Tropical birds are declining in the Hainan Island of China☆

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ABSTRACT

Understanding biodiversity trends and the factors that influence those trends is crucial for effectively reducing global biodiversity loss. The biodiversity trends in tropical areas are largely unclear. In addition, habitat loss and fragmentation and illegal wildlife use threaten biodiversity in these regions. Herein, we report abundance and species richness trends for birds in the tropical Hainan Province (Island), China, based on extensive transect surveys conducted in 10 km × 10 km grid cells across the majority of the island between 1997–1998 and 2012–2013. We also quantified the effects of changes associated with land use, natural forests, agriculture, human populations, protected areas and wildlife-rearing farms on the abundance of birds. We found that 145 bird species suffered significant declines in abundance and species richness between two time periods of the survey. 28 species exhibited a decline in abundance, while 33 species showed an increase. Other species showed no significant changes in abundance. More common bird species declined more rapidly than less abundant species. The abundance trend in a grid cell (log ratio: 2012–2013/1997–1998) increased with the proportion of protected area but decreased with a reduction in natural forests and an increase in the number of wildlife-rearing farms. These results suggest that the avian decline on Hainan Island is mainly due to deforestation and illegal wildlife use. To slow this decline, it is necessary to increase protected area coverage and networks on the island, strengthen natural forest protection, and reinforce regulations to reduce habitat destruction and illegal wildlife use by wildlife-rearing farms.

1. Introduction

Despite increased conservation efforts in recent decades, global biodiversity is still experiencing a rapid decline (Hoffmann et al., 2010; Newbold et al., 2015; Pimm et al., 2014). Understanding the areas where local biodiversity is declining and the factors that contribute to such declines is crucial for effectively reducing global biodiversity loss. Tropical regions harbour the richest biodiversity; however, these regions are experiencing rapid environmental changes, such as declining forest cover, increasing habitat loss and hunting, and increasing air and water pollution due to increased human population densities (Cheng, 1993; Pimm and Raven, 2000; Lee et al., 2004; Bradshaw et al., 2009; Abernethy et al., 2013; Corlett, 2007). However, the current level of global biodiversity monitoring is less extensive in tropical areas than in temperate regions (Collen et al., 2008; Collen et al., 2009; Bradshaw et al., 2009; Abernethy et al., 2013; Ahumada et al.,

2013). Studies based on limited data show that populations of terrestrial vertebrates have declined at a more rapid rate in tropical areas than in other regions since 1970 (Collen et al., 2008; Collen et al., 2009; Whitfield et al., 2007). However, relatively few studies have quantified the effects of environmental changes on vertebrate trends in tropical areas (Ahumada et al., 2011, 2013; Craigie et al., 2010).

Hainan Island is a tropical province that located in China's southern tropical fringe. The province supports a rich biodiversity and the local population has a long history of using wildlife for food and Traditional Chinese Medicine (TCM) (Lau et al., 2010; Liang et al., 2011; Wan et al., 2015; Zhou et al., 2005). Over the past 20 years, both fast economic growth and rapid expansion of nature reserves have occurred in the province (Wang et al., 2013; Wu et al., 2011). The construction of factories, residential homes, roads and other infrastructure has expanded into farmlands and natural habitats. As a result, the tropical natural forest coverage rate has decreased at an alarming rate (Zhang et al., 2010). Large portions of these deforested natural areas have been replaced with plantation forests of rubber and eucalyptus forests for the papermaking industry (Barr and Cossalter, 2004; Zhai et al., 2014; Zhai et al., 2012). This fast economic growth is also facilitating wildlife consumption. For example, there were no wildlife-rearing farms in Hainan before 1988; however, Hainan contained at least 14 registered

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wildlife-rearing farms by 1998 and at least 500 farms by 2012. These farms provide various types of terrestrial wildlife for markets, including wild birds, mammals, snakes, turtles, lizards, frogs and salamanders (Gong et al., 2006; Lau et al., 2010; Lee et al., 2004). To mitigate the threats caused by converting, degrading and fragmenting natural habitats and farmland, in addition to harvesting, local governments have made great efforts to preserve land for nature reserves. For example, the number (and area) of terrestrial nature reserves increased from 22 (approximately 800 km^2, covering 2.7% of the land area of the island) in 1988 to 32 (2400 km^2, covering 7.4% of the land area) in 2014 (Ministry of Environment Protection of the People's Republic of China, http://sts.mep.gov.cn/zrbhq/). Given the threats faced by biodiversity in Hainan and in spite of encouraging developments in conservation, there is a need for better understanding of Hainan's biodiversity trends. In addition, the roles of habitat change, human population growth, wildlife-rearing farms and protected area expansion should be assessed in the context of affecting biodiversity trends.

In this study, we report species richness and bird abundance trends and identify the factors that affect these trends on Hainan Island. Birds are among the best known groups of major organisms (BirdLife International, 2004). Extensively studied and robust, long-term and large-scale datasets are available (Inger et al., 2015). Moreover, the conservation status of all birds on Hainan Island has been assessed according to the IUCN Red List Categories and Criteria (IUCN, 2015). Among the 437 bird species on the island (unpublished data, Jiang), 1.8% are categorized as critically endangered (e.g., Spoon-billed Sandpiper (*Eurynorhynchus pygmeus*, 0.7% as endangered (e.g., White-eared Night-heron (*Gorsachius magnificus*), Black-faced Spoonbill (*Platalea minor*) and Spotted Greenshank (*Tringa guttifer*)) and 1.6% as vulnerable (e.g., Hainan Partridge (*Arborophila ardens*), Hainan Leaf-warbler (*Phylloscopus hainanus*) and Pale-capped Pigeon (*Columba punicea*)). Additionally, birds provide important goods and services to humankind and play irreplaceable roles in ecosystems (Newbold et al., 2012; Sekercioglu et al., 2004). Bird population declines have important ecological and economic consequences, including seed dispersal and pollination disruption in some plants, carrion consumption reductions, loss of natural pest control (pest insects and rodents), loss of socioeconomic resources and other unpredictable consequences (Sekercioglu et al., 2004; Gangoso et al., 2013; Wenny et al., 2011; Whelan et al., 2008). We conducted extensive transect surveys on the abundance and species richness of birds covering most of Hainan Island in the periods of 1997–1998 and 2012–2013, providing a good opportunity to assess the abundance and species richness trends of birds on the island.

To understand the impact of development on bird populations, we first examined the abundance and species richness trends of birds in two periods. Our large scale data that covers a large proportion of Hainan's area allow us to compare and assess the changes in abundance and richness of bird over time. We compared the proportion of species that have declined between the various abundance groups and feeding guilds as well as between birds with different migratory behaviours. Then, we quantified the relative effects of habitat change, human population growth and expanding nature reserves on bird abundance trends during the two periods.

