest for forest birds, although Oriental Magpie-robin is a notable exception among the non-forest complement (Figure 3.6). For most species the modelled range does not extend beyond the BirdLife range. Exceptions are Pied Triller, which shows a significant range extension to the east, and Horsfield's Babbler, whose modelled range is almost the inverse of the BirdLife range. The modelled and BirdLife ranges for Ruby-throated Bulbul, Yellow-eared Barbet and Sunda Pied Fantail are similar and are restricted to forest land cover.

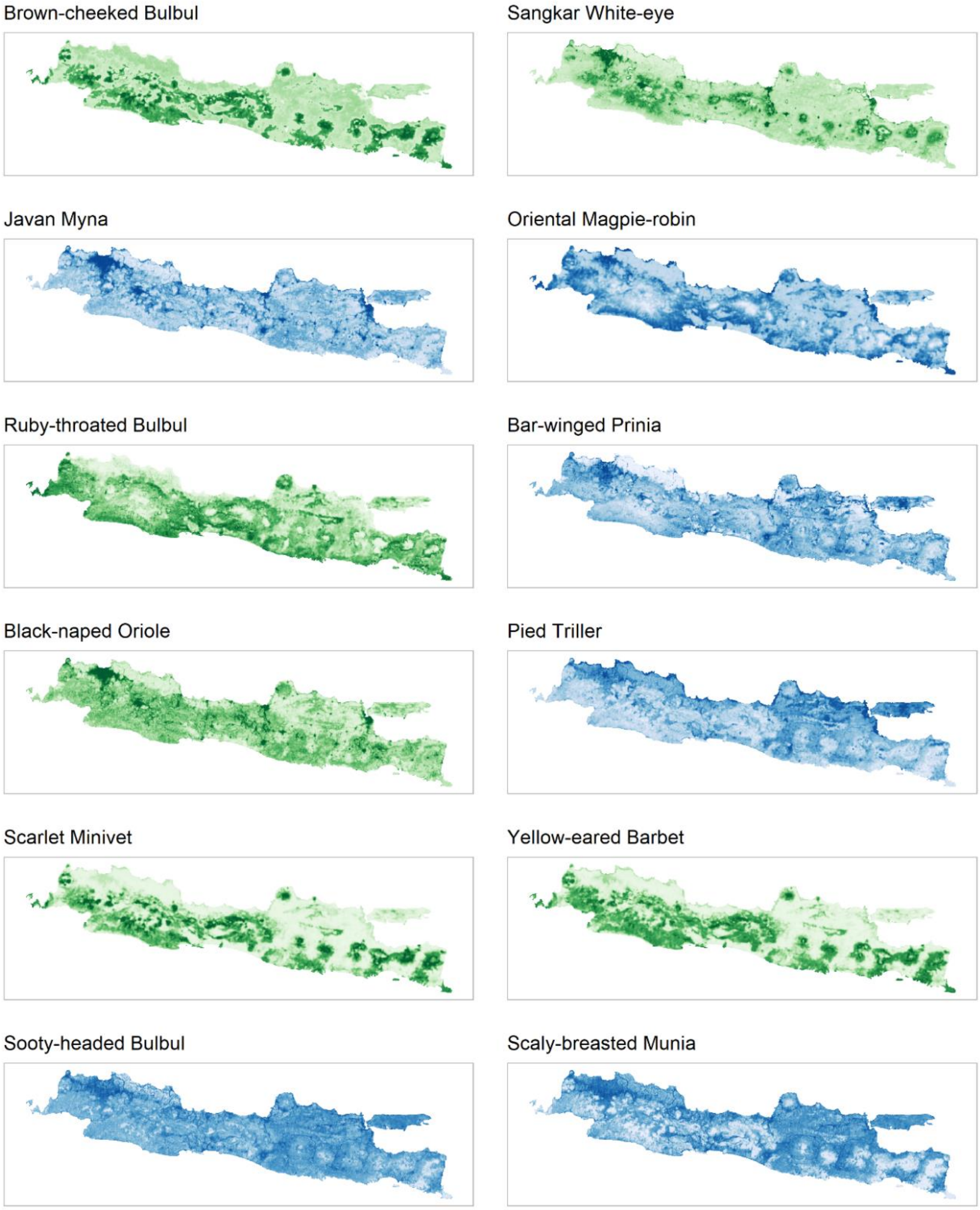
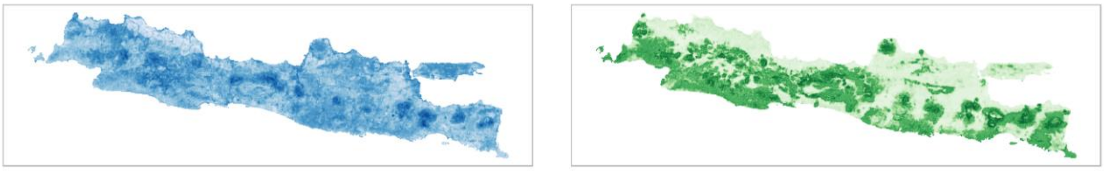
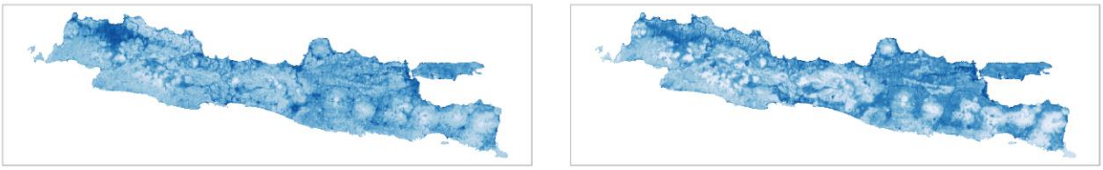
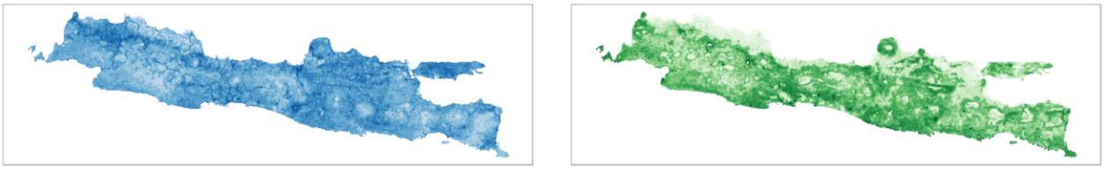


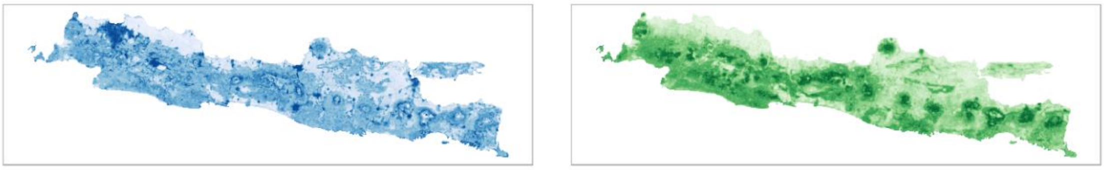
Figure 3.5a Page 1 of 2-page figure.


Long-tailed Shrike
Javan Banded Pitta


Common Iora
Sunda Pied Fantail


Yellow-vented Bulbul
Plain Prinia


Common Tailorbird
Black-naped Monarch


Coppersmith Barbet
Horsfield's Babbler

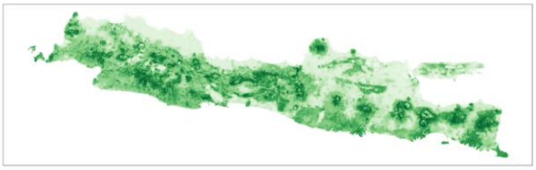
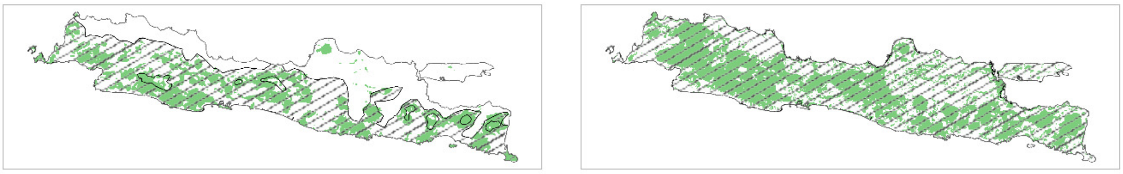
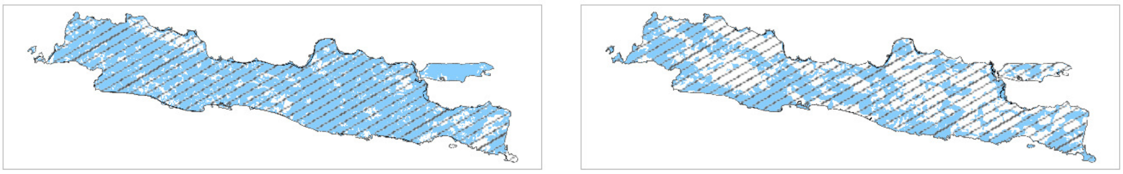

Black-winged Flycatcher-shrike

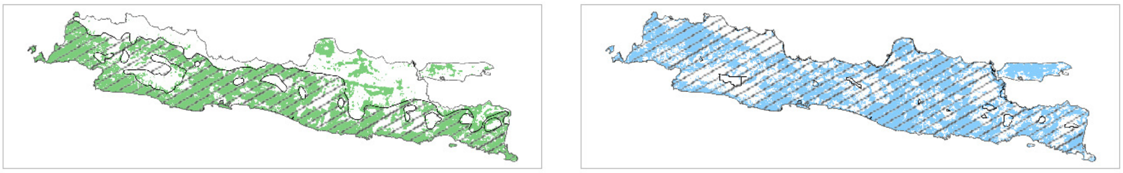
Figure 3.5b Modelled suitability for species on a 0–1 scale with darker shades indicating higher suitability, and maps coloured green for forest and blue for non-forest species. Species are presented in descending order of trapping threat rank (see Table 3.1) and are considered forest or non-forest species following the habitat preferences presented in the same table.


Brown-cheeked Bulbul

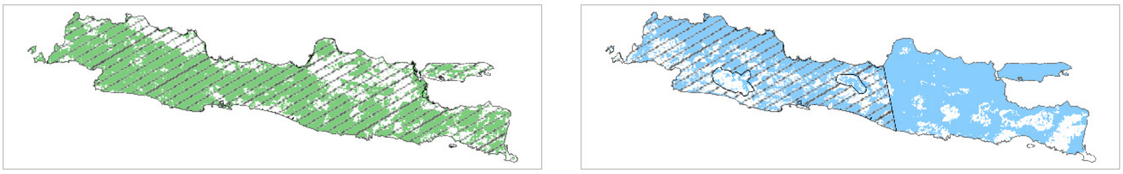
Sangkar White-eye


Javan Myna

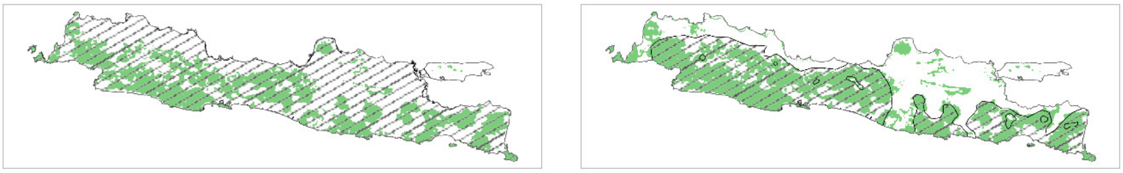
Oriental Magpie-robin


Ruby-throated Bulbul

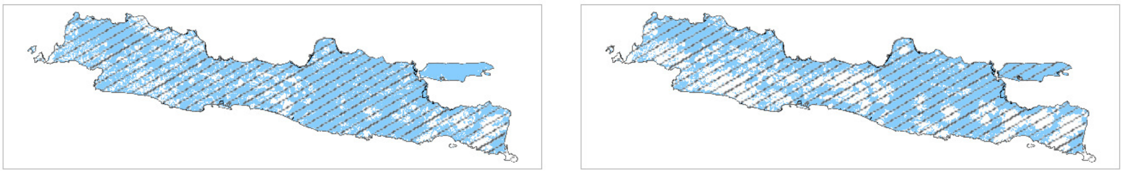
Bar-winged Prinia


Black-naped Oriole

Pied Triller

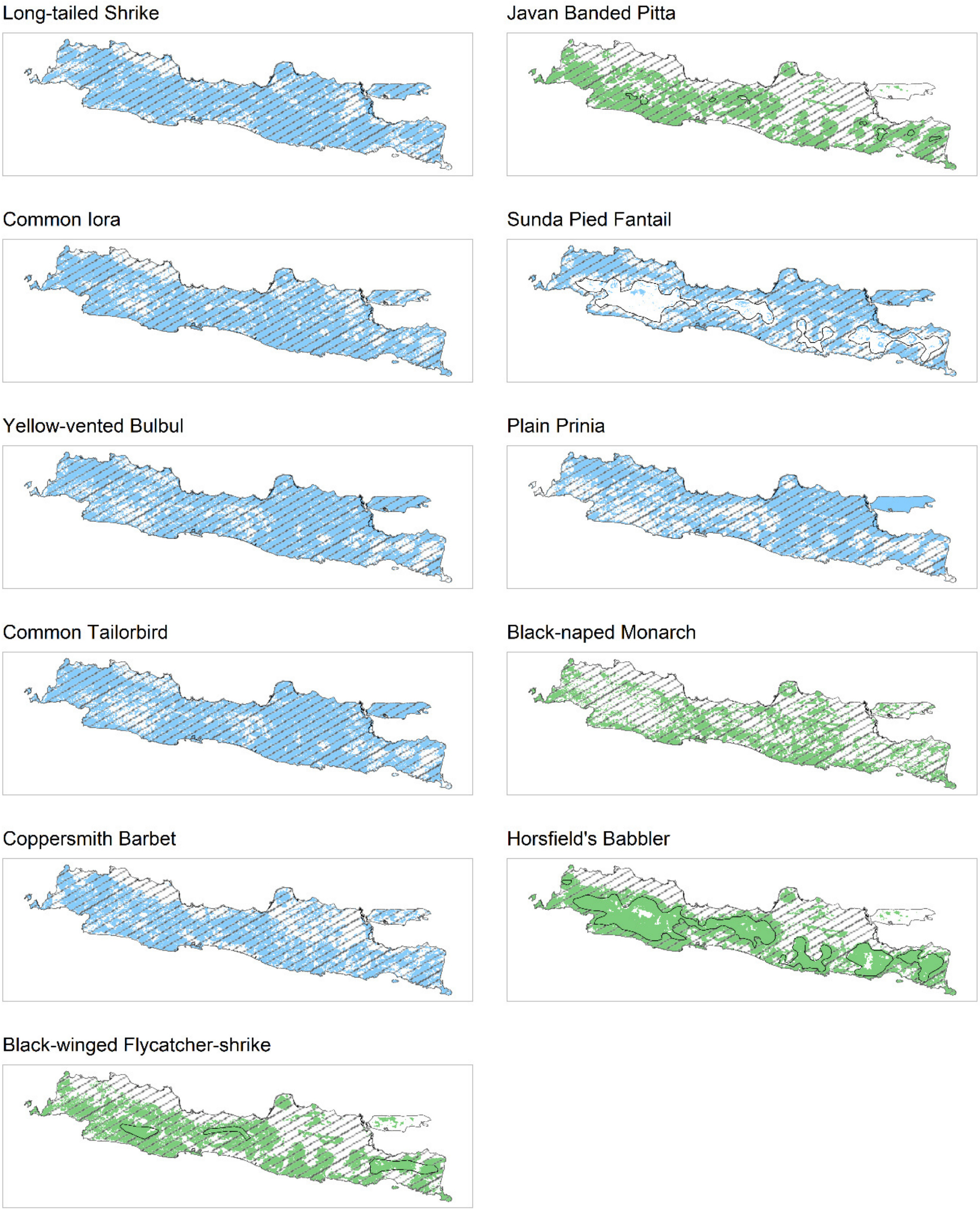

Scarlet Minivet

Yellow-eared Barbet


Sooty-headed Bulbul

Scaly-breasted Munia

Figure 3.6a Page 1 of 2-page figure.



Long-tailed Shrike

Javan Banded Pitta

Common Iora

Sunda Pied Fantail

Yellow-vented Bulbul

Plain Prinia

Common Tailorbird

Black-naped Monarch

Coppersmith Barbet

Horsfield's Babbler

Black-winged Flycatcher-shrike

Figure 3.6b Current distribution maps for study species coloured green for forest species and blue for non-forest species, with an overlay (grey hatch) of the distribution maps provided by HBW and BirdLife International (2022b). Species are presented in descending order of threat ranking (see Table 3.1).

3.3.3 The influence of bird ownership on species distributions

There were no strong patterns in the current distributions of species in relation to the areas of high bird ownership. In Figure 3.7, the species are plotted in descending order of trapping threat rank, and there is no pattern following this order to suggest that those species considered most at risk from trapping exhibit a different distribution to those least at risk. Areas from which the modelled species are ‘missing’ (present in the HBW and BirdLife range but absent from the current modelled distribution) tend to be closer to areas of high bird ownership for forest birds, but the inverse is generally true for non-forest birds, with Oriental Magpie-robin and Coppersmith Barbet exceptions to this trend.

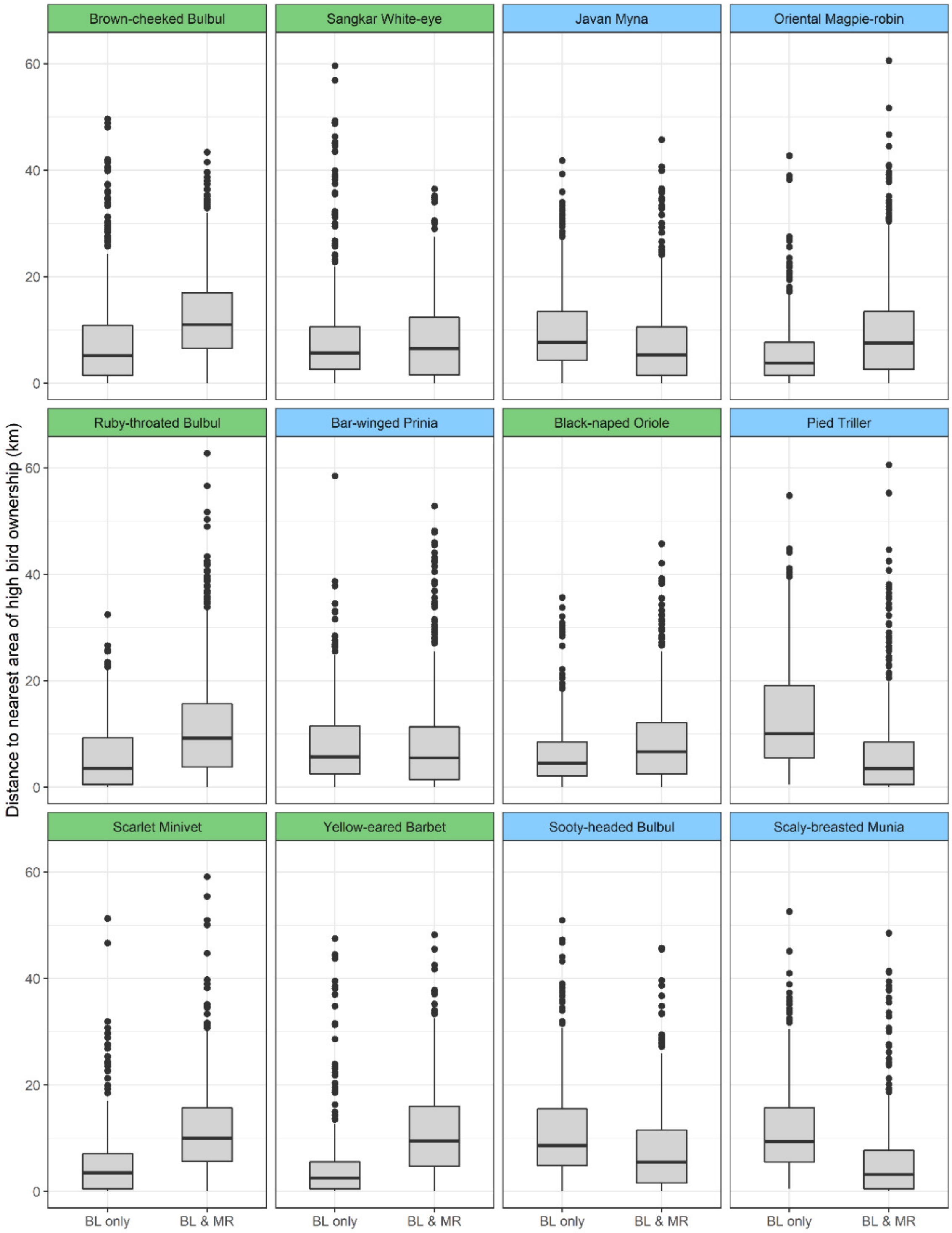


Figure 3.7a Page 1 of 2-page figure.

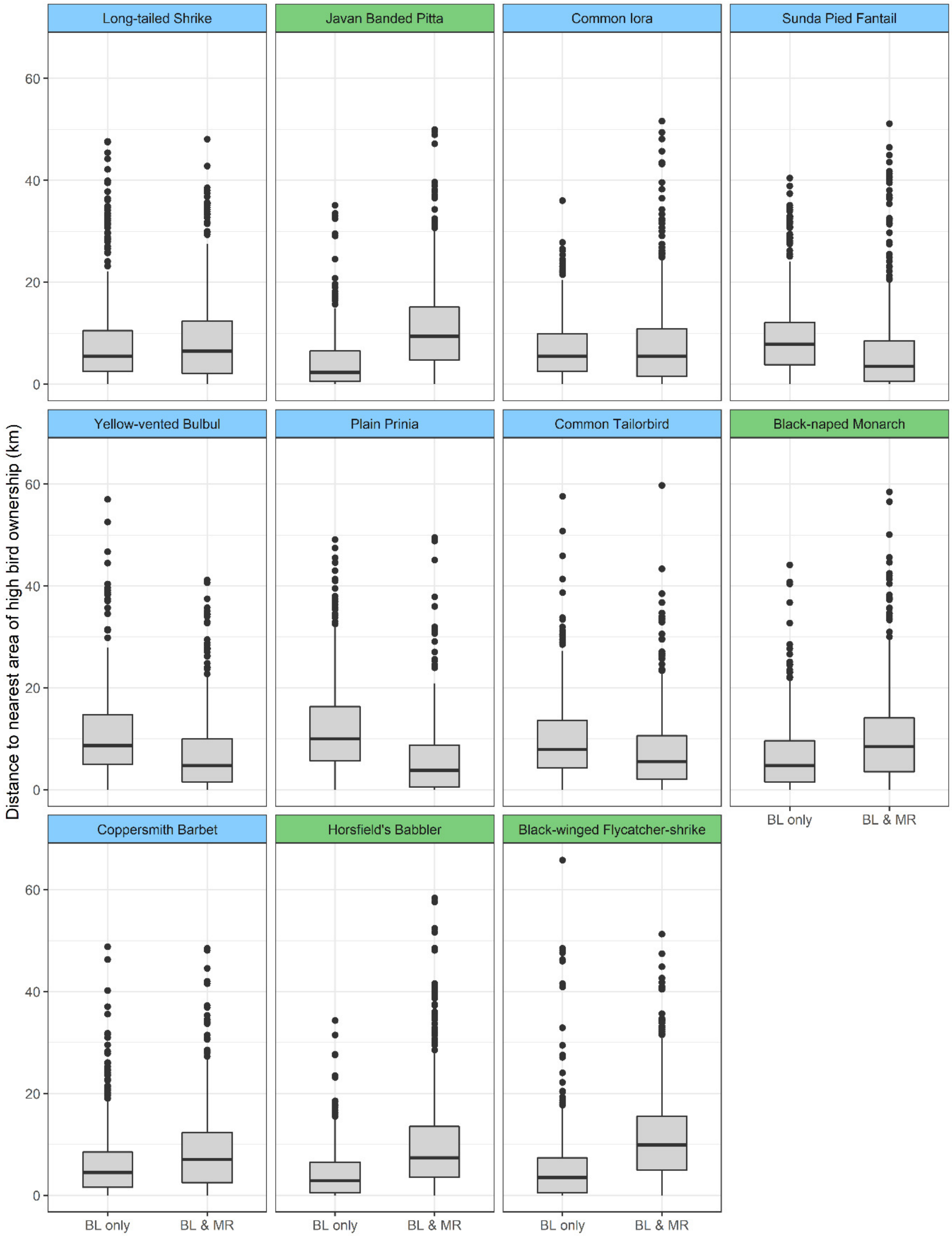


Figure 3.7 Distance to areas of high bird ownership (km) for a random sample of points from areas within the BirdLife (BL) range but outside the modelled range (MR) for each study species (*n* = 500), and from areas within both the BirdLife and modelled range (*n* = 500). Plot headers are colour-coded for forest (green banner) and non-forest (blue banner) species according to the habitat preferences in Table 3.1.

3.3.4 Range loss, degree of protection, and high priority areas

Modelled species richness was highest in the southern and central areas of Java, while patchier areas of relatively high richness were distributed in the east and west of the island (Figure 3.8). These areas can be characterised as lowland with high landscape-scale habitat diversity and a high proportion of forest cover. Areas where modelled species richness was low in this region tended to be either at high altitude or with a high proportion of cultivated land.

Modelled species richness was lowest in the northern parts of Java, particularly in the lowlands in the north of East Java, where the largest expanses of continuous cultivated land exist. Habitat diversity was not particularly low in these areas, but the dominant habitat types were cultivated land and grass/shrubland.

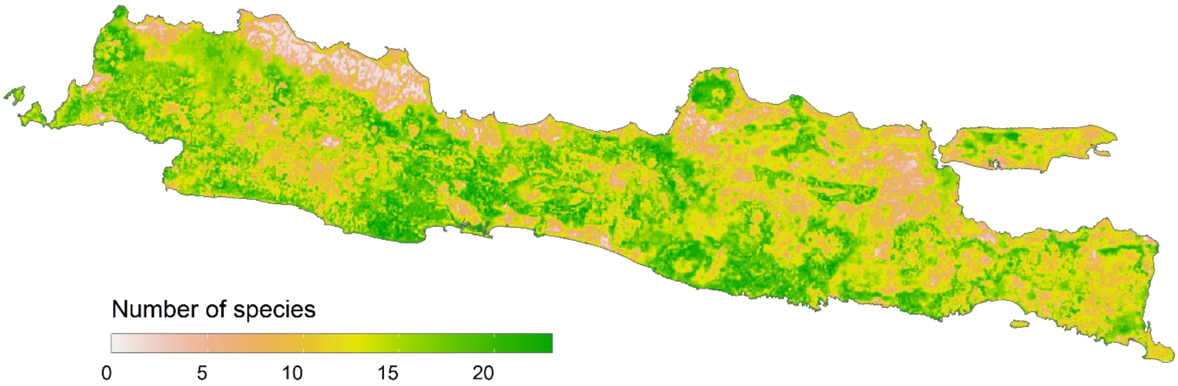


Figure 3.8 Model-predicted species richness in each cell (1-km resolution) based on the modelled range for each species (the maximum number of species in a cell is 23).

The modelled range for all species except Pied Triller was smaller than the BirdLife range, and on average covered an area 70% that of the BirdLife range. The average protected area coverage for all species was 4.3%, and the average for species categorised as Near Threatened, Vulnerable or Endangered was 5.2% (Table 3.4). Of the eight species with a current modelled distribution that was less than 60% of the BirdLife range, six are forest species. There was no relationship between the species trapping threat ranking (see Table 3.1) and the modelled range as a percentage of the BirdLife range (*rho* = 0.02, df = 21, *p* = 0.92).

Table 3.4 Modelled range compared to the HBW and BirdLife International range (2022b) and protected area coverage. Species are presented in descending order of trapping threat rank (see Table 3.1).

Species and IUCN Red List status	BirdLife range (km^2)	Modelled range (km^2)	Modelled range as % of BirdLife range	Protected area coverage (%)
Brown-cheeked Bulbul EN	81,993	39,628	48.3	11.3
Sangkar White-eye VU	130,111	68,089	52.3	5.3
Javan Myna VU	124,402	107,443	86.4	2.2
Oriental Magpie-robin LC	130,116	67,420	51.8	4
Ruby-throated Bulbul VU	78,599	66,641	84.8	4.3
Bar-winged Prinia NT	123,470	75,634	61.3	2.9
Black-naped Oriole LC	129,983	92,628	71.3	4.4
Pied Triller LC	64,127	95,602	149.1	1.5
Scarlet Minivet LC	125,993	51,385	40.8	9.4
Yellow-eared Barbet LC	72,453	62,607	86.4	7.8
Sooty-headed Bulbul LC	125,622	99,155	78.9	1.5
Scaly-breasted Munia LC	130,102	85,303	65.6	0.4
Long-tailed Shrike LC	130,116	100,692	77.4	3.8
Javan Banded Pitta LC	123,896	63,510	51.3	7.8
Common Iora LC	130,116	96,999	74.5	2.5
Sunda Pied Fantail LC	101,907	67,538	66.3	2.6
Yellow-vented Bulbul LC	130,009	99,292	76.4	1.3
Plain Prinia LC	125,622	86,139	68.6	0.6
Common Tailorbird LC	130,116	100,494	77.2	1.8
Black-naped Monarch LC	130,116	52,859	40.6	4
Coppersmith Barbet LC	130,116	77,269	59.4	5.4
Horsfield's Babbler LC	87,942	78,426	89.2	6.3
Black-winged Flycatcher-shrike LC	118,753	68,004	57.3	7.1

Areas with high modelled species richness cover 35,902 km^2 of Java (27.2% of total), but these are not well aligned with the protected area network across Java—only 2.7% of this area falls within the existing protected area network (Figure 3.9). The largest contiguous areas with high modelled species richness are in the southern part of Central Java (especially areas 1 and 3 in Figure 3.9), while there are some smaller and more isolated areas of high modelled species richness in the north of Java (area 2 in Figure 3.9). In the west and east of Java the distribution of areas with high modelled species richness is patchier, while the northern coast of Java has few areas with high modelled species richness.

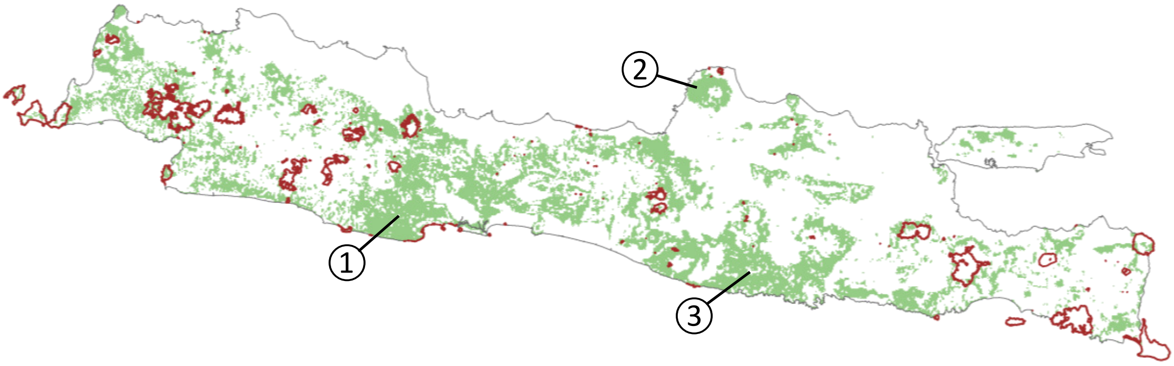


Figure 3.9 Areas of ‘high modelled species richness’ (green shade)—where at least 70% of the modelled species (17/23) are predicted to co-occur in a cell (1-km resolution)—overlaid by protected areas (IUCN categories I–VI) across Java (red polygons). Potential priority areas (see Section 3.2.7) are indicated by the circled numbers and cover the following regencies: **(1)** Ciamis, West Java; **(2)** Jepara and Pati, Central Java; **(3)** Wonogiri, Central Java, and Pacitan and Trenggalek, East Java.

3.4 Discussion

Unsustainable trapping of wild birds for the cage-bird trade has triggered an Asian songbird crisis that threatens a large number of species across Java, the island at the epicentre of the trade (Shepherd and Cassey, 2017; Sykes, 2017; Nijman et al., 2019; Collar and Wirth, 2022). Lowland birds may be particularly vulnerable to the synergistic threats of habitat loss and trapping because, although they may have bigger ranges than montane species, they have experienced far greater habitat loss and are consequently far more accessible to trappers (Harris et al., 2017; Symes et al., 2018a; Higginbottom et al., 2019; Romero-Muñoz et al., 2019). Indeed, devastating declines have been documented for some of Java’s lowland birds (Jepson, 2016; Bergin et al., 2018; van Balen and Collar, 2021; Squires et al., 2022). However, for effective prioritisation of conservation action sufficient data are required to calculate metrics associated with extinction risk such as the extent of occurrence and area of occupancy (Harris and Pimm, 2008; IUCN Standards and Petitions Working Group, 2022), and for the vast majority of Java’s lowland birds these data are missing (BirdLife International, 2022). Here, I modelled the current distributions for a suite of Java’s lowland birds using a combination of land cover, bird ownership and other relevant environmental predictors to begin to address this knowledge gap. The most important variables in the models were those related to land cover and its configuration in the landscape, while there was relatively little signal of a relationship between the current distribution of species and the proxy of trapping pressure I used—the bird ownership layer. Clearly, there would be no birds to trap without suitable habitat for them to survive and reproduce in. Habitat loss and degradation is an important issue across Indonesia (Hughes, 2017), with widespread forest loss due to agricultural expansion and industrialisation (Brockerhoff et al., 2008; Margono et al., 2014), and lowland forest replacement with plantations (Danielsen et al., 2009; Gaveau et al., 2016). With the highest human population density anywhere in Indonesia, the accessible lowlands of Java have historically suffered massive forest loss, such that lowland forest cover is now estimated to be under 3% (Margono et al., 2014). In addition, it has been suggested that the widespread application of agricultural chemicals and urbanisation are implicated in significant wildlife losses across Java (Ismail, 2014; Firman, 2017; van Balen and Collar, 2021). It is thus not surprising that these factors play an important role in shaping the distribution of Java's lowland birds, nor is it surprising that those with the smallest ranges are the forest species.

There are a number of explanations why no clear signature of bird trapping pressure on the predicted distribution of species was found. It is possible that the proxy of bird trapping used did not adequately describe the actual pattern of bird trapping across Java. It is difficult to measure direct trapping effects on bird abundance and diversity, even at relatively small scales, as shown by a study of Sumatran birds that failed to find trapping effects along a 5-km remoteness gradient (Harris et al., 2017). Owing to the understandable lack of direct bird trapping data, here bird ownership data were substituted to generate a bird ownership density layer for Java, from which the distance of each cell in the modelling area from cells with high bird ownership was used as a proxy for trapping pressure. This was based on the assumption that areas with high trapping pressure are in close proximity to areas with highest ownership levels, itself based on assumptions that, all else being equal, consumers prefer to travel short distances to purchase birds and trappers prefer to minimise the distance they travel to trap and sell birds. As the value and prestige of species increases, trappers would be expected to travel further to trap birds (Pires and Clarke, 2011), and consumers would travel to particular markets to purchase such birds. Nevertheless, across Java the vast majority of birds sold are relatively low value (Chng et al., 2015), so the basic premise that areas with most bird trapping are close to areas of high ownership seems reasonable.

Another reason why it was difficult to detect a signal of trapping in the bird distributions is because I searched for effects of unsustainable trapping by looking for range contractions. Unlike other major threats to species, such as habitat loss, which may be more likely to lead to rapid local range contractions, it is possible that trapping leads to range-wide population declines or more fine-scale fragmentation, which may not translate into a detectable range contraction at the resolution of my models (Osuri et al., 2020). It has been suggested that overexploitation alone is unlikely to lead to local extirpation because of the increased costs (i.e. trapping effort) involved in catching the last remaining individuals (Brook et al., 2008). A counter-argument is that as species become rarer their value to humans increases, making it worth trappers' efforts to continue searching for the last individuals (Courchamp et al., 2006), and while the market value and rarity of birds are linked (Harris et al., 2017), in reality this is only likely to occur where a species has particular appeal to buyers and is not easily 'substituted' by an alternative species that is easier to trap and cheaper to acquire. Proneness to extinction must also be species-specific, depending on ecological traits such as nesting preferences, productivity, survival rate and habitat preferences among others (McKinney, 1997). Nevertheless, even species with favourable traits, such as high fecundity, have been exploited elsewhere to near extinction, warning against complacency in the conservation of apparently common species that are overexploited (Sadovy and Cheung, 2003). Even if range contractions do occur, there is probably a relatively long time lag between serious range-wide faunal depletion and local extirpation, so other methods in addition to SDMs should be used to monitor species suspected of being negatively affected by trade.

While the current distributions of the modelled species are in some cases much smaller than the predicted extent of occurrence maps produced by HBW and BirdLife International (2022b), these results should be interpreted cautiously. As previously stated, species throughout the tropics remain poorly documented (Collen et al., 2008), and for the vast majority efforts to map distributions accurately have not yet been made. In place of this, HBW and BirdLife (2022b) have used locality data, expert opinion, broad biome-level land-cover data and elevational ranges to produce extent of occurrence maps for all species to support extinction risk assessments. For some of these species, the scale of the area covered by these maps spans all or most of Southeast Asia. It is therefore unsurprising that the current distribution maps for species presented here differ for some species, given that they cover a much smaller area and are more similar to an area of occupancy than an extent of occurrence metric.

The current distributions of the species studied appear to be incongruent with the formally protected area network across Java, meaning that for the large proportion of Java's avifauna that occurs in the lowlands, the vast majority of which are not formally protected, other conservation strategies will be needed (UNEP-WCMC and IUCN, 2022). Although the species modelled here are not a random selection of Java's lowland birds and therefore may not be representative of the Javan avifauna, they do cover a broad range of families so this should be considered a pilot exercise for a much-needed larger programme of monitoring and evaluation in the future. Most of the species studied had a distribution that included large areas of human-modified habitats, and this applies even to the forest species, as Java's lowland primary forest has been almost completely destroyed and exists in small fragments, while most other lowland forest is either heavily disturbed or actively managed as plantation woodland (Margono et al., 2014). Most of Java's lowlands can therefore be considered 'working landscapes', and as such are not suited to strict land-based nature designations that typically exclude people (Jonas et al., 2014).

The future for many of Java's lowland birds lies alongside people, so it is essential that socio-ecological conditions enable both people and wildlife to flourish, also recognising that promoting functional diversity of bird communities is vital for people due to the ecosystem services birds provide including pollination, pest control and seed dispersal (Şekercioğlu, 2012). Encouraging the management of such working landscapes using wildlife-friendly methods is likely to be the best way to conserve Java's lowland birds in most cases (Edwards et al., 2010, 2011), and in some cases these types of landscapes may rival protected areas for the biodiversity they harbour (Karp et al., 2019). Nevertheless, the importance of retaining remnant forests to protect biodiversity cannot be overstated (Anand et al., 2010), and the high biological value of the low-to-mid elevation forests surrounding protected areas in West Java has already been demonstrated (Higginbottom et al., 2019). Across Java, the vast tracts of plantation woodland (mostly of teak *Tectona grandis*) and agriculture offer ample opportunity to establish 'other effective area-based conservation measures' (OECMs; Jonas et al., 2014) with the involvement of local communities (Whitten et al., 1996). There are no data available for OECMs in Java, so it would be prudent to assume that few, if any, exist, except where certain conditions apply over relatively small areas such as temples or mines guarded by security personnel. For lowland birds that are already rare, enterprises to ensure the survival of small pockets of habitat and species would provide support for ongoing and future captive breeding and reintroduction efforts, and provide protection to small populations of vulnerable birds (Dolman et al., 2015; Eaton et al., 2015; Nijman et al., 2018). In addition, robust distributional baselines need to be established for all of Java's lowland birds at any risk from trapping and habitat loss, and ideally population monitoring should be initiated, and may be achievable through citizen science efforts. Identifying the areas of lowland Java most suitable for the initiation of community-based conservation schemes should follow, and these areas will need to be identified based on their natural characteristics as well as socio-economic conditions and the strength of local attitudes related to bird trapping and conservation projects.

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Chapter 4

4 Controlling trapping, overgrazing and invasive vegetation is key to saving Java's last population of the Black-winged Myna

Abstract

The Black-winged Myna (*Acridotheres melanopterus*) is an Endangered passerine endemic to the islands of Java and Bali, Indonesia. Illegal trapping to supply the cage-bird trade has led to its near-total extinction, with the global population estimated to number fewer than 100 individuals. I estimated the current range and population size of the species at Baluran National Park, which supports Java's last known population, and used species distribution modelling to evaluate potential suitability of currently unoccupied areas across the park to identify priorities for management intervention. I estimate that the Black-winged Myna population numbers 179 individuals (95% CI: 111–288; density: 14.3 ± 3.5 individuals km^{-2}) and that its current range is 12.3 km^2. My model indicated that some 72 km^2 of the park (30% of total area) has potentially suitable habitat for the species, and I infer that the principal cause for the disparity between its current and potential range is trapping, compounded by savanna loss and degradation due to illegal domestic cattle grazing and the spread of invasive thorny acacia (*Vachellia nilotica*). The partial clearance of acacia in recent years appears to have assisted a modest population recovery by the myna. Its further population growth and range expansion in Baluran depends on effective management of illegal poaching, further clearance of acacia, and easing domestic cattle grazing pressure on areas of savanna, particularly through engagement with communities living inside the park. Any actions that increase the size of the Black-winged Myna population are likely to benefit other threatened savanna-dependent wildlife in the park, notably banteng (*Bos javanicus*) and Green Peafowl (*Pavo muticus*). While my models and recommendations may be applicable to other protected areas in Java, and indeed other threatened myna species, trapping and habitat change may have site-specific dimensions, especially outside of protected areas, and thus demand local bespoke solutions.

Abstrak (Bahasa Indonesia)

Jalak putih (Acridotheres melanopterus) adalah burung berkicau endemik di pulau Jawa dan Bali, Indonesia, yang berstatus Terancam. Selama beberapa dekade terakhir, penangkapan ilegal untuk memasok perdagangan burung dalam sangkar telah menyebabkan kepunahan dengan populasi global diperkirakan berjumlah kurang dari 100 individu. Kami melakukan estimasi sebaran dan ukuran populasi spesies saat ini di Taman Nasional Baluran, yang mendukung populasi liar terakhir yang diketahui di Jawa, serta menggunakan pemodelan distribusi spesies untuk mengevaluasi potensi kesesuaian area yang tidak ditempati Jalak putih di seluruh Taman Nasional untuk mengidentifikasi prioritas intervensi pengelolaan. Estimasi populasi Jalak putih di TN Baluran berjumlah 179 individu (95% CI: 111–288; kepadatan: 14,3 ± 3,5 individu km^{-2}) dan sebarannya saat ini adalah 12.3 km2. Pemodelan kesesuaian habitat menunjukkan bahwa sekitar 72 km2 dari taman nasional (30% dari total luas) berpotensi sesuai untuk spesies tersebut, dengan kesimpulan bahwa penyebab utama perbedaan antara sebaran saat ini dan potensi sebaran adalah penangkapan, ditambah dengan degradasi dan hilangnya savana akibat penggembalaan sapi lokal ilegal serta penyebaran akasia berduri yang invasif (Vachellia nilotica). Namun, pembukaan sebagian akasia berduri tampaknya telah membantu pemulihan populasi secara sederhana. Pertumbuhan populasi dan perluasan sebaran jalak putih di Baluran, tergantung pada pengelolaan perburuan liar yang efektif, pengurangan akasia berduri yang terus berlanjut, serta mengurangi tekanan penggembalaan ternak lokal di daerah savanna terutama melalui keterlibatan masyarakat yang tinggal di dalam kawasan. Setiap tindakan untuk meningkatkan jumlah Jalak putih, kemungkinan besar akan menguntungkan satwa liar lain yang berstatus terancam dan bergantung pada savana di dalam kawasan, terutama banteng (Bos javanicus) dan merak hijau (Pavo muticus). Sementara model dan rekomendasi ini mungkin bersifat umum dalam penerapannya pada kawasan lindung lainnya di Jawa, serta spesies Jalak lainnya yang terancam, namun masalah penangkapan dan perubahan habitat yang terjadi bersamaan mungkin bersifat spesifik lokasi, terutama di luar kawasan konservasi, dan mungkin pada gilirannya menuntut solusi lokal yang tepat.

4.1 Introduction

Biological diversity is being eroded at an unprecedented rate and wildlife trade is a main underlying cause, contributing to enormous declines in species abundance, loss of ecosystem function, and increased risks to human health through zoonotic diseases (Dirzo et al., 2014; Pimm et al., 2014; Benítez-López et al., 2017; Aguirre et al., 2020). Nearly a fifth of all extant vertebrate species are traded, mostly in and from the tropics, with birds and mammals being disproportionately affected (Wyler and Sheikh, 2008; Barber-Meyer, 2010; Scheffers et al., 2019). Southeast Asia, one of the most biodiverse regions on earth, has among the highest proportion of threatened species for most higher classes of animals (Myers et al., 2000; Sodhi et al., 2010; Hughes, 2017). Throughout the region, the trade in wild-caught songbirds—prized for their vocal ability, plumage, rarity and cultural significance—is having a massive effect on wild populations (Nijman, 2010; Lee et al., 2016; Symes et al., 2018; Indraswari et al., 2020). The resulting 'Asian Songbird Crisis' has left many species facing extinction, while for many others the damage trade has wrought on their populations is still poorly understood due to insufficient monitoring (Eaton et al., 2015; Shepherd and Cassey, 2017; Bergin et al., 2018; Marshall et al., 2020). Indonesia, particularly its most populous island of Java, is widely regarded as the epicentre of the bird trade in Southeast Asia, with millions of birds sold annually at markets irrespective of their legal status and an estimated 70 million cage-birds kept in one-third of Java's 36 million households (Chng and Eaton, 2016; Harris et al., 2017; Marshall et al., 2020). Half of Indonesia's 64 globally threatened songbirds (order Passeriformes) are threatened primarily by trade, and most of them occur on Java (IUCN, 2022).

There are several patterns of decline exhibited by species under heavy pressure from habitat loss and trapping: some show dampened population densities across their range (Laaksonen and Lehikoinen, 2013) while others collapse into just a few strongholds (Abram et al., 2015; Annorbah et al., 2016). The role of formally protected areas in the conservation of endangered wildlife is also varied, ranging from absolutely critical (Ghosh-Harihar et al., 2019; Prakash et al., 2019) to relatively secondary (Agardy et al., 2003; Kamp et al., 2015). A number of conservation strategies may be useful for songbirds in Indonesia. Some species, such as the Bali Myna (*Leucopsar rothschildi*), survive almost exclusively in formally protected areas (Jepson, 2016) but other species survive in refuges outside of protected areas (Kurniandaru, 2008; Yong et al., 2018). The latter, which may fall under the umbrella of ‘other effective area-based conservation measures’ (OECMs; Jonas et al., 2014), can include temples and other culturally important sites, small islands, tourist facilities and privately guarded sites, where work with local communities/authorities underpins the maintenance of socio-ecological conditions that support the survival of key species, intentionally or otherwise (Negi, 2010; Li et al., 2014; Dolman et al., 2021).

The Black-winged Myna (*Acridotheres melanopterus*), endemic to the Indonesian islands of Java, Bali and Madura, now primarily survives in formally protected areas, but was once widespread in the lowlands, predominantly savannas and cultivated areas up to 1,200 m in West Java and reportedly 2,400 m in East Java (Feare & Craig 1998; Collar et al. 2001). It has been present in both domestic and international trade for decades, despite its protection under Indonesian law since 1979 (Minister of Agriculture, Decree no. 757/Kpts/Um/12/1979). It is, however, domestic trade that is largely responsible for the precipitous decline of the wild population, which began in the 1960s but was most pronounced in the 1990s, and the sharp decline in numbers traded in the 2000s gave a clear indication that wild populations were vanishing because of trapping (Collar et al., 2001, 2012; Eaton et al., 2015; Shepherd et al., 2016; Nijman et al., 2018).

Although small numbers may persist in recently unsurveyed areas including some nature reserves, the only known wild population of Black-winged Myna left on Java occurs in Baluran National Park, East Java (Winnasis et al., 2020; eBird, 2021), while a small number persist at two sites in Bali, with 35 birds at Bali Barat National Park (Brillianti et al., 2019) and 12 at another unspecified site (Eaton et al., 2015). The small number of birds known from two sites near Jakarta, Java (Eaton et al., 2015) are unlikely to persist (TMS pers. obs.). At Baluran, the population has been extremely low over the past decade: the largest flocks observed in 2009 and 2010 numbered 25 and 12 individuals, respectively (Winnasis et al., 2011; Eaton et al., 2015), although in 2016 a flock of 37 was recorded (BirdLife International, 2021). Accordingly, the global population size of wild Black-winged Mynas is considered to be below 100 individuals, probably around 85. This circumstance indicates a clear and urgent need to carry out a thorough ecological assessment of the species to inform its conservation management strategy (Lee et al., 2016). I therefore sought to (1) document its current distribution and estimate its population size within Baluran National Park; (2) use species distribution modelling to identify potentially suitable areas that should be prioritized for appropriate management; and (3) identify the barriers to population expansion in different parts of the park and recommend interventions that can break these down.

4.2 Methods

4.2.1 Study area

Baluran National Park (BNP; 7°50′S 114°22′E) is situated on the north-eastern tip of Java, with a land area of 264 km^2 (Figure 4.1). It was first established as Baluran Game Reserve in 1937 by the Dutch colonial government owing to the large mammals found there—banteng (*Bos javanicus*), Javan rusa (*Rusa timorensis*), feral water buffalo (*Bubanus bubanus*), dhole (*Cuon alpinus*), Javan leopard *(Panthera pardus melas*) and the now extinct Javan tiger (*Panthera tigris sondaica*) (Whitten et al., 1996). The park is in one of the driest parts of Java, receiving <1,500 mm of rainfall a year, most falling between December and February (Winnasis et al., 2011); a pronounced May–October dry season, in combination with fire and herbivory, maintains the savanna-like landscape in the north and east of the park (Pennington et al., 2018). Mount Baluran (1,247 m), a dormant volcano, dominates the centre of the park and is cloaked in tropical evergreen and dry deciduous forest.

The residents of Karang Tekok village to the north-west and Wonorejo village to the south-east have always utilized BNP to trap and hunt wildlife and to collect wood, seeds, fodder, honey, and tamarind fruit (*Tamarindus indica*), and they also start fires both accidentally and deliberately (Whitten et al., 1996; Sabarno, 2002). Overall, hunting and trapping still represent the greatest threat to wildlife in BNP: the Green Peafowl (*Pavo muticus*), an Endangered species, has declined at BNP, partly because adults and chicks are trapped and sold either alive as pets or dead as food (Winnasis et al., 2011). At least 23 other bird species are known to have been trapped within BNP, mostly alive using mist-nets (Winnasis et al., 2011), and hunters have been prosecuted for poaching East Javan langurs (*Trachypithecus auratus*), a globally Vulnerable species (Nijman, 2020). Five guard posts control the main entry points and regular patrols are conducted, but a public road bisects the west side and the park's long coastline offers great ease of access from the sea.

The savanna in the north of BNP has been significantly altered since 1975, when a commercial license was awarded to replace native trees with an agati or turi (*Sesbania grandiflora*) plantation, a pulpwood species used in paper production (Pudyatmoko et al., 2018). The plantation workers have remained in the park despite the license expiring in 2000, and now graze herds of cattle across the savanna and grow crops along the coast (Wianti, 2014). More cattle kept by residents of Karang Tekok village on the park's boundary also enter the northern savanna to graze daily (Prijono, 2014). In total, almost four thousand cattle and over a thousand goats subsist on the northern savanna (Prijono, 2014; Pudyatmoko, 2017). The park's savanna is also threatened by the spread of thorny acacia (*Vachellia nilotica*), which was planted at Bekol in 1969 to prevent fire from spreading into teak (*Tectona grandis*) plantations (Sutomo et al., 2016).

4.2.2 Classifying the park's habitats

I generated a contemporary land-cover map for BNP based on cloud-free LANDSAT 8 imagery (30 m resolution) from October 2018. Some recent burns on the image were removed using neighbour-based interpolation. Training data were obtained using field data collected in September–November 2018 by classifying land-cover types visually with descriptions employed in previous land-cover maps (Appendix 4.1), and by Google Earth image interpretation, which was used to increase the sample size of the smallest classes to address the potential training data imbalance (Millard and Richardson, 2015). I selected the following nine land-cover types for the classification, modified from the latest BNP land-cover map (Baluran National Park, 2008) and using relevant descriptions (Appendix 4.1): open savanna, savanna woodland, dry deciduous woodland/shrubland, dry deciduous forest, thorny acacia scrub, teak plantation, tropical evergreen forest, beach forest, and mangrove forest. The difficult terrain on Mount Baluran made field data collection impossible there, but the tropical evergreen forest on its slopes could be delimited from Google Earth images and was therefore included in the training data.

Training data were used to build a random forest (RF) classification model (Breiman, 2001). The environmental variables I used were bands 1–7 of the LANDSAT 8 imagery; normalized difference vegetation index (NDVI); the Global Land Analysis and Discovery (GLAD) laboratory's Global Forest Canopy Height 2019 product (Potapov et al., 2021); wetness, brightness and greenness indices; and topography (elevation, slope and aspect) calculated from the Shuttle Radar Topography Mission (SRTM) digital elevation model with 30 m resolution (van Zyl, 2001). Analysis was conducted in R (R Core Team, 2022) using packages 'raster' v3.3.13 (Hijmans, 2020), 'randomForest' v.4.6.14 (Breiman, 2001) and 'sf' v0.9-7 (Pebesma, 2018). As a dimension reduction procedure, I ran the RF classification 25 times and recorded the five most important variables for each iteration. A Spearman's rank correlation analysis was then used to measure pair-wise correlations and if any of the five most important variables were highly correlated ($r_s > 0.90$) with lower-ranked variables, the latter were removed (Millard and Richardson, 2015). The model was tuned by selecting the number of trees to grow and variables sampled at each split that minimized the out-of-bag error rate. The final RF classifier grew 1,000 trees and two variables were randomly sampled at each split.

4.2.3 Population size and range estimation

To estimate the current range of the Black-winged Myna in BNP I included all records from line transect sampling and incidental observations during the fieldwork period from March 2018 to March 2019, as well as one observation by a proficient local birdwatcher (Heru Fitriadi) and 19 observations by experienced members of the Copenhagen Zoo project staff at BNP, who carried out a survey in November 2017. I removed six outlying records from the current range estimate (see Figure 4.1) because they were over 4 km from the main cluster of observations, and were never of more than two birds which, given the sociality of mynas, suggests they were exploratory movements by dispersing birds. Moreover, I visited the area of these sightings monthly during the fieldwork period and only recorded mynas on two occasions, indicating that the southern area of the park did not form part of the species' home range. I constructed a minimum convex polygon around the remaining occurrences (Burgman and Fox, 2003) in R with a 200-m buffer to provide a final range estimate.

Based on the pre-existing land-cover map a total of 36 distance sampling line transects with a combined length of 73.5 km were run across BNP to sample each land-cover type except tropical rainforest, which I considered unsuitable for Black-winged Mynas (Collar et al., 2001). Line transects were distributed following a stratified sampling approach (Buckland, 2004), whereby more transects were located in the east of the park to focus survey effort on the area expected to contain most of the Black-winged Myna population (Winnasis et al., 2011). Within what were classified as open habitat types (open savanna and savanna woodland) line transects were assigned randomly; in woodland (classified as closed habitat), transects followed narrow tracks made by humans or large mammals, as attempts to penetrate the dense understory, especially during the wet season, risked disturbing the target species before detection (Buckland, 2001). Line transects outside the estimated Black-winged Myna range were excluded from the distance analysis and subsequent population density estimate because no encounters were recorded on any of these. Most transects were walked in both wet and dry seasons, but eight were only walked in the dry season. Twenty-one transects with a combined length of 29.1 km (mean length: 1.4 km) were included in the final analysis, giving a total effort of 50.6 km (Figure 4.1).

Distance sampling was undertaken at the end of the wet season from 28 March to May 30 2018 (May was included in wet season sampling because the vegetation is still lush and the understory dense), and in the dry season from 5 October to 21 November 2018. Sampling was conducted between 06h00 and 12h00 by one of two experienced observers (TMS, PGA) following the standard protocol for line-transect distance sampling (Buckland et al., 2008). Transects were walked at 1.5 km h^{-1} and each was preceded by a 200-m ‘burn in’—the first section of a transect (not included in its reported length) that is walked but encountered birds are not recorded in an attempt to equalize the effects of observer disturbance across the transect (Buckland, 2004). Birds seen only in flight were ignored, but those seen taking off or landing (using the habitat) were recorded. Laser rangefinders (Hawke LRF 400) were used to measure distances and targeted the nearest visible object when there was no clear line of sight to the bird (Buckland et al., 2008). Analysis was carried out using package ‘Distance’ v1.0.2 (Miller et al., 2019) in R (R Core Team, 2022) following standard distance sampling methods (Buckland, 2001). Detection data were right-truncated at 135 m after discarding the furthest 5% of detections from the transect. Group sizes for aural-only detections were replaced by the average size of all known groups. I generated a two-level open–enclosed habitat covariate for the detection function. The final detection function model was selected based on minimum Akaike Information Criterion (AIC) and after checking the detection function fitted the observed data (see Appendix 4.2 and Appendix 4.3).

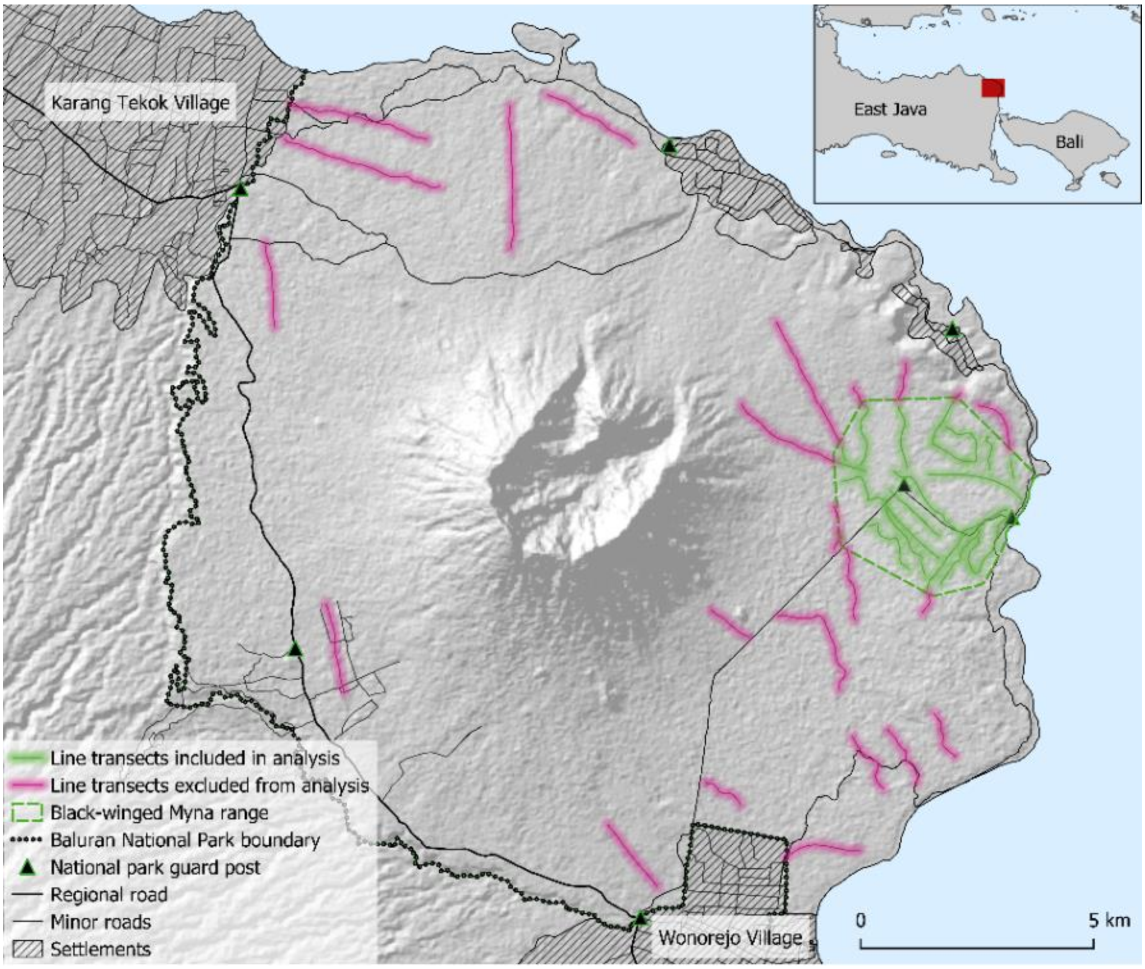

Figure 4.1 Baluran National Park, East Java, showing the location of all line transects, of which those in green were included in the Black-winged Myna population density analysis because they were inside the species' estimated range (green dashed line).

4.2.4 Estimating habitat suitability

I used species distribution modelling (SDM) to estimate the habitat suitability for Black-winged Myna within each 30 m pixel across BNP (Araújo and Guisan, 2006). Three environmental covariates were used in the models: an NDVI layer generated from LANDSAT 8 imagery; the land-cover classification raster covering the park (Figure 4.2); and a habitat openness layer, for which each pixel represented the proportion of open vs. enclosed habitat pixels in the 0.56 km^2 (25 × 25 pixels) surrounding the target pixel. Black-winged Myna occurrences (n = 339) were filtered to include just one per pixel, resulting in 291 pixels containing presences across the raster layer.

I used R packages 'raster' v3.3-13 and 'biomod2' v3.4.6 (Thuiller et al., 2020) to model Black-winged Myna distribution. During the data formatting procedure in biomod2, I set the number of pseudo-absences (PAs) to 5,000 and created five sets of PAs (biomod2 generated a total of 24,122 unique PAs) using the 'disk' algorithm with the minimum distance to presences set to 75 m (Chefaoui and Lobo, 2008; Thuiller et al., 2020). I ran three different SDM algorithms available in biomod2 on the five presence/PAs datasets: generalized linear model (GLM), generalized additive model (GAM), and maximum entropy (MaxEnt). Models were calibrated using 80% of the data and the remainder were used to evaluate model performance. Every algorithm was run five times with each PA dataset (total number of runs for each algorithm = 25).

SDM algorithms were evaluated using area under the curve (AUC) values calculated in biomod2. All of the algorithms performed well (AUC > 0.80) and were used to produce full models without data partitioning. Variable importance values on a scale from 0 to 1 were calculated for each algorithm, higher values indicating greater influence in the model. I then aggregated the models by algorithm and generated algorithm-specific projections of habitat suitability across BNP in order to calculate algorithm-specific model averages, which were used to evaluate how well individual algorithms discriminated between presences and PAs based on Tjur's R^2 (Tjur, 2009).

For the final output, an ensemble of the projections was generated with the SDM algorithms using mean-weighting based on the algorithms' AUC value. This raster map provided raw suitability values for each pixel in BNP on a scale of increasing suitability from 0 to 1. I generated a binary raster of unsuitable and suitable habitat using a threshold value, which was the minimum suitability value at a pixel including a Black-winged Myna presence after excluding the lowest 10th percentile of suitability values from pixels with presences. Final maps from the analysis were generated using QGIS v3.10.3.

4.3 Results

Black-winged Mynas were recorded a total of 339 times during fieldwork, with all but six records in the Bekol, Balanan and Bama areas in the east of the park (Figure 4.2 and Figure 4.4). Based on the occurrence data gathered, I estimate that the current range for Black-winged Myna covers 12.3 km^2 (<5% of the park's area). The land-cover types with most occurrences were open savanna (37.8%), followed by savanna woodland and dry deciduous woodland/shrubland (both 20.6%), thorny acacia scrub (19.8%; but see the discussion), beach forest (0.9%) and dry deciduous forest (0.3%); the full land-cover classification map is shown in Figure 4.2. Of the outlying occurrences excluded from the range estimation, three were in savanna 4 km south of Bekol, and three were near the cultivated land adjacent to Wonorejo village, 6 km south of the current range. The largest groups recorded were seen gathering before dusk, with the highest record being of 97 individuals on 26 September 2018, when birds flew to roost in ten groups (maximum single group size = 25 individuals).

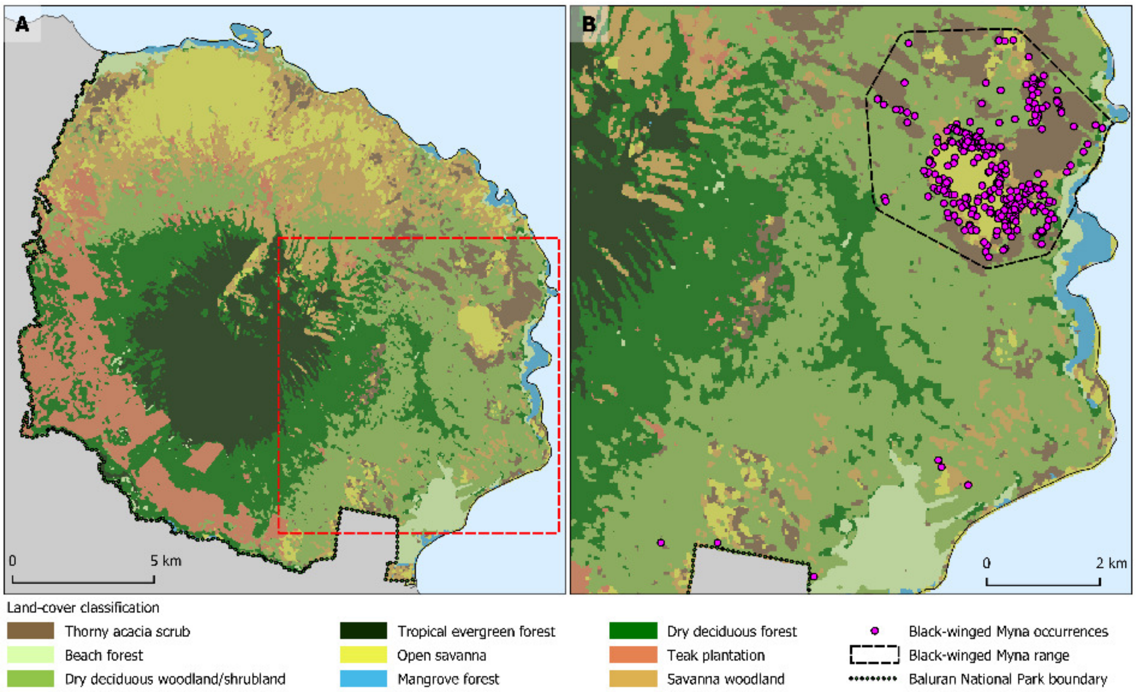

Figure 4.2 (**A**) Land-cover classification for Baluran National Park; and (**B**) all Black-winged Myna occurrences (magenta circles) recorded and the occupied range (black dashed line) overlaid on land-cover types. The extent of (**B**) is shown by the red dashed line in (**A**).

Black-winged Mynas (median group size = 2) were detected on 56 occasions during line-transect distance sampling over the wet (n = 30) and dry (n = 26) seasons. The average encounter rate was 2.9 ± 0.7 individuals km^{-1}, with the highest encounter rate in savanna woodland followed by open savanna, and the lowest in dry deciduous woodland/shrubland and thorny acacia scrub; birds were not detected in the other land-cover types during line-transect distance sampling (Table 4.1). Detection probabilities were described best by a uniform key function with one cosine adjustment term (Appendix 4.2 and Appendix 4.3). Population density was highest in savanna woodland (34.4 individuals km^{-2}, 95% CI: 13.5–88.0) and lowest in dry deciduous woodland/shrubland (10.9, 95% CI: 4.6–25.9), and the overall population density in the current range was 14.3 individuals km^{-2} (95% CI: 8.8–23.1) (Table 4.1). I estimate the overall population size to be 179 individuals (95% CI: 111–288).

Table 4.1 Estimated Black-winged Myna population density and abundance within its estimated range for each land-cover type in which birds were detected during line-transect distance sampling.

Land-cover type	Area within range (km^2)	Encounter rate individuals km^{-1} ± SE	Density individuals (95% CI)	Abundance individuals (95% CI)
Savanna woodland	0.8	5.8 ± 2.7	34.4 (13.5–88.0)	28 (11–70)
Open savanna	2.1	3.6 ± 1.8	21.0 (7.8–57.0)	44 (16–120)
Dry deciduous woodland/shrubland	5.3	1.9 ± 0.8	10.9 (4.6–25.9)	58 (24–137)
Thorny acacia scrub	4.1	2.0 ± 0.8	12.0 (5.6–25.6)	49 (23–105)
Total/overall	12.3	2.9 ± 0.7	14.3 (8.8–23.1)	179 (111–288)

All the SDM algorithms performed well based on AUC values (Table 4.2). The Tjur's R^2 values for all three algorithms were similar and showed a high level of discrimination between pixels with occurrences and pseudo-absences. Variable importance values for models produced by each algorithm showed that land-cover type had the greatest influence in models produced by all algorithms (Appendix 4.4), and relative differences in variable importance for the models produced by GAM and GLM were similar. The influence of land-cover type and habitat edge was similar in the MaxEnt model, while NDVI had a relatively small influence.

Table 4.2 Calculated AUC, sensitivity, specificity and Tjur's R2 values of SDM algorithms used to estimate Black-winged Myna habitat suitability. Values of AUC, sensitivity and specificity are averages ± SD across the five different datasets included, each comprising Black-winged Myna presences (n = 291) and 5,000 pseudo-absences. Tjur's R2 values were calculated from model averages for each SDM algorithm after projecting the models across the raster surface for the study area. The highest values for each metric are shown in bold.

SDM algorithm	AUC	Sensitivity	Specificity	Tjur's R^2
GLM	**0.88 ± 0.02**	86.9 ± 6.43	**76.3 ± 6.95**	0.46
GAM	0.87 ± 0.01	**89.3 ± 5.13**	72.4 ± 6.58	**0.48**
MaxEnt	0.87 ± 0.02	87.5 ± 5.27	73.4 ± 6.75	0.46

The final ensemble model had a Tjur's R^2 value of 0.47 and discriminated well between areas I expected to be suitable (savanna-type land-covers) and unsuitable (closed-canopy areas) for Black-winged Myna. Large areas of potentially suitable habitat (values > 0.6) are predicted across the north of the park, where open savanna and savanna woodland dominate, while there are smaller areas of suitable habitat surrounding the savanna in the south-east of the park (Figure 4.3). The central areas of large open savannas are deemed less suitable than the edges. The areas of highest suitability adjoining the current range of Black-winged Myna are distributed to the north and north-west.