2. Materials and methods

2.1. Study area

Hainan Island (18°10′–20°10′N and 108°37′–111°03′E, Fig. 1) is located in the southernmost terrestrial portion of China, encompassing an area of approximately 33,900 km^2. It is the largest tropical island in China and is located within the Indo-Burma biodiversity hotspot (Francisco-Ortega et al., 2010; Myers et al., 2000). The island includes 18 counties or cities. It was separated from the mainland 65 million years ago due to terrane drifting and rotation (Liang, 2013). The island
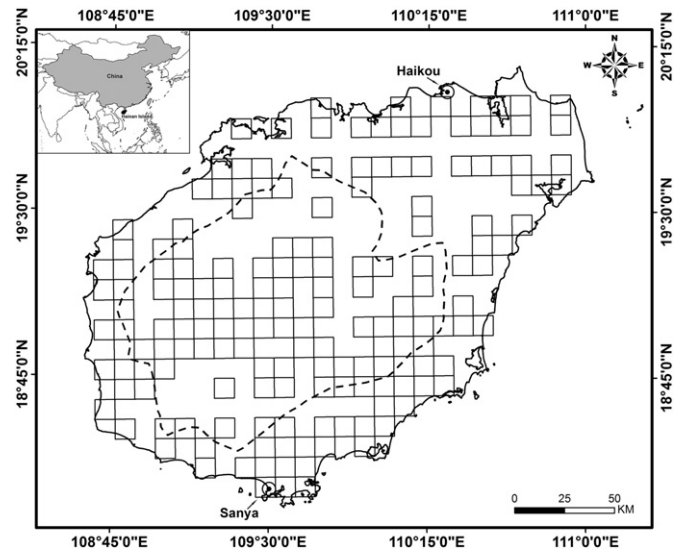

Fig. 1. Hainan Island and 172 sample grids (10 km × 10 km) used in analysis. Dotted line indicate boundary between two terrestrial ecoregions (inner one: Hainan Island monsoon rain forests; outer one: South China-Vietnam subtropical evergreen forests).

has a complex topography, being high and rugged in the central mountains (38.7% of total area) and low and flat in the surrounding areas (61.3%). The highest peak on the island has an elevation of 1867 m. The climate is typical of the tropical ocean monsoon zone and is highly seasonal, with a dry season extending from November to April and a rainy season from May to October. The average annual temperature ranges from 16 °C in January to 26–29 °C in July. The annual precipitation is more than 1600 mm (Francisco-Ortega et al., 2010). Rainfall mainly comes from typhoons in the Pacific Ocean during the rainy season. The rainfall is distributed unevenly across the island. The western part of the island is characterized as dry, while the eastern part is humid. The natural vegetation in the area is dominated by evergreen seasonal rainforest and deciduous seasonal rainforest. The island is biogeographically divided into two ecoregions: the Hainan Island Monsoon Rain Forest Ecoregion (HMRF) in the middle, mountainous area and the South China-Vietnam Subtropical Evergreen Forests Ecoregion (SCSEF) on the coastal plains (Olson et al., 2001) (Fig. 1). At least 430 species have been recorded on the island, accounting for 32% of the total bird species in China.

2.2. Bird abundance and richness

We sampled bird abundance and richness using the line transect method. The method was originally designed at a 5 km × 5 km grid cell resolution across the entire island (1372 grid cells, excluding grid cells with land areas of <50% located offshore) for periods of 1997–1998 and 2012–2013. We adopted the 5 km × 5 km grid cell resolution for the field survey according to protocols of the first (or second) State Terrestrial Vertebrates Survey (State Forestry Bureau Survey Designing Institute, 1995; 2011). During the 1997–1998 survey period, we established line transects at different longitudinal intervals (Fig. A1). Each line transect was approximately 5 km long and 50 m wide, with 25 m on each side of the transect (Lemoine et al., 2007). Two nearby line transects at a similar longitude were established at intervals of 15 km (between the midpoints of the two transects). Line transects at a similar latitude were established at intervals of 5–10 km (some grid cells were not surveyed due to inaccessibility). The transect density on the southern part of the island was higher than that on the northern part of the island due to more heterogeneous topography in the south. During the 2012–2013 period, we increased our sampling efforts using a short line transect that was 3 km long and 50 m wide. We divided the entire island into two zones: the coastal zone and inland zone

(Fig. A2). In the inland zone, we sampled large numbers of parallel 5 km × 5 km grid cells along different longitudinal intervals. Two nearby grid cells at similar longitudes were established at intervals of 10–15 km (between the midpoints of the two squares). Grid cells at similar latitudes were established at intervals of 10–20 km. Within each grid cell, we fixed three parallel line transects. The middle transect passed through the centre of the square and two nearby transects were separated by an interval of 1 km. The starting points for all three transects within a grid cell were set at random along a given longitude. In the coastal zone, we established a large number of parallel line transects (3 km) at different longitudinal intervals, as in the previous period (Fig. A2). Additionally, we randomly established numerous line transects in accessible areas of nature reserves (with the area of transects covering >2% of the total area of the reserve). Two nearby transects within a reserve were separated by intervals of 500 m or more (Fig. A2).

In both periods, each transect was surveyed by a team consisting of two people. To ensure the quality and consistency of the survey methods, all participants were ornithologists or bird experts who are familiar with the avian species in Southern China. Most of these experts (at least one member of a team) participated in both survey periods. All investigators were trained before the field surveys to ensure consistency. The training emphasized the survey methods, including species identification, distance estimates and recording methods. Surveys were mainly conducted during the breeding season (March–October). Each line transect was surveyed once. Line transects were visited in the early morning between 0630 and 1030 and traversed at a speed of approximately 1 km per hour. Following the methods of Buckland et al. (2005), we visually estimated the perpendicular distance from each detection to the midline of a line transect (within approximately 1 m). Information recorded along each transect included the location of each detection, species name and number of birds. Bad weather was avoided during the survey periods. For grids within each county or city, the dates when the surveys were conducted were as close as possible between the two periods (i.e., the differences in the exact dates between the two periods were usually less than 15 days).

Although we attempted to consistently conduct surveys during the two periods, we acknowledge that line transects were only surveyed once in each period and only in two periods. Additionally, increased survey effort in 2012–2013 led to the detection of many understory and ground-dwelling birds. These issues are shortcomings of this study. However, increased survey efforts were conducted during each period due to the length and abundance of line transects, which encompassed different habitats over most of the island. Thus, these surveys reliably reflect the overall bird trends on the island.

Making direct comparisons based on the bird abundance and richness trends of the two periods would be difficult due to variable line transect characteristics. Moreover, it is likely false absence of rare species may be large at a grid cell resolution of 5 km × 5 km. Therefore, we made comparisons between the two periods based on the same length of a resampled transect segment (0.5 km in length) at 10 km × 10 km grid cells. This method ensured that survey efforts during the two periods were comparable and reduced the bias from the false absence (low detectability) of rare species at the finer resolution (Fig. 1). The segment with the same length was resampled to standardise the survey efforts during both periods (Burnham et al., 1980). Within a grid cell, each transect was divided into several segments. We counted the abundance of a species recorded within a segment, and calculated pooled abundance and species richness of all birds recorded for the segment. Segments of less than 0.5 km were excluded from the calculations. The number of segments ranged from 2 to 240 per grid (mean = 23.5, SD = 26.6) in the 1997–1998 period and from 2 to 226 per grid (mean = 44.7, SD = 36.0) in the 2012–2013 period. We calculated the average abundance (or richness) per segment by averaging the pooled abundance (or species richness) of birds in each segment across all grid cells sampled. Grids with fewer than two segments in any

period were excluded from further analysis due to low sampling efforts (Fig. 1).

We determined the trends in pooled abundance and species richness in a grid cell using the Living Planet index (LPI), which has been widely used to reflect changing trends of terrestrial vertebrates (Collen et al., 2009). We estimated the pooled abundance and species richness trends according to the following formula:

$d = \log_{10}(N_t/N_{t-1})$ where N is the pooled abundance per segment within a grid and t is the period (Collen et al., 2009). We calculated d by only considering species that were observed in both periods. Furthermore, we removed nocturnal species from our analysis because the morning survey time may underestimate the abundance of these birds. We also excluded irregularly occurring birds, such as dispersant and vagrant species, from the analysis because their associated trends would provide little ecological value.

2.3. Factors affecting avian abundance trends

Based on previous studies (McKinney, 2001; Newbold et al., 2012; Wang et al., 2013), we evaluated the effects of the following variables on bird abundance trends.

2.3.1. Land use
Land use change is one of the most crucial anthropogenic pressures affecting bird abundance (Jetz et al., 2007; Newbold et al., 2012; Sala et al., 2000). We estimated the effects of natural forest, cropland and human infrastructure (roads, residential areas and factories) on the bird abundance in 2012–2013 and between the two periods. The variations in natural forest, cropland and human infrastructure areas were calculated as the proportions of each area per grid in 1997–1998 minus those proportions in 2012–2013. Because natural forest area data are collected at 5-year intervals by the State Forestry Administration, we used data from 1995 to approximate the 1997–1998 period and data from 2010 to approximate the 2012–2013 period. The natural forest data were obtained from the maps of the National Forest Resources Inventory of China (http://www.forestry.gov.cn/gjslzyqc.html). The resolution of the 1995 natural forest data was lower than that of the 2010 data (small patches of natural forest could not be identified in 1995); therefore, we digitized the maps and re-edited the 1995 data to include patches of natural forests identified in 2010 to maintain consistency. The croplands (Newbold et al., 2015) and infrastructure (Newbold et al., 2015; Rytwinski and Fahrig, 2011) data were obtained from the Data Centre for Resource and Environmental Sciences of the Chinese Academy of Science (RESDC) (http://www.resdc.cn) at a resolution of 1 km^2.

2.3.2. Human population density and annual growth rate
Increasing the human population density increases the risk of avian extinction (Kerr and Currie, 1995; McKinney, 2001). Human density data from 1995 and 2010 were collected from RESDC (http://www.resdc.cn) (at the resolution of 1 km × 1 km grid cells). The average human density per grid cell (10 km × 10 km) was calculated as the average human density across 1 km × 1 km grid cells. The annual growth rate of the human population in a grid cell (10 km × 10 km) was calculated according to the following formula (Caughley, 1997):

$$Gr = \log\left(\sqrt[15]{\frac{d_{2010}}{d_{1995}}} - 1 \right)$$

where Gr is the annual growth rate of the human population density, and d_{1995} and d_{2010} are the human population densities in 1995 and 2010, respectively.

2.3.3. Wildlife-rearing farms

During the last two decades, the number of wildlife-rearing farms has increased rapidly in Hainan. These farms have become an important industry in the province, providing live wildlife to meet the demand throughout China. These wildlife-rearing farms can occupy large areas of wetlands or croplands. Some of the animals on these farms are likely captured from the wild. This capture process may affect wild bird populations. According to the regulations in China, a wildlife-rearing farm requires a permit from the local forestry administration when it begins operation (Law of the People's Republic of China on the Protection of Wildlife, 1989). We obtained information regarding the registered wildlife-rearing farms from the local forestry administration in 2012. Then, we visited each farm to confirm that it still was running and determined its location using Global Positioning System (GPS). We also counted the number of wildlife-rearing farms that were operating in each grid cell.

2.3.4. Areas of nature reserves and the time since reserve establishment

Nature reserves are the most important biodiversity conservation tools in Hainan (Wang et al., 2013; Wu et al., 2011). The area of a nature reserve and the time since it was established may affect the bird abundance. In this study, we included different types of nature reserves, including terrestrial reserves, inland wetland reserves and coastal mangrove reserves. Information about the reserves, including their dates of establishment, areas and locations, was collected from different sources in 2012 (Ministry of Environmental Protection of the People's Republic of China, http://sts.mep.gov.cn/zrbhq/; Forestry Department of Hainan Province). The geographical distribution of the reserves was digitized at a resolution of 1:50,000. We calculated the areas of the reserves using ArcGIS v10.2. The time since a reserve was established was calculated by subtracting the year of establishment from 2012. When a grid cell (10 km × 10 km) contained two or more reserves, the longest establishment time was used.

2.3.5. Geographic influence

Topographic heterogeneity can affect bird species richness (Davies et al., 2007; Jetz and Rahbek, 2002). We used the topographic heterogeneity index to represent topographic heterogeneity. The index was measured based on the altitudinal range. We calculated the average range of elevation values found within each grid cell (10 km × 10 km) based on the WorldClim 1 km DEM.