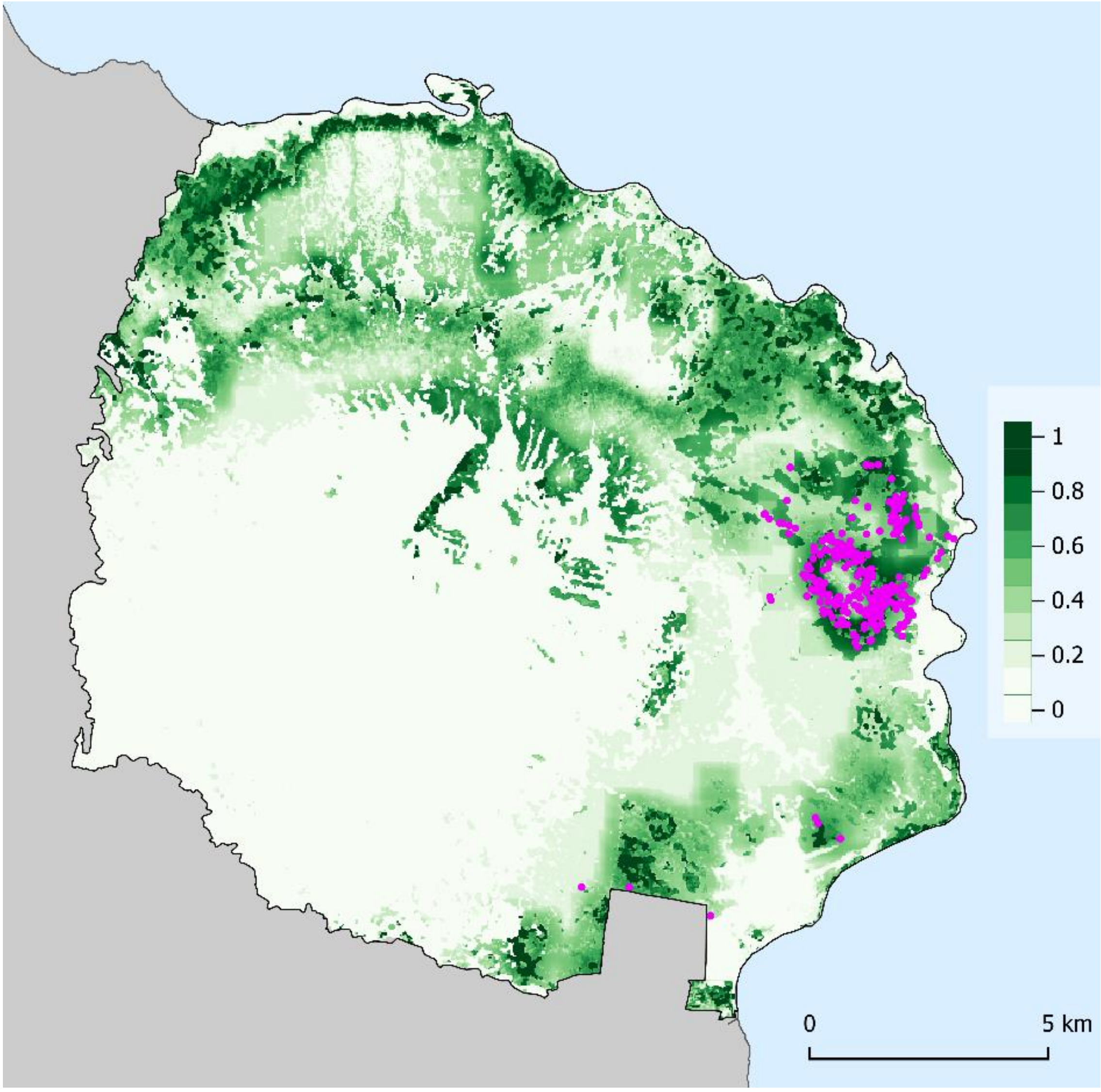

Figure 4.3 Baluran National Park habitat suitability map for Black-winged Myna, with values on a scale of increasing suitability from 0 to 1. Magenta circles show the actual Black-winged Myna occurrences that were used in species distribution modelling.

I then delimited the potentially suitable area for Black-winged Myna across BNP from the suitability raster using the calculated threshold (Figure 4.4). This final output indicated that there are an estimated 72.1 km^2 of potentially suitable habitat for Black-winged Mynas, mainly to the north-west of the current range and mostly within open savanna and savanna woodland. Of the potentially suitable habitat, 89% lies within 5 km of the coast at elevations below 300 m. Much of the potentially suitable area is close to the roads and settlements that occur in the north of BNP.

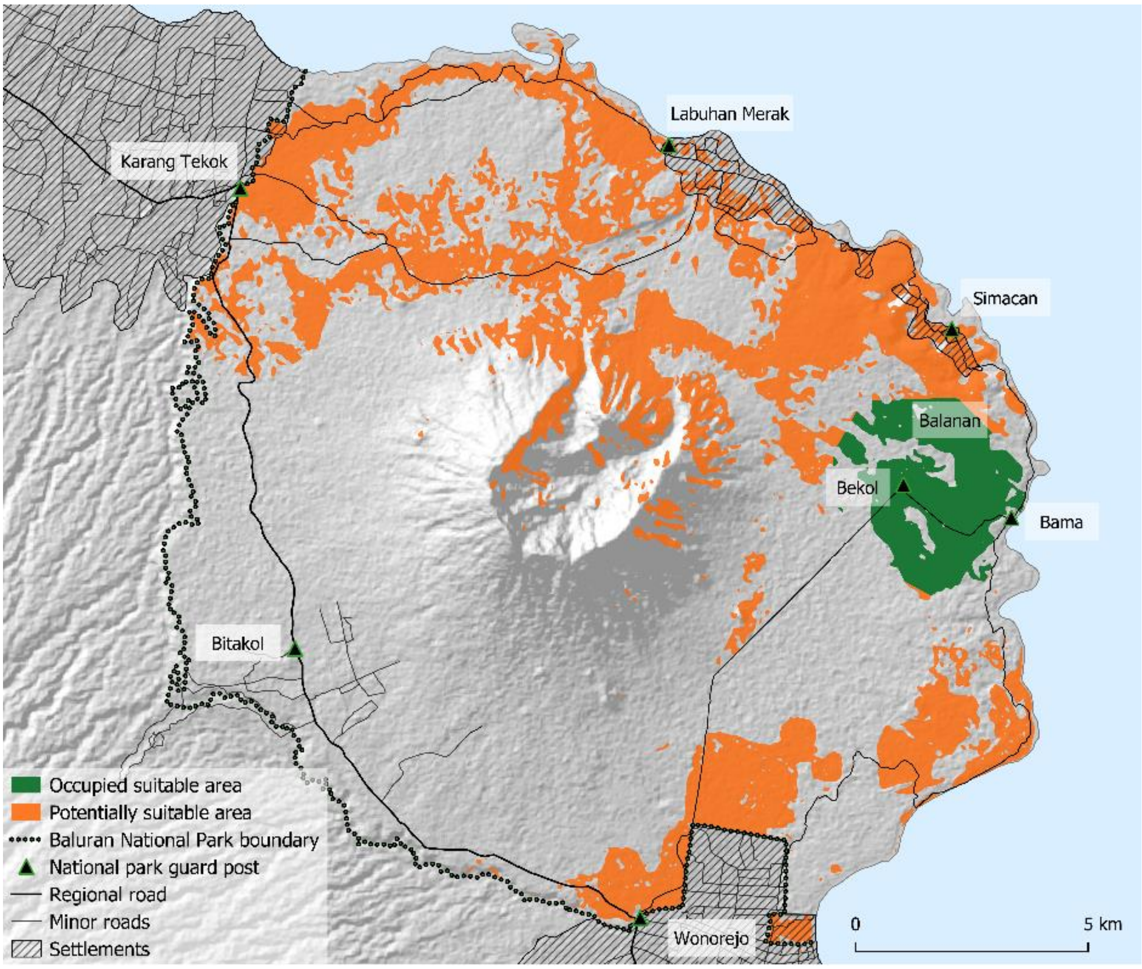

Figure 4.4 Baluran National Park, showing predicted suitable habitat for Black-winged Mynas that is currently occupied and unoccupied.

4.4 Discussion

The once widespread but now Endangered Black-winged Myna has been extirpated from localities throughout its range on Java and Bali after decades of overexploitation for the cage-bird trade (Eaton et al., 2015; Shepherd et al., 2016; Nijman et al., 2018). This first comprehensive assessment of its status at Baluran National Park, the home of the only known wild population on Java, reveals that fewer than 200 individuals are confined to 12.3 km^2 of savanna and shrubland in the east of the park, an area six times smaller than the 72.1 km^2 of potentially suitable habitat identified by my species distribution model. While some of the potentially suitable habitat is located on Mount Baluran at higher elevations than the species currently occurs at in the park, it is within the species' historically reported altitudinal range of 0–2,400 m (Collar et al., 2001). Although formal population assessments have not been published, a comparison of my population estimate with the maximum flock size of 37 birds recorded in 2016 (BirdLife International, 2021) suggests that there may have been recent population growth. The model delineates areas where management for Black-winged Mynas should be prioritized, and if conditions could be improved across the entire potentially suitable habitat, with a population density matching the average estimated for its current range, BNP might be able to support a population in excess of 1,000 Black-winged Mynas. However, there remain at least three significant barriers to such population recovery—trapping, overgrazing and invasive thorny acacia.

There are, however, several difficulties associated with building distributional models for species with small remnant populations that were once widespread and have declined due to multiple and concurrent factors. These difficulties may help to explain why the species is currently not occupying habitat identified by my model as suitable. First, it is possible that part of the population occupies suboptimal habitat constituting a demographic 'sink', where mortality exceeds productivity (Howe et al., 1991), in which case my model may have considered suboptimal habitat suitable for the species. Second, characterizing trapping pressure is difficult and direct indices are seldom available (but see Biddle et al., 2021). Metrics of remoteness (distance from roads or human settlements) may explain, to some degree, such anthropogenic pressures at a large scale (Benítez-López et al., 2017; Shaney et al., 2017; Symes et al., 2018), but fail to account for spatially discrete forms of anthropogenic protection that benefit some species such as guarding (Demerdzhiev et al., 2014); tourism, as on Komodo (Reuleaux et al., 2020); community-based conservation (Watson et al., 2007); and special land status such as sacred groves (Plieninger et al., 2020). The mynas at BNP appear to benefit from a combination of guarding and tourism (or even research) activities, for which spatial data reflecting the complexity of the situation were not available. Third, the myna population in BNP is clumped, so projecting this distribution onto other parts of the—albeit relatively small—park needs to be done with care, as habitat and other relationships may not hold in other areas. This caveat extends to other protected areas in East Java (e.g., Meru Betiri and Alas Purwo National Parks), and especially to non-protected areas, where habitat associations and trapping pressures may be different. In fact, the lack of studies when the species was still common means that its true habitat preferences remain uncertain, but they may have included a wider range of habitats than those present in BNP, although probably not closed-canopy woodland. Nevertheless, the model built for Baluran is a starting point for use in other protected areas in East Java, and the habitat associations resemble those of the nearby remnant populations of Black-winged Myna around Bali Barat National Park (TMS pers. obs.).

The most likely constraint on the size of the myna population in BNP is trapping for the domestic pet trade. Black-winged Mynas have been heavily trapped and traded over four or five decades for cage-bird markets across Java (Collar et al., 2001; Shepherd et al., 2016; Nijman et al., 2018); such is their value that seven years ago 151 individuals were stolen in a highly organized raid on a well-guarded conservation breeding centre (Tritto and Sözer, 2014). Recently, mynas explicitly identified as trapped in Baluran have been openly advertised for sale online (Bruslund et al., 2021). Evidence from this study suggests that the species persists only in the small area where I found them because birds there are afforded protection against trapping. This likely comes from three sources: the heavy park staff presence at Bekol, which houses an office, conservation breeding enclosure and guard post; tourist activity, which is centered around Bekol and Bama and which saw visitors rise from 39,874 in 2013 to 245,901 in 2020 (Padmanaba et al. 2017; BNP unpublished data); and the presence of park staff, contract workers and the Copenhagen Zoo project team performing savanna restoration at Balanan. Elsewhere in the park trappers may be less constrained, owing to the much less frequent presence of guards and tourists.

A second constraint on Black-winged Mynas at BNP is savanna degradation and disturbance resulting from the 5,000 domestic livestock grazing and browsing some 56 km^2 of the park's northern savannas (Pudyatmoko, 2017), representing 21% of the park's total area. Just over half of the 4,000 cattle and all of the goats are kept by inhabitants of the settlements along BNP's northern coast from Labuhan Merak to Simacan, with the remainder kept by residents of Karang Tekok village (Pudyatmoko, 2017). Most of the livestock belong to a few members of the local elite and are loaned to keepers under a *gaduh* system, whereby owners take the profit from cattle sold for slaughter and keepers retain calves born on pasture, with a ten-fold difference in annual profit in favour of owners (Wianti, 2014; Pudyatmoko et al., 2018). Low-intensity grazing can be beneficial for many starlings and mynas because it maintains a low sward height (Fuller et al. 2013), providing birds with access to surface and topsoil invertebrates (Heldbjerg et al., 2016; van Balen and Collar, 2021), while manure increases food availability by promoting plant and invertebrate abundance and diversity (McNaughton, 1985; Steinauer and Collins, 1995; Plantureux et al., 2005). Indeed, like other *Acridotheres*, Black-winged Mynas commonly associate with large herbivores, feeding on invertebrates from the ground they disturb (Collar et al., 2001). However, the current high intensity of grazing by domestic livestock in the northern savannas of BNP probably greatly exceeds former natural levels based on the maximum recent population sizes for the entire park of water buffalo (1,293 in 1984; Suhadi, 2009) and banteng (267 in 2000; Winnasis et al., 2011), and the graziers who accompany livestock are a source of disturbance to wildlife (Pudyatmoko, 2017). The resulting soil impoverishment (Dormaar and Willms, 1998; Villamil et al., 2001) and atrophied biodiversity (Olff and Ritchie, 1998; Dhaou et al., 2010) associated with intensive grazing seriously reduces the habitat suitability for Black-winged Myna and other savanna-dependent wildlife.

That the recent clearance of thorny acacia within the Bekol area has coincided with an apparent population upturn of Black-winged Mynas (my population estimate of around 179 individuals is substantially higher than the maximum flock sizes reported in the 2010s) suggests that thorny acacia is a poor habitat for the species, despite the relatively high usage registered in my results, caused by birds perching in acacias fringing open foraging areas. Thorny acacia spreads rapidly, replacing savanna with impenetrable thickets and reducing the food available to savanna-dependent wildlife (Kriticos et al., 1999; Dhileepan, 2009; Zahra et al., 2020). Dense stands had engulfed the savanna at Bekol, Balanan and Kramat by 1993, covering an estimated 12 km^2 (Schuurmans 1993 in Setiabudi et al., 2013), and despite restoration attempts since 1985, with some of Bekol savanna successfully cleared (Zahra et al., 2020), by 2014 the overall acacia cover had increased (Sutomo et al., 2020). Since 2016, thorny acacia clearance at Balanan has restored 3.6 km^2 of grassland (Copenhagen Zoo, 2021), which has been used by Black-winged Mynas, including for breeding (TMS pers. obs.). My habitat classification indicated that there are at least another 12 km^2 of thorny acacia within BNP in monospecific stands, most of which are close to the current Black-winged Myna range and Simacan settlement (Figure 4.4); efforts to clear this are therefore a priority and could double the habitat immediately available to the species.

If birds (re)colonize this area, guard patrols must of course follow. Such protection could be supported by the mapping of nesting and particularly roosting areas, which are probably where most trapping is done; well-protected nest-boxes could be deployed to encourage dispersing birds, as practiced in early reintroduction attempts (Tritto, 2014), particularly as the myna’s natural cavity nest sites may have been reduced in number by the removal of dead trees for fuelwood near settlements (TMS pers. obs.). Patrolling also needs to target beaches, land access points, and the northern settlements where mynas and Baluran's other key species are suspected of being smuggled out of the park (Winnasis et al., 2011).

A crucial underpinning of any conservation management of Black-winged Mynas will be engagement with the park's human communities, especially those at Simacan and along BNP's northern coast. Such work has been instrumental in protecting species facing similar anthropogenic pressures elsewhere: the Philippine Cockatoo (*Cacatua haematuropygia*) was trapped almost to extinction until a conservation program trained and employed ex-trappers as wardens and involved local communities in wildlife monitoring (Widmann et al., 2006). Communities living inside BNP could likewise be recruited to support a nestbox and monitoring scheme for Baluran's mynas in a type of payment for ecosystem services arrangement (Ferraro, 2011). In 2016 BNP established a 20 km^2 'special use zone' in a bid to settle a long-standing dispute over the communities' land rights (Mulyana et al., 2010; Wianti, 2014; Pudyatmoko et al., 2018), which suggests that local goodwill might allow such a project to be implemented. Nevertheless, further negotiations are essential to achieve a significant but equitable reduction in grazing pressure in the 56 km^2 of highly degraded northern savanna, for example, by keeping cattle in enclosures and providing alternative livelihoods (Pudyatmoko et al. 2018).

Protected areas are understandably the first option considered when seeking to preserve species, because they provide a pre-existing legal, geographical, organizational and social framework for the endeavour, and often also because they are the last places where the species of concern survive. With the western form of Black-winged Myna, nominate *melanopterus*, apparently extinct in the wild and the Bali form *tertius* not known to number more than 35 inside Bali Barat National Park, Baluran National Park represents by far the most important opportunity to save the Black-winged Myna from the trapping pressure that is driving it to extinction. The cases of the Javan Pied Starling (*Gracupica jalla*), now almost certainly extinct in the wild (van Balen and Collar, 2021), and the Bali Myna, once thought extinct in the wild and now surviving only through intensely managed reintroductions (Jepson, 2016), serve as examples of what the near future could hold for the Black-winged Myna without effective action.

At Baluran, there is scope to increase its small population by habitat restoration and enhancement, elevated protective vigilance and strong community engagement, building on established models trialled elsewhere in the world. Nevertheless, open-country starlings and mynas tend to make medium-distance movements to forage and explore (Bruun and Smith, 2003; Minderman et al., 2010; Astudillo et al., 2019), so protected areas cannot be expected to harbour them indefinitely. In the longer term, therefore, such species, especially if under pressure from trapping, will have to be conserved by management strategies that embrace adaptation and improvisation, taking advantage of the various types of security provided by mining or geothermal operations (Randriamamonjy et al., 2015; Devenish et al., 2022), religious sites (Colding and Folke, 1997), tourist resorts (Moritz et al., 2017) and organic farming, all of which may to some extent be leveraged to create appropriate socio-ecological conditions to allow them to survive within working landscapes. Other interventions proposed for exploited wider-ranging species include demand reduction (Burivalova et al., 2017; Marshall et al., 2020), commercial breeding (Jepson et al., 2011) and better enforcement of trade laws (Nijman, 2010), and all of these measures could be applied in the case of the Black-winged Myna, whose recovery in Baluran could also be abetted by supplementations of captive-bred birds. What can be achieved at Baluran in the next decade may therefore point the way for many species recoveries, not just of threatened songbirds and not just in protected areas, across Indonesia and indeed the world.

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Appendices

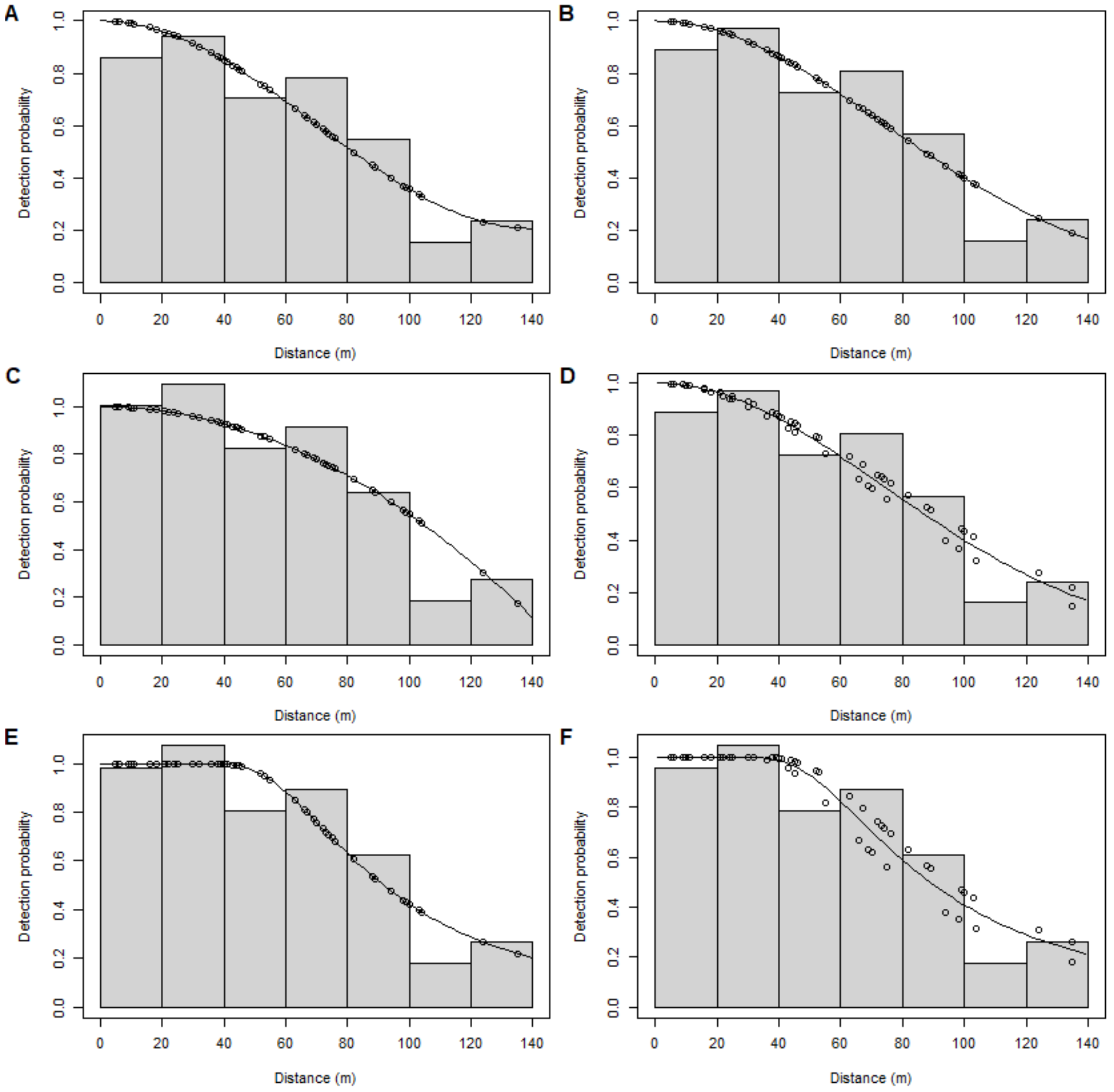
Appendix 4.1 Definitions of the land-cover types used in this study and how they relate to the previous land-cover map for Baluran National Park.

Land-cover type in this study	Baluran land-cover map (2008) class	Location	Characteristics	Sources
Open savanna	Savanna	Mostly lowland, some on volcano rim	Almost treeless with continuous grass layer	(Pennington et al., 2018)
Savanna woodland	Savanna	Lowland	Continuous grass layer with open canopy woodland of *Vachellia leucophloea*, *Azadirachta indica*, *Ziziphus rotundifolia* and occasional *Corypha utan* palms	(Winnasis et al., 2011; Pennington et al., 2018)
Dry deciduous woodland/ shrubland	Bush	Lowland	Mixture of scrub, some small open patches with a grass layer and closed-canopy deciduous forest with *Schoutenia ovata*, *Tamarindus indica*, *Albizia procera*, *Vachellia leucophloea*	(Winnasis et al., 2011; Dexter et al., 2018)
Dry deciduous forest	Secondary forest	Lowest slopes of Mount Baluran	Mostly closed-canopy forest on lowest slopes of Mt. Baluran that receives slightly higher precipitation than dry deciduous forest, although with similar species and including *Sterculia foetida*	(Winnasis et al., 2011; Dexter et al., 2018)
Thorny acacia scrub	Invasive acacia	Open areas near savanna	Closed-canopy monoculture of *Vachellia nilotica*, almost bare understorey owing to its bark tannins having an allelopathic effect on other plants	(Sutomo et al., 2016)
Teak plantation	Plantation forest	Mostly alongside regional road west of Mt. Baluran	Monoculture of mostly mature teak *Tectona grandis* with sparse understorey	(Whitten et al., 1996)
Tropical evergreen forest	Primary forest	> 400 m on Mt. Baluran and in small patches around wet creeks in lowland	Closed-canopy forest with evergreen plants including *Homalium foetidum*, *Aleurites moluccana*, *Drypetes ovalis*	(Nijman, 2004; Winnasis et al., 2011)
Beach forest	Secondary forest	Behind mangrove forest, in coastal areas	Closed-canopy forest on sandy soils with *Terminalia catappa*, *Barringtonia racemosa*, *Hibiscus tiliaceus, Pandanus tectorius* and the palms *Corypha utan* and *Borassus flabellifer*	(Winnasis et al., 2011)
Mangrove forest	Mangrove forest	Intertidal areas	Mangrove specialists including *Rhiziphora* spp., *Sonneratia alba*, *Avicennia alba*	(Winnasis et al., 2011)

Appendix 4.2 Comparison of uniform, half-normal and hazard-rate detection functions with and without adjustment terms or habitat openness as a covariate. The best model did not include the habitat openness 2-level covariate. C-vM: Cramér-von Mises goodness-of-fit test; $\hat{P}_a$: average detectability; se($\hat{P}_a$): standard error of average detectability; ΔAIC: difference in Akaike Information Criterion compared to the best model.

Key function	**Formula**	**C-vM *p*-value**	$\hat{P}_a$	**se($\hat{P}_a$)**	**ΔAIC**
Uniform with one cosine adjustment term	NA	0.960	0.595	0.055	0.000
Half-normal	~1	0.983	0.610	0.072	0.203
Uniform with two simple polynomial adjustment terms	NA	0.670	0.702	0.042	1.481
Hazard-rate	~1	0.991	0.652	0.094	2.059
Half-normal	~openness	0.981	0.610	0.072	2.175
Hazard-rate	~openness	0.988	0.644	0.096	4.006

Appendix 4.3 Detection functions for Black-winged Myna recorded at Baluran National Park (open circles are individual detections). **(A)** Uniform key function with one cosine adjustment term, **(B)** half-normal key function, **(C)** uniform key function with two simple polynomial adjustment terms, **(D)** half-normal key function with habitat openness as a covariate, **(E)** hazard-rate key function, and **(F)** hazard-rate key function with habitat openness included as a covariate. In both **(D)** and **(F)**, the two slopes of open circles show the influence of the two-level habitat openness covariate: the detection probability was slightly higher further away from the observer in open habitats (slope of open circles above the line).



Appendix 4.4 Variable importance for habitat suitability models generated using different SDM algorithms for Black-winged Myna. The most important variable in the averaged model for each algorithm is in bold.

Explanatory variable	GLM	GAM	MaxEnt
Habitat edge	0.20	0.20	0.45
Land-cover type (factor with nine levels)	**0.79**	**0.80**	**0.51**
NDVI	0.15	0.15	0.14

Chapter 5

5 Assessing the near-future population viability of the Critically Endangered Bali Myna *Leucopsar rothschildi*

Abstract

Population viability analysis is a simulation tool for determining the probable trajectory of a population that can be used to quantify extinction risk, identify threats, and compare management interventions. The iconic Bali Myna (*Leucopsar rothschildi*), Bali's only endemic bird species, is Critically Endangered after decades of overexploitation for the cage-bird trade. Despite decades of conservation efforts, in the 2000s there were probably no birds left in the wild. Since then, various management interventions at Bali Barat National Park (BBNP), including consistent annual releases of a greater number of captive-bred birds and a revamped artificial nestbox scheme, have led to population growth. To plan for the next decade of conservation management, I modelled the free-living Bali Mynas at BBNP aiming to explore the effects of (1) changes to population supplementation and (2) an increase in the level of illegal trapping (still the main threat). A baseline model was validated using population census and captive-bred release data from 2012–21, and the model was projected to 2032. The population was predicted to increase under current levels of supplementation, and stopping supplementation in five years had only a small effect. Two trapping methods (nest poaching, with only chicks taken, and mist-netting, with chicks and adults taken) and three trapping volumes (low, 40 birds year^{-1}; medium, 80; and high, 120) were modelled. The population was resilient to the lowest level of offtake with or without population supplementation. Mist-netting appeared more damaging than nest poaching, but the highest level of trapping produced a high risk of population decline irrespective of method used. On current trajectory, the population is approaching self-sustainability in the next 5–10 years, as long as trapping levels do not increase. The supplementation programme at BBNP could either be scaled back or repurposed as a translocation project to repopulate vacant parts of the myna's range. The model was parameterised with the few data available for the species, so its findings are provisional. Nevertheless, it provides insights that appear robust. With the species now living alongside local communities, an opportunity exists to enlist their support and to work on generating new data to improve the model.

5.1 Introduction

Population viability analysis (PVA) is a useful tool for determining the probable trajectory of a population of an animal or plant species based on a combination of species-specific life-history and demographic data and the environmental variables likely to affect the population over time across the range it occupies. Conservation biologists use PVA to quantify overall extinction risk, identify particular risk factors (parameters influencing population declines), itemise management measures to improve the sustainability of the population and set numerical thresholds by which to gauge the success of such measures (Akçakaya and Sjögren-Gulve, 2000; Beissinger and McCullough, 2002).

The broad requirements population modelling has for both biological and environmental data make it both rigorous and versatile, meaning that it can be used in many ways to provide a solid scientific basis for conservation management decisions. In the realm of bird conservation it has been used to guide the management of populations under threat from alien invasive predators (Hegg et al., 2013), synanthropic corvids (Peery and Henry, 2010), disease transmission (Iverson et al., 2016), volcanic eruptions (Oppel et al., 2014), stochastic drought (Beissinger, 1995), windfarms over time (Carrete et al., 2009), targeted human exploitation (Valle et al., 2018), incidental human exploitation (Lu and Sun, 2011), Allee effects caused by habitat degradation (Bazzano et al., 2014), multiplicities of factors (García-Ripollés and López-López, 2011) and unexplained declines (Fantle-Lepczyk et al., 2018). It can help decide whether to elect for ex situ intervention (Dolman et al., 2015), what impact egg and nestling removal might have on donor populations in reintroduction programmes (Margalida et al., 2015), what parameters (e.g. carrying capacity and sex ratio) to influence in reintroduction programmes (Zhang et al., 2021), how many individuals to supplement a population over what time-frame (Bernardo et al., 2014), when to cease such supplementations (Schaub et al., 2009), and how to choose between interventions based on returns on investment (Sebastián-González et al., 2011).

However, it is widely acknowledged that the accuracy of population model projections is constrained by the quality and quantity of the data used to parameterise the model, and the construction of the model itself (Boyce, 1992; Coulson et al., 2001; Chaudhary and Oli, 2020). The data required to identify the demographic causes of changes to the population growth rate poses a significant challenge, especially to the study of species with small or declining numbers for which a clear understanding of population dynamics is of immediate applied value (Heppell et al., 2000; Sim et al., 2011). For many such species, long-term studies of reproductive success and survival from which to calculate demographic rates and their temporal variance are unavailable, in which case values often have to be assumed even though they are difficult to validate and may produce misleading results (Banks et al., 2010). In some cases, the construction of models does not accurately reflect the life history of the study species, with simple accounting issues in vital rates, particularly the fertility coefficient, prevalent in the literature (Kendall et al., 2019). Incorporating uncertainty into models inevitably diminishes the confidence they can be allowed, but it makes for more truthful assessments of probability and possibility (Parysow and Tazik, 2002; Oppel et al., 2014). The uncertainties inherent in population models have led to a shift away from quantifying an extinction risk over a given timeframe (e.g. the minimum viable population concept, Shaffer and Samson, 1985) towards a focus on sensitivity analyses, which are used to assess the relative effects of different threats or management treatments on population growth and persistence (Reed et al., 2002).

The Bali Myna (*Leucopsar rothschildi*) is a Critically Endangered species confined to Bali Barat National Park (BBNP) in north-west Bali, Indonesia, where it became so rare through the conversion of its open monsoon woodland habitat and, especially in the past fifty years, intensive trapping for trade that it now survives in the wild only through the release of captive-bred birds (Collar et al., 2001; Sutedi, 2012; Hernowo, 2017; Yuni et al., 2022). This programme of supplementation (and indeed likely complete replacement) of the wild population by birds of captive origin has not been documented in great detail; there are no data on the ages, sex or relatedness of releasees. Moreover, although birds have responded favourably to the provision of nestboxes and food in the vicinity of release sites, there has been no systematic post-release monitoring of birds in terms of survival, dispersal, habitat use or, apart from a single recent study (Yuni et al., 2022), productivity. These gaps in information notwithstanding, developing a population model is now desirable to begin to understand the degree to which the free-flying Bali Mynas at BBNP have acquired the characteristics of a self-sustaining population and how they might further be supported by management measures in the near future. In my view the value of building a preliminary model, which can of course be improved as the missing data are gathered, is greater than the value of waiting for possibly many years in order to obtain those missing data. In large part the real value of making an early start on a population model is to demonstrate more clearly which aspects of the biology and environment of the Bali Myna need particular scientific attention, as well as to inform management practices such as the current programme of population supplementation with captive-bred birds.

Important practical questions related to the viability of the free-flying population of Bali Mynas are (1) how sensitive is that population to changes in the supplementation regime? (2) What are the effects of trapping method and volume on the population? And (3) what are the compound effects of differential supplementation volume, trapping volume, and trapping method? To address these questions, I developed a matrix population model to explore the population's projected growth and persistence under scenarios relevant to each question.

5.2 Methods

5.2.1 Model development

The development of the matrix population model was undertaken in R (R Core Team, 2022; see Appendix 5.1) using packages 'popbio' v.2.7 (Stubben and Milligan, 2007), 'mvtnorm' v.1.1-3 (Genz and Bretz, 2009), and 'msm' v.1.6.9 (Jackson, 2011). Because I aimed to assess the short-term prospects of the population, model projections were made 10 years into the future and no density-dependent factors limiting population growth were included. The dataset used to validate the model covered the period 2012–21. Like most other members of the Sturnidae—exceptions being the Common Starling (*Sturnus vulgaris*) (Lack, 1948) and species considered invasive in parts of their range due to human-assisted introductions (e.g., Common Myna *Acridotheres tristis*; Lermite et al., 2021)—life-history data for the Bali Myna in the wild is lacking. The absence of age-specific mortality or fecundity data for the Bali Myna precluded the use of an age-structured matrix model, meaning I instead developed a stage-based Lefkovitch model (Lefkovitch, 1965; Caswell, 2001). A female-only model was constructed because females typically limit reproductive output and including both sexes can lead to underestimates of extinction risk (Brook et al., 2000). The initial population size used in the model was fixed at 15 adults, with no first-year birds, given that the population had been even smaller in the preceding years and that it is likely all these birds were released from the captive-breeding programme and thus had already lived for more than one year. The model was parameterised using data collected from BBNP (Yuni et al., 2022), captive Bali Mynas (Earnhardt et al., 2009; Ross et al., 2021), and estimated or assumed values based on studies of other starling species.

Bali Mynas moult into their adult plumage and can join the breeding population at the end of their first year (Collar et al., 2001; Ross et al., 2021). I therefore included two life-stages in the model: first-year and after first-year birds (adults). A time-step of one-year, spanning one pre-breeding census to the next, was used because Bali Mynas have a defined breeding season which coincides with the wet monsoon in north-west Bali from November to May (Yuni et al., 2022). Stochastic threats to the population in the form of demographic and environmental fluctuations were incorporated in the model by including an arbitrary small amount of variance (SD) of 0.05 to the mortality and fecundity rates (see Table 5.1).