2.4. Statistical analyses

The variables above are likely to be useful predictors of bird abundance changes. To improve linearity, 10 variables were $\log_{10}(x + 1)$ transformed. These variables include changes in the proportions of natural forests, crop fields and human infrastructure; areas of natural forest, crop fields, human infrastructure in 2010 and nature reserves in 2012; the time since reserve establishment; the number of wildlife-rearing farms; and the human population density. The annual growth rate of the human population was $\log_{10}(x + 2)$ transformed and the topographic heterogeneity was $\log_{10}(x)$ transformed. We examined the species richness and abundance differences for all birds per segment using the paired t-test. We tested the abundance difference for each species per segment in a grid between the two periods using the Wilcoxon signed-rank test. We tested if the distribution of proportions of species with increased or decreased was even among abundance quartiles (Inger et al., 2015), among diet guilds and between resident and migratory birds using the Chi-square test. We ranked the abundance of each species based on average abundance of a species per segment across all grid cells in 1997–1998 survey period. Then we divided the rank studied into 4 abundance quartiles (Inger et al., 2015). Based on the feeding preferences observed during the majority of the year, we classified the diet guilds of the birds as Invertebrate (invertebrates), VertFishScav (vertebrates, fish and carrion), Omnivore (omnivores),

FruiNect (fruit and nectar) and PlantSeed (plants and seeds) (Wilman et al., 2014). Based on their migratory behaviours, we classified the birds into residents and migrants. The migrants included birds that wintered or summered away from Hainan Island (Zheng, 2011).

We adopted an information-theoretic approach to select the best candidate models in our analysis (Burnham and Anderson, 2002). If present, spatial autocorrelation can cause parameter estimation bias. We addressed this issue using two approaches. First, we determined whether an ecoregion can be used as a block-based random factor in a generalized linear mixed model (GLMM) to account for local spatial dependence of grids within two ecoregions by examining residual spatial autocorrelation (Li et al., 2015). We found there were no differences in the residuals between two ecoregions (Fig. A3), suggesting that including ecoregion as a random effect would not improve the parameter estimates of the models. Therefore, we treated ecoregion as a fixed variable. Second, we determined whether multiple linear regression models performed better than spatial generalized least-squares models (GLS) using grid cell coordinates to account for the spatial correlation structure and based on different link functions (exponential, Gaussian, linear, spherical and ratio) (Dormann et al., 2007; Chiron et al., 2009). We also compared these models to a GLS model built using standard multiple regression that assumes spatial independence. We selected the best models based on the lowest AIC_c (Akaike's information criterion corrected for small sample sizes) (Burnham and Anderson, 2002; Chiron et al., 2009). Because multiple linear regression models ($\Delta AIC_c \leq 2$) performed better than all the GLS models (Table A7), we constructed multiple linear regression models for multimodel inference using information theory to evaluate the relative impact of each variable on the bird abundance trend (Burnham and Anderson, 2002). The global model includes the bird abundance trend as a response variable and changes in the proportions of natural forest, human infrastructure and cropland; areas of natural forest, human infrastructure and cropland in 2010; annual growth rate of the human population; number of rearing farms in 2012; area of nature reserve in 2012; time since a reserve was established; topographic heterogeneity; and ecoregion (binary variable) as predictors. The full set of candidate models (total 2^{13}–1 = 8191 models) considering all possible combinations of the 13 variables were ranked based on AIC_c. The relative importance of a variable was evaluated by summing the Akaike weights of all the models including that variable. We reported the most strongly supported models that were within 2 AIC units ($\Delta AIC_c \leq 2$). We also provided supported models that were within 6 AIC units ($\Delta AIC_c \leq 6$) (Table A8) (Richards, 2008). All of the analyses were conducted using R v3.2.0 (R Development Core Team, 2015). The GLS models used the *gls* function in the *nlme* package (Pinheiro et al., 2016). The *dredge* and *model.avg* functions in the *MuMIn* package were used for model averaging (Bartoń, 2015).

Collinearity between variables was generally weak. Most pair-wise correlation coefficients were below 0.7 ($|r| < 0.7$) (Table A6). Model fit statistics such as AIC_c were not affected by the low collinearity (Burnham and Anderson, 2002).

3. Results

Overall, 262 species were recorded during the 1997–1998 period and the 2012–2013 period. The number of bird species recorded increased from 198 species in 1997–1998 to 214 species in 2012–2013 (Table A1). Forty-eight species were recorded only in the 1997–1998 period, including rare species such as the Chinese Egret (*Egretta eulophotes*), Black-faced Spoonbill (*P. minor*), Pallid Harrier (*Circus macrourus*), Far Eastern Curlew (*Numenius madagascariensis*) and Saunders's Gull (*Larus saundersi*). Sixty-four species were recorded only in the 2012–2013 period, including rare species such as the Oriental Honey-Buzzard (*Pernis ptilorhyncus*), Chinese Sparrowhawk (*Accipiter soloensis*), Pale-capped pigeon (*C. punicea*) and Blue-rumped Pitta (*Pitta soror*).

3.1. Changes in bird abundance and species richness between the two periods

There were 172 grids (10 km × 10 km) with two or more segments in both periods. A Wilcoxon rank-sum test showed that no differences existed among the longitudes, latitudes or elevations at the midpoints of the segments within a grid cell between the two periods ($z = 14,838$, $p = 0.9607$ for longitude; $z = 14,830$, $p = 0.9676$ for latitude; $z = 14,256$, $p = 0.5615$ for elevation), indicating that the locations and elevation of segments were unlikely to affect comparisons on the abundance and species richness of birds.

We observed 150 species in 172 grid cells in both periods. Of these species, the White-browed Laughingthrush (*Garrulax sannio*), Tickell's Leaf-warbler (*Phylloscopus affinis*), Black-winged Stilt (*Himantopus himantopus*), Fork-tailed Swift (*Apus pacificus*) and Asian Dollarbird (*Eurystomus orientalis*) are dispersants and vagrants to Hainan Island. These species were excluded from subsequent analyses.

For the remaining 145 species in 172 grid cells, both the pooled abundance (Table A4) and species richness (Table A5) per segment in a grid cell were lower in 2012–2013 than in 1997–1998 (paired *t*-test, $t = 5.05$, $df = 171$, $p < 0.001$ for abundance; $t = 5.80$, $df = 171$, $p < 0.001$ for species richness) (Fig. 2a). Moreover, the abundance (Table A4) and species richness (Table A5) of both resident and migratory birds were lower in 2012–2013 than in 1997–1998 (resident birds: $t = 4.13$, $df = 171$, $p < 0.001$ for abundance; $t = 3.72$, $df = 171$, $p < 0.001$ for species richness; migratory birds: $t = 4.15$, $df = 171$, $p < 0.001$ for abundance; $t = 8.19$, $p < 0.001$ for species richness) (Fig. 2b). Furthermore, lower abundance (Table A4) and species richness (Table A5) values were detected in grids both with and without reserves in 2012–2013 than in 1997–1998 (grids with reserves: $t = 2.49$, $df = 83$, $p = 0.01$ for abundance; $t = 3.79$, $p < 0.001$ for species richness; grids without reserves: $t = 4.55$, $df = 69$, $p < 0.001$ for abundance; $t = 4.45$, $p < 0.001$ for species richness) (Fig. 2c). Additionally, the abundance (Table A4) and species richness (Table A5) values in both ecoregions were lower in 2012–2013 than in 1997–1998 (HMRF: $t = 3.69$, $df = 69$, $p = 0.002$ for abundance; $t = 3.22$, $p = 0.002$ for species richness; SCSEF: $t = 3.64$, $df = 101$, $p < 0.001$ for abundance; $t = 4.83$, $p < 0.001$ for species richness) (Fig. 2d).

Twenty-eight species exhibited decreased abundances in 2012–2013 compared to 1997–1998, whereas thirty-three species displayed increased abundances (Table A2). However, the decreased abundances observed for the species that declined in 2012–2013 was the main reason for the overall lower abundance and species richness values for all birds (145 species) in 2012–2013 compared with 1997–1998. For the species that declined, the abundance decreased from 3.0 birds per segment in 1997–1998 to 1.0 bird per segment in 2012–2013 (Fig. 2e). In contrast, for the species that increased, the abundance only increased from 0.6 birds per segment in 1997–1998 to 1.2 birds per segment in 2012–2013 (Table A4).

The proportions of the species that declined were not significantly different between migrant and resident birds (Chi-square test, $df = 1$, $\chi^2 = 2.7928$, $p = 0.0947$) or among feeding guilds ($df = 4$, $\chi^2 = 3.6250$, $p = 0.4591$). However, they were significantly different among abundance quartiles ($df = 3$, $\chi^2 = 25.8000$, $p < 0.001$) (Fig. 3a, b, c). Approximately 17 species in quartile 1 (36 species) showed decreased abundances in 2012–2013, accounting for 47.2% of the species in this abundance quartile. This proportion gradually decreased from quartile 1 to quartile 4, with 25.0% (9/36 species) of the species declining in quartile 2, 5.6% (2/36 species) declining in quartile 3 and no species declining in quartile 4 (37 species) (Fig. 3a).

However, the proportions of the species with increased abundances were not significantly different among the abundance quartiles ($df = 3$, $\chi^2 = 1.9437$, $p = 0.5842$) or feeding guilds ($df = 4$, $\chi^2 = 1.6862$, $p = 0.7932$). However, they were significantly different between the resident and migratory birds ($df = 1$, $\chi^2 = 8.7798$, $p < 0.001$) (Fig. 3d, e, f). Approximately 32 species of resident birds exhibited an increased abundance, accounting for 29.6% of the residents, whereas only one species (2.7% = 1/37 species) of migratory bird displayed an increased abundance (Fig. 3f).

3.2. Factors that affect bird abundance

The six most highly supported models ($\Delta AIC_c \leq 2$) contained nine variables (Table 1): the change in the proportion of natural forest loss, change in the proportion of human infrastructure areas, area of human infrastructure in 2010, annual growth rate of the human population, human population in 2010, area of nature reserves in 2012, number of wildlife-rearing farms in 2012, topographic heterogeneity and ecoregions. However, these models displayed high model selection uncertainties ($W_i = 0.12$–0.30).

Model averaging showed that the changes in the proportion of natural forest loss (relative importance value = 0.99), the number of wildlife-rearing farms in 2012 (0.92) and the area of nature reserves in 2012 (0.75) were the most important factors affecting bird abundance. The model-averaged 95% confidence intervals of these three predictors never overlapped with zero. Bird abundance decreased ($\beta < 0$) with a corresponding increase in the proportion of natural forest loss and the number of wildlife-rearing farms in 2012, but increased ($\beta > 0$) with the area of nature reserves (Table 2). Other factors had comparatively minor effects (relative importance value ranged from 0.25 to 0.72), and the model-averaged 95% confidence intervals of these predictors overlapped with zero.

4. Discussion

Our results demonstrate that tropical birds exhibited declining trends of abundance and species richness on Hainan Island over the last 15 years. Although some species, such as the Grey-cheeked Fulvetta (*Alcippe morrisonia*) and Eurasian Tree Sparrow (*Passer montanus*), showed an increased abundance, declines in the abundances of more common species, like Japanese White-eye (*Zosterops japonicus*) and Light-vented Bulbul (*Pycnonotus sinensis*), have resulted in overall reductions in the overall abundance and species richness of birds on the island. Changes in the area of natural forest, the number of wildlife-rearing farms and area of nature reserves in 2012 were explanatory variables included in the most highly supported models. The variables were important predictors for bird abundance trends in model averaging. These results suggest that the decline of the overall bird abundance was associated with the number of wild-rearing farms, an increase in the area of nature reserves and a loss of natural forest.

The declines of bird abundance detected in this study were statistically sound and ecologically meaningful. Declining bird abundance trend on the island likely does not reflect the annual fluctuations due to climate variations in the two survey periods, as the mean temperature and precipitation in both periods were comparable to typical years (Table A3). Additionally, we calculated the abundance of birds based on averaging the densities of individual species in each grid. Therefore, particularly low or high population densities of individual birds in a specific year are unlikely to have markedly impacted our average abundance values (Lemoine et al., 2007).