The Bali Myna population in and around Bali Barat National Park is isolated and occupies a continuous area, with birds moving between locations several kilometres apart (T.M.S. pers. obs.); it was therefore considered a single closed biological population without immigration or emigration (Squires et al., in review). The number of captive-bred birds released from 2012–21 was known, but because these numbers have varied and gradually risen, I used the average number of birds released annually over the last five years (59 individuals) as the baseline continued supplementation rate. The finite rate of population growth λ was set to 1.37 and calculated using the known population growth over the period 2012–21 using the following equation (Caswell, 2001):

$$\lambda = \sqrt[x]{\frac{Nt + x}{Nt}}$$

where $Nt + x$ and Nt are the Bali Myna population size at two different times separated by x number of years.

The maximum recorded lifespan of Bali Mynas in captive populations is more than 20 years (Ross et al., 2021). However, the age structure of captive populations reveals that this is an extreme value and that the maximum age almost never exceeds 16 years; this is also the maximum breeding age for females (Ross et al., 2021). Given that birds in the wild face competition for limited resources and predation pressure, I used this lower maximum age of 16 years in the model; in reality, the mortality rate I applied made it unlikely that birds would reach this age anyway, rendering the parameter’s impact on the model outcome minimal.

There is no information available on the mortality rate of Bali Mynas. Assumed values were used in the model based on the assumptions that (1) annual mortality would be considerably lower than the values reported for the Common Starling of 55% (Coulson, 1960) owing to differences in r and K selection pressures in temperate latitudes and the tropics respectively (Martin, 2004), and that (2) mortality would be higher in first-year birds than adults (Loery et al., 1987). I included a first-year survival of 55% following values reported in Ross et al. (2021) and an adult survival of 80%.

Bali Mynas establish stable pair-bonds for at least a single breeding season, as evidenced by their nodding display and allopreening behaviour (Harrison, 1963; Collar et al., 2001). In the absence of specific data for a free-living Bali Myna population, I assumed an even sex ratio at birth. I calculated the proportion of mature females that breed using data collected in and before the 2019–20 breeding season. Then it was estimated that 30 females bred (Yuni et al., 2022), while the census conducted beforehand reported a total of 256 mynas (Bali Barat National Park, 2021), from which I subtracted the number released in the year preceding the census (n = 108), assuming they would be unlikely to have had time to form a pair and breed. Of the 148 remaining birds, two-thirds were assumed to be mature adults (n = 98), following the assumption used in the IUCN Red List effective population size calculations when no other data are available (R. W. Martin pers. comm.). After applying the sex ratio (0.5) to that total, I estimated that the population in 2019–20 comprised 49 mature adult females, meaning approximately 61% successfully bred (30/49). Between-year fluctuations in breeding were accounted for by adding variance to the coefficient (SD = 0.1).

Bali Mynas can successfully fledge multiple broods during a single breeding season (Collar et al., 2001). In the model, I attributed a random number of successful nesting attempts to each breeding female based on the known distribution of successful attempts (mean = 1.6 attempts, SD = 0.9) from the 2019–20 breeding season (Yuni et al., 2022). To calculate the number of fledged mynas in a year I multiplied the known average number of chicks fledged in the 2019–20 breeding season (1.6) by the number of successful breeding attempts. Using this ‘basic’ fecundity coefficient, I generated rates for three classes of breeding Bali Myna according to the following assumptions: (1) the fecundity of wild-born Bali Mynas would be greater than the basic value, which was calculated from a population comprising a large number of captive-bred birds (Yuni et al., 2022); (2) captive-bred birds would have lower fecundity in their first year following release, as some of these birds would be unlikely to pair up before the beginning of the breeding season; (3) birds at the end of their first year, and thus breeding for the first time, would exhibit lower fecundity than the basic value.

The life-history traits used in the baseline model are summarised in (Table 5.1). The model was validated using data from the annual pre-breeding census undertaken by BBNP following a long-established protocol (van Balen, 1995; Bali Barat National Park, 2021). For each different scenario, the model was run 1,000 times.

Table 5.1 Values used to parameterise the baseline Bali Myna population model.

Model parameter	Value used
Number of model runs	1,000
Number of populations modelled	1
Number of years modelled	20 (2012–2032)
Immigration rate	None
Emigration rate	None
Life stages	juvenile (1st year); adult (after 1st year)
Maximum lifespan	16 years
Baseline λ (finite rate of population growth)	1.37
Annual first-year survival (SD)	55% (0.05)
Annual adult survival (SD)	80% (0.05)
Reproductive system	Monogamy
Age of first reproduction	1 year
Maximum age of reproduction	16 years
Sex ratio at birth (female: male)	0.5
Proportion of adult females breeding (SD)	0.61 (0.1)
Number of breeding attempts per year	1–4
Number of breeding attempts per breeding adult female per year (SD)	1.6 (0.9)
Mean number of chicks fledged per attempt	1.6
Number of captive-bred adults released (2012–2021)	10, 24, 14, 12, 22, 28, 42, 76, 80, 68
Penalty to fecundity of birds at the end of their first year	30%
Number of captive-bred birds released in years predicted (2021 onwards)	59 adults
Penalty to survival rate of released birds for first year post-release	30%
Penalty to fecundity of released birds for first year post-release	30%
Carrying capacity	None used
Inbreeding depression	No inbreeding depression assumed
Initial population size	Juvenile = 0, adult = 15

5.2.2 Population supplementation scenarios

To assess the effect of altering the population supplementation regime at BBNP on the Bali Myna population, I parameterised the model using two scenarios that differed from the baseline model (i.e., continued supplementation). In the first scenario, releases stopped after 5 more years of supplementation at the baseline model rate (59 birds year^{-1}), and in the second supplementation halted immediately after the model validation period (i.e., 2022, the first projected year).

5.2.3 Trapping method and volume scenarios

Trapping is the main threat facing the Bali Mynas at BBNP (Squires et al., in review). Nest poaching may always have been the commonest form of trapping, as suggested by the relatively greater number of females entering markets in the past (poachers typically operate at night when females are normally sitting in the nest and males roosting outside nearby) (van Balen et al., 2000). Moreover, artificial nestboxes and other relatively accessible and conspicuous locations, such as bee boxes (Yuni et al., 2022), occupied by birds for breeding are obvious targets for poachers; and some well-known natural nest cavities are repeatedly affected (H. Kusumanegara, pers. comm.). However, mist-netting is also known to occur around the park on cultivated land (T.M.S. pers. obs.). I therefore included these two trapping methods as scenarios to model their likely effect on the population and used three different volumes of trapping: low, involving a 10% annual offtake of the current population (40 birds year^{-1}); medium, with a 17.5% annual offtake (80 birds year^{-1}; and high, with a 25% annual offtake (120 birds year^{-1}). For the mist-netting scenario, I assumed that birds from each life stage would be randomly affected. In the nest-poaching scenario only chicks were affected. I ran the simulation model with these scenarios and two of the population supplementation scenarios ('continued supplementation' and 'no future supplementation') to assess the compound effect of the different scenarios implemented.

5.3 Results

5.3.1 Model validation

The modelled population trend shows a strong similarity with the known pre-breeding population size from 2012–21, with an average difference between the modelled trend and known population sizes of 8% across the validation period (Figure 5.1). The greatest differences in population size between the simulated and validated trend occurred in 2015 and 2018, with a 14% and 13% difference respectively, but in the final two years of the validation period the simulated population trend was within 6% of the validated census data, amounting to an absolute difference of no more than 25 individuals.

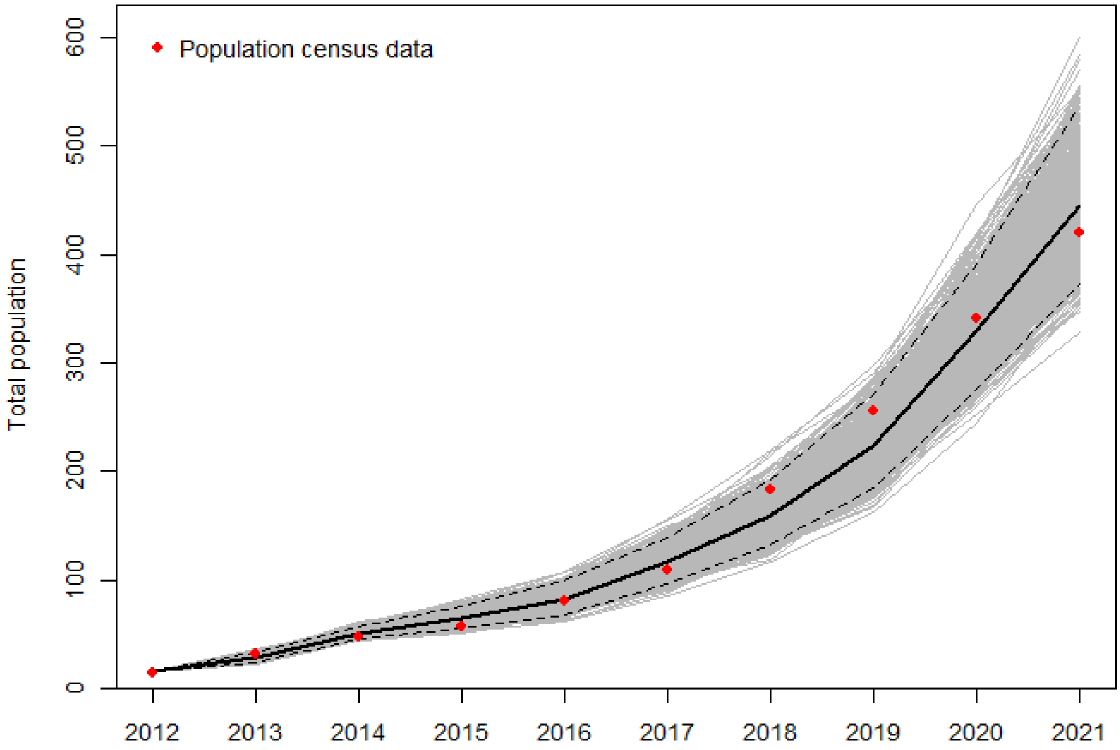


Figure 5.1 Population census data (red points) collected at Bali Barat National Park compared with the baseline matrix population model population trajectories after 1,000 simulation runs (grey lines). The solid and dashed black lines are the mean simulated population trend and 95% confidence intervals respectively.

5.3.2 Impact of population supplementation

Continuing to supplement the population over the entire modelling period at the baseline level resulted in population growth with high certainty (Figure 5.2). Ceasing supplementation immediately resulted in slower population growth, and the mean predicted population size after 10 years was 37% smaller than if supplementation was continued (2,021 vs. 3,184 individuals), and 17% smaller than the population if supplementation stopped after five years (2,639 vs. 3,184 individuals). Ceasing population supplementation after five years likely involved almost no risk (<1%) that the population would remain below 1,000 individuals after 10 years, whereas this risk increased to 5% if supplementation stopped immediately.

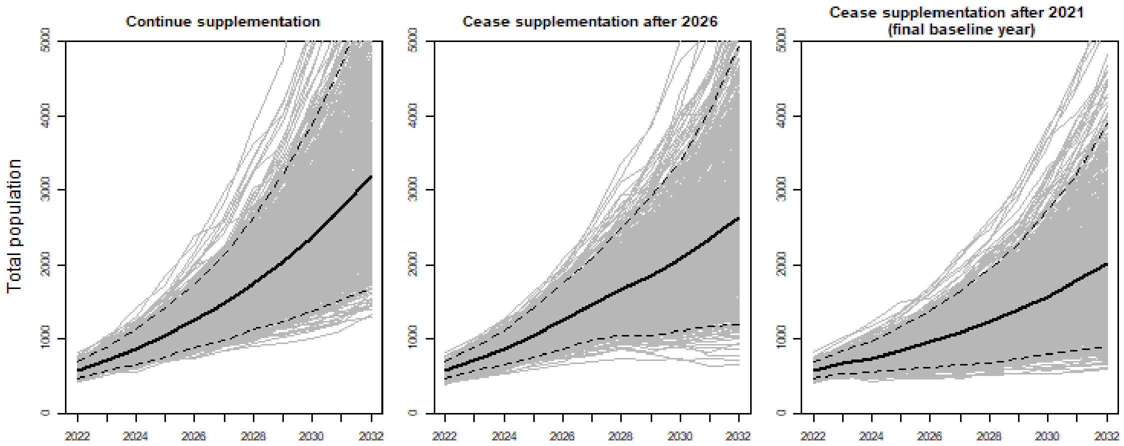


Figure 5.2 Simulated Bali Myna population trajectories (grey lines) using the baseline model with continued supplementation (59 birds year^{-1}), supplementation ceasing after 2026 (five years of releases at 59 birds year^{-1} past the final baseline year), and supplementation ceasing immediately (after 2021). Solid and dashed lines show the simulated population trend and 95% confidence intervals respectively.

5.3.3 Effects of future trapping

There were strong differences in the projected Bali Myna populations depending on the type and volume of trapping imposed and whether or not supplementation continued (Table 5.2 and Figure 5.3). The modelled population was more sensitive to mist-netting than nest poaching, as the former involves the offtake of adults as well as juveniles. The population was resilient to trapping of both types with continued supplementation, except that under a high volume of mist-netting there was a sharp increase in the risk of the population remaining below 1,000 individuals. At the medium or high volume of trapping and without supplementation the population showed limited growth in most scenarios and exhibited an elevated risk of declining and remaining below 1,000 individuals. The benefit of supplementing the population increased as trapping volume increased.

Table 5.2 Summary of modelled population projections based on scenarios used to examine the effects of nest-poaching and mist-netting with and without population supplementation.

	Nest poaching scenario		**Mist-netting scenario**	
Trapping volume	Population size after 10 years: supplemented vs. unsupplemented (% reduction)	% risk of population <1,000 after 10 yrs: supplemented vs. unsupplemented	Population size after 10 years: supplemented vs. unsupplemented (% reduction)	% risk of population <1,000 after 10 yrs: supplemented vs. unsupplemented
Low (40 birds year^{-1})	2,739 vs. 1,582 (42%)	<1% vs. 18%	2,435 vs. 1,295 (47%)	<1% vs. 37%
Medium (80 birds year^{-1})	2,225 vs. 1,156 (48%)	<1% vs. 48%	1,734 vs. 654 (62%)	13% vs. 79%
High (120 birds year^{-1})	1,769 vs. 719 (59%)	13% vs. 76%	1,072 vs. 155 (86%)	52% vs. 97%

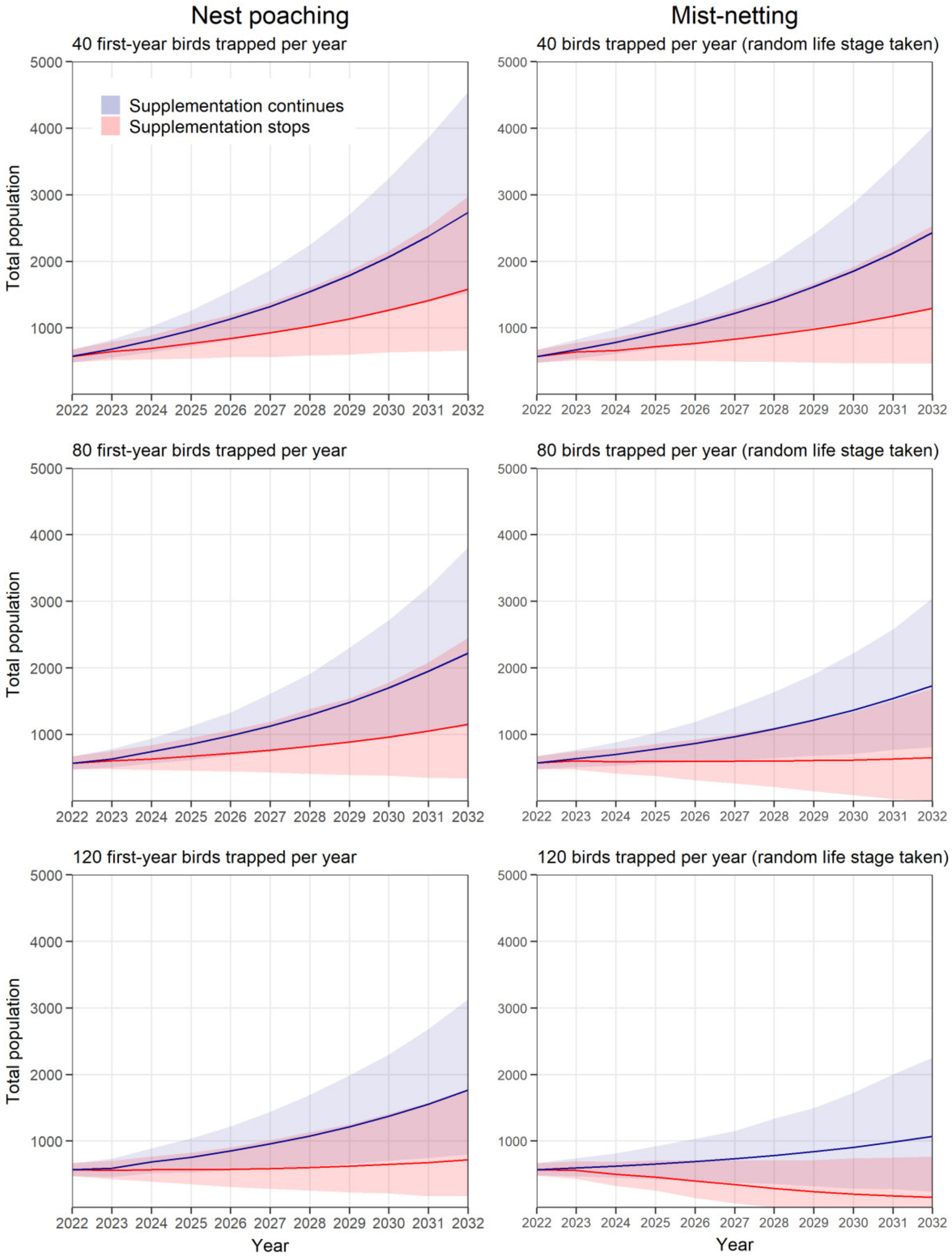


Figure 5.3 Simulated trajectories of the Bali Myna population when affected by nest poaching (left side), in which only first-year birds are taken, and mist-netting (right side), with first-year and adult birds affected. The lines and shaded areas show the population trend and 95% confidence intervals respectively after 1,000 simulation runs. Each plot shows the effect of continuing (blue line) or ceasing (red line) population supplementation with captive-bred birds.

5.4 Discussion

Until the last decade, conservation efforts, including periodic releases of captive-bred birds, failed to re-establish a viable Bali Myna population at BBNP, such that in 2006 the species may have briefly gone extinct in the wild (Jepson, 2016). Since 2012 however, the free-living population has shown a strong increase, evidently owing in part to more consistent releases of larger numbers of captive-bred birds and in part to these birds being released in new locations that benefited from the presence of more suitable habitat, many nestboxes, and park staff (Bali Barat National Park, 2021; Squires et al., in review). My near-future demographic model shows that if conditions remain stable the population at BBNP will continue to increase and the need for supplementation will steadily diminish to the point when its cost outweighs the benefit. This is a conclusion that I feel is sufficiently important and robust to merit the public trial of time, but to reach it I used a model whose weaknesses I must acknowledge and itemise.

First, the survival rates used in the model were based on assumptions and the fecundity rates were based on breeding productivity data collected over a single breeding season (Yuni et al., 2022). While in some cases life-history parameters can be assumed from closely-related surrogate species (Banks et al., 2010), I had no alternative data to draw on, partly because no tropical members of the Sturnidae family have been the subject of population modelling, but also because the Bali Myna, as the sole representative of the genus Leucopsar, is taxonomically unique within the Sturnidae (Lovette et al., 2008) and evidently—albeit in ways not yet understood—strongly specialised (see Yuni et al., 2022). On the other hand, it was not critical for these rates to be completely accurate because they were held constant across all models, which is a real strength of PVA (Beissinger and McCullough, 2002; Perkins et al., 2008).

Inbreeding depression, which can significantly impact species with small populations (Crnokrak and Roff, 1999) but is often overlooked in population models (Frankham et al., 2014), was deliberately excluded from consideration. No studies of this have ever been undertaken on the relatively large captive Bali Myna population from which most, if not all, BBNP's free-flying birds are descended and supplemented (Bali Barat National Park, 2021).

A limitation of stage-based matrix models is that they are a simplified representation of the population, and that all individuals within a stage-class are assumed to have the same demographic rates (Boyce, 1992). The solution to this is for more detailed long-term mark–recapture studies to discern age- or stage-specific demographic rates to increase the model's ability to simulate the population's structure. However, even long and detailed mark-recapture datasets can still have high levels of uncertainty associated with survival estimates (Tinoco et al., 2019).

The current level of trapping at BBNP is, understandably, not known, but the offtake was implicitly included in my model because it was validated using annual population censuses from the last decade. My projections indicate that, currently, trapping pressure is sufficiently low for the population to continue to grow. However, the numerical range that demarcates sustainable from unsustainable harvest can be disconcertingly narrow, especially when expressed as percent offtake of the initial population (Valle et al., 2018), indicating the need for caution and ever better data. My model also confirmed a common finding in PVA (Mortensen and Reed, 2016), that the trapping of adults has a stronger impact on population trajectories than the taking of juveniles. Although I simplified the effects of mist-netting and nest poaching for the modelling procedure, in reality both practices probably involve adults being trapped, particularly if nest poachers, who typically operate at night, take the sitting adult female as well as the young (Sieber, 1983; van Balen et al., 2000), thereby skewing the sex ratio and greatly limiting population growth.

My model is also unable to accommodate extraneous factors that could affect one or more of the vital rates in a density-dependent manner, even at relatively small population sizes. Nest site availability is one such factor. The felling of large mature trees in BBNP and their replacement with trees still too young to bear natural cavities necessitated a nestbox scheme to compensate (Whitten et al., 1996; Yuni et al., 2022). However, Bali Mynas are territorial when breeding, so a density-dependent effect on the reproductive output of the population could result from insufficient numbers and dispersion of both natural cavities and nestboxes. New release sites and new locations to which Bali Mynas have naturally spread need careful checking to ensure that there are suitable nest cavities in the vicinity, if need be through the provision of nestboxes.

Captive breeding programmes for the recovery of endangered species are an expensive option fraught with potential problems, but are an effective tool in the last line of defence against species extinctions (Snyder et al., 1996). At BBNP, the captive breeding and release programme has formed a major component of recent conservation efforts and has contributed to the strong growth of the wild population (Sutedi, 2012;

Squires et al., in review). While my model suggests that supplementation could be discontinued in five years with almost no impact on the projected population growth, the decision to do so must give due consideration to the likelihood of the rate of trapping to increase in the near future, especially if birds living outside the park in anthropogenic landscapes face a higher risk of trapping.

Over recent years, the increase in availability of captive-bred Bali Mynas in markets due to commercial breeding and BBNP's innovative bird loaning scheme has led to the black market price for an individual bird falling from US$2,000 or more in the 1990s (PHPA and BirdLife International-IP, 1997) to US$752–1,278 in the late 2010s (Nijman et al., 2017). Despite this, however, the average monthly salary for an Indonesian agricultural worker in 2020 was around IDR 2 million or US$144 (Badan Pusat Statistik, 2022), so the risk of trapping is likely to remain high. Nevertheless, the direct protection afforded Bali Mynas by national park staff combined with the penalties in place for illegally capturing, trading, and keeping the birds should, in theory at least, form a considerable deterrent against trapping (Didarali et al., 2022). Perhaps, however, a greater deterrent may come from the communities amongst whom the species has in the past decade begun to appear (Knapp, 2012). The combination of the myna's extraordinary beauty, iconic nature, and tiny range represents a potential source of pride in and therefore protection by those communities. There has certainly never been a better opportunity to secure the Bali Myna's future than now, through the engagement of the human communities that find themselves hosting the species in north-west Bali, both through outreach programmes at schools and public space, and through the creation of alternative livelihoods (Squires et al., in review). Moreover, if the trust of local people can be won, the opportunity also exists for an intensive community-supported research programme that can begin to deliver the data from which the viability of the myna's population can be determined with real rigour.