Deforestation is one of the largest threats to birds in China and tropical areas (Castelletta et al., 2000; Li and Wilcove, 2005; Newbold et al., 2014). Natural forests are the most important habitats and refuges for birds. Forests play an irreplaceable role in supplying food resources, nesting conditions, breeding sites and other requirements for bird survival (Gibson et al., 2011). The loss and degradation of natural forests can result in dramatically declined bird populations (Gibson et al., 2011). Although logging on Hainan Island was banned in 1993 (Zhang et al., 2000), rubber and pulp plantations have developed and expanded rapidly in recent decades (Barr and Cossalter, 2004; Zhai et al., 2012; Zhai et al., 2014; Zhang et al., 2000). Such plantations have destroyed large areas of natural forest. Several more common species that are
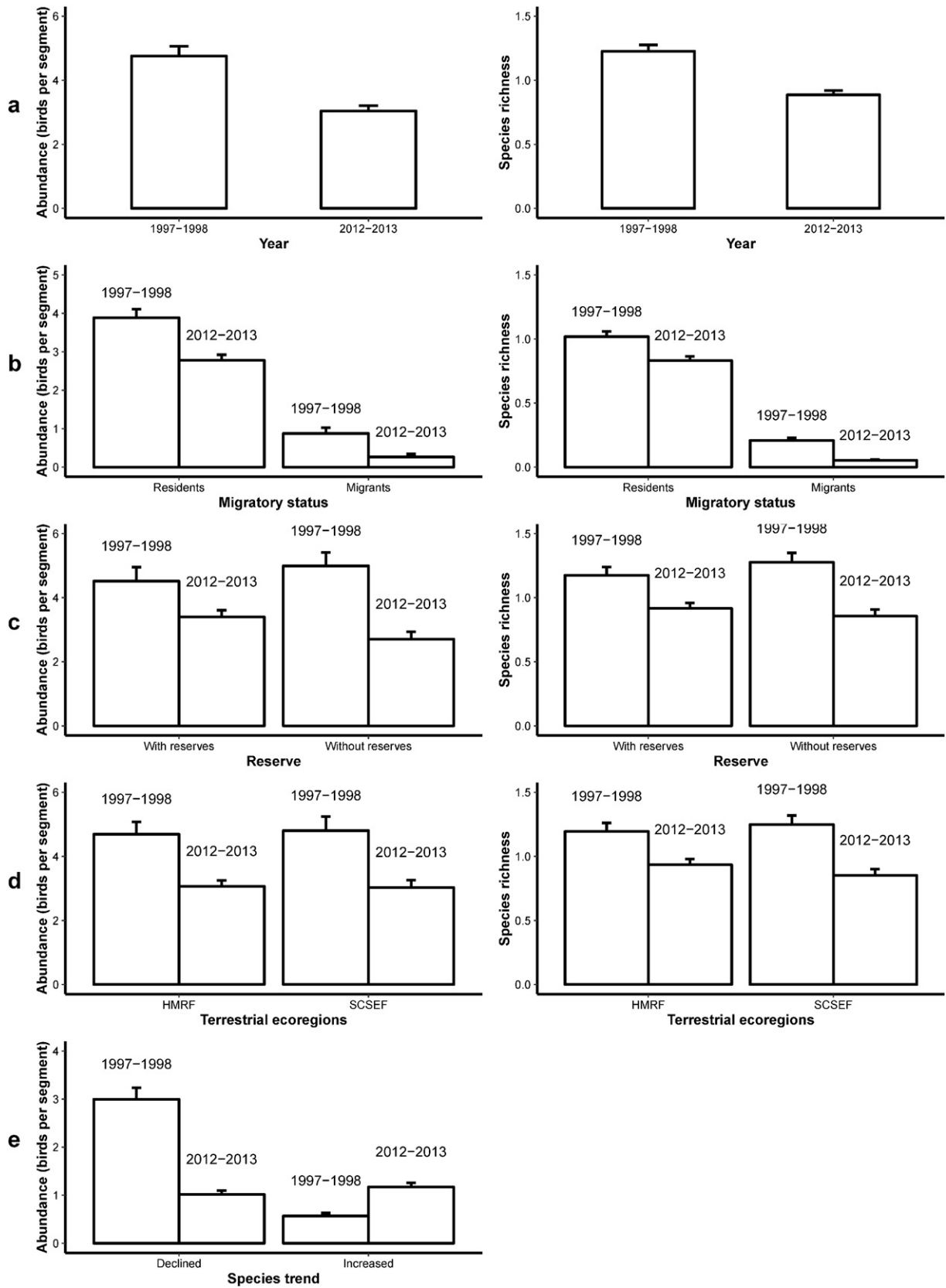
Fig. 2. Comparisons on the pooled abundance and species richness of birds between two periods on Hainan Island. a: all the grids; b: migratory status; c: grids with nature or without reserves; d: two terrestrial ecoregions; e:. increased and declined species.
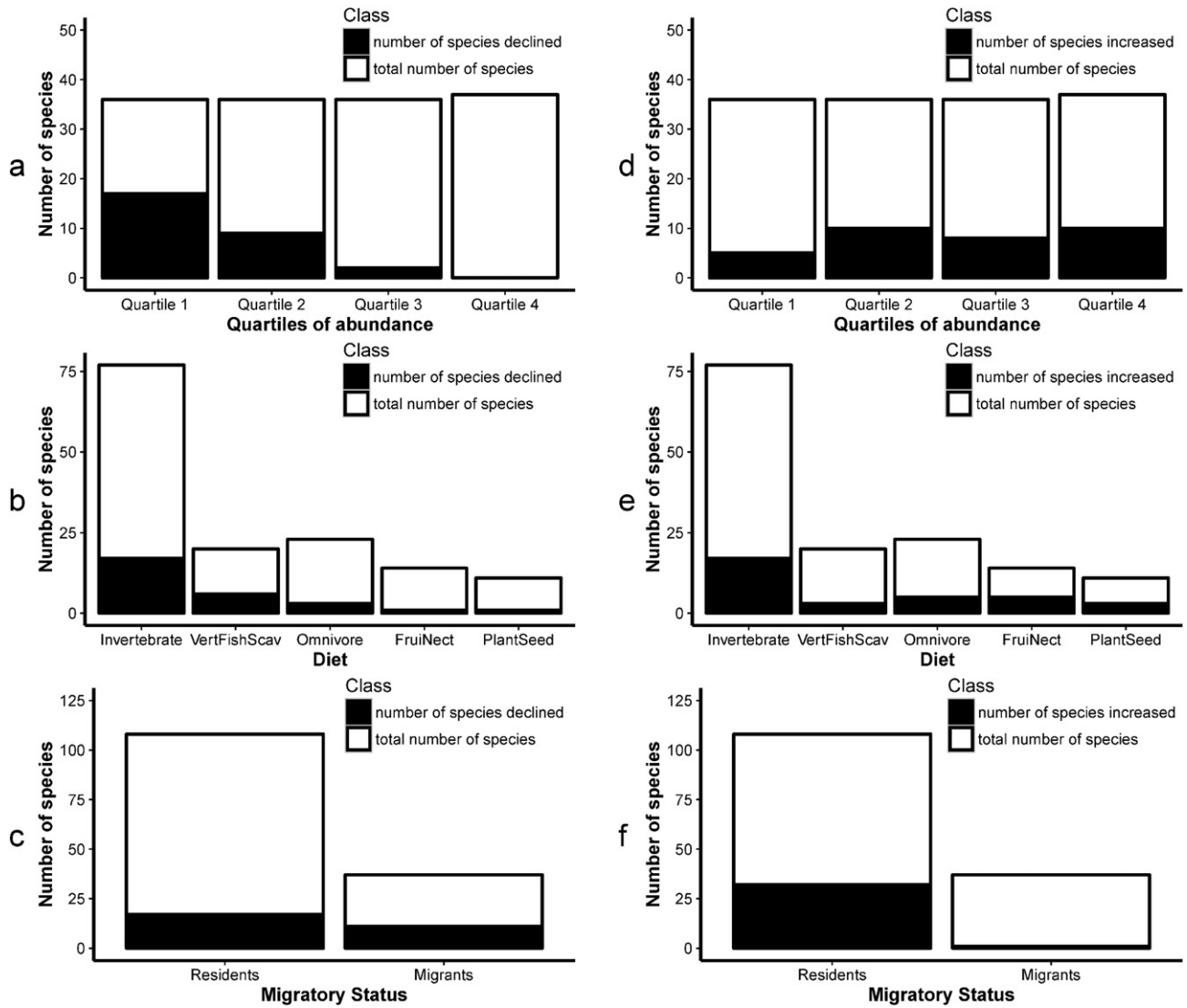
Fig. 3. Comparisons on distributions of number of bird species with increased or declined abundance. a and d: abundance quartiles; b and e: diet guilds (Invertebrate: invertebrates; VertFishScav: vertebrates and fish and carrion; Omnivore: omnivores; FruiNect: fruit and nectar; PlantSeed: plant and seed); c and f: migratory status.

often found in natural forests, such as the Inornate Warbler (*Phylloscopus inornatus*) and Indian Cuckoo (*Cuculus micropterus*), have declined in abundance, due in part to the loss of natural forest.

The negative relationship between the number of wildlife-rearing farms and the bird abundance trend suggests that wildlife-rearing farms have had a detrimental impact on the abundance of avian species

Table 1
The top 6 linear regression models (lm) (ΔAIC$_c$ ≤2) which investigating the effects of habitat, human activities, protecting activities and abiotic variables on the decline of bird abundance on Hainan Island.

Variable	1	2	3	4	5	6	R^2
Number of rearing farms in 2012	●	●	●	●	●	●	0.0402
Ecoregions	●	●	●	●	●	●	0.0002
Area of human infrastructure in 2010			●				0.0186
Change in proportion of area of human infrastructure				●			0.0005
Change in proportion of natural forest loss	●	●	●	●	●	●	0.0735
Human population in 2010						●	0.0186
Annual growth rate of human population		●					0.0131
Topographic heterogeneity					●		0.0233
Area of nature reserve in 2012	●	●	●	●	●	●	0.0359
ΔAIC	0.00	1.19	1.52	1.63	1.73	1.78	
AICc	187.35	188.54	188.87	188.98	189.08	189.13	
Wi	0.30	0.17	0.14	0.13	0.13	0.12	
R^2	0.152	0.156	0.155	0.154	0.154	0.154	

●, indicate the variable is included in the model; ΔAIC, the difference between each model and the highest ranked model; AIC$_c$, Akaike's information criterion adjusted for small sample sizes; W$_i$ (Akaike weights), the probability that a model is best given the particular set of models considered; R^2, amount of variation that is explained by factors. Models are ranked in the order of increasing of AIC$_c$.

Table 2
The summary of model averaging based on lm models (8191 models) using habitat, human activities, protecting activities and abiotic variables to explain the decline of bird abundance.

Variables	β	SE	95% CI (Lower, upper)	Relative importance
Change in proportion of natural forest loss	−0.9328	0.2645	**−1.45, −0.41**	0.99
Change in proportion of area of human infrastructure	1.0351	1.4848	−1.90, 3.97	0.30
Change in proportion of area of cropland	0.0074	0.4974	−0.97, 0.99	0.25
Area of natural forest in 2010	0.0041	0.0949	−0.18, 0.19	0.27
Area of human infrastructure in 2010	0.0585	0.1217	−0.18, 0.3	0.29
Area of cropland in 2010	−0.0798	0.1241	−0.32, 0.16	0.32
Annual growth rate of human population	−0.3022	0.2574	−0.81, 0.21	0.41
Human population in 2010	0.0616	0.0827	−0.10, 0.22	0.35
Number of rearing farms in 2012	−0.3243	0.1239	**−0.57, −0.08**	0.92
Area of nature reserve in 2012	1.0917	0.5366	**0.03, 2.15**	0.75
Time since a reserve was established	0.0063	0.0928	−0.18, 0.19	0.27
Topographic heterogeneity	−0.0773	0.1257	−0.33, 0.17	0.31
Ecoregions	0.1623	0.0816	**0.00, 0.32**	0.72

Model-averaged 95% confidence interval excluded zero value are showed in bold; β, model-averaged regression coefficients.

on the island. Two reasons may be largely responsible for this negative impact. First, wildlife-rearing farms play key roles in supplying wildlife for markets. Similar to other areas in South Asia (Corlett, 2007; Li and Li, 1998; Harris et al., 2016), illegal wildlife hunting and trade for food, TCM and pet are widespread in South China, including in provinces such as Guangxi, Guang Dong and Hainan (Gong et al., 2006; Lee et al., 2004; Li and Li, 1996, 1998; Liang et al., 2011; Zhou et al., 2005). Most illegal hunting activities are not for subsistence but rather for wildlife trade, because hunters can obtain high economic benefits from wildlife trade (Li and Li, 1996, 1998; Li et al., 2000; Lee et al., 2004). Many farms are involved in illegally hunting wildlife or purchasing wildlife captured by illegal hunters. These animals are sold in wildlife markets for substantial amounts (Chinese Forestry Yearbook Editors, 1999; 2000; Li and Li, 1998; Lee et al., 2004). For example, the Forestry Bureau of Hainan Province confiscated 15,288 animals that were illegally traded in 2009. These trade involved hundreds of illegal hunters and wildlife-rearing farms (Chinese Forestry Yearbook Editors, 2010). The confiscated animals included birds, mammals and reptiles. A large number of the farms within a grid may increase the illegal hunting pressure on wild birds and negatively impact the grid-based bird abundance. Second, wildlife-rearing farms can occupy large expanses of wetlands and croplands, destroying habitats for some waterbirds and farmland birds. Several waterbird populations that have declined, including the Cinnamon Bittern (*Ixobrychus cinnamomeus*) and White-throated Kingfisher (*Halcyon smyrnensis*), may be partly due to the loss of wetland from construction of the farms. A more recent study revealed that pet trade was a significant predictor of bird species decline in Indonesia (Harris et al., 2016). More research is needed for understanding how wildlife-rearing farms are linked to the bird decline in Hainan Island.

The 145 species observed in both periods are widely hunted in South China. For example, approximately 81% of these species are kept as pets or used for food, medicine or raw materials (Table A2). Furthermore, approximately 70% of these species are commonly found near Hainan Island in wildlife markets of Guangzhou and Shenzhen (Lee et al., 2004). This suggests that the demand for these birds is considerable in large cities. For birds with declined abundance, approximately 79% (22/28) of the species were kept as pets or used for food, medicine or raw materials (Table A2). In addition, 71% (20/28) were present in wildlife markets (Table A2). The prevalent wildlife trade on bird species implied that overhunting is likely a main factor that contributes to species declines.