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Appendices

Appendix 5.1 R script used to develop and implement Bali Myna matrix population model.

```
#' PROJECT: Bali myna population viability analysis
#' CONTENTS:
#' - building a matrix population model for validation and projection of future scenarios of
population demographics
#'  DEPENDENCIES:
#'  - Code documents
#'  - Data files
#' AUTHOR: tom squires
# PREAMBLE =========================================
rm(list=ls())
# Directories ------------------------------------------------------------------
# Define directories in relation to project directory
Dir.Base <- getwd()
Dir.Data <- file.path(Dir.Base, "data")
Dir.Exports <- file.path(Dir.Base, "exports")
# Create directories
Dirs <- c(Dir.Data, Dir.Exports)
CreateDir <- sapply(Dirs, function(x) if(!dir.exists(x)) dir.create(x))

## FUNCTION =======================================================
# implementing a stage-based female-only matrix population model (MPM) to simulate the Bali
Myna population trajectory over the modelling period for different scenarios

balimynaMPM <- function(
  # default settings for the Bali myna MPM
lambda = 1.374, # = initial estimate of population increase, i.e., 10 root of (N in 2021 / N in 2012)
= (420/15)^(1/10)
stages = c("Juvenile","Adult"), # life stages
T1 = 1, # length of lifestage 1 (juvenile)
T2 = 15,
# length of lifestage 2 (adult) max lifespan in Ross et al. 2021 "Population Analysis  &
# Breeding and Transfer Plan: Bali Myna (Leucopsar rothschildi) AZA Species Survival
# Plan® Green Program"
firstYear = 2012, # first year of modelling period
historicalYears = 10, # number of years with historical data
projectionYears = 10, # number of years to project into future
nJuvsInit = 0, # initial number of juveniles in the population at t = 0
nAdultsInit = 15, # initial number of adults in the population at t = 0
nsimul = 1000, # number of times to run the simulation
Sjuv_mean = 0.55, # mean juvenile survival
Sjuv_sd = 0.05, # variation in juvenile survival
Sadult_mean = 0.8, # mean adult survival
Sadult_sd = 0.05, # variation in adult survival
female.ratio = 0.5, # birth sex ratio
female.breeders_mean = 0.61,
# mean proportion of females breeding based on Yuni et al. 2022 breeding females = 30,
# BBNP 2021 census = 256 with the 108 birds released since Oct 2018 removed = 148. 148
```

```
# * 0.66 (R. Martin's 2/3 rule for adult/juv in IUCN Red List effective population size calcs)  # =
97.6 * 0.5 (sex ratio) = 48.8. 30/48.8 females breeding = 0.62
female.breeders_sd = 0.1, # variation in proportion of females breeding
F.penalty.juv = 0.7,
# penalty to first year birds during first breeding season because their fecundity is probably
# lower than adults
chicksPerAttempt = 1.6,
attemptsMin = 1,
attemptsMax = 4,
attemptsMean = 1.6,
attemptsSD = 0.9,
# Fmean = 1.6, # mean number of chicks per brood (not used)
# Fsd = 1.1, # variation in number of chicks per brood (not used
 supplementation = ceiling(c(10,24,14,12,22,28,42,76,80,68)),
# known supplementation levels from BBNP (2021)
future.supplementation = 59,
# continuing supplementation at the average number released over the last five years
supplementation_penalty_S = 0.3,
# penalty to adult survival of birds released previous yr; lower values = higher pen
supplementation_penalty_F = 0.3,
# penalty to fecundity of birds released previous yr; lower values = higher pen
delayedSupplementation = FALSE,
supplementationDelay = 5,
densityDependence = FALSE, # if TRUE,  nest site limitation is used to impose D-D
N_nestSites = 150,
# current number of nest sites, used as baseline from which to increase the nest sites using
# the growth rate
nestSiteGrowthRate = 1.05, # rate of nest site increase per year
max_nest_sites = 600,
# arbitrary maximum number of nest sites estimated for the study region to be able to put
# a density-dependent term in the model
# trapping scenario
trapping = FALSE,
trappingType = "nest", # options = "nest" or "mist-net"
trappedN = 40 # total number trapped (males + females)
)
{
#####--------------------------------#####
## load packages
#####--------------------------------#####
install.load.package <- function(x){
if(!require(x, character.only = TRUE))
install.packages(x, repos = 'http://cran.us.r-project.org')
require(x, character.only = TRUE)
}
package_vector <- c("popbio","mvtnorm","msm")
sapply(package_vector, install.load.package)
#####--------------------------------#####
## establish some model values based on the input parameters
#####--------------------------------#####
# define length of modelling period
yearsModelled <- historicalYears + projectionYears
lastYear <- firstYear + yearsModelled
```

```
# create vector of number of nest sites available in each year
nest_sites <- round(N_nestSites * nestSiteGrowthRate^(0:yearsModelled), digits = 0)
# number of nest sites increases each year by x% (installation of more nest boxes)

## create empty matrix to be filled during each ITERATION of the simulation (n = 1000)
N <- matrix(0, nrow = length(stages), ncol = yearsModelled + 1)
rownames(N) <- stages # name the rows
colnames(N) <- firstYear:lastYear # name the cols with actual years

# the first year of the model contains the known population figures
N[,1] <- ceiling(c(nJuvsInit,nAdultsInit) * female.ratio)

# create a matrix to contain the population size in each year and run of simulation
NTot <- matrix(0, nrow = nsimul, ncol = yearsModelled + 1)
rownames(NTot) <- 1:nsimul # name rows
colnames(NTot) <- firstYear:lastYear # name cols with actual years

# create a matrix to contain the proportion of birds that were juvenile in each year
propJuv <- matrix(0, nrow = nsimul, ncol = yearsModelled + 1)
rownames(propJuv) <- 1:nsimul # name rows
colnames(propJuv) <- firstYear:lastYear # name cols with actual years

# numbers supplemented needs to be converted into females only:
supplementationFemales <- supplementation * female.ratio
future.supplementationFemales <- future.supplementation * female.ratio

#####--------------------------------#####
## FUNCTIONS -----------------------------
#####--------------------------------#####

# juvenile survival (growth into adult stage)
juvenile.survival.growth <- function(){
# juvenile survival = 0 because this life stage lasts for t = 1 (they either become adults or
# die)
s1 <- round(rtnorm(n = 1, mean = Sjuv_mean, sd = Sjuv_sd,lower = 0, upper = 1), digits = 3)
# generate a random annual survival rate from the survival rate for juveniles (mean and
# sd)

p1 <- round(s1 * (1 - ((((s1/lambda)^T1) - ((s1/lambda)^(T1-1))) / (((s1/lambda)^T1)-1))), digits =
3)
g1 <- round(s1 * ((((s1/lambda) ^T1) - ((s1/lambda) ^(T1-1))) / (((s1/lambda)^T1)-1)), digits = 3)

result <- matrix(c(s1, p1, g1),
          nrow = 1,
          dimnames = list("",c("s1","p1","g1")))
 }

## adult survival -------------------------------------------------------
adult.survival <- function(){
s2 <- round(rtnorm(n = 1, mean = Sadult_mean, sd = Sadult_sd, lower = 0, upper = 1), digits = 3)
p2 <- round(s2 * (1 - ((((s2 / lambda) ^ T2) - ((s2 / lambda) ^ (T2 - 1))) / (((s2 / lambda) ^  T2) -
1))), digits = 3)
# this is a correction to the adult survival rate to account for the loss of individuals that
# reach the maximum life expectancy of the species
```

```
# modify survival for adults that were released in t - 1
p2_released <- round(p2 * supplementation_penalty_S, digits = 3)
# penalty to survival of previous year's released birds

p2_adjusted <- (p2_released * prop.released) + (p2 * (1 - prop.released))
# combine the survival rates for wild and released adults based on relative proportions
}

## adult fecundity --------------------------------------------------------
fecundity <- function(){
# A few important points about this function:
# 1. number of attempts is drawn from the number of breeding females, which is then
# multiplied by the fixed parameter of chicks per attempt taken from Yuni et al 2022)
# 2. my census data are pre-breeding. So, birds with age 0 in the current timestep do not
# breed until the next timestep because the next breeding season follows the next census.
# This means juvenile females should be excluded from fecundity calculations, unlike if it
# was a post-breeding census.
# 3. as it is a pre-breeding census, my fecundity rates should NOT be adjusted down to
# account for mortality (birds breed immediately after census - presumed that there is
# almost no mortality). Fecundity rates should NOT be subjected to stage-specific survival # rate.
# 4. it is likely that birds breeding for the first time are not as productive as full adults, partly
# because a lot of the 1st year males may not even breed. So a penalty should be applied to
# the fecundity rate of 1st year birds (in addition to the survival rate penalty)

  if(female.breeders_mean < 1){
   female_breeders <- rtnorm(n = 1,
                   mean = female.breeders_mean,
                   sd = female.breeders_sd,
                   lower = 0,
                   upper = 1) # randomly select % of females that breed
  } else{female_breeders <- 1}

N_female_breeders <- round((N_adults_female + N_juvs_female) * female_breeders, digits = 0)
    if(N_female_breeders > 0){
     N_attempts <- sum(floor(rtnorm(n = N_female_breeders,
                    mean = attemptsMean,
                    sd = attemptsSD,
                    lower = attemptsMin,
                    upper = attemptsMax))) # total number of breeding attempts
  } else{N_attempts <- 0}
  N_chicks <- N_attempts * chicksPerAttempt * female.ratio # female chicks per attempt is fixed based on productivity data from 2019
  if(N_chicks > 0 & N_female_breeders > 0){
  f2_basic <- N_chicks / N_female_breeders
# fecundity of released birds is penalised and subject to the survival rate of first year birds
# (PREBREEDING CENSUS - chicks must survive the year first)

f2_released <- f2_basic * supplementation_penalty_F * g1

# fecundity of all 'wild' birds is boosted because my baseline data come from a supplemented population
 f2_boosted  <- f2_basic * 1.5 * g1
# prebreeding so include first year survival in the fertility coefficient
```

```
  f2_year1    <- f2_basic * 1.5 * F.penalty.juv * g1
# prebreeding so include juvenile survival rate
  f2_adjusted  <-  (f2_released  *  prop.released)  +  (f2_year1  *  prop.juv)  +  (f2_boosted  *
prop.wildAdFemales)
  f2 <- f2_adjusted
   } else{
      f2 <- 0
   }
    if(densityDependence == TRUE){
    if(N_female_breeders > max_nest_sites) {N_female_breeders <- max_nest_sites}
# number of breeding females cannot be higher than the max number of nest sites
    dd_f2 <- f2 * (1 - (N_female_breeders/max_nest_sites) ^ 10)
# if you decrease the power here, the density dependent effect is strengthened, i.e. females #
are affected by the lack of nest sites sooner (due to increased intraspecific competition for #
nest sites, a likely occurrence for Bali Myna)
    f2 <- dd_f2
   }
   return(f2)
  }

# trapping scenario
trappingScenario <- function(){
# one of these two options should be used, otherwise throw an error
   if(trappingType != "mist-net" & trappingType != "nest"){
    stop("trappingType should be either nest or mist-net")
}
# convert trappedN into females only (female-only model)
trappedFemales <- trappedN * female.ratio

# calculate number of juveniles and adult females to remove from the population. There is a
# check here to make sure the population cannot go below zero
   if(trappingType == "mist-net"){

# for mist-net trapping, a random proportion of birds trapped are assumed to be juveniles,
# and it changes each year
proportionJuv <- round(runif(1, min = 0.2, max = 0.8), digits = 1)

# number of juveniles trapped
    nJuvsTrapped <- round(trappedFemales * proportionJuv, digits = 0)
    nJuvs <- if(nJuvsTrapped > N_juvs_female){N_juvs_female}else{nJuvsTrapped}
#number of adults trapped
    nAdultsTrapped <- round(trappedFemales * (1 - proportionJuv), digits = 1)
    nAdults <- if(nAdultsTrapped > N_adults_female) {
    N_adults_female
    } else{
    nAdultsTrapped
    }
    result <- c(nJuvs, nAdults)
   }
   if(trappingType == "nest"){
    nJuvs <- if(N_juvs_female < trappedFemales) {N_juvs_female
    } else{
      trappedFemales
    }
```

```
     nAdults <- 0
     result <- c(nJuvs, nAdults)
    }
    return(result)
  }

####------------------------------------------####
### run model using FOR loops
####------------------------------------------####

  for (j in 1:nsimul) {

    for (t in 1:historicalYears) {

# this loop is for the first 10 historical years (2012-2021) of data taken from BBNP's 2021
# Bali Myna report

# establish the number of juvs, females, and released females in last year
    N_juvs_female   <- N[1, t] # number of juveniles at the start of this year
    N_adults_female <- N[2, t] # number of adults at the start of this year
    N_released      <- if(t > 1) {supplementationFemales[t-1]} else {N_adults_female * 0.5}
# for the first year of modelling half of the adult females are considered released individuals
# (actually, in 2011 four adults were released, so that is about right )

# establish proportion of all birds that are each type:
prop.wildAdFemales <- (N_adults_female - N_released) / (N_adults_female + N_juvs_female)
prop.released <- N_released/(N_adults_female + N_juvs_female)
prop.juv <- N_juvs_female/(N_juvs_female + N_adults_female)

# juvenile survival and growth into adult life stage
     juvenile_survival_metrics <- juvenile.survival.growth()
     p1 <- juvenile_survival_metrics[1, "p1"]
     g1 <- juvenile_survival_metrics[1, "g1"]

# adult survival
     p2 <- adult.survival()

# fecundity
     f2 <- fecundity()
# construct the transition matrix for period t
     TMat <- matrix(c(p1, g1, f2, p2), nrow = 2, ncol = 2) # create the transition matrix
     rownames(TMat) <- stages
     colnames(TMat) <- stages

# matrix of supplementation
     supMat <- matrix(0, nrow = 2, ncol = 1) # matrix to store number of birds supplemented
     rownames(supMat) <- stages
     supMat[,1] <- c(0,supplementationFemales[t])

     N[ ,t+1] <- (TMat %*% N[,t]) + supMat[ ,1]
     NTot[j, t+1] <- round(sum(N[, t+1]) / female.ratio, digits=0)
# convert the number of females back to total number of adults based on female: male ratio

     propJuv[j, t+1] <- round(N[1, t+1] / sum(N[ , t+1]), digits=2)
```

```
  }
  for(t in (historicalYears + 1):yearsModelled){

# future years of the modelling period
# establish the number of juveniles, adults, and released females were in the modelled
# population in the last year
   N_juvs_female <- N[1, t] # number of juveniles at the start of this year
   N_adults_female <- N[2, t] # number of adults at the start of this year
   N_released <- future.supplementationFemales

   prop.wildAdFemales <- (N_adults_female - N_released) / (N_adults_female +
N_juvs_female)
   prop.released <- N_released/N_adults_female # proportion of all birds that are released
   prop.juv <- N_juvs_female / (N_juvs_female + N_adults_female)

# juvenile metrics
   juvenile_survival_metrics <- juvenile.survival.growth()
   p1 <- juvenile_survival_metrics[1, "p1"]
   g1 <- juvenile_survival_metrics[1, "g1"]

# adult survival
   s2 <- adult.survival()

# fecundity
   f2 <- fecundity()

# trapped birds
   trapped_birds <- if(trapping == TRUE){trappingScenario()} else{c(0,0)}

# construct the transition matrix for period t
   TMat <- matrix(c(p1, g1, f2, p2), nrow = 2, ncol = 2)
   rownames(TMat) <- stages
   colnames(TMat) <- stages

# add supplemented birds and include delay if the option is selected in function input
   if(delayedSupplementation == TRUE){
    if(t <= historicalYears + supplementationDelay){
     futureSup <- matrix(c(0, future.supplementationFemales), nrow = 2)
    } else{
     futureSup <- matrix(c(0, 0), nrow = 2)
    }
   } else{
   futureSup <- matrix(c(0, future.supplementationFemales), nrow = 2)
   }
# add year of data to N and then NTot
   N[, t + 1] <- (TMat %*% N[ , t]) + futureSup[,1] - trapped_birds
   NTot[j, t+1] <- round(sum(N[ ,t+1]) / female.ratio, digits = 0)
   propJuv[j, t+1] <- round(N[1, t + 1] / sum(N[ , t + 1]), digits = 2)
  }
# this does the sum to fill in the first year because that did not get covered by the loop
  NTot[j, 1] <- sum(N[,1]) / female.ratio
  propJuv[j, 1] <- round(N[1,1] / sum(N[, 1]), digits = 2)
 }
 result <- list(NTot, propJuv)
```

```
  names(result)[1] <- "NTot"
  names(result)[2] <- "proportionJuveniles"
  return(result)
}

### The script used to run the models with the settings shown in this chapter and produce the
figures is shown below.

# read in the baseline MPM function, making sure you use the correct directory depending on
# where function script is saved!
source("./R/bali-myna_MPM_penalised_supplementation.R")
require(ggplot2)
require(egg) # install packages if you do not have them already

####-------------------------------------------------####
# basic plot of the entire simulation, with all simulation runs shown in grey and a mean trend
line in black
####-------------------------------------------------####
plot_MPM <- function(NTot = model_name,
                     title_name = "",
                     y_axis = "t",
                     list_element = 1, # this is needed because function output is a list - the first element
of the basic function is the NTot matrix with the model data
                     plot_future_only = FALSE, # if true you get rid of first 10 years
                     confidence_intervals = TRUE,
                     y_upper_limit = 4000,
                     y_axis_label = "Total population"
                     ) {
NTot <- NTot[[list_element]] # first element of results list is the NTot matrix

  if(plot_future_only == TRUE){
    NTot <- NTot[ ,11:length(NTot[1,])] # this is to filter out the historical years
  }
  plot(NTot[1,],
       xaxt = "n",
       xaxs = "r",
       yaxs = "i",
       yaxt = y_axis,
       type = "l",
       lwd = 1,
       lty = 1,
       xlab = expression(bold("")),
       ylab = y_axis_label,
       ylim = c(0, y_upper_limit),
       col="grey72")

    for (i in 2:nrow(NTot)){
      lines(NTot[i,], type = "l", lwd = 1, lty = 1,col="grey72")
    }
# plots the results of each of the simulations through a loop. The x axis is purposefully blanked
# to fill with the appropriate years instead of the model years
    axis(1, at=1:length(NTot[1,]),
         labels = c(as.numeric(names(NTot[1,])[1]):as.numeric(names(NTot[1,])[length(NTot[1,])])))
    trend <- colMeans(NTot)
```

```
# calculates the mean of totals from each simulation for each given year i.e. mean trend of all
population trajectories
    lines(trend, type = "l", lwd = 2, lty = 1)

# plots the mean trend, the 0.5 is to get it out of being a female-only total
    if(confidence_intervals == TRUE){
      q5 <- apply(NTot, MARGIN = 2, function(x) quantile(x, probs = 0.025))
      q95 <- apply(NTot, MARGIN = 2, function(x) quantile(x, probs = 0.975))
      lines(q5, type = "l", lwd = 1, lty = 2)
      lines(q95, type = "l", lwd = 1, lty = 2)
      }
    title(title_name)
  }

####----------------------------------------------------####
# function for plotting the historic and model data (MODEL VALIDATION)
####----------------------------------------------------####
plot_model_validation <- function(NTot = model_name) {
    NTot <- NTot[[1]] # the first element of the results list is the NTot matrix
  validation <- NTot[,1:10]
    years <- names(validation[1,])
    nyears <- length(validation[1,])

    censusData <- matrix(c(1:10, 15,32,48,57,81,109,184,256,341,420),ncol = 2)
  plot(validation[1,],
       xaxt = "n",
       xaxs = "r",
       yaxs = "i",
       type = "l",
       lwd = 1,
       lty = 1,
       xlab = expression(bold("")),ylab = "Total population",
       ylim = c(0,max(c(max(validation), max(censusData)))*1.05),col="grey72")
  for(i in 2:length(validation[,1])){
      lines(validation[i,], type = "l", lwd = 1, lty = 1,col="grey72")
       }
  axis(1, at = 1:nyears, labels=c(years[1]:years[nyears]))
  trend <- colMeans(validation)
  q5 <- apply(validation, MARGIN = 2, function(x) quantile(x, probs = 0.025))
  q95 <- apply(validation, MARGIN = 2, function(x) quantile(x, probs = 0.975))
     lines(trend, type = "l", lwd = 2, lty = 1)
     lines(q5, type = "l", lwd = 1, lty = 2)
     lines(q95, type = "l", lwd = 1, lty = 2)
    points(x = censusData[,1], y = censusData[,2], type = "p", col = "red", pch = 16)
   legend(x = "topleft",
        inset = 0.02,
        legend = "Population census data",  # Legend texts
        pch = 16,
        col = "red",
        bty = "n",
        lty = 0)
    }
```

```
# show the options available in the function
args(balimynaMPM)

####-----####
#### Model 1 - baseline model using the default settings
####-----####
mod1_baseline <- balimynaMPM()
plot_MPM(mod1_baseline, plot_future_only = TRUE)
plot_model_validation(NTot = mod1_baseline)

# what is the final mean and median population size?
mean(mod1_baseline[[1]][,"2032"])
median(mod1_baseline[[1]][,21])
## compare simulated vs. validated population sizes for validation period
        simulation_est <- round(as.numeric(colMeans(mod1_baseline[[1]])),digits = 0)
        simulation_est <- simulation_est[1:10]
        census <- c(15,32,48,57,81,109,184,256,341,420)
        validated_data_difference <- (1-simulation_est/census) * 100
        validation_diff_abs <- round(abs(validated_data_difference),digits = 1)
        validation_diff_abs
        mean(validation_diff_abs)
# use table() to check how many population simulations reached certain status by x date
table(mod1_baseline[[1]][,"2032"] < 1000)

## how many birds are juvenile?
mod1_meanPropJuvenile <- apply(mod1_baseline[[2]][,11:20],
                    MARGIN = 2,
                    function(x) mean(x))
mod1_meanPropJuvenile
mean(mod1_meanPropJuvenile)
# result shows % of birds juvenile in the future years modelled

####--------------------------------------------####
#### FIGURE 1. Model validation
####--------------------------------------------####
png(filename = "./exports/figures/bali_myna_pva/fig1_model_validation.png",
width = 700,
height = 500,
pointsize = 14)
par(mfrow = c(1,1),
   mar = c(3,4,3,3))
plot_model_validation(NTot = mod1_baseline)
dev.off()

####-----####
## model 2 > SUPPLEMENTATION CEASES
####-----####
mod2_supplementation_stops <- balimynaMPM(future.supplementation = 0)

## how many birds are juvenile?
mod2_meanPropJuvenile <- apply(mod2_supplementation_stops[[2]][,11:21],
                    MARGIN = 1,
                    function(x) mean(x))
mean(mod2_meanPropJuvenile)
```

```
  # result shows % of birds juvenile in the future years modelled
table(mod2_supplementation_stops[[1]][,"2032"] < 1000) # ~31% chance of being below 1000
individuals
mean(mod2_supplementation_stops[[1]][,"2032"])

  ####-----####
  ## model 2b > SUPPLEMENTATION CEASES AFTER 5 YEARS
  ####-----####
mod2b_supplementation_stops_delayed <- balimynaMPM(delayedSupplementation = TRUE)
table(mod2b_supplementation_stops_delayed[[1]][,"2032"] < 1000) # % chance of population
being below 1000 at the end of the modelling period
mean(mod2b_supplementation_stops_delayed[[1]][,"2032"])

####------------------------------------------####
#### FIGURE 2. Baseline model with supplementation vs. without supplementation vs. delayed
end to supplementation
####------------------------------------------####
  png(filename = "./exports/figures/bali_myna_pva/fig2_baseline_model_95CI_v2.png",
     width = 750,
     height = 300,
     pointsize = 12)

 par(oma=c(0,3,0,0),mar=c(3,2,3,1.5),mfrow=c(1,3))

  plot_MPM(NTot = mod1_baseline,
         title = "Continue supplementation",
         plot_future_only = TRUE,
         confidence_intervals = TRUE,
         y_upper_limit = 5000,
         y_axis_label = ""
         )
  # abline(h = mean(mod1_baseline[[1]][,21]), col = "red", lty = "dashed")
  plot_MPM(NTot = mod2b_supplementation_stops_delayed,
         title = "Stop supplementation \nafter 5 years",
         plot_future_only = TRUE,
         confidence_intervals = TRUE,
         y_upper_limit = 5000,
         y_axis_label = ""
  )
   plot_MPM(NTot = mod2_supplementation_stops,
          title = "Stop supplementation",
          plot_future_only = TRUE,
          confidence_intervals = TRUE,
          y_upper_limit = 5000,
          y_axis_label = ""
   )

 mtext(text="Total population",side=2,line=1,outer=TRUE, cex = 1)
 dev.off()

####------------------------------------------####
### SCENARIO: Trapping birds (above background rate in baseline model)
####------------------------------------------####
# two types of trapping:
```

```
# 1. nest trapping - only first year birds removed from pop
# 2. mist-netting - adults and first years taken in random proportions
# nest trapping (supplementation continues)
nest_trapping_sim <- c(40, 80, 120)
nest_trapping_list <- list()
nest_trapping_out <- matrix(0,
                    nrow = length(nest_trapping_sim),
                    ncol = length(mod1_baseline[[1]][1,]))
for(scenario in 1:length(nest_trapping_sim)) {
  res <- balimynaMPM(trapping = TRUE,
               trappingType = "nest",
               trappedN = nest_trapping_sim[scenario])
  nest_trapping_list[[scenario]] <- res[[1]]
  nest_trapping_out[scenario,] <- colMeans(nest_trapping_list[[scenario]])
  }
par(mfrow = c(1,1))
plot(nest_trapping_out[1,],
    xaxt = "n",
    xaxs = "r",
    yaxs = "i",
    type = "l",lwd = 1,
    lty = 1,
    xlab = expression(bold("")),
    ylab = "Total population",
    ylim = c(0,5000))
for(i in 2:length(nest_trapping_out[,1])){
  lines(nest_trapping_out[i,], type = "l", lwd = 1, lty = 1)
}
# nest trapping AND supplementation STOPS
nest_trapping_noSupp_list <- list()
nest_trapping_noSupp_out <- matrix(0,
                    nrow = length(nest_trapping_sim),
                    ncol = length(mod1_baseline[[1]][1,]))

for(scenario in 1:length(nest_trapping_sim)) {
  res <- balimynaMPM(trapping = TRUE,
               trappingType = "nest",
               trappedN = nest_trapping_sim[scenario],
               future.supplementation = 0)
  nest_trapping_noSupp_list[[scenario]] <- res[[1]]
  nest_trapping_noSupp_out[scenario,] <- colMeans(nest_trapping_noSupp_list[[scenario]])
}
  plot(nest_trapping_noSupp_out[1,],
    xaxt = "n",
    xaxs = "r",
    yaxs = "i",
    type = "l",lwd = 1,
    lty = 1,
    xlab = expression(bold("")),
    ylab = "Total population",
    ylim = c(0,4000))

for(i in 2:length(nest_trapping_noSupp_out[,1])){
  lines(nest_trapping_out[i,], type = "l", lwd = 1, lty = 1)
```

```
}
# mist-net trapping simulation (supplementation continues)
 mistNet_trapping_sim <- c(40,80,120)
 mistNet_trapping_list <- list()
 mistNet_trapping_out <- matrix(0,
                            nrow = length(mistNet_trapping_sim),
                            ncol = length(mod1_baseline[[1]][1,]))
   for(scenario in 1:length(mistNet_trapping_sim)) {

   res <- balimynaMPM(trapping = TRUE,
                   trappingType = "mist-net",
                   trappedN = mistNet_trapping_sim[scenario])

   mistNet_trapping_list[[scenario]] <- res[[1]]
   mistNet_trapping_out[scenario,] <- colMeans(mistNet_trapping_list[[scenario]])
   }
 mistNet_trapping_fig <- mistNet_trapping_out[,11:21]
 png(filename = "./exports/figures/bali_myna_pva/mist-net-trapping_scenario.png")

 plot(mistNet_trapping_fig[1,],
     xaxt = "n",
     xaxs = "r",
     yaxs = "i",
     type = "l",lwd = 1,
     lty = 1,
     xlab = expression(bold("")),
     ylab = "Total population",
     ylim = c(0,4000),
     main = "Population trapped with mist-nets, random take of juvs and adults")

 for(i in 1:length(mistNet_trapping_fig[,1])){
   lines(mistNet_trapping_fig[i,], type = "l", lwd = 1, lty = 1)
 }
 dev.off()

## mist-net trapping simulation (supplementation STOPS)
 mistNet_trapping_sim <- c(40,80,120)
 mistNet_trapping_noSupp_list <- list()
 mistNet_trapping_noSupp_out <- matrix(0,
                               nrow = length(mistNet_trapping_sim),
                               ncol = length(mod1_baseline[[1]][1,]))

 for(scenario in 1:length(mistNet_trapping_sim)) {

   res <- balimynaMPM(trapping = TRUE,
                   trappingType = "mist-net",
                   trappedN = mistNet_trapping_sim[scenario],
                   future.supplementation = 0)

   mistNet_trapping_noSupp_list[[scenario]] <- res[[1]]
   mistNet_trapping_noSupp_out[scenario,]                                    <-
colMeans(mistNet_trapping_noSupp_list[[scenario]])

}
```

```
  mistNet_trapping_noSupp_fig <- mistNet_trapping_noSupp_out[,11:21]
    png(filename = "./exports/figures/bali_myna_pva/mist-netting_scenario.png")
    plot(mistNet_trapping_noSupp_fig[1,],
      xaxt = "n",
      xaxs = "r",
      yaxs = "i",
      type = "l",lwd = 1,
      lty = 1,
      xlab = expression(bold("")),
      ylab = "Total population",
      ylim = c(0,4000),
      main = "Population trapped with mist-nets, random take of juvs and adults")
    for(i in 2:length(mistNet_trapping_noSupp_fig[,1])){
    lines(mistNet_trapping_noSupp_fig[i,], type = "l", lwd = 1, lty = 1)
  }
    dev.off()

#####--------------------######
# summary stats to show mean population and risk of population being below 1000 individuals
# after ten years

  mean(nest_trapping_list[[1]][,"2032"])
  mean(nest_trapping_noSupp_list[[3]][,"2032"])

  mean(mistNet_trapping_list[[3]][,"2032"])
  mean(mistNet_trapping_noSupp_list[[3]][,"2032"])

  table(nest_trapping_list[[3]][,"2032"] < 1000)
  table(nest_trapping_noSupp_list[[1]][,"2032"] < 1000)

  table(mistNet_trapping_list[[3]][,"2032"] < 1000)
  table(mistNet_trapping_noSupp_list[[3]][,"2032"] < 1000)

#####-------------------#####
#### FIGURE 3: Plot trapping scenarios side by side. Columns are trapping type, rows are
trapping intensity, and the trend and 5/95 percentiles are shown for supplemented and non-
supplemented population in the same plot. Result should be a 3 by 2 grid of plots
#####-------------------#####
# function to create the data frames I need to make the plot
filterSim <- function(name1,
                 name2,
                 list.element = 1,
                 conf.width = 0.025
                 ){
# establish continued supplementation data frame
res1 <- name1[[list.element]]
res1 <- res1[,11:length(res1[1,])]
df1 <- data.frame(meanPop = colMeans(res1),
             lower = apply(res1,
                   MARGIN = 2,
                   function(x) quantile(x,
                              probs = conf.width)),
            upper = apply(res1,
```

```
                        MARGIN = 2,
                        function(x) quantile(x,
                                             probs = 1-conf.width)),
             Year = 2022:2032,
             supp = "Supplementation continues"
             )

# establish supplementation stops data frame
res2 <- name2[[list.element]]
res2 <- res2[,11:length(res2[1,])]

df2 <- data.frame(meanPop = colMeans(res2),
              lower = apply(res2,
                         MARGIN = 2,
                         function(x) quantile(x, probs = conf.width)),
              upper = apply(res2,
              MARGIN = 2,
              function(x) quantile(x, probs = 1-conf.width)),
              Year = 2022:2032,
              supp = "Supplementation stops"
   )
b <- list(df1,df2)
df <- do.call(rbind, b)
return(df)
}

## create the dataframes for plotting
nest_low <- filterSim(name1 = nest_trapping_list,
                  name2 = nest_trapping_noSupp_list,
                  list.element = 1,
                  conf.width = 0.05)
mist_low <- filterSim(name1 = mistNet_trapping_list,
                  name2 = mistNet_trapping_noSupp_list,
                  list.element = 1,
                  conf.width = 0.05)
nest_mid <- filterSim(name1 = nest_trapping_list,
                  name2 = nest_trapping_noSupp_list,
                  list.element = 2,
                  conf.width = 0.05)
mist_med <- filterSim(name1 = mistNet_trapping_list,
                  name2 = mistNet_trapping_noSupp_list,
                  list.element = 2,
                  conf.width = 0.05)
nest_high <- filterSim(name1 = nest_trapping_list,
                  name2 = nest_trapping_noSupp_list,
                  list.element = 3,
                  conf.width = 0.05)
mist_high <- filterSim(name1 = mistNet_trapping_list,
                  name2 = mistNet_trapping_noSupp_list,
                  list.element = 3,
                  conf.width = 0.05)

# and now generate plots for each model
#### plot 1 ####
```

```
p1 <-   ggplot() +
  geom_ribbon(data = nest_low[nest_low$supp == "Supplementation continues", ],
            aes(x = Year, ymin = lower, ymax = upper, fill = supp),
            alpha = 0.1) +
  geom_ribbon(data = nest_low[nest_low$supp == "Supplementation stops", ],
            aes(x = Year, ymin = lower, ymax = upper, fill = supp),
            alpha = 0.15) +
  geom_line(data = nest_low[nest_low$supp == "Supplementation stops", ],
          aes(x = Year, y = meanPop),
          col = "red") +
  geom_line(data = nest_low[nest_low$supp == "Supplementation continues", ],
          aes(x = Year, y = meanPop),
          col = "blue4") +
  scale_y_continuous(breaks = seq(1000,5000,1000),
                  expand = c(0,0),
                  name = "Total population") +
  scale_x_continuous(breaks = seq(2022,2032,1),
                  name = NULL,
                  expand = c(0,0)) +
  coord_cartesian(ylim = c(0,5000)) +
  ggtitle("Nest poaching",
        subtitle = "40 first-year birds trapped per year")+
  theme_bw() +
  theme(plot.margin = margin(0.4,0.5,0.2,0.2, "cm"),
      panel.grid.minor = element_blank(),
      axis.ticks.length=unit(.15, "cm"),
      legend.justification = c(0,1),
      legend.position = c(0.05,0.95),
      legend.title = element_blank(),
      legend.key.size = unit(0.5, "cm"),
      legend.text = element_text(size=12),
      axis.text = element_text(size = 10),
      axis.title.y = element_text(size = 12,
                            margin = margin(0,5,0,0)),
      title = element_text(size = 14),
      plot.title = element_text(hjust = 0.5),
      plot.subtitle = element_text(size = 12)) +

  scale_fill_manual(values = c("Supplementation continues" = "blue4",
                          "Supplementation stops" = "red"),
                  labels = c("Supplementation continues",
                          "Supplementation stops"))
#### plot 2 ####
p2 <- ggplot() +
  geom_ribbon(data = mist_low[mist_low$supp == "Supplementation continues", ],
            aes(x = Year, ymin = lower, ymax = upper, fill = supp),
            alpha = 0.1) +
  geom_ribbon(data = mist_low[mist_low$supp == "Supplementation stops", ],
            aes(x = Year, ymin = lower, ymax = upper, fill = supp),
            alpha = 0.15) +
  geom_line(data = mist_low[mist_low$supp == "Supplementation stops", ],
          aes(x = Year, y = meanPop),
          col = "red") +
  geom_line(data = mist_low[mist_low$supp == "Supplementation continues", ],
```

```
           aes(x = Year, y = meanPop),
           col = "blue4") +
  scale_y_continuous(breaks = seq(1000,5000,1000),
                 expand = c(0,0),
                 name = NULL) +
  scale_x_continuous(breaks = seq(2022,2032,1),
                 name = NULL,
                 expand = c(0,0)) +
  coord_cartesian(ylim = c(0,5000)) +
  ggtitle("Mist-netting",
        subtitle = "40 birds trapped per year (random life stage taken)") +
  theme_bw() +
  theme(plot.margin = margin(0.4,0.5,0.2,0.2, "cm"),
      axis.ticks.length=unit(.15, "cm"),
      legend.position = "none",
      panel.grid.minor = element_blank(),
      title = element_text(size = 14),
      plot.title = element_text(hjust = 0.5),
      plot.subtitle = element_text(size = 12)) +
  scale_fill_manual(values = c("Supplementation continues" = "blue4",
                          "Supplementation stops" = "red"),
                 labels = c("Supplementation continues",
                          "Supplementation stops"))

#### plot 3 ####
p3 <-   ggplot() +
  geom_ribbon(data = nest_mid[nest_mid$supp == "Supplementation continues", ],
            aes(x = Year, ymin = lower, ymax = upper, fill = supp),
            alpha = 0.1) +
  geom_ribbon(data = nest_mid[nest_mid$supp == "Supplementation stops", ],
            aes(x = Year, ymin = lower, ymax = upper, fill = supp),
            alpha = 0.15) +
  geom_line(data = nest_mid[nest_mid$supp == "Supplementation stops", ],
           aes(x = Year, y = meanPop),
           col = "red") +
  geom_line(data = nest_mid[nest_mid$supp == "Supplementation continues", ],
           aes(x = Year, y = meanPop),
           col = "blue4") +
  scale_y_continuous(breaks = seq(1000,5000,1000),
                 expand = c(0,0),
                 name = "Total population") +
  scale_x_continuous(breaks = seq(2022,2032,1),
                 name = NULL,
                 expand = c(0,0)) +
  coord_cartesian(ylim = c(0,5000)) +
  ggtitle("80 first-year birds trapped per year") +
  theme_bw() +
  theme(plot.margin = margin(0.4,0.5,0.2,0.2, "cm"),
      axis.ticks.length=unit(.15, "cm"),
      legend.position = "none",
      axis.text = element_text(size = 10),
      axis.title.y = element_text(size = 12,
                            margin = margin(0,5,0,0)),
      plot.title = element_text(size = 12),
```

```
      panel.grid.minor = element_blank()) +
  scale_fill_manual(values = c("Supplementation continues" = "blue4",
                              "Supplementation stops" = "red"),
                    labels = c("Supplementation continues",
                              "Supplementation stops"))

#### plot 4 #####
p4 <- ggplot() +
  geom_ribbon(data = mist_med[mist_med$supp == "Supplementation continues", ],
              aes(x = Year, ymin = lower, ymax = upper, fill = supp),
              alpha = 0.1) +
  geom_ribbon(data = mist_med[mist_med$supp == "Supplementation stops", ],
              aes(x = Year, ymin = lower, ymax = upper, fill = supp),
              alpha = 0.15) +
  geom_line(data = mist_med[mist_med$supp == "Supplementation stops", ],
            aes(x = Year, y = meanPop),
            col = "red") +
  geom_line(data = mist_med[mist_med$supp == "Supplementation continues", ],
            aes(x = Year, y = meanPop),
            col = "blue4") +
 scale_y_continuous(breaks = seq(1000,5000,1000),
                    expand = c(0,0),
                    name = NULL) +
  scale_x_continuous(breaks = seq(2022,2032,1),
                    name = NULL,
                    expand = c(0,0)) +
  coord_cartesian(ylim = c(0,5000)) +
  ggtitle("80 birds trapped per year (random life stage taken)") +
  theme_bw() +
  theme(plot.margin = margin(0.4,0.5,0.2,0.2, "cm"),
      axis.ticks.length=unit(.15, "cm"),
      legend.position = "none",
      panel.grid.minor = element_blank(),
      plot.title = element_text(size = 12)) +
  scale_fill_manual(values = c("Supplementation continues" = "blue4",
                              "Supplementation stops" = "red"),
                    labels = c("Supplementation continues",
                              "Supplementation stops"))

#### plot 5 ####
p5 <-   ggplot() +
  geom_ribbon(data = nest_high[nest_high$supp == "Supplementation continues", ],
              aes(x = Year, ymin = lower, ymax = upper, fill = supp),
              alpha = 0.1) +
  geom_ribbon(data = nest_high[nest_high$supp == "Supplementation stops", ],
              aes(x = Year, ymin = lower, ymax = upper, fill = supp),
              alpha = 0.15) +
  geom_line(data = nest_high[nest_high$supp == "Supplementation stops", ],
            aes(x = Year, y = meanPop),
            col = "red") +
  geom_line(data = nest_high[nest_high$supp == "Supplementation continues", ],
            aes(x = Year, y = meanPop),
            col = "blue4") +
  scale_y_continuous(breaks = seq(1000,5000,1000),
```

```
            expand = c(0,0),
            name = "Total population") +
  scale_x_continuous(breaks = seq(2022,2032,1),
            name = "Year",
            expand = c(0,0)) +
  coord_cartesian(ylim = c(0,5000)) +
  ggtitle("120 first-year birds trapped per year") +
  theme_bw() +
  theme(plot.margin = margin(0.4,0.5,0.2,0.2, "cm"),
     panel.grid.minor = element_blank(),
     axis.ticks.length=unit(.15, "cm"),
     legend.position = "none",
     axis.text = element_text(size = 10),
     axis.title.y = element_text(size = 12,
                    margin = margin(0,5,0,0)),
     axis.title.x = element_text(size = 12,
                    margin = margin(5,0,0,0)),
     plot.title = element_text(size = 12)) +
  scale_fill_manual(values = c("Supplementation continues" = "blue4",
                  "Supplementation stops" = "red"),
            labels = c("Supplementation continues",
                  "Supplementation stops"))

#### plot 6 ####
p6 <- ggplot() +
  geom_ribbon(data = mist_high[mist_high$supp == "Supplementation continues", ],
        aes(x = Year, ymin = lower, ymax = upper, fill = supp),
        alpha = 0.1) +
  geom_ribbon(data = mist_high[mist_high$supp == "Supplementation stops", ],
        aes(x = Year, ymin = lower, ymax = upper, fill = supp),
        alpha = 0.15) +
  geom_line(data = mist_high[mist_high$supp == "Supplementation stops", ],
       aes(x = Year, y = meanPop),
       col = "red") +
  geom_line(data = mist_high[mist_high$supp == "Supplementation continues", ],
       aes(x = Year, y = meanPop),
       col = "blue4") +
  scale_y_continuous(breaks = seq(1000,5000,1000),
            expand = c(0,0),
            name = NULL) +
  scale_x_continuous(breaks = seq(2022,2032,1),
            name = "Year",
            expand = c(0,0)) +
  coord_cartesian(ylim = c(0,5000)) +
  ggtitle("120 birds trapped per year (random life stage taken)") +
  theme_bw() +
  theme(plot.margin = margin(0.4,0.5,0.2,0.2, "cm"),
     axis.ticks.length=unit(.15, "cm"),
     legend.position = "none",
     panel.grid.minor = element_blank(),
     axis.title.x = element_text(size = 12,
                    margin = margin(5,0,0,0)),
     plot.title = element_text(size = 12)) +
  scale_fill_manual(values = c("Supplementation continues" = "blue4",
```

```
                          "Supplementation stops" = "red"),
                 labels = c("Supplementation continues",
                          "Supplementation stops"))
#### arrange plots and save ####
Fig3 <- ggarrange(p1,p2,p3,p4,p5,p6, ncol = 2)
ggsave("./exports/figures/bali_myna_pva/Fig3_scenarios.png",  plot = Fig3,
      width = 9, height = 12, units = "in")
```

Chapter 6

6 General conclusions and recommendations for future work

6.1 Introduction

Wildlife trade is a multi-billion dollar industry and one of the leading contributors to the massive global declines in biodiversity that have precipitated a global biodiversity crisis (Bush et al., 2014; Sykes, 2017; Cardoso et al., 2021; Morton et al., 2021). Birds are one of the most heavily traded taxonomic groups, with nearly a third of extant species identified as threatened by trade (Butchart, 2008). Across Southeast Asia more than 1,000 bird species have been identified as traded (Nijman, 2010; Chng et al., 2015; Harris et al., 2017; Marshall et al., 2020b), and there are accordingly high levels of bird ownership across the region, but particularly in Indonesia (Nijman, 2010; Marshall et al., 2020b). Commercial interest in songbirds (passerines) is high across Southeast Asia due to the value people attribute to various species' singing ability, and they are kept as pets, status symbols, and for use in training or competition for songbird contests, a popular activity that can involve substantial prizes for winners (Jepson and Ladle, 2005; Jepson et al., 2011; Marshall et al., 2020a). While some traded birds are captive-bred (Marshall et al., 2020a), the vast majority are wild-caught and this overexploitation has precipitated an Asian Songbird Crisis (Sykes, 2017). In response, an Asian Songbird Trade Specialist Group (ASTSG) comprising concerned experts from institutions both within and outside the region was established to coordinate effective conservation efforts to prevent imminent extinctions of songbirds (Shepherd and Cassey, 2017). There are five main themes to the work being carried out by members of the ASTSG: field research, genetic research, conservation breeding and reintroduction, trade and legislation, and education and community engagement.

In this thesis, I aimed to understand the ecology and management needs of some of the passerines most threatened by the cage-bird trade across Java and Bali, Indonesia, to guide in situ conservation actions. In Chapter 2 I addressed the existing gaps in biological data coverage that hampers efforts to generate robust baseline information on the distribution and abundance of birds across Java and Bali by implementing a month-long birdwatching citizen science event, which had the dual objective of engaging Indonesian society in citizen science and generating a large bird occurrence dataset. This dataset was utilised in Chapter 3 to generate baseline distribution models for a suite of Java's lowland birds and I investigated the importance of habitat, trade and protected areas in shaping their current distributions. In Chapter 4 I assessed the population size and ecology of one of Java's most threatened starlings, the Black-winged Myna (*Acridotheres melanopterus*), at its last remaining stronghold on Java at Baluran National Park. And finally, In Chapter 5 I evaluated the near-future viability of the largest remaining population of the iconic Bali Myna (*Leucopsar rothschildi*), Bali's only endemic bird species, which has suffered serious declines and a probable extinction in the wild owing to trapping and historic habitat loss in its range in the coastal areas of north-west Bali.

6.2 Summary of key chapter findings

BigMonth2020 citizen science project fills gaps in bird distribution data

I addressed the need to obtain data for bird monitoring across Java by designing and implementing a citizen science event (Collar et al., 2012). Citizen science remains rare in biodiverse yet data-poor countries, and this represents a missed opportunity for generating data for biodiversity monitoring and promoting the public stewardship of nature (Chandler et al., 2017; Pocock et al., 2019). I documented and analysed BigMonth2020, a month-long birdwatching event across Java and Bali, publicised through social media and incentivised with grants and competitions. A huge number of bird records were submitted to the 'Burungnesia' phone app during the event, resulting in a massive increase in spatial coverage of data, more than doubling the number of atlas grid squares containing at least some data. Three quarters of Java and Bali's avifauna (excluding vagrants) were recorded and this included a large number of threatened species, many of which were recorded in new areas. The event was far more inclusive in terms of female participation than other bird-related pastimes in Indonesia, such as bird-keeping and songbird contests, and the vast majority of participants were under 30-years old, most of whom were graduates and members of birdwatching clubs. The project cost less than US$10,000 to run, and serves as a model for rapidly establishing a distributional baseline for monitoring biodiversity trajectories.

Predicting current distributions of lowland birds in Java

Data generated from BigMonth2020 were combined with other citizen science bird datasets available for Java (eBird, Burungnesia and the Indonesian Bird Atlas) to assess the current distributions of a suite of Java's lowland birds. The current distribution for most species was relatively patchy and often significantly smaller than the extent of occurrence. Among the environmental variables used in modelling, land-cover-based predictors were ultimately the most important in the models for most species. The lack of convergence between the current distribution of the modelled species with Java's formally protected areas suggests that future conservation for these and other lowland birds, which are likely to come under increasing anthropogenic pressure, will need to involve other effective area-based conservation measures (OECMs). My findings highlight the considerable value of continued citizen science efforts across Java, and indeed elsewhere in data-poor yet biodiverse regions, and the importance of further survey work to discover areas of high biodiversity value.

After addressing data gaps in order to examine large-scale patterns of bird distributions across Java in Chapters 2 and 3, in Chapters 4 and 5 I focused on two of the species most threatened by the cage-bird trade in Java and Bali, the Black-winged Myna and Bali Myna.

Conservation management for the Black-winged Myna

I documented the plight of the Endangered Black-winged Myna, a Java and Bali endemic that has been trapped to near extinction, with a global population estimated to number fewer than 100 individuals. I estimated the current range and population size of the species at Baluran National Park, which supports Java's last known population, and used species distribution modelling to evaluate the potential suitability of currently unoccupied areas across the park to identify priorities for management intervention. I estimated that the Black-winged Myna population numbers 179 individuals and that its current range is restricted to a small area of savanna and dry deciduous woodland, while my model indicated that a considerable extra portion has potentially suitable habitat for the species. I inferred that the principal cause for the disparity between its current and potential range is trapping, compounded by savanna loss and degradation due to illegal domestic cattle grazing and the spread of invasive thorny acacia (*Vachellia nilotica*). The recent partial clearance of acacia appears to have assisted a modest population recovery by the myna. Its further population growth and range expansion in Baluran depends on effective management of illegal poaching, further clearance of acacia, and easing domestic cattle-grazing pressure on areas of savanna, particularly through engagement with human communities living inside the park. Any actions that increase the size of the Black-winged Myna population are likely to benefit other threatened savanna-dependent wildlife in the park, notably banteng (*Bos javanicus*) and Green Peafowl (*Pavo muticus*).

Measuring the viability of the Bali Myna population to plan for the future

Population viability analysis is a simulation tool for determining the probable trajectory of a population that is used to quantify extinction risk, identify threats, and compare management interventions (Beissinger and McCullough, 2002). Despite decades of conservation efforts, in the 2000s there were probably no Bali Mynas left in the wild (Jepson, 2016). Since then, various management interventions at Bali Barat National Park (BBNP) have led to population growth. To plan for the next decade of conservation management, I modelled the Bali Myna population at BBNP aiming to explore the effects of (1) changes to population supplementation and (2) an increase in trapping intensity. A baseline model was validated using population census and captive-bred release data from the last ten years and the model was projected ten years into the future. The population was predicted to increase under current levels of supplementation, while stopping supplementation in five years had only a small effect. I also modelled the differential effects of two likely trapping methods used by poachers and three trapping volumes. The population was resilient to low levels of trapping with and without population supplementation, but under high levels of trapping the population declined steeply irrespective of the trapping method used. On current trajectory, I estimated that the population will approach self-sustainability in the next 5–10 years. The supplementation programme at BBNP could either be scaled back or repurposed as a translocation project to repopulate vacant parts of the myna’s range, and nest-boxes could be used as a potential tool to support population growth. The model was parameterised with the limited data available for the species, so its findings are provisional. Nevertheless, it provides insights that appear robust. With the species now living alongside local communities, there is an opportunity to enlist their support and work on generating new data to improve the model.

Conclusion

Understanding the ecology and management of birds threatened by the cage-bird trade is vital to prevent imminent extinctions and the loss of important ecosystem services they provide (Şekercioğlu, 2012). The lack of baseline data for most species seriously hinders and in many cases precludes efforts to monitor birds in the region (Lee et al., 2016; Collar and Wirth, 2022). In this thesis I highlighted the potential for citizen science to deliver extensive biological data and to promote environmental stewardship among participants, who are also given opportunities to gain skills and meet like-minded people (Bela et al., 2016; Pocock et al., 2019). Additionally, I have sought to demonstrate how these data can be used to establish the current distributions of species hitherto largely ignored in the scientific literature. Finally, I have conducted species-specific field research on two severely threatened mynas involving the interplay of distributional evidence, habitat selection and trade pressures, and in the process highlighted some difficulties associated with studying the ecology of species close to extinction, but most importantly provided recommendations that may help protect these species in the near-future.

6.3 A family crisis: conservation of Indonesia’s threatened and little known sturnids

Around 30 species of starlings and mynas occur across Indonesia, with the total varying depending on the taxonomic authority followed; here I have used the HBW and BirdLife International taxonomy (2022). Of these species, three are Critically Endangered (Nias Hill Myna *Gracula robusta*, Bali Myna and Javan Pied Starling *Gracupica jalla*), two Endangered (Black-winged Myna and Tenggara Hill Myna *Gracula venerata*), two Vulnerable (Pale-bellied Myna *A. cinereus* and Javan Myna *A. javanicus*), five Near Threatened, and 18 Least Concern (BirdLife International, 2022). One species, the Common Myna (*A. tristis*), is the only non-native introduced species and is unique in being the only sturnid whose population is thought to be increasing owing to its rapid adaptation to alien environments (Baker and Moeed, 1987). For the remainder, the threats of habitat destruction and fragmentation, and rampant illegal trapping for the cage-bird trade are driving population declines, although for many of the species worryingly little information is available (BirdLife International, 2022). Some species have been introduced elsewhere and their populations have expanded rapidly, but have become increasingly difficult to find in their native range; perversely, Javan Mynas are probably trapped in Peninsular Malaysia and used to supply the cage-bird trade back in Java (Arazmi et al., 2022; BirdLife International, 2022).

Sturnids are popular cage-birds owing to their vocal ability—especially their powers of mimicry—and the bold colours and striking patterns of their plumage and in some cases bare facial skin (Feare and Craig, 1998). The Common Hill Myna (*Gracula religiosa*) is one of the most popular cage-birds across Southeast Asia due to its remarkable ability to mimic sounds including human speech, and despite its large range it was already becoming scarce in parts of Indonesia in the 2000s due to trapping (Shepherd, 2006). The trade in sturnids generally across Southeast Asia appears to differ somewhat from other traded passerines, and in some ways is akin to the trade in larger-bodied species such as cockatoos (Reuleaux et al., 2022). Owing to their distinctive features, they tend to serve as status symbols for the powerful and wealthy, and are not part of the 'cut-flower market' of cage-birds found in other groups such as munias (family Estrildidae). In some cases, species now threatened by the largely domestic trade may have initially been popularised as status symbols due to their popularity among European and North American collectors (Jepson, 2016). Since the widespread depletion of birds on Java, trapping pressure appears to be radiating out from the island, and sturnids from Papua, Sulawesi, and other islands are becoming increasingly threatened (Indraswari et al., 2020). This poses a serious risk to sturnids because many are highly gregarious, so their populations can be depleted rapidly if trappers target them (Feare and Craig, 1998; Ng et al., 2021).

The taxonomy of the sturnids is complex and the impact of taxonomic changes can have far-reaching conservation implications. Splitting or lumping species can have a major impact on the extinction risk category in which a species falls, with the extinction risk increasing when species are split (IUCN Standards and Petitions Working Group, 2022). This is important because extinction risk assessments are used as a guide for conservation planning, and they may have a strong influence on whether or not conservation work is ultimately funded. Another issue arising from taxonomic confusion is that some taxa are being released back into the wild in the range of other taxa, for example, as escapes from bird-keepers or following confiscation from illegal smuggling, with ensuing hybridisation and genetic mixing. Taxonomic changes have been a serious issue for the Javan Pied Starling, which, until being split from Asian Pied Starling

(*Gracupica contra*) in 2016, was not a focus of conservation efforts, but its plight proves to be unbelievably serious (Reuleaux et al., 2018; Collar and Wirth, 2022). Finally, there is also the insidious interest among some bird breeders in oddities and hybrids, with much genetic mixing occurring in places, causing serious problems for conservationists attempting to establish conservation breeding programmes and release confiscated individuals back into the wild (Sadanandan et al., 2020; Nijman et al., 2022).

Considering the severity of the threats faced by many sturnids across Indonesia, surprisingly little is known about the ecology and conservation status of most species, and this needs to be urgently addressed (Feare and Craig, 1998; BirdLife International, 2022). For the majority of species there are no population estimates available in the literature (except Callaghan et al., 2021, but the enormous confidence limits around estimates mean they are not usable), despite most being suspected of population declines. The majority of the *Aplonis* starlings are common and widespread so are not priorities for assessment, except the Purple-chinned Starling (*Aplonis circumscripta*), which is thought to be approaching threatened status due to logging in its small range (BirdLife International, 2022). However, there are glaring shortfalls in knowledge for other sturnids that need to be addressed urgently to prevent declines similar to those that have occurred on Java. The Sulawesi Myna (*Basilornis celebensis*) is being traded in relatively large numbers (Shepherd and Leupen, 2021), yet is assessed as Least Concern with no recent information available on the population size and little information concerning its ecological preferences (Feare and Craig, 1998; Billerman et al., 2022; BirdLife International, 2022). Mynas in the genera *Streptocitta* and *Basilornis* that occur in Sulawesi, Maluku, and the nearby islands are poorly understood and all are thought to be suffering population declines through a combination of trapping and habitat loss (BirdLife International, 2022). The Grosbeak Starling (*Scissirostrum dubium*), also from Sulawesi, is being traded in large numbers but its population status is unknown (Marshall et al., 2020b). The same goes for two Papuan sturnids, the Yellow-faced Myna (*Mino dumontii*) and Golden Myna (*M. anais*), which are both traded with unknown effects on their populations (BirdLife International, 2022).

Recommendation: It is strongly recommended that survey work for the aforementioned sturnids is carried out urgently to determine the current distribution, population size and abundance. Data gleaned from surveys should aim to feed into the next IUCN Red List extinction assessments. Projects on Sulawesi and its nearby islands, as well as West

Papua should be prioritised. A study on Sulawesi's endemic and threatened sturnids would be an ideal topic for a PhD project that could study the ecology of the species and assess the impact of threats facing them.

Important ecological questions need to be addressed for sturnids to support their conservation management and potential future reintroductions. An example of the type of species-specific information that is currently lacking is provided by the Bali Myna, which is endemic to the coastline of north-west Bali: its dependency on a small length of coastline with mangrove forest is still not definitively known. In this thesis, I hypothesised that it may depend on food resources found in mangrove trees to feed its young during the breeding season, a specialisation specific to the reproductive season recorded in other starlings (Bruun and Smith, 2003). However, due to the absence of strong evidence supporting this theory, there have been a number of inland releases of Bali Myna, yet none to date have had any lasting success (BirdLife International, 2022). Indeed, the only other release where the population may be approaching self-sustainability is on Nusa Penida, where birds also occur close to the coastline and are rarely recorded further than a few kilometres inland (Dijkman, 2007). Thus, a lack of ecological knowledge can lead to a waste of resources for conservation work, and this could be avoided through targeted research.

6.4 Future conservation for birds affected by the Asian Songbird Crisis

The cage-bird trade has undoubtedly been negatively affecting some species for decades already, but the magnitude of the effects on wild birds across Southeast Asia have only begun to be documented in real detail over the past 10–20 years (Collar et al., 2012; Collar and Butchart, 2014; Eaton et al., 2015; Collar and Wirth, 2022). Unfortunately, the situation appears to be worsening across the region: in 2015, when the first Asian Songbird Crisis Summit was held, 12 species were listed as priorities for immediate conservation action (Sykes, 2017), yet that number has ballooned to 43 in 2022 following the latest assessment carried out by the Asian Songbird Trade Specialist Group (ASTSG), albeit this was in part a result of taxonomic changes to account properly for conservation units that should be kept separate regardless of their status as species or subspecies (ASTSG, 2022).

Work undertaken by members of the ASTSG to date has included taxonomic studies to resolve confusion over some taxa (Ng et al., 2017, 2021); the establishment and continuation of conservation breeding programmes for a growing number of the most threatened species, with future reintroductions expected in the near-future (Collar and Wirth, 2022); bird market surveys to measure the scale of the trade and species being targeted (Chng and Eaton, 2018; Chng et al., 2018); documentation of confiscations by the Indonesian authorities of illegally traded birds (Indraswari et al., 2020), although it has been argued that much more needs to be done with regard to enforcement of Indonesian laws designed to protect wildlife (Shepherd et al., 2013; Rentschlar et al., 2018; Shepherd and Leupen, 2021); household surveys that have revealed extraordinary levels of bird ownership across Java (Jepson, 2010; Marshall et al., 2020b); characterisation of bird-keeping user groups to support demand reduction actions (Marshall et al., 2020a); and species-specific field research for some of the most threatened species (Chapters 4 and 5 of this thesis).

Looking to the future, there is still much work needed to reduce the impact of the cage-bird trade on wild birds across Indonesia and promote population recoveries for some of the worst affected species. In the following sections of this chapter, I make some general recommendations for possible future conservation work related to the topics covered in this thesis.

Assessing the distribution and abundance of birds across Indonesia

At the broad scale, essential data on the distribution and abundance of most species is still lacking, meaning that setting baselines from which to begin a robust scheme of monitoring and evaluation is still not possible (BirdLife International, 2022).

Recommendation: A larger citizen science event building on the work carried out to implement BigMonth2020 (documented in Chapter 2) should be developed for all of Indonesia. This might first target the islands where this is logistically most feasible and suitable conservation NGOs can be identified to lead on such efforts. Generating distributional data with greater spatial coverage is important in order to establish current distributions for species. It is vital to begin monitoring population trajectories for all bird species, but especially traded ones. To achieve this, a more detailed recording protocol will be needed that accounts for survey effort and other possible biases. Following a protocol similar to the highly successful one adopted in southern Africa may be possible (Harrison et al., 2008), although depending on the number of skilled participants willing to carry out such work, this may be a goal for the near-future, after the event has been established for some iterations and there is a base of skilled participants to implement the more detailed protocol.

For BigMonth2020, the members of Birdpacker, a tour-guide business and conservation NGO based in Malang, Java, were instrumental in its organisation and promotion. Expanding on the strategy used for BigMonth2020, I envisage an annual or biannual citizen science event lasting for one week or a month during which participants are asked to go and record birds in their local area. Given the size and spread of Indonesia, logistics will difficult for one organisation to cover, so Birdpacker could be at the centre of an organisational hub, with partner organisations identified on each of the other major islands across Indonesia to lead efforts in their region, as they will be best placed to promote the event among local participants. For example, Planet Indonesia, a conservation NGO based in Kalimantan that carries out important projects to conserve wildlife and works closely with local communities (Miller et al., 2020), might be a suitable partner to coordinate efforts in Kalimantan. This type of project would need to be built up gradually, but could be an effective means to initiate population monitoring for a wide range of species, and would have the positive impact of growing a large and connected grassroots network of Indonesian conservationists.

Ecological fieldwork for rare and endangered species

For some of the species most threatened by the cage-bird trade there are important gaps in ecological knowledge that hinder efforts to effectively conserve them and/or carry out reintroduction work. In part this is simply due to a lack of research, but it is also true that there are unique difficulties associated with studying rare species affected by a threat like trapping—the last places where overexploited birds are found are unlikely to be the best places for them ecologically, but rather the places where the threat has taken longest to reach them (Caughley, 1994). Despite these difficulties, ecological research is clearly needed for many species to determine population size and abundance, nesting and feeding preferences (Collar et al., 2012; Eaton et al., 2015; Lee et al., 2016; Reuleaux et al., 2018).

Recommendation: Ecological fieldwork would be beneficial for a number of the priority species identified by the ASTSG for which knowledge is still relatively limited regarding habitat use, foraging, and reproductive ecology (ASTSG, 2022). Although this is not an exhaustive list, ecological field research would be especially useful for the following species:

- Brown-cheeked Bulbul (*Alophoixus bres*)
- Ruby-throated Bulbul (*Rubigula dispar*)
- Orange-headed Thrush (*Geokichla citrina*) on Java and Bali (ssp. *rubecula*)
- Javan White-eye (*Zosterops flavus*)
- Javan Heleia (*Heleia javanica*)
- White-bellied Fantail (*Rhipidura euryura*)
- Javan Leafbird (*Chloropsis cochinchinensis*)
- Sumatran Leafbird (*Chloropsis media*)
- Greater Green Leafbird (*Chloropsis sonnerati*)
- Bare-throated Whistler (*Pachycephala nudigula*)
- Sulawesi sturnids (covered in previous section)

Promoting community-based conservation

Many of the bird species affected by the cage-bird trade are distributed predominantly outside protected areas and exist alongside people. This presents conservation challenges (e.g. increased likelihood of trapping) as well as opportunities to develop small bottom-up community projects that seem to be proving more successful than the much more expensive large integrated conservation and development projects (ICDPs) (Horwich and Lyon, 2007). Community conservation has been implemented successfully elsewhere in Asia for the conservation of pangolins (Khatiwada et al., 2020) and tigers (Neelakantan et al., 2021), and has had relatively strong results so far on Nusa Penida for the Bali Myna (D. Donato, pers. comm.). Community conservation projects could be carried out in isolation of protected areas, but, as is the case at Bali Barat National Park, they may also benefit from being co-ordinated alongside national park authorities to expand species' ranges beyond protected areas through cooperative enterprises that involve local communities and reduce the potential threat of trapping (Squires et al., in review).

Recommendation: Identify communities where community conservation projects could be initiated in order to protect birds occurring outside protected areas; this could be on the edge of existing protected areas, so that national park authorities could lend their expertise to the project and provide better buffer areas for the wildlife within the protected area. NGOs such as Planet Indonesia are already implementing this type of work in Kalimantan (Miller et al., 2020), so if similar projects on Java are created without support from national park authorities it is recommended that project leaders seek advice from them or other experts in the field. It has been reported that small scale community conservation initiatives have had more success to date than top–down efforts that are larger in scale but more expensive (Horwich and Lyon, 2007). It is therefore recommended that, at least initially, small community conservation projects with relatively limited scope are first attempted to provide a blueprint for other projects to follow.

Advancing reintroduction ecology

Species reintroductions will become an increasingly important component of the response to the Asian Songbird Crisis, as conservation breeding efforts develop and sufficient numbers of individuals of the most threatened species become available for release into the wild (Collar et al., 2012; Collar and Butchart, 2014). However, there are important ethical considerations regarding the individual birds to be aware of when undertaking a species reintroduction (Thulin and Röcklinsberg, 2020), so planning for reintroduction is essential and they should, as far as possible, follow the guidelines drawn up by experts in the field (IUCN/SSC, 2013). Some of the reintroductions that have been implemented to date across Indonesia have not followed these guidelines closely enough. For example, the reintroduction of the Bali Myna has faced many issues, not least that trapping in and around the release site was never properly addressed, so that released birds in some cases were swiftly entering the cage-bird trade (Jepson, 2016). Finally, it is problematic that the outcome of reintroductions in the past have not always been properly documented, although there are some examples of good reporting of outcomes (Collins et al., 1998; Owen et al., 2014). A lack of reporting of outcomes not only represents a serious missed opportunity to improve our knowledge of reintroductions, but can create bad feeling among project stakeholders that have invested their resources to support work.

Recommendations: The minimum requirements before carrying out a reintroduction should include (1) an assessment of socio-economic and ecological conditions to assess the suitability of potential release sites based on the existing knowledge for the species; (2) local communities should be engaged to measure attitudes towards the release, address concerns people may have, and foster support for the project; and (3) any necessary security or patrol arrangements should be put in place. The possibility for experimentation during releases, such as testing the differential effect of soft- and hard-releases on outcomes, may also be explored to increase learning opportunities (Armstrong and Seddon, 2008).

After any reintroduction has taken place, it is recommended that the released population is subject to post-release monitoring to measure the population size and trend, survival rate, and reproductive success. This could be achieved by implementing cost-effective methods such as marking individuals using colour rings, and if nest-boxes can be installed then the next generations of birds could be tracked using this method and breeding productivity measured (Berger-Tal and Saltz, 2014). Finally, it is strongly encouraged that the results of reintroductions, however small the release, are documented in order to improve transparency and contribute to the growing science of reintroductions for passerines. It should be noted that these recommendations are documented in greater detail in the IUCN guidelines for reintroductions (IUCN/SSC, 2013).

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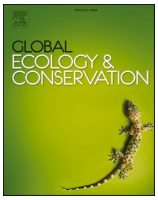

Citizen science rapidly delivers extensive distribution data for birds in a key tropical biodiversity area

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ABSTRACT

Citizen science projects remain rare in biodiverse yet data-poor countries, contributing to a shortfall in data for biodiversity monitoring and promoting public stewardship of nature. We document and analyse BigMonth2020, a month-long birdwatching event across Java and Bali, publicised through social media and incentivised with grants and competitions. Over 20,000 lists containing 100,000 bird records were submitted to the 'Burungnesia' phone application. Spatial coverage extended to 71% of the islands' 3408 atlas grid squares (6.9 × 6.9 km), including 1613 previously undocumented squares, with 353 bird species recorded, representing 74% of Java and Bali's avifauna excluding vagrants; 27 threatened species were recorded, with new records for 204 grid squares. Almost 25% of contributors were female, 72% were under 30 years old, and most were graduates and members of birdwatching clubs. The project cost less than US$10,000 to run, and serves as a model for rapidly establishing a distributional baseline for monitoring biodiversity trajectories in the tropics.

1. Introduction

Obtaining broad-scale ecological data to evaluate species distributions and their responses to environmental change requires resources unavailable to most researchers (Dickinson et al., 2010). Citizen science is a practical way to bridge the resource gap, with projects typically mobilising volunteers to gather and/or classify data following a protocol developed by experts (Dickinson et al., 2012). Ecology-related citizen science projects vary widely, ranging from online exercises (Shamir et al., 2014; Swanson et al., 2015; Rosenthal et al., 2018) to field surveys (Preston, 2013; Gillings et al., 2019). Scientists benefit from citizen science by obtaining large datasets with higher coverage, the volunteers experience direct involvement in science and enhance their skills (Dickinson et al., 2010), and wider society benefits, as volunteers often share their knowledge, increasing levels of scientific literacy and environmental advocacy among peers (Johnson et al., 2014).

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Citizen science has recently proliferated in developed countries but remains rare in developing countries (Chandler et al., 2017). This is problematic for conservation, since biodiversity hotspots predominantly coincide with data-poor, highly threatened areas (Brooks et al., 2006; Fisher and Christopher, 2007). Barriers to citizen science in developing countries include low awareness of opportunities (for both participants and institutions) (Pocock et al., 2019), low appreciation of its environmental and societal value (Chandler et al., 2017), and low levels of expertise, time, money and perceived personal benefits (Requier et al., 2020). By way of counterbalance, the global rise in smartphone ownership and internet coverage in many developing countries gives citizen science both practicality and appeal (August et al., 2015; Taylor and Silver, 2019).

Indonesia is one of the most biodiverse nations on earth, but habitat loss through land-use change is a major threat to wildlife and habitats, while illegal trapping of wild birds has triggered an 'Asian Songbird Crisis' (Margono et al., 2014; Lee et al., 2016; Hughes, 2017). This trade affects at least 32 threatened species in Indonesia and many common species (Eaton et al., 2015; BirdLife International, 2021), with households in Java, Indonesia's most populous island, keeping some 74 million cage-birds (Marshall et al., 2020). Baseline distribution data for widespread Javan species are now urgently required to identify future changes. To date, such data have been gathered by Indonesian birdwatchers for the first Indonesian Bird Atlas ('Atlas Burung Indonesia'; Taufiqurrahman et al., 2016), and through eBird (Sullivan et al., 2014). However, these data are predominantly gathered in urban centres, ecotourism hotspots and protected areas, leaving large intervening spaces. To develop baseline distribution models for common birds, data need to cover the range of habitats and land-use types within the study area (Phillips et al., 2006).

To this end, we developed 'BigMonth2020′, a citizen science project held in Java and Bali during January 2020 which aimed to engage Indonesian society, expand the coverage of bird distribution data, and incidentally contribute to the Indonesian Bird Atlas. Here we outline the scope and design of BigMonth2020, the data collection and validation protocols followed, and the promotional campaign and incentive scheme intended to attract participation. We then assess the bird data collected for their novelty, composition and quality, and examine the demographics of those who contributed to BigMonth2020. Finally, we review the project's outcomes and the benefits and pitfalls of a citizen science event, providing lessons learnt for similar initiatives and the continuation of the work on Java and Bali.

2. Methods

2.1. Inception

We developed an outline plan for an incentive-driven inclusive birdwatching event, to be promoted primarily through social media, and enlisted the involvement of two Indonesian partner organisations: the Indonesian Ornithologists' Union (IdOU), whose members are predominantly academics or work for conservation NGOs, and Birdpacker, a grassroots birdwatching community whose citizen science phone application 'Burungnesia' (*burung*, Indonesian 'bird'; *-nesia* from Indonesia) was released in 2016. This application enables birdwatchers to submit georeferenced bird lists in support of efforts to produce *Atlas Burung Indonesia*, the country's first national bird atlas.

2.2. BigMonth2020

The event's scope was limited to Java and Bali to ensure its logistics were manageable and it lasted a full month to maximise data accumulation within the constraint of limited administrative resources. This gave contributors ample opportunity to log data yet was short enough to maintain social media interest. We timed the event to coincide with university and national holidays, when participants had more free time and were dispersed from large urban centres.

A competition was promoted via social media. We purchased ornithological equipment (binoculars, telescope, field guides, etc.) as prizes for various categories, including the overall top-ten contributors of bird lists, the best social media influencer, and the best photograph. We also established a small grant scheme, administered online with simple bank transfers, to cover transport and subsistence for trips to under-recorded areas. After the third week, we identified the five largest remaining unrecorded areas and offered grants to people to visit them. A total of IDR 27M (US$1850) was divided among 51 applicants, in grants ranging from US$6.80 for one person on a day trip to US$200 for a seven-day trip by eleven students. Overall operational costs, including a small team assembled by Birdpacker to administer the various aspects of the event (i.e. social media, data handling and expert validation), were covered by US $7000 from the Oriental Bird Club (OBC) and US$400 by Idea Wild. Other indirect project-related costs included the incidental funding of TMS and SJM, as well as the in-kind cost of developing and running the data-logging application.

2.3. Promotion

In November 2019, partners from Manchester Metropolitan University (MMU) and Birdpacker presented BigMonth2020 at the annual Indonesian birdwatchers' conference. Thereafter, promotion was carried out on social media. We posted Indonesian-language promotions via Facebook and Instagram, and English-language promotions via Twitter. We directly contacted 33 naturalist clubs (22 of them university societies), eleven Indonesian NGOs, two Indonesian zoos, and the European (EAZA) and North American (AZA) zoo associations via email and social media. Thirty-four organisations became official supporters and their logos featured on promotional material.

Social media promotions began with a digital project poster (Fig. 1) on Facebook, Instagram and Twitter two weeks before the event, followed by information about its aims and objectives, the data collection protocol, and the competition rules. A BigMonth2020 webpage provided tutorials for data collection and input, as well as identification guides for lookalike species. Once BigMonth2020 commenced, social media posts were made almost daily on Instagram and Facebook, providing updates on the prize competition, data accumulation, unusual findings, and priority grid squares. Many participants shared our promotions or created their own content, increasing the project's reach (Fig. A.1). The MMU partners visited the Birdpacker team in Malang, East Java, in mid-January and collected data alongside students, NGO staff and government officials, while the OBC chairman recorded data with members of the Birdpacker team in East Java.

2.4. Data collection

Participants were asked to focus their efforts on low-elevation land (< 800 m altitude) outside protected areas, because the largest gaps in data occur in these relatively accessible areas, and to use the Burungnesia phone application to submit their data; we did promote the use of eBird but no participants chose to use this during BigMonth2020. Participants compiled lists of bird species recorded at a unique location as either presence-only or count data (sample data and full protocol in Appendix B). Participants were encouraged to search for birds around the start location for at least one hour and to begin a new list if they travelled 3 km away from the start point. As Burungnesia currently lacks the functionality to record extensive list metadata, we could not obtain data on survey

218 likes

birdpacker Bagi yang nonton live dadakan kita pagi ini pasti paham kalau kita baru saja kedatangan paket khusus! Ini dia penampakan... more

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Fig. 1. From top left clockwise: BigMonth2020 promotional poster used on social media; Instagram promotion of competition prizes; BigMonth2020 competition winner being awarded his prize; and young participants wearing their BigMonth2020 T-shirts with field guides they won as prizes.

effort (distance travelled, survey duration) or list completeness (i.e. whether all species encountered were recorded).

We increased the resolution of the Indonesian Bird Atlas grid system (0.25° × 0.25°, WGS 84) by dividing each square into 16 cells (0.0625° × 0.0625°; 6.9 × 6.9 km), resulting in 3408 grid squares. Using existing data from eBird and the Indonesian Bird Atlas, we categorised squares as unvisited (no bird lists) or visited at two levels (1–5 bird lists, > 5 bird lists). Data for the Indonesian Bird Atlas were collected manually until the Burungnesia application was released in 2016; here, both datasets are combined and referred to as 'Indonesian Bird Atlas data'. BigMonth2020 participants could download the map as a *.kmz* file. Trips to unvisited squares were incentivised using a weighted point-scoring system for the competition, with extra points awarded if five bird lists were submitted from an unvisited grid square. The map and grid square status were updated every three days and a new download made available.

2.5. Data validation

Six experts validated submitted data throughout and after the event. A bird list was flagged for further review if (1) a location description did not match the GPS coordinates; (2) the habitat description did not match the habitat depicted on Google Earth; or (3) a species record was deemed unusual in terms of location, time of year or habitat. For flagged records, the observer was asked for supporting evidence, and depending on the response the record was either retained in or removed from the database. Records without coordinates were omitted. All records were adjusted so that taxonomy followed HBW and BirdLife International (2019).

2.6. Participant questionnaire

An Indonesian-language questionnaire (Appendix C) was posted online to learn more about the event's participants, with a free BigMonth2020 T-shirt (Fig. 1) offered to all respondents. Participants provided demographic data (age, education level, employment status) and information on their birdwatching expertise, motivations and perceptions of conservation issues.

2.7. Ethical statement

The questionnaire was administered by Universitas Atma Jaya Yogyakarta, followed their research guidelines and conformed to standards in BSA (2017). It explained its objectives at the start and participants provided informed consent by answering the questions. Questionnaire data were accepted from adults only (> 18 years) and anonymised before analysis.

2.8. Data analysis

The species recorded were classified into six functional groups to examine differences in data recording. Birds were categorised as either raptors, aerial feeders or waterbirds based on taxonomy and feeding strategy, while all species outside those categories were grouped by preferred habitat (woodland, open/agriculture or scrub/savanna) using BirdLife Data Zone information (BirdLife International, 2020; see Table D.1). We calculated Shannon's evenness (E_H) (Peet, 1974) for the six classes to measure the within-group relative abundance of records for each species, with values ranging from 0 (group dominated by few taxa) to 1 (records evenly distributed among taxa).

To identify participant attributes associated with high survey effort and data composition, we fitted two generalised linear models (GLMs) using the dataset of 134 participants' questionnaire responses combined with the bird data they submitted to Burungnesia. The survey effort model used the number of grid squares visited as the dependent variable, while the data composition model used a 'rarity recording' metric, calculated following August et al. (2020): every species was ranked according to the number of times it was recorded and assigned a rarity value from 1 = most common to 100 = most rare; we then subtracted the median rarity value across all observations in the dataset from the median rarity value across all records for the participant, so that negative values of the metric indicate that the participant recorded common species more frequently than expected and positive values show that the participant recorded rare species more frequently than expected. The predictors used in both models were age in years, gender, occupation (formal employment, freelance-type work, student), birdwatching experience in years, and bird club membership; the number of grid squares visited was included as a predictor in the data composition model. All statistical analyses were conducted in R (4.0.2, R Core Team,

Table 1
Summary statistics for two existing citizen science bird distribution datasets for Java and Bali and the BigMonth2020 dataset.

Characteristic	eBird	Indonesian Bird Atlas	BigMonth2020
Years covered	1970–2020	2003–2020	2020
Number of records	180,975	39,011	102,887
Number of bird lists	11,666	4130	22,055
Species recorded (threatened)	517 (39)	469 (38)	353 (27)
Median species recorded per bird list (IQR)	9 (3–19)	6 (2–13)	4 (3–6)
Grid squares (exclusive to dataset)	594 (67)	827 (135)	2417 (1613)
Contributors	1241	483	218[a]
Median number of lists per contributor	4 (1–11)	2 (1–6)	8.5 (3–27)

[a] unique Burungnesia users who submitted data. Some people recorded in groups and the total number of participants is estimated at 373. IQR: interquartile range.

2020) using package 'MASS' (Venables and Ripley, 2002).

3. Results

3.1. Data accumulation

A total of 22,055 bird lists were submitted across Java and Bali during BigMonth2020 comprising 102,887 bird records (Table 1). The daily number of bird lists submitted grew throughout the event, punctuated by peaks in submissions at weekends (average 55% increase vs. preceding weekdays) (Fig. 2). The difference in data accumulation between the first (2564 lists, 11.6%) and last (8470 lists, 38.4%) seven-day period was particularly sharp.

During BigMonth2020 218 unique users submitted data to Burungnesia, although the total number of participants was an estimated 373 because some worked in groups with only one member submitting data. Ten contributors collected 72% of all bird lists (16,090), 25 contributors submitted over 100 bird lists, and 99 submitted at least ten. A median of four species (IQR 3–6 species) per bird list was slightly lower than the eBird and Indonesian Bird Atlas datasets (Table 1).

3.2. Data coverage

At least one bird list was recorded in 2417 (70.9%) of the 3408 grid squares across Java and Bali (Fig. 3). Data were initially concentrated around major cities, but coverage steadily expanded to remoter areas (Fig. E.1). Many low-elevation agricultural areas were surveyed for the first time. Coverage was greatest in Central Java and least in remote parts of West and East Java, with limited road access and a higher proportion of forested uplands.

Prior to BigMonth2020, bird occurrence data were available for 1092 (32% overall; eBird 17.4%; Indonesian Bird Atlas 24.2%) of the grid squares across Java and Bali (Table 1). BigMonth2020 extended bird distribution data to a further 1613 (47.2%) grid squares, representing a 147% increase in coverage. Combined with eBird and the Indonesian Bird Atlas data, total coverage is now 79.3% of grid squares (Fig. 4). Coverage increased by over 50% for 72 species and over 100% for 37 (Table D.1).