Nature reserves play a key role in minimizing environmental threats to species (Beale et al., 2013), as both birds and their habitats can be protected from negative human activities in these areas. Correspondingly, bird abundance in a grid was positively correlated to the area of nature reserves. More resident species exhibited increased abundances than did migratory on the island. This result is potentially because resident birds benefit more from local nature reserves, while migratory

birds would also be affected by pressures outside Hainan Island. For example, rare species such as the Hainan Partridge (*A. ardens*) (VU) was mainly found in nature reserves, displayed increased abundance trends in 2012–2013 (Table A2).

Previous studies on Hainan Island have mainly concentrated on the bird species richness and the associated determinants at limited temporal (e.g., seasonal) and spatial scales (Yang et al., 2009; Zou et al., 2012). For the first time, our work revealed that tropical birds have suffered from an overall decline in abundance and richness on the island. A recent study that examined Europe-wide trends in avian abundance showed that common birds are declining rapidly, whereas less abundant species are increasing in abundance due to changes associated with agricultural intensification (Inger et al., 2015). Similarly, we found that more common species declined more rapidly than less abundant species on Hainan Island (Fig. 3c). The low proportion of species with decreased abundance in abundance quartile 3 and 4 were unlike due to less abundant species in two quartiles, which might reduce the statistical power to detect the trend. Compared with those in quartile 1 and 2, similar number of species with increased abundance was detected in quartile 3 and 4, suggesting that there was no problem of the low statistical power with quartile 3 and 4. The pattern that we detected may be partly because more abundant species are subjected to greater hunting pressures. For example, approximately 86% (31/36) of the species in abundance quartile 1 and quartile 2 were traded or used, while 79% (28/36) of species in quartile 3 and 51% (19/37) of species in quartile 4 were traded or used (Table A2). Due to wide distributions and high abundances of common species, decreases in these species would lead to a disproportionate loss of functional diversity among bird communities and reduce the resilience of ecosystems to environmental change (Gaston, 2010; Gaston and Fuller, 2007; Inger et al., 2015; Newbold et al., 2012).

Studies suggest that different feeding guilds have different vulnerabilities due to habitat loss and fragmentation. Insectivores, frugivores and nectarivores are considered more vulnerable to habitat loss and fragmentation due to their relatively poor abilities to disperse among patches (Chang et al., 2013; Newbold et al., 2012; Newbold et al., 2014; Robinson and Sherry, 2012; Sekercioglu et al., 2004; Sigel et al., 2006; Sodhi et al., 2004). Conversely, omnivores are likely more tolerant to such disturbances (Sigel et al., 2006). Similar to what has been observed for birds in Europe (Inger et al., 2015), we detected no differences in the numbers of species showing declines or increases among feeding guilds. The pattern that feeding guilds did not show differences in increased or declined species is likely associated with overhunting, which is a major threat to birds on the island. Hunting pressures are extensive and do not favour particular bird species, causing evenly distributed abundance reductions among different feed guilds.

While predictive factors only explained approximately 15% of the variations in the abundances of birds in this study (Table 1), a portion of the unexplained variation may have arisen due to the explanatory

variables that were not quantified, or those that were not well documented (e.g., habitat fragmentation, climate change and alien species invasions). Habitat fragmentation affects bird population dynamics differently than does the loss of forest area. Fragmentation has been shown to result in migratory forest bird declines in Japan (Kurosawa and Askins, 2003). The fragmentation of bird habitat on Hainan Island has increased due to deforestation and habitat conversion since 1988 (Wang and Liu, 2013). Furthermore, studies have suggested that climate change has negatively impacted bird abundances in Europe (Lemoine et al., 2007). The average air temperature in Hainan has increased 0.6–0.8 °C over the last century (Stocker et al., 2014). Biological invasions may also have detrimental effects on bird abundance due to habitat modification, food supply changes, increased predation, enhanced competition and the spread of disease by alien species (Gurevitch and Padilla, 2004; LaDeau et al., 2007; Mack et al., 2000). Although we did not detect any alien birds during the two survey periods, the plantation of introduced tree species, such as *Acacia mangium* and *Eucalyptus spp.*, has expand rapidly on Hainan Island, A study suggests that *Spartina alterniflora* invasions have changed mudflats into *Spartina* meadows in the Yangtze River estuary, China (Li et al., 2009), and altered local bird composition. Hence, the effects of these factors on bird abundances on the island warrant further study.

The results of these surveys may have important implications for bird conservation on Hainan Island. Policymakers on Hainan Island generally give more attention to threatened species due to their high extinction risks associated with human disturbances (Wang et al., 2013). However, 'common' species that have declined on the island must also be protected. The 3 most abundant birds (i.e., the Japanese White-eye (*Z. japonicus*), Light-vented Bulbul (*P. sinensis*) and Barn Swallow (*Hirundo rustica*)) in 1997–1998 all declined in abundance (Table A2). Yet, these species remain common in Hong Kong, South China (Kwok and Corlett, 2000; Corlett, 2005). Bird conservation in Hong Kong may provide useful lessons for conserving common species in Hainan Island. For example, the plantation that is replacing the natural secondary forest in Hainan island should be restricted due to the fact that many bird species cannot breed in the plantation (Kwok and Corlett, 2000; Corlett, 2005). The local government of Hainan Island should implement a long-term bird monitoring programme to know where birds are declining rapidly. Joint efforts by government agencies and Non-Governmental Organizations (NGOs) for conservation may foster educational campaigns targeted to bird conservation (Ma et al., 2013).

We found that the loss of natural forests was a driver of bird declines on the island. Therefore, conserving natural forests is crucial for managing bird diversity. Stricter regulations prohibiting or reducing deforestation are necessary to promote the conservation of birds and other wildlife. Additionally, the results of this study suggest that an increase in the area of nature reserves should decrease the rate of bird species decline. Currently, the area of nature reserves (7.4% of the land area) on Hainan is small compared to the area of reserves (14.9% of the land area) throughout China (Ren et al., 2015). Thus, designating more natural forest areas for nature reserves may be an effective approach for conserving bird diversity. In addition, local governments should strictly enforce wildlife-rearing regulations to prohibit the illegal capture of wildlife for wildlife-rearing farms and wildlife trade. In addition, strict entrance permits must be required for wildlife-rearing farms. To reduce the threats imposed by wildlife-rearing farms on local wildlife, the potential impacts of new wildlife-rearing farms on wildlife habitats should be evaluated before the farms are established.

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