3.3. Data composition

There were 353 bird species recorded during BigMonth2020, representing 74% of those known from Java and Bali excluding vagrants (Lepage, 2020). Cave Swiftlet (*Collocalia linchi*), Eurasian Tree Sparrow (*Passer montanus*) and Sooty-headed Bulbul (*Pycnonotus aurigaster*) made up 16.2%, 10.2% and 6.0% of the 102,887 records, respectively. Twenty-seven species on the IUCN Red List

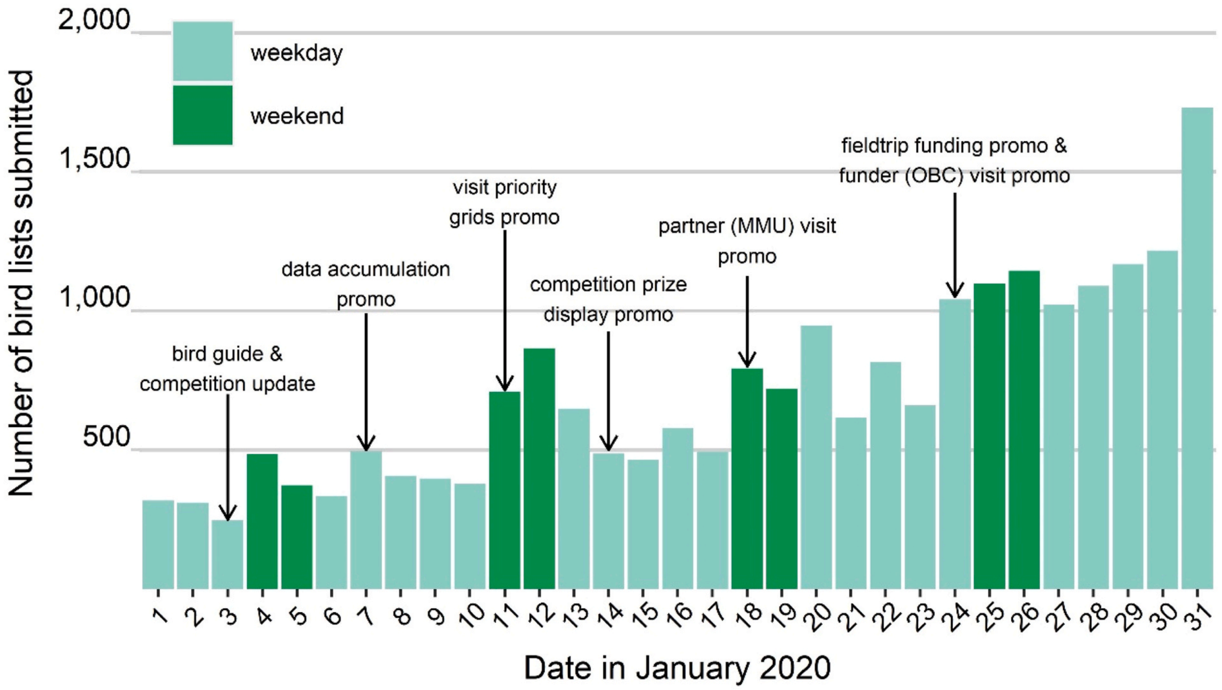

Fig. 2. Daily number of bird lists uploaded to the Burungnesia phone application during BigMonth2020. Annotations show the timing and content of popular social media posts by Birdpacker. Submission peaks on 13 and 20 January are probably data-reporting lags from weekends, and the peak on the final day is probably contributors entering data before the competition cut-off time.

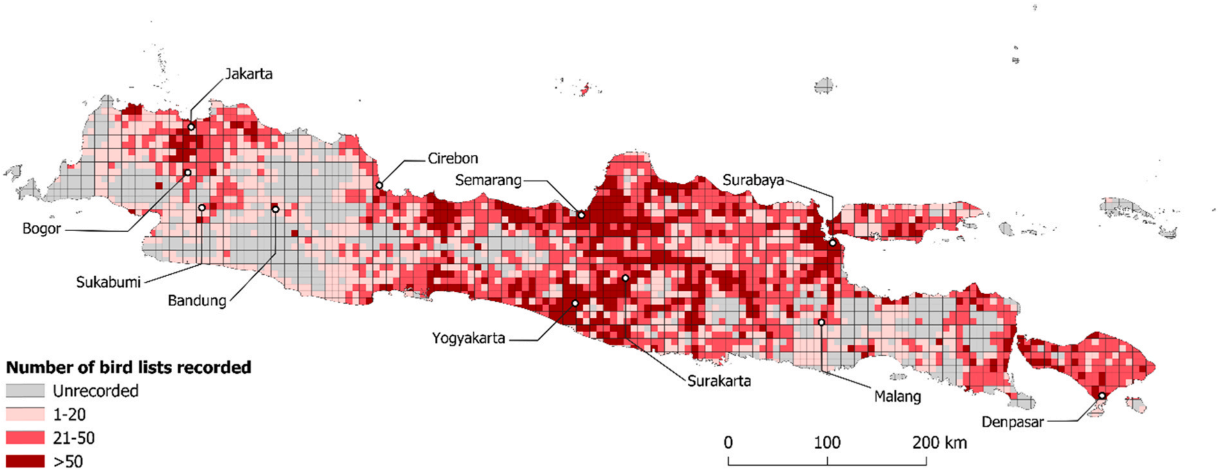

Fig. 3. Data coverage for BigMonth2020. Grid squares (6.9 × 6.9 km) are coloured according to the number of bird lists recorded within them. Major cities in Java and Bali are shown.

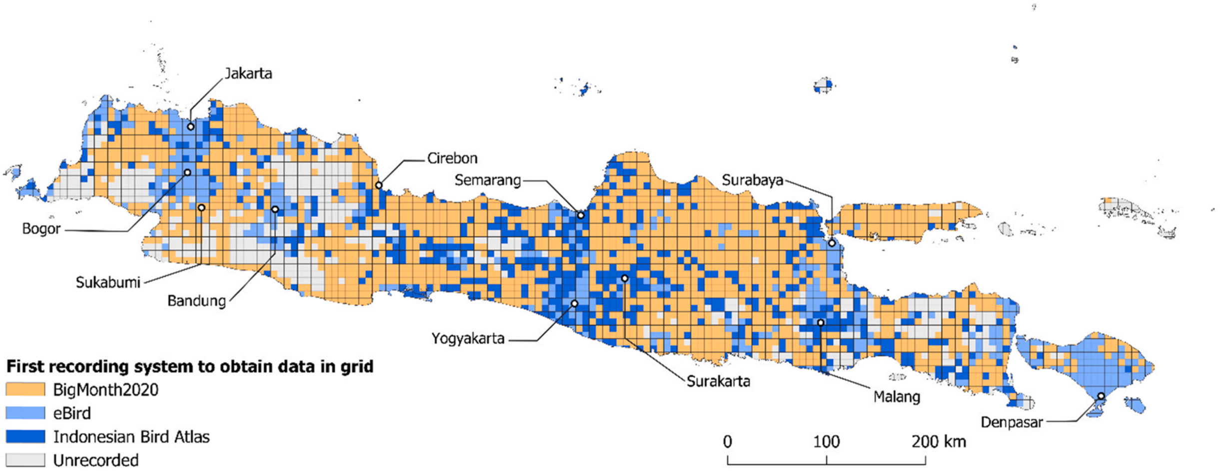

Fig. 4. Bird distribution data coverage for Java and Bali. Grid squares (6.9 × 6.9 km) are coloured according to which recording system was first to obtain data there: BigMonth2020 (n = 1613; 47.3%); Indonesian Bird Atlas (n = 575; 16.9%); eBird (n = 514; 15.1%); unrecorded (n = 706; 20.7%).

(14 Vulnerable, 9 Endangered, 4 Critically Endangered) were recorded, ten of which are significantly affected by the cage-bird trade (BirdLife International, 2020). Six threatened species were recorded on > 20 lists: Javan Myna (*Acridotheres javanicus*) (142 lists), Javan Coucal (*Centropus nigrorufus*) (101), Sangkar White-eye (*Zosterops melanurus*) (66), Ruby-throated Bulbul (*Rubigula dispar*) (62), Milky Stork (*Mycteria cinerea*) (33) and Java Sparrow (*Lonchura oryzivora*) (23). Threatened species were recorded for the first time in 204 grid squares, and seven species were recorded for the first time in at least ten squares, with grid square coverage for these increasing by 15.5–69.8% (Table D.1).

Table 2
Summary of bird data recorded during BigMonth2020, with species grouped based on taxonomy and feeding strategy (raptors, waterbirds and aerial feeders) or preferred habitat (woodland birds, birds of open country/agricultural areas, and scrub/savanna birds).

Group	Species	Percentage of all species	Percentage of records	Threatened species	Evenness (E_H)
Open country/agriculture	31	8.8	31.4	3	0.57
Woodland	172	49.0	26.7	15	0.50
Aerial feeders	15	4.3	24.1	0	0.42
Scrub/savanna	21	6.0	9.3	2	0.56
Waterbirds	91	25.9	7.9	6	0.66
Raptors	21	6.0	0.6	1	0.70

Species of open country and farmland were most frequently observed (Table 2), with Eurasian Tree Sparrow, Scaly-breasted Munia (*Lonchura punctulata*) and Javan Munia (*L. leucogastroides*) comprising 70.8% of these records. Nearly half the species inhabit woodland but accounted for only a quarter of observations. Aerial feeders were over-represented in the dataset (4.3% of all species recorded accounting for 24.1% of all observations), as were scrub and savanna birds; waterbirds and raptors were under-represented.

3.4. *Data quality*

During data validation, 845 bird lists (3.8%) were flagged for review. Data from 494 (58.5%) lists were retained in the database following verification, 253 (29.9%) were retained with updated location or species data, and 98 (11.6%) were removed for lack of supporting evidence. Some easily misidentified species commonly required review, notably tailorbirds (*Orthotomus* spp.): 19 of 60 records of Ashy Tailorbird (*O. ruficeps*) were reviewed, of which 13 were accepted with evidence, five re-identified as Olive-backed Tailorbird (*O. sepium*) and one as Common Tailorbird (*O. sutorius*).

3.5. *Participant characteristics*

Of the estimated 373 participants, 188 (50.4%) answered the questionnaire, all of whom were Indonesian. Of these, 23.4% were female and 71.8% were under 30 years old. Most respondents lived in East Java (28.2%), Yogyakarta (21.8%) and Central Java (14.9%), with fewer in West Java (12.8%), Jakarta (5.3%), Banten (2.1%) and Bali (2.1%), and the remainder (12.8%) lived elsewhere in Indonesia. Most were members of a bird club (67%) and discovered BigMonth2020 through their club (39%) or social media (20% Instagram; 9% Facebook); 36.2% owned a camera but not binoculars, 30.3% had both a camera and binoculars, 16.5% used binoculars alone and 17% had no equipment. Top-ranking motives for their participation in BigMonth2020 were ‘contributing to conservation’ (74% of respondents) followed by ‘seeing new bird species’ (64%) (Fig. F.1a). The cage-bird trade and habitat loss were considered equally important threats to birds in Java, followed by climate change (Fig. F.1b). The number of grid squares visited by participants (sampling effort) was significantly higher for participants with more birdwatching experience ($z = 2.79 \pm 0.03$, $p < 0.01$) and who were male ($z = 2.32 \pm 0.28$, $p = 0.02$). In terms of rarity recording, participants who visited more grids during BigMonth2020 tended to record common birds more frequently than expected ($t = -2.44 \pm 0.006$, $p < 0.01$). Full GLM parameters are provided in Appendix G.

4. Discussion

BigMonth2020 demonstrates the viability of citizen science in Indonesia and could be replicated in other countries where citizen science projects are scarce and biodiversity seriously under-recorded (Meyer et al., 2015). Over 300 Indonesians (Appendix H) generated a dataset comprising over 100,000 bird records, half of which were collected in previously unsurveyed areas (see https://bigmonth2020.shinyapps.io/shiny_app/).

4.1. *Data coverage and composition*

BigMonth2020 has more than doubled bird distribution data coverage on Java and Bali, extending to almost 80% of grid squares. Sampling biases related to contributor distribution are a common and expected feature of citizen science data (Dennis and Thomas, 2000; Romo et al., 2006), and the spatial distribution of data here broadly reflects the accessibility of squares and the distribution of contributors, the most prolific of whom mainly lived in Central Java, Yogyakarta and East Java. Consequently only one in ten bird lists were submitted in western Java (Banten, Jakarta and West Java provinces) despite half Java’s population residing there (Badan Pusat Statistik, 2016). While inaccessible upland areas in western Java account for the largest remaining gaps in data coverage, some accessible areas close to urban centres were unvisited. In part this is because Central Java and Yogyakarta possess more bird clubs, which are associated with the region’s cluster of biology-focused universities. It may also reflect cultural differences in interest in birds between the Sundanese in western Java and the Javanese in central and eastern Java (Jepson and Ladle, 2005). Moreover, it could be linked to the rapid urbanisation of western Java (Firman, 2017), producing a human–nature disconnect and reduction in pro-environmental feeling (Cleary et al., 2020).

The considerable increase in data coverage for many commoner species will enable us to develop robust distribution models to establish a distributional baseline against which to monitor the stability of the environment, as changes in the distribution of common species representative of major habitat types can reveal patterns of wider ecosystem health (Caro and O’Doherty, 1999). Distribution models for common species, which contribute most to patterns of overall species richness (Vázquez and Gaston, 2004), could be used to identify areas of relatively high biodiversity value in under-recorded regions of Java. Estimating the distribution of rare and threatened species is another important aspect of biodiversity monitoring (BirdLife International, 2021), and BigMonth2020 delivered valuable data for 27 threatened species, for nine of which we obtained at least the minimum number of records needed to build accurate distribution models (Proosdij et al., 2016). However, some of Java’s Critically Endangered species, such as Black-winged Myna (*A. melanopterus*) and Javan Pied Starling (*Gracupica jalla*), were conspicuous absentees from the dataset, highlighting the disastrous declines of some species due to bird trapping in the region.

4.2. Participation and demographics

BigMonth2020 engaged with over 300 Indonesian citizens (Appendix H), a level of participation comparable to similar schemes in Africa and Taiwan (Ko et al., 2014; APLORI, 2020). BigMonth2020 had more participants under 30 years old than projects in countries where birdwatching has a longer tradition with a wider spectrum of cohorts (Wright et al., 2015; MacPhail and Colla, 2020). It also attracted people who were not already birdwatchers, suggesting that such events can promote engagement with nature and conservation issues. Although this demographic may present challenges relating to capacity to participate (e.g. less disposable income, limited transport) and data quality (i.e. less birdwatching experience, limited access to equipment), it indicates a growing community of nature enthusiasts who could rapidly become a significant body of conservation advocates. Retaining participants is, however, critical if BigMonth2020′s baseline is to serve its purpose, because participant expertise can be expected to increase over time, especially if project goals and data use are effectively communicated (Forrester et al., 2017). This is best achieved by continuing to appeal to peoples' varied initial motivations for participating (Clary and Snyder, 1999; Bruyere and Rappe, 2007).

Birds have a deep cultural significance in Indonesia (Jepson and Ladle, 2005), but bird-keeping and songbird competitions are almost exclusively male-dominated activities (Marshall et al., 2020); encouragingly, however, a quarter of questionnaire respondents for BigMonth2020 were female. Nevertheless, female participants visited fewer grid squares than average, suggesting that gender-specific barriers to participation still exist and initiatives to encourage female participation are warranted. Even so, we speculate that birdwatching could develop as an inclusive pursuit in Indonesia, irrespective of sex, age or social class, and events like BigMonth2020 are ideally placed to promote this. The distribution of contributors to BigMonth2020 mirrors the prevalence of bird-keeping across Java (Jepson and Ladle, 2009; Marshall et al., 2020), suggesting that people from bird-keeping households could be attracted to birdwatching and conservation as an alternative means to enjoy birds, thereby helping to reduce the threat from the cage-bird trade.

4.3. Project design and data collection

Some adjustments to the sampling strategy we used for BigMonth2020 could help address the spatial bias and remaining gaps in data coverage. Besides the bias we introduced by asking volunteers to visit low-elevation unprotected areas, survey bias was linked to human population density and accessibility, a common problem when *ad hoc* sampling is used, reflecting the trade-off between protocol complexity (data quality) and ease of participation (Bird et al., 2014; Geldmann et al., 2016). While it should be minimised, spatial bias does not preclude the accurate estimation of species distributions (Johnston et al., 2020). Moreover, the uptake of our small incentives to explore under-recorded areas suggests that further such incentives to visit grid squares remote from major roads would help reduce the current spatial bias.

Some issues related to the data collection protocol can be addressed by modifications to the data-logging application. First, the number of taxa recorded per bird list for BigMonth2020 was low relative to other reporting systems for the same area, suggesting that either sampling effort per list (not recorded) or bird detection frequency was lower. The design of the competition, kept simple to promote engagement, probably contributed to this by awarding points for every bird list submitted, thereby encouraging low sampling effort; this is corroborated by our finding that participants who submitted most data tended to record commoner birds more frequently than expected. Requiring a minimum sampling effort for every bird list could resolve this issue, and highlights the need to design incentives carefully. Second, contributors may not have reported all species they encountered—possibly ignoring common species or those posing identification challenges (Snäll et al., 2011; Tulloch et al., 2013)—so inferring species absence was not possible. If absences and sampling effort are known, biases can be accounted for statistically, so these metadata should be required by future versions of the application (Fink et al., 2020). Finally, we manually validated photographic evidence requested from users after bird records were flagged. To expedite this process in future, users should be able to attach photographic evidence to their records during data submission, and the proportion of correctly identified photographs could be used as a data quality metric (Vantieghem et al., 2017).

4.4. Biodiversity monitoring in Java

BigMonth2020 delivered high geographic coverage of the study region and valuable distribution data for most of Java's bird species. The immediate aim following the event is to widen the network of citizen science birdwatchers and improve the utility of the data collected, in order to establish distributional baselines for birds across Indonesia. Extending survey effort beyond the populous islands of Java and Bali poses a logistical challenge given Indonesia's geography, but, beyond simply replicating the efforts described here, in more remote regions organisers could seek to engage with local stakeholders and integrate forms of traditional and indigenous knowledge into the project (Leach and Fairhead, 2002). It would also be desirable to extend the monitoring protocol to better enable the calculation of population trends for common birds from the dataset, which as previously discussed is not possible with the data collected in BigMonth2020. To achieve this, a repeated samples protocol is needed consisting of a random selection of fixed sites, stratified by habitat type, to be surveyed at regular intervals. Meanwhile, the *ad hoc* sampling adopted for Bigmonth2020 would be retained because it is inclusive, offers training opportunities for less experienced volunteers, and helps recruit, retain and involve more casual participants (Higby et al., 2012). Finally, spatiotemporal data coverage can be extended and duplication of effort avoided by establishing closer connections with existing initiatives including the Asian Waterbird Census (International Waterbird Census, 2020), eBird and Raptor Watch (Yuda, 2017).

5. Conclusion

We have demonstrated the potential of citizen science to address gaps in biodiversity distribution data coverage that are unlikely to be filled by traditional fieldwork, as well as its ability to engage with a young demographic, not all of whom were seasoned bird-watchers. Our approach was based on a tailored incentive scheme and targeted social media promotion campaign and stimulated a data collection approach built on existing local efforts. We have identified key aspects of the incentive scheme and data collection protocol that can be adapted to improve data quality, and what would be required to monitor population trends as well as distributions. Considering the popularity of citizen science among funders (Gura, 2013) and the benefits that can be derived from it (McKinley et al., 2017), we hope that the findings and processes reported here will prove a basis, guide and stimulus to similar endeavours across the tropics.

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Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.gecco.2021.e01680. The following supplementary data are available: social media posts by BigMonth2020 participants (Appendix A), the data recording protocol using the Burungnesia application (Appendix B), the feedback questionnaire (Appendix C), a summary of species data recorded during BigMonth2020 (Appendix D), spatiotemporal data accumulation during BigMonth2020 (Appendix E), questionnaire responses related to motivations to participate in BigMonth2020 and threats to Javan birds (Appendix F), GLM parameters (Appendix G), and contributors to BigMonth2020 data collection (Appendix H). Pre- and post-BigMonth2020 data coverage for species recorded (excluding 'sensitive species') can be viewed at https://bigmonth2020.shinyapps.io/shiny_app/.

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RESEARCH ARTICLE

Controlling trapping, overgrazing and invasive vegetation is key to saving Java's last population of the Black-winged Myna

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ABSTRACT

The Black-winged Myna (*Acridotheres melanopterus*) is an Endangered passerine endemic to the islands of Java and Bali, Indonesia. Illegal trapping to supply the cage-bird trade has led to its near-total extinction, with the global population estimated to number fewer than 100 individuals. We estimated the current range and population size of the species at Baluran National Park, which supports Java's last known population, and used species distribution modeling to evaluate potential suitability of currently unoccupied areas across the park to identify priorities for management intervention. We estimate that the Black-winged Myna population numbers 179 individuals (95% CI: 111–288; density: 14.3 ± 3.5 individuals km^{-2}) and that its current range is 12.3 km^2. Our model indicated that some 72 km^2 of the park (30% of total area) has potentially suitable habitat for the species, and we infer that the principal cause for the disparity between its current and potential range is trapping, compounded by savanna loss and degradation due to illegal domestic cattle grazing and the spread of invasive thorny acacia (*Vachellia nilotica*). The partial clearance of acacia in recent years appears to have assisted a modest population recovery by the myna. Its further population growth and range expansion in Baluran will depend on effective management of illegal poaching, further clearance of acacia, and easing domestic cattle grazing pressure on areas of savanna, particularly through engagement with communities living inside the park. Any actions that increase the size of the Black-winged Myna population are likely to benefit other threatened savanna-dependent wildlife in the park, notably banteng (*Bos javanicus*) and Green Peafowl (*Pavo muticus*). While our models and recommendations may be applicable to other protected areas in Java, and indeed other threatened myna species, trapping and habitat change may have site-specific dimensions, especially outside of protected areas, and thus demand local bespoke solutions.

Keywords: Acridotheres melanopterus, Asian songbird crisis, Baluran, Black-winged Myna, threatened species, Indonesia, Java

LAY SUMMARY

- The Black-winged Myna, confined to Java and Bali in Indonesia, is threatened with extinction due to illegal trapping for the cage-bird trade. Baluran National Park supports the last known population on Java.
- We estimated the number of mynas in the park, mapped where they occur, and assessed their habitat to determine how much of it is currently unoccupied.
- There are ~180 Black-winged Mynas in the park, indicating recent population growth. However, they occupy <20% of the potentially suitable habitat, restricting further population growth.
- Trapping is the foremost factor holding back the mynas, but overgrazing by domestic livestock and invasion by thorny acacia negatively affect the potentially suitable habitat.
- Thorny acacia eradication must continue. Working with the human communities living in the park is key to alleviating the grazing pressure and addressing the trapping issue.

Pengawasan penangkapan, penggembalaan berlebihan serta vegetasi infasif adalah kunci penyelamatan populasi terakhir Jalak putih di Jawa

ABSTRAK

Jalak putih (*Acridotheres melanopterus*) adalah burung berkicau endemik di pulau Jawa dan Bali, Indonesia, yang berstatus Terancam. Selama beberapa dekade terakhir, penangkapan ilegal untuk memasok perdagangan burung dalam sangkar telah menyebabkan kepunahan dengan populasi global diperkirakan berjumlah kurang dari 100 individu. Kami melakukan estimasi sebaran dan ukuran populasi spesies saat ini di Taman Nasional Baluran, yang mendukung populasi liar terakhir yang diketahui di Jawa, serta menggunakan pemodelan distribusi spesies untuk mengevaluasi potensi kesesuaian area yang tidak ditempati Jalak putih di seluruh Taman Nasional untuk mengidentifikasi prioritas intervensi pengelolaan. Estimasi populasi Jalak putih di TN Baluran berjumlah 179 individu (95% CI: 111–288; kepadatan: 14,3 ± 3,5 individu km^{-2}) dan sebarannya saat ini adalah 12.3 km^2. Pemodelan kesesuaian habitat menunjukkan bahwa sekitar 72 km^2 dari taman nasional (30% dari total luas) berpotensi sesuai untuk spesies tersebut, dengan kesimpulan bahwa penyebab utama perbedaan antara sebaran saat ini dan potensi sebaran adalah penangkapan, ditambah dengan degradasi dan hilangnya savana akibat penggembalaan sapi lokal ilegal serta penyebaran akasia berduri yang invasif (*Vachellia nilotica*).

Namun, pembukaan sebagian akasia berduri tampaknya telah membantu pemulihan populasi secara sederhana. Pertumbuhan populasi dan perluasan sebaran jalak putih di Baluran, tergantung pada pengelolaan perburuan liar yang efektif, pengurangan akasia berduri yang terus berlanjut, serta mengurangi tekanan penggembalaan ternak lokal di daerah savanna terutama melalui keterlibatan masyarakat yang tinggal di dalam kawasan. Setiap tindakan untuk meningkatkan jumlah Jalak putih, kemungkinan besar akan menguntungkan satwa liar lain yang berstatus terancam dan bergantung pada savana di dalam kawasan, terutama banteng (*Bos javanicus*) dan merak hijau (*Pavo muticus*). Sementara model dan rekomendasi ini mungkin bersifat umum dalam penerapannya pada kawasan lindung lainnya di Jawa, serta spesies Jalak lainnya yang terancam, namun masalah penangkapan dan perubahan habitat yang terjadi bersamaan mungkin bersifat spesifik lokasi, terutama di luar kawasan konservasi, dan mungkin pada gilirannya menuntut solusi lokal yang tepat.

Kata kunci: Acridotheres melanopterus, Jalak putih, krisis burung penyanyi Asia, Status Terancam, Indonesia, Jawa, Baluran

INTRODUCTION

Biological diversity is being eroded at an unprecedented rate and wildlife trade is a main underlying cause, contributing to enormous declines in species abundance, loss of ecosystem function, and increased risks to human health through zoonotic diseases (Dirzo et al. 2014, Pimm et al. 2014, Benítez-López et al. 2017, Aguirre et al. 2020). Nearly a fifth of all extant vertebrate species are traded, mostly in and from the tropics, with birds and mammals being disproportionately affected (Wyler and Sheikh 2008, Barber-Meyer 2010, Scheffers et al. 2019). Southeast Asia, one of the most biodiverse regions on earth, has among the highest proportion of threatened species for most higher classes of animals (Myers et al. 2000, Sodhi et al. 2010, Hughes 2017). Throughout the region, the trade in wild-caught songbirds—prized for their vocal ability, plumage, rarity, and cultural significance—is having a massive effect on wild populations (Nijman 2010, Lee et al. 2016, Symes et al. 2018, Indraswari et al. 2020). The resulting "Asian Songbird Crisis" has left many species facing extinction, while for others the damage trade has wrought on their populations is still poorly understood due to insufficient monitoring (Eaton et al. 2015, Shepherd and Cassey 2017, Bergin et al. 2018, Marshall et al. 2020). Indonesia, particularly its most populous island of Java, is widely regarded as the epicenter of the bird trade in Southeast Asia, with millions of birds sold annually at markets irrespective of their legal status and an estimated 70 million cage-birds kept in one-third of Java's 36 million households (Chng and Eaton 2016, Harris et al. 2017, Marshall et al. 2020). Almost half of Indonesia's 64 globally threatened songbirds (order Passeriformes) are threatened primarily by trade, and most of them occur on Java (IUCN 2021).

There are several patterns of decline exhibited by species under heavy pressure from habitat loss and trapping: some show dampened population densities across their range (Laaksonen and Lehikoinen 2013) while others collapse into just a few strongholds (Abram et al. 2015, Annorbah et al. 2016). The role of formally protected areas in the conservation of endangered wildlife is also varied, ranging from absolutely critical (Prakash et al. 2019, Ghosh-Harihar et al. 2019) to relatively secondary (Agardy et al. 2003, Kamp et al. 2015). A number of conservation strategies may be useful for songbirds in Indonesia. Some species, such as the Bali Myna (*Leucopsar rothschildi*), survive almost exclusively in formally protected areas (Jepson 2016) but other species survive in refuges outside of protected areas (Kurniandaru 2008, Yong et al. 2018). Such refuges, which may fall under the umbrella of "other effective area-based conservation measures" (OECMs; Jonas et al. 2014), can include temples and other culturally important sites, small islands, tourist facilities, and privately guarded sites, where work with local communities/authorities underpins the maintenance of socio-ecological conditions that support the survival of key species, intentionally or otherwise (Negi 2010, Li et al. 2014, Dolman et al. 2021).

The Black-winged Myna (*Acridotheres melanopterus*), endemic to the Indonesian islands of Java, Bali, and Madura, now primarily survives in formally protected areas, but was once widespread in the lowlands, predominantly savannas and cultivated areas up to 1,200 m in West Java and reportedly 2,400 m in East Java (Feare and Craig 1998, Collar et al. 2001). It has been present in both domestic and international trade for decades, despite its protection under Indonesian law since 1979 (Minister of Agriculture, Decree no. 757/Kpts/Um/12/1979). It is, however, domestic trade that is largely responsible for the precipitous decline of the wild population, which began in the 1960s but was most pronounced in the 1990s. The sharp decline in numbers traded in the 2000s gave a clear indication that wild populations were vanishing because of trapping (Collar et al. 2001, 2012, Eaton et al. 2015, Shepherd et al. 2016, Nijman et al. 2018).

Although small numbers may persist in recently unsurveyed areas including some nature reserves, the only known wild population of Black-winged Myna left on Java occurs in Baluran National Park, East Java (Winnasis et al. 2020, eBird 2021), while a small number persist at two sites in Bali, with 35 birds at Bali Barat National Park (Brillianti et al. 2019) and 12 at another unspecified site (Eaton et al. 2015). The small number of birds known from two sites near Jakarta, Java (Eaton et al. 2015) are unlikely to persist (T.M.S. personal observation). At Baluran, the population has been extremely low over the past decade: the largest flocks observed in 2009 and 2010 numbered 25 and 12 individuals, respectively (Winnasis et al. 2011, Eaton et al. 2015), although in 2016 a flock of 37 was recorded (BirdLife International 2021). Accordingly, the global population size of wild Black-winged Mynas is considered to be below 100 individuals, probably ~85. This circumstance indicates a clear and urgent need to carry out a thorough ecological assessment of the species to inform its conservation management strategy (Lee et al. 2016). We therefore sought to (1) document its current distribution and estimate its population size within Baluran National Park; (2) use species distribution modeling to identify potentially suitable areas that should be prioritized for appropriate management; and (3) identify the barriers to population expansion in different parts of the park and recommend interventions that can break these down.

METHODS

Study Area

Baluran National Park (BNP; 7°50′S 114°22′E) is situated on the north-eastern tip of Java, with a land area of 264 km^2 (Figure 1). It was first established as Baluran Game Reserve in 1937 by the Dutch colonial government owing to the large mammals found there—banteng (*Bos javanicus*), Javan rusa (*Rusa timorensis*), feral water buffalo (*Bubanus bubanus*), dhole (*Cuon alpinus*), Javan leopard *(Panthera pardus melas*) and the now extirpated Sunda Island tiger (*Panthera tigris sondaica*) (Whitten et al. 1996). The park is in one of the driest parts of Java, receiving <1,500 mm of rainfall a year, most falling between December and February (Winnasis et al. 2011); a pronounced May−October dry season, in combination with fire and herbivory, maintains the savanna-like landscape in the north and east of the park (Pennington et al. 2018). Mount Baluran (1,247 m), a dormant volcano, dominates the center of the park and is cloaked in tropical evergreen and dry deciduous forest.

The residents of Karang Tekok village to the north-west and Wonorejo village to the south-east have always utilized BNP to trap and hunt wildlife and to collect wood, seeds, fodder, honey, and tamarind fruit (*Tamarindus indica*), and they also start fires both accidentally and deliberately (Whitten et al. 1996, Sabarno 2002). Overall, hunting and trapping still represent the greatest threat to wildlife in BNP: the Green Peafowl (*Pavo muticus*), an Endangered species, has declined there, partly because adults and chicks are trapped and sold either alive as pets or dead as food (Winnasis et al. 2011). At least 23 other bird species are known to have been trapped within BNP, mostly alive using mist-nets (Winnasis et al. 2011), and hunters have been prosecuted for poaching East Javan langurs (*Trachypithecus auratus*), a globally Vulnerable species (Nijman 2020). Five guard posts control the main entry points and regular patrols are conducted, but a public road bisects the west side and the park's long coastline offers great ease of access from the sea.

The savanna in the north of BNP has been significantly altered since 1975, when a commercial license was awarded to replace native trees with an agati or turi (*Sesbania grandiflora*) plantation, a pulpwood species used in paper production (Pudyatmoko et al. 2018). The plantation workers have remained in the park despite the license expiring in 2000, and now graze herds of cattle across the savanna and grow crops along the coast (Wianti 2014). More cattle kept by residents of Karang Tekok village on the park's boundary also enter the northern savanna to graze daily (Prijono 2014). In total, ~4,000 cattle and over a 1,000 goats subsist on the northern savanna (Prijono 2014, Pudyatmoko 2017). The park's savanna is also threatened by the spread of thorny acacia (*Vachellia nilotica*), which was planted at Bekol in 1969 to prevent fire from spreading into teak (*Tectona grandis*) plantations (Sutomo et al. 2016).

Classifying the Park's Habitats

We generated a contemporary land-cover map for BNP based on cloud-free LANDSAT 8 imagery (30-m resolution) from October 2018. Some recent burns on the

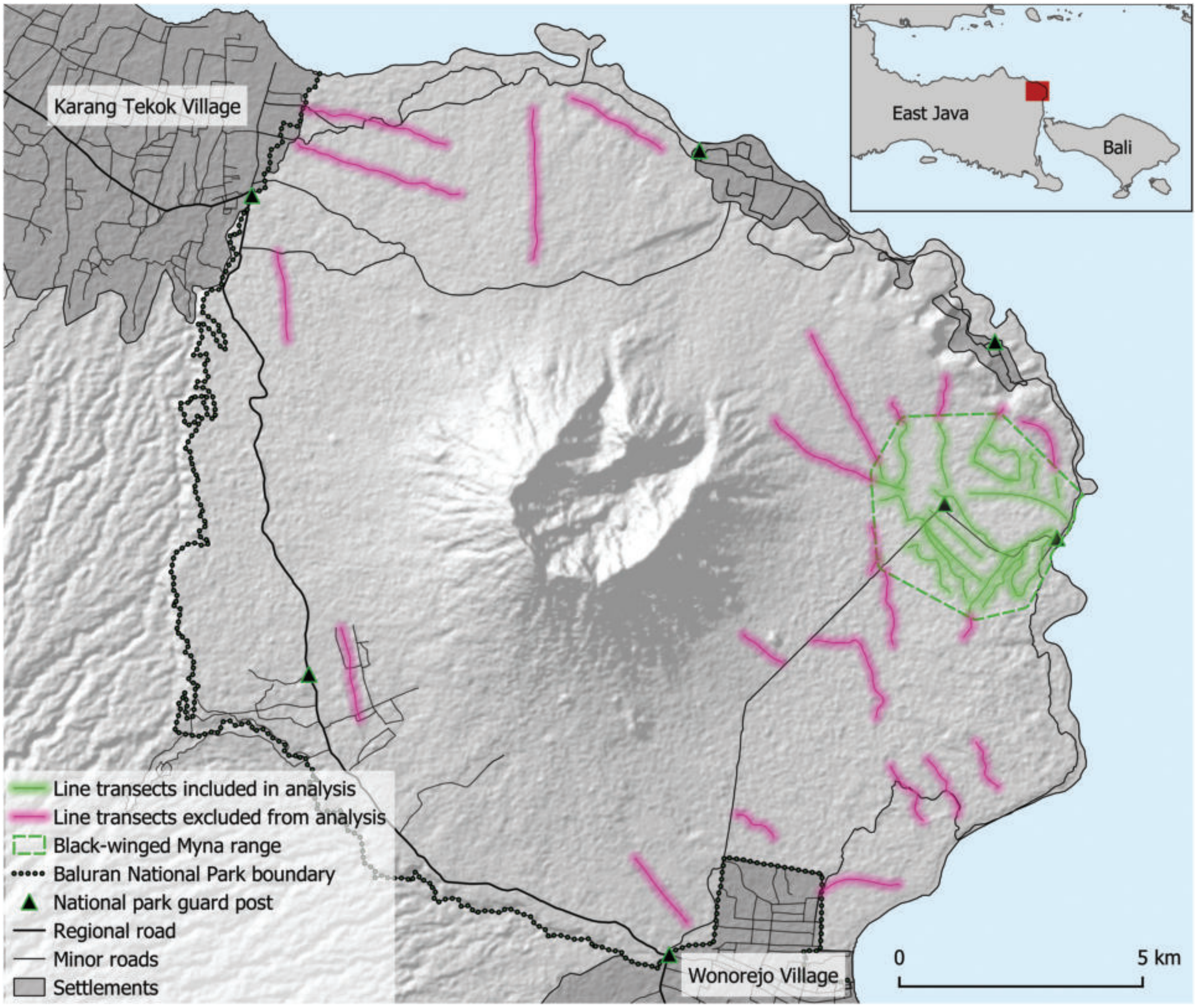

FIGURE 1. Baluran National Park, East Java, showing the location of all line transects, of which those in green were included in the Black-winged Myna population density analysis because they were inside the species' estimated range (green dashed line).

image were removed using neighbor-based interpolation. Training data were obtained using field data collected in September–November 2018 by classifying land-cover types visually with descriptions employed in previous land-cover maps (Supplementary Material Table 1), and by Google Earth image interpretation, which was used to increase the sample size of the smallest classes to address the potential training data imbalance (Millard and Richardson 2015). We selected the following 9 land-cover types for the classification, modified from the latest BNP land-cover map (Baluran National Park 2008) and using relevant descriptions (Supplementary Material Table 1): open savanna, savanna woodland, dry deciduous woodland/shrubland, dry deciduous forest, thorny acacia scrub, teak plantation, tropical evergreen forest, beach forest, and mangrove forest. The difficult terrain on Mount Baluran made field data collection impossible there, but the tropical evergreen forest on its slopes could be delimited from Google Earth images and was therefore included in the training data.

Training data were used to build a random forest (RF) classification model (Breiman 2001). The environmental variables we used were bands 1–7 of the LANDSAT 8 imagery; normalized difference vegetation index (NDVI); the Global Land Analysis and Discovery (GLAD) laboratory's Global Forest Canopy Height 2019 product (Potapov et al. 2021); wetness, brightness and greenness indices; and topography (elevation, slope, and aspect) calculated from the Shuttle Radar Topography Mission (SRTM) digital elevation model with 30-m resolution (van Zyl 2001). Analysis was carried out in R (R Core Team 2020) using packages *raster* v3.3.13 (Hijmans 2020), *randomForest* v.4.6.14 (Breiman 2001), and *sf* v0.9-7 (Pebesma 2018). As a dimension reduction procedure, we ran the RF classification 25 times and recorded the five most important variables for each iteration. A Spearman's rank correlation analysis was then used to measure pair-wise correlations and if any of the 5 most important variables were highly correlated ($r_s > 0.90$) with lower-ranked variables, the latter were removed (Millard and Richardson 2015). The model was tuned by selecting the number of trees to grow and variables sampled at each split that minimized the out-of-bag error rate. The final RF classifier grew 1,000 trees and 2 variables were randomly sampled at each split.

Population Size and Range Estimation

To estimate the current range of the Black-winged Myna in BNP we included all records from line transect sampling and incidental observations during the fieldwork period from March 2018 to March 2019, as well as one observation by a proficient local birdwatcher (Heru Fitriadi) and 19 observations by experienced members of the Copenhagen Zoo project staff at BNP, who carried out a survey in November 2017. We removed 6 outlying records from the current range estimate (see Figure 1) because they were over 4 km from the main cluster of observations, and were never of more than 2 birds which, given the sociality of mynas, suggests they were exploratory movements by dispersing birds. Moreover, we visited the area of these sightings monthly during the fieldwork period and only recorded mynas on two occasions, indicating that the southern area of the park did not form part of the species' home range. We constructed a minimum convex polygon around the remaining occurrences (Burgman and Fox 2003) in R with a 200-m buffer to provide a final range estimate.

Based on the pre-existing land-cover map a total of 36 distance sampling line transects with a combined length of 73.5 km were run across BNP to sample each land-cover type except for tropical rainforest, which we considered unsuitable for Black-winged Mynas (Collar et al. 2001). Line transects were distributed following a stratified sampling approach (Buckland 2004), whereby more transects were located in the east of the park to focus survey effort on the area expected to contain most of the Black-winged Myna population (Winnasis et al. 2011). Within what were classified as open habitat types (open savanna and savanna woodland) line transects were assigned randomly; in woodland (classified as closed habitat), transects followed narrow tracks made by humans or large mammals, as attempts to penetrate the dense understory, especially during the wet season, risked disturbing the target species before detection (Buckland 2001). Line transects outside our estimated Black-winged Myna range were excluded from the distance analysis and subsequent population density estimate because no encounters were recorded on any of these. Most transects were walked in both wet and dry seasons, but eight were only walked in the dry season. Twenty-one transects with a combined length of 29.1 km (mean length: 1.4 km) were included in the final analysis, giving a total effort of 50.6 km (Figure 1).

Distance sampling was undertaken at the end of the wet season and beginning of the dry season from March 28 to May 30, 2018 (hereafter referred to as wet season sampling because in May the vegetation is still lush and the understory dense), and in the dry season from October 5 to November 21, 2018. Sampling was conducted between 0600 and 1200 hr by one of two experienced observers (T.M.S., P.G.A.) following the standard protocol for line-transect distance sampling (Buckland 2001, Buckland et al. 2008). We walked transects at ~1.5 km h^{-1} and each was preceded by a 200-m 'burn in'—the first section of a transect (not included in its reported length) which is walked but encountered birds are not recorded in an attempt to equalize the effects of observer disturbance across the transect (Buckland 2004). Birds seen only in flight were ignored, but those seen taking off or landing (i.e. using the habitat) were recorded. We used laser rangefinders (Hawke LRF 400) to measure distances and targeted the nearest visible object when no clear line of sight was available (Buckland et al. 2008). Distance sampling analysis followed standard methods (Buckland 2001) and was carried out in R (R Core Team 2020) using package *Distance* v1.0.2 (Miller et al. 2019). Detection data were right-truncated at 135 m after discarding the furthest 5% of detections from the transect. Group sizes for aural-only detections were replaced by the average of all groups of known size. We generated a two-level open vs. enclosed habitat covariate for the detection function. The final model for the detection function was selected based on the minimum Akaike Information Criterion (AIC) value and after confirming that the detection function curve fitted the observed data (Supplementary Material Table 2 and Figure 1).

Estimating Habitat Suitability

We used species distribution modeling (SDM) to estimate the habitat suitability for Black-winged Myna within each 30-m pixel across BNP (Araújo and Guisan 2006). Three environmental covariates were used in the models: an NDVI layer generated from LANDSAT 8 imagery; the land-cover classification raster covering the park (Figure 2); and a habitat openness layer, for which each pixel represented the proportion of open vs. closed habitat pixels in the 0.56 km^2 (25 × 25 pixels) surrounding the target pixel. Black-winged Myna occurrences ($n = 339$) were filtered to include just one per pixel, resulting in 291 pixels containing presences across the raster layer.

We used R packages *raster* v3.3-13 and *biomod2* v3.4.6 (Thuiller et al. 2020) to model Black-winged Myna distribution. During the data formatting procedure in *biomod2*, we set the number of pseudo-absences (PAs) to 5,000 and created 5 sets of PAs (*biomod2* generated a total of 24,122 unique PAs) using the "disk" algorithm with the minimum

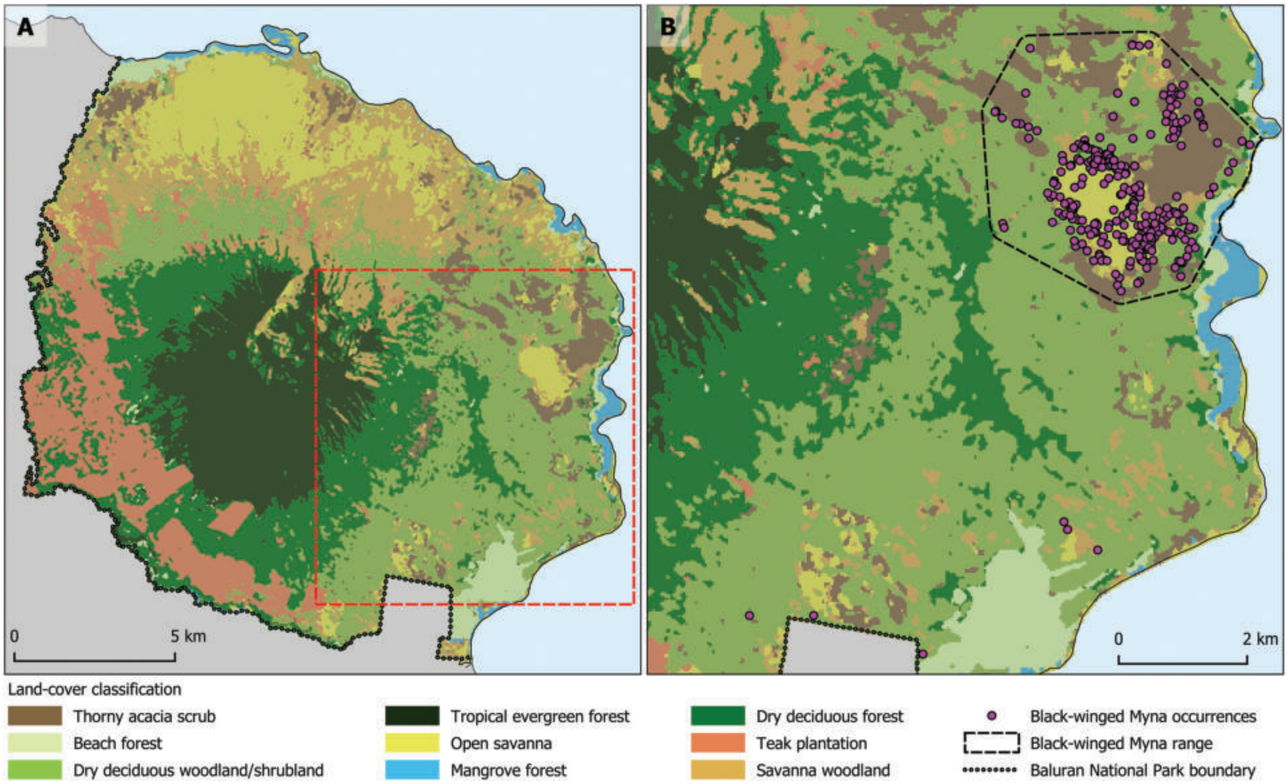

FIGURE 2. (**A**) Land-cover classification for Baluran National Park; and (**B**) all Black-winged Myna occurrences (magenta circles) recorded and the occupied range (black dashed line) overlaid on land-cover types. The extent of (**B**) is shown by the red dashed line in (**A**).

distance to presences set to 75 m (Chefaoui and Lobo 2008, Thuiller et al. 2020). We ran three different SDM algorithms available in *biomod2* on the 5 presence/PAs datasets: generalized linear model (GLM), generalized additive model (GAM), and maximum entropy (MaxEnt). Models were calibrated using 80% of the data and the remainder were used to evaluate model performance. Every algorithm was run 5 times with each PA dataset (total number of runs for each algorithm = 25).

SDM algorithms were evaluated using area under the curve (AUC) values calculated in *biomod2*. All of the algorithms performed well (AUC > 0.80) and were used to produce full models without data partitioning. Variable importance values on a scale from 0 to 1 were calculated for each algorithm, higher values indicating greater influence in the model. We then aggregated the models by algorithm and generated algorithm-specific projections of habitat suitability across BNP to calculate algorithm-specific model averages, which were used to evaluate how well individual algorithms discriminated between presences and PAs based on Tjur's R^2 (Tjur 2009).

For the final output, an ensemble of the projections was generated with the SDM algorithms using mean-weighting based on the AUC value. This raster map provided raw suitability values for each pixel in BNP on a scale of increasing suitability from 0 to 1. We generated a binary raster of unsuitable and suitable habitat using a threshold value, which was the minimum suitability value at a pixel including a Black-winged Myna presence after excluding the lowest 10th percentile of suitability values from pixels with presences. Final maps from the analysis were generated using QGIS v3.10.3.

RESULTS

Black-winged Mynas were recorded a total of 339 times during fieldwork, with all but 6 records in the Bekol, Balanan, and Bama areas in the east of the park (Figures 2 and 4). Based on the occurrence data gathered, we estimate that the current range for Black-winged Myna covers 12.3 km^2 (<5% of the park's area). The land-cover types with most occurrences were open savanna (37.8%), followed by savanna woodland and dry deciduous woodland/shrubland (both 20.6%), thorny acacia scrub (19.8%; but see the discussion), beach forest (0.9%), and dry deciduous forest (0.3%); the full land-cover classification map is shown in Figure 2. Of the outlying occurrences excluded from the range estimation, three were in savanna 4 km south of Bekol, and three were near the cultivated land adjacent to Wonorejo village, 6 km south of

TABLE 1. Estimated Black-winged Myna population density and abundance within its estimated range for each land-cover type in which birds were detected during line-transect distance sampling.

Land-cover type	Area within current range (km^2)	Encounter rate individuals (km^{-1} ± SE)	Density individuals (95% CI)	Abundance individuals (95% CI)
Savanna woodland	0.8	5.8 ± 2.7	34.4 (13.5–88.0)	28 (11–70)
Open savanna	2.1	3.6 ± 1.8	21.0 (7.8–57.0)	44 (16–120)
Dry deciduous woodland/shrubland	5.3	1.9 ± 0.8	10.9 (4.6–25.9)	58 (24–137)
Thorny acacia scrub	4.1	2.0 ± 0.8	12.0 (5.6–25.6)	49 (23–105)
Total/overall	12.3	2.9 ± 0.7	14.3 (8.8–23.1)	179 (111–288)

the current range. The largest groups recorded were seen gathering before dusk, with the highest record being of 97 individuals on 26 September 2018, when birds flew to roost in 10 groups (maximum single group size = 25 individuals).

Black-winged Mynas (median group size = 2) were detected on 56 occasions during line-transect distance sampling over the wet (n = 30) and dry (n = 26) seasons. The average encounter rate was 2.9 ± 0.7 individuals km^{-1}, with the highest encounter rate in savanna woodland followed by open savanna, and the lowest in dry deciduous woodland/shrubland and thorny acacia scrub; birds were not detected in the other land-cover types during line-transect distance sampling (Table 1). Detection probabilities were described best by a uniform key function with one cosine adjustment term (Supplementary Material Table 2 and Figure 1). Population density was estimated to be highest in savanna woodland (34.4 individuals km^{-2}, 95% CI: 13.5–88.0) and lowest in dry deciduous woodland/shrubland (10.9, 95% CI: 4.6–25.9), and the overall population density was 14.3 individuals km^{-2} (95% CI: 8.8–23.1) (Table 1). We estimate the overall population size to be 179 individuals (95% CI: 111–288).

All the SDM algorithms performed well based on AUC values (Table 2). The Tjur's R^2 values for all three algorithms were similar and showed a high level of discrimination between pixels with occurrences and pseudo-absences. Variable importance values for models produced by each algorithm showed that land-cover type had the greatest influence in models produced by all algorithms (Supplementary Material Table 3), and relative differences in variable importance for the models produced by GAM and GLM were similar. The influence of land-cover type and habitat edge was similar in the MaxEnt model, while NDVI had a relatively small influence.

The final ensemble model had a Tjur's R^2 value of 0.47 and discriminated well between areas we expected to be suitable (savanna-type land-covers) and unsuitable (closed-canopy areas) for Black-winged Myna. Large areas of potentially suitable habitat (values of >0.6) are predicted across the north of the park, where open savanna and savanna woodland dominate, while there are smaller areas of suitable habitat surrounding the savanna in the south-east of the park (Figure 3). The central areas of large open savannas are deemed less suitable than the edges. The areas of highest suitability adjoining the current range of Black-winged Myna are distributed to the north and north-west.

We then delimited the potentially suitable area for Black-winged Myna across BNP from the suitability raster using the calculated threshold (Figure 4). This final output indicated that there are an estimated 72.1 km^2 of potentially suitable habitat for Black-winged Mynas, mainly to the north-west of the current range and mostly within open savanna and savanna woodland. Of the potentially suitable habitat, 89% lies within 5 km of the coast at elevations below 300 m. Much of the potentially suitable area is close to the roads and settlements that occur in the north of BNP.

DISCUSSION

The once widespread but now Endangered Black-winged Myna has been extirpated from localities throughout its range on Java and Bali after decades of overexploitation for the cage-bird trade (Eaton et al. 2015, Shepherd et al. 2016, Nijman et al. 2018). This first comprehensive assessment of its status at Baluran National Park, the home of the only known wild population on Java, reveals that fewer than 200 individuals are confined to 12.3 km^2 of savanna and shrubland in the east of the park, an area six times smaller than the 72.1 km^2 of potentially suitable habitat identified by our species distribution model. While some of the potentially suitable habitat is located on Mount Baluran at higher elevations than the species currently occurs at in the park, it is within the species' historically reported altitudinal range of 0–2,400 m (Collar et al. 2001). Although formal population assessments have not been published, a comparison of our population estimate with the maximum flock size of 37 birds recorded in 2016 (BirdLife International 2021) suggests that there may have been recent population growth. Our model delineates areas where management for Black-winged Mynas should be prioritized, and if conditions could be improved across the entire potentially suitable habitat, with a population density matching the average estimated for its current range, BNP might be able to support a population in excess of 1,000

TABLE 2. Calculated AUC, sensitivity, specificity, and Tjur's R^2 values of SDM algorithms used to estimate Black-winged Myna habitat suitability. Values of AUC, sensitivity, and specificity are averages ± SD across the five different datasets included, each comprising Black-winged Myna presences (n = 291) and 5,000 pseudo-absences. Tjur's R^2 values were calculated from model averages for each SDM algorithm after projecting the models across the raster surface for the study area. The highest values for each metric are shown in bold.

SDM algorithm	AUC	Sensitivity	Specificity	Tjur's R^2
GLM	**0.88 ± 0.02**	86.9 ± 6.43	**76.3 ± 6.95**	0.46
GAM	0.87 ± 0.01	**89.3 ± 5.13**	72.4 ± 6.58	**0.48**
MaxEnt	0.87 ± 0.02	87.5 ± 5.27	73.4 ± 6.75	0.46

Black-winged Mynas. However, there remain at least three significant barriers to such population recovery—trapping, overgrazing, and invasive thorny acacia.

There are several difficulties associated with building distributional models for species with small remnant populations that were once widespread and have declined due to multiple and concurrent factors. These difficulties may help to explain why the species is currently not occupying habitat identified by our model as suitable. First, it is possible that part of the population occupies suboptimal habitat constituting a demographic "sink", where mortality exceeds productivity (Howe et al. 1991), in which case our model may have considered suboptimal habitat suitable for the species. Second, characterizing trapping pressure is difficult and direct indices are seldom available (but see Biddle et al. 2021). Metrics of remoteness (distance from roads or human settlements) may explain, to some degree, such anthropogenic pressures at a large scale (Benítez-López et al. 2017, Shaney et al. 2017, Symes et al. 2018), but fail to account for spatially discrete

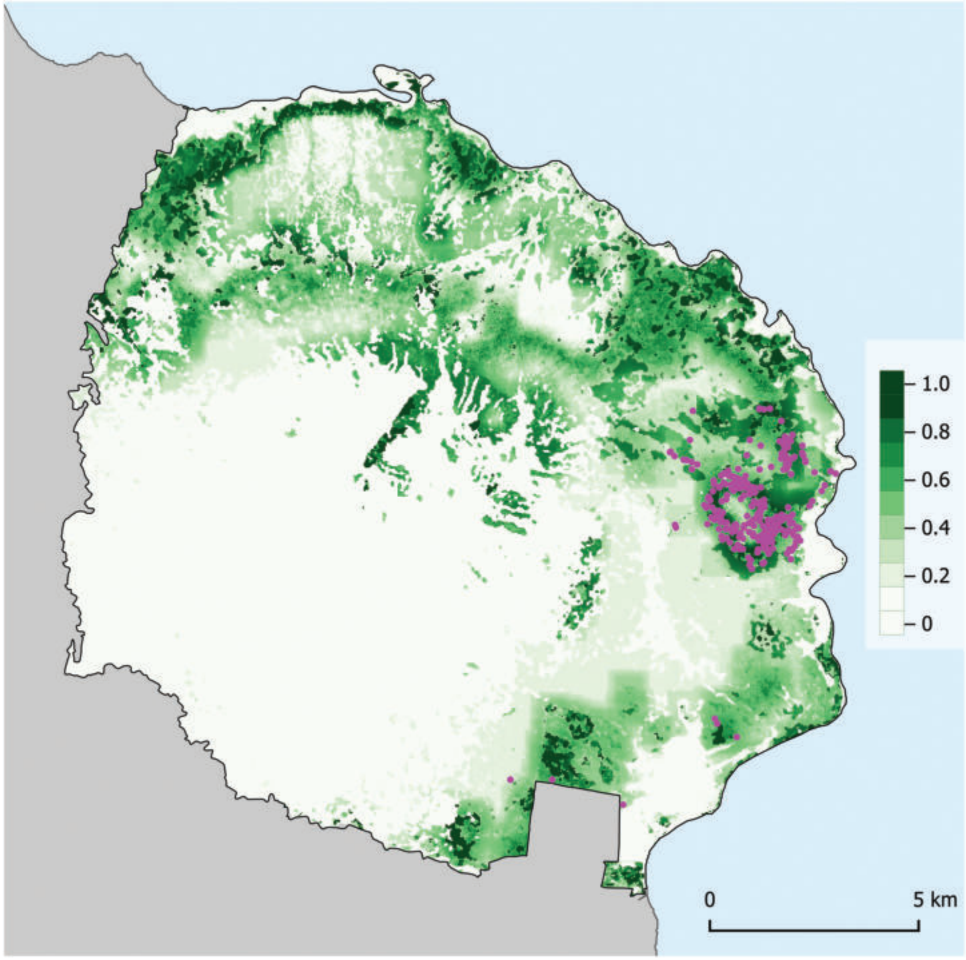

FIGURE 3. Baluran National Park habitat suitability map for Black-winged Myna, with values on a scale of increasing suitability from 0 to 1. Magenta circles show the actual Black-winged Myna occurrences that were used in species distribution modeling.

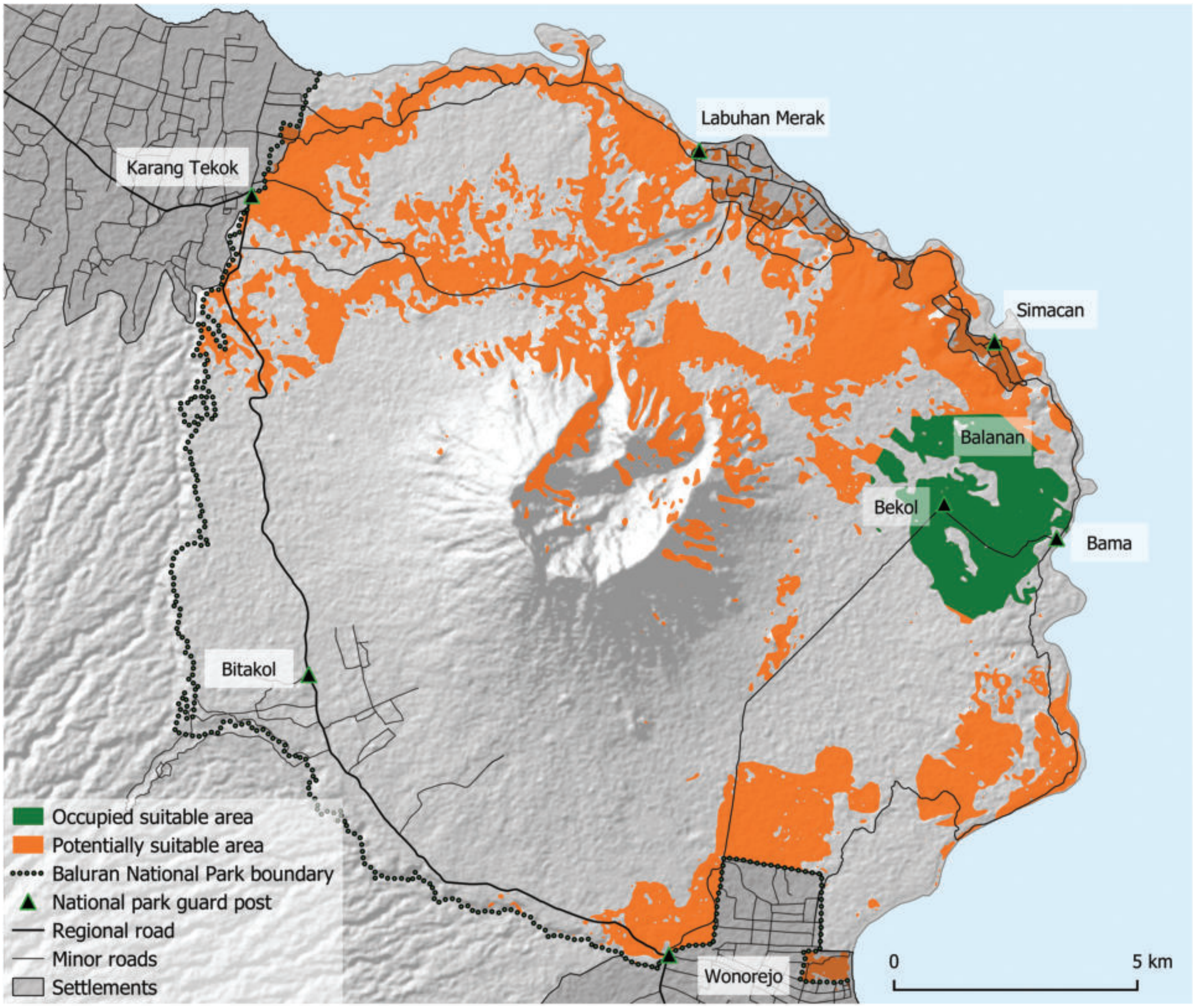

FIGURE 4. Baluran National Park, showing predicted suitable habitat for Black-winged Mynas that is currently occupied and unoccupied.

forms of anthropogenic protection, such as guarding, that benefit some species (Demerdzhiev et al. 2014); tourism, as on Komodo (Reuleaux et al. 2020); community-based conservation (Watson et al. 2007); and special land status such as sacred groves (Plieninger et al. 2020). The mynas at BNP appear to benefit from a combination of guarding and tourism (or even research) activities, for which spatial data reflecting the complexity of the situation were not available. Third, the myna population in BNP is clumped, so projecting this distribution onto other parts of the—albeit relatively small—park needs to be done with care, as habitat and other relationships may not hold in other areas. This caveat extends to other protected areas in East Java (e.g., Meru Betiri and Alas Purwo National Parks), and especially to non-protected areas, where habitat associations and trapping pressures may be different. In fact, the lack of studies when the species was still common means that its true habitat preferences remain uncertain, but they may have included a wider range of habitats than those present in BNP, although probably not closed-canopy woodland. Nevertheless, the model built for Baluran is a starting point for use in other protected areas in East Java, and the habitat associations resemble those of the nearby remnant populations of Black-winged Myna around Bali Barat National Park (T.M.S. personal observation).

The most likely constraint on the size of the myna population in BNP is trapping for the domestic pet trade. Black-winged Mynas have been heavily trapped and traded over 4 or 5 decades for cage-bird markets across Java (Collar et al. 2001, Shepherd et al. 2016, Nijman et al. 2018); such is their value that 7 years ago 151 individuals were stolen in a highly organized raid on a well-guarded conservation breeding center (Tritto and Sözer 2014). Recently, mynas explicitly identified as trapped in Baluran have been openly advertised for sale online (Bruslund et al. 2021). Evidence from our study suggests that the species persists only in the small area where we found them because birds there are afforded protection against trapping. This likely comes from three sources: the heavy park staff presence at Bekol, which houses an office, conservation breeding enclosure and guard post; tourist activity, which is centered around Bekol and Bama and which saw visitors rise from 39,874 in 2013 to 245,901 in 2020 (Padmanaba et al. 2017; A.P. personal observation); and the presence of park staff, contract workers and the Copenhagen Zoo project team performing savanna restoration at Balanan. Elsewhere in the park trappers may be less constrained, owing to the much less frequent presence of guards and tourists.

A second constraint on Black-winged Mynas at BNP is savanna degradation and disturbance resulting from the 5,000 domestic livestock grazing and browsing some 56 km^2 of the park's northern savannas (Pudyatmoko 2017), representing 21% of the park's total area. Just over half of the 4,000 cattle and all of the goats are kept by inhabitants of the settlements along BNP's northern coast from Labuhan Merak to Simacan, with the remainder kept by residents of Karang Tekok village (Pudyatmoko 2017). Most of the livestock belong to a few members of the local elite and are loaned to keepers under a *gaduh* system, whereby owners take the profit from cattle sold for slaughter and keepers retain calves born on pasture, with a ten-fold difference in annual profit in favor of owners (Wianti 2014, Pudyatmoko et al. 2018). Low-intensity grazing can be beneficial for many starlings and mynas because it maintains a low sward height (Fuller et al. 2003), providing birds with access to surface and topsoil invertebrates (Heldbjerg et al. 2016, van Balen and Collar 2021), while manure increases food availability by promoting plant and invertebrate abundance and diversity (McNaughton 1985, Steinauer and Collins 1995, Plantureux et al. 2005). Indeed, like other *Acridotheres*, Black-winged Mynas commonly associate with large herbivores, feeding on invertebrates from the ground they disturb (Collar et al. 2001). However, the current high intensity of grazing by domestic livestock in the northern savannas of BNP probably greatly exceeds former natural levels based on the maximum recent population sizes for the entire park of water buffalo (1,293 in 1984; Suhadi 2009) and banteng (267 in 2000; Winnasis et al. 2011), and the graziers who accompany livestock are a source of disturbance to wildlife (Pudyatmoko 2017). The resulting soil impoverishment (Dormaar and Willms 1998, Villamil et al. 2001) and atrophied biodiversity (Olff and Ritchie 1998, Dhaou et al. 2010) associated with intensive grazing seriously reduces the habitat suitability for Black-winged Myna and other savanna-dependent wildlife.

That the recent clearance of thorny acacia within the Bekol area has coincided with an apparent population upturn of Black-winged Mynas (our population estimate of ~179 individuals is higher than the previous global population estimate of fewer than 100) suggests that thorny acacia is a poor habitat for the species, despite the relatively high usage registered in our results, caused by birds perching in acacias fringing open foraging areas. Thorny acacia spreads rapidly, replacing savanna with impenetrable thickets and reducing the food available to savanna-dependent wildlife (Kriticos et al. 1999, Dhileepan 2009, Zahra et al. 2020). Dense stands had engulfed the savanna at Bekol, Balanan, and Kramat by 1993, covering an estimated 12 km^2 (Schuurmans 1993 in Setiabudi et al. 2013), and despite restoration attempts since 1985, with some of Bekol savanna successfully cleared (Zahra et al. 2020), by 2014 the overall acacia cover had increased (Sutomo et al. 2020). Since 2016, thorny acacia clearance at Balanan has restored 3.6 km^2 of grassland (Copenhagen Zoo 2021), which has been used by Black-winged Mynas, including for breeding (T.M.S. personal observation). Our habitat classification indicated that there are at least another 12 km^2 of thorny acacia within BNP in monospecific stands, most of which are close to the current Black-winged Myna range and Simacan settlement (Figure 4); efforts to clear this are therefore a priority and could double the habitat immediately available to the species.

If birds (re)colonize this area, guard patrols must of course follow. Such protection could be supported by the mapping of nesting and particularly roosting areas, which are probably where most trapping is done; well-protected nest-boxes could be deployed to encourage dispersing birds, as practiced in early reintroduction attempts (Tritto 2014), particularly as the myna's natural cavity nest sites may have been reduced in number by the removal of dead trees for fuelwood near settlements (T.M.S. personal observation). Patrolling also needs to target beaches, land access points, and the northern settlements where mynas and Baluran's other key species are suspected of being smuggled out of the park (Winnasis et al. 2011).

A crucial underpinning of any conservation management of Black-winged Mynas will be engagement with the park's human communities, especially those at Simacan and along BNP's northern coast. Such work has been instrumental in protecting species facing similar anthropogenic pressures elsewhere: the Philippine Cockatoo (*Cacatua haematuropygia*) was trapped almost to extinction until a conservation program trained and employed ex-trappers as wardens and involved local communities in wildlife monitoring (Widmann et al. 2006). Communities living inside BNP could likewise be recruited to support a nest box and monitoring scheme for Baluran's mynas in a type of payment for ecosystem services arrangement (Ferraro 2011). In 2016 BNP established a 20 km^2 "special use zone" in a bid to settle a long-standing dispute over the communities' land rights (Mulyana et al. 2010, Wianti

2014, Pudyatmoko et al. 2018), which suggests that local goodwill might allow such a project to be implemented. Nevertheless, further negotiations are essential to achieve a significant but equitable reduction in grazing pressure in the 56 km^2 of highly degraded northern savanna, for example, by keeping cattle in enclosures and providing alternative livelihoods (Pudyatmoko et al. 2018).

Protected areas are understandably the first option considered when seeking to preserve species, because they provide a pre-existing legal, geographical, organizational, and social framework for the endeavor, and often also because they are the last places where the species of concern survive. With the western form of Black-winged Myna, nominate *melanopterus*, apparently extinct in the wild and the Bali form *tertius* not known to number more than 35 inside Bali Barat National Park, Baluran National Park represents by far the most important opportunity to save the Black-winged Myna from the trapping pressure that is driving it to extinction. The cases of the Javan Pied Starling (*Gracupica jalla*), now almost certainly extinct in the wild (van Balen and Collar 2021), and the Bali Myna, once thought extinct in the wild and now surviving only through intensely managed reintroductions (Jepson 2016), serve as examples of what the near future could hold for the Black-winged Myna without effective action.

At Baluran, there is scope to increase the Black-winged Myna population by habitat restoration and enhancement, elevated protective vigilance, and strong community engagement, building on established models trialed elsewhere in the world. Nevertheless, open-country starlings and mynas tend to make medium-distance movements to forage and explore (Bruun and Smith 2003, Minderman et al. 2010, Astudillo et al. 2019), so protected areas cannot be expected to harbor them indefinitely. In the longer term, therefore, such species, especially if under pressure from trapping, will have to be conserved by management strategies that embrace adaptation and improvisation, taking advantage of the various types of security provided by mining or geothermal operations (Randriamamonjy et al. 2015, Devenish et al. 2021), religious sites (Colding and Folke 1997), tourist resorts (Moritz et al. 2017) and organic farming, all of which may to some extent be leveraged to create appropriate socio-ecological conditions to allow the birds to survive within working landscapes. Other interventions proposed for exploited wider-ranging species include demand reduction (Burivalova et al. 2017, Marshall et al. 2020), commercial breeding (Jepson et al. 2011), and better enforcement of trade laws (Nijman 2010), and all of these measures could be applied in the case of the Black-winged Myna, whose recovery in Baluran could also be abetted by supplementations of captive-bred birds. What can be achieved at Baluran in the next decade may therefore point the way for many species recoveries, not just of threatened songbirds and not just in protected areas, across Indonesia and indeed the world.

SUPPLEMENTARY MATERIAL

Supplementary material is available at *Ornithological Applications* online.

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Conflict of interest: T.M.S, N.J.C., C.D., A.O., N.L.W., and S.J.M. declare that they have no conflicts of interest. A.P. declares that she is employed by Baluran National Park (BNP).

Data deposits: Analyses reported in this article can be reproduced partly using the data provided by Squires et al. (2022). The full dataset can be shared by the authors upon reasonable request from bona fide researchers. The full dataset used in this paper is not available online due to the threatened status of the study species.

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