RAFFLES BULLETIN OF ZOOLOGY **71**: 572–582
Date of publication: 2 October 2023
DOI: 10.26107/RBZ-2023-0043
http://zoobank.org/urn:lsid:zoobank.org:pub:E1EC9C54-F518-4090-84D3-9A224B8A975F

Long-term changes in avian relative abundances in relation to human disturbance in a tropical dry forest in central Myanmar

Myint Myint **Soe**[1], Daphawan **Khamcha**[2], Dusit **Ngoprasert**[1], Tommaso **Savini**[1], Naruemon **Tantipisanuh**[2] & George A. **Gale**[1*]

Abstract. Tropical dry forests, including dry dipterocarp forests of Asia, are in jeopardy from increasing human pressures, especially agricultural encroachment, hunting, and probably climate change. Quantifying the impacts of these pressures and implementing effective management solutions have been hampered by a scarcity of regional long-term wildlife monitoring data. Our study examined long-term (22-year, 1999–2020) changes in the relative abundances of six avian guilds at a small-scale (30 survey points) in Chatthin Wildlife Sanctuary in Myanmar's Central Dry Zone. We focused on common feeding guilds that have significant effects on ecosystem processes, including seed dispersal (frugivores), insect abundance regulation (bark-gleaning insectivores [woodpeckers], foliage-gleaning insectivores, sallying insectivores, and terrestrial insectivores), and seed predation (granivores). We also used data from five focal species within these guilds to further assess long-term trends. Trends in relative abundances were analysed in relation to time (survey year) and forest disturbance. Although the forest within a 300-m radius of the survey points appeared to be only moderately disturbed, relative abundances of frugivores, granivores, and terrestrial insectivores declined over the study period. Woodpeckers declined in response to specific forest disturbance incidents, but did not show long-term declines. Foliage-gleaning and sallying insectivore abundances did not change significantly. Among individual focal species, blossom-headed parakeet (*Psittacula roseata*) and white-browed fantail (*Rhipidura aureola*) abundances declined. The parakeet and fantail declines are particularly salient because these two species are typically associated with dry forest, although the parakeet was probably highly impacted by poaching. In contrast, relative abundances of the more generalist common woodshrike (*Tephrodornis pondicerianus*) increased over time, golden-fronted leafbird (*Chloropsis aurifrons*) likely also increased, while rufous treepie (*Denmdrocitta vagabunda*) did not show a clear trend. Although our monitoring data on human disturbance was limited, most declines were probably a consequence of increased human use inside the sanctuary rather than climate change, indicating significant cumulative effects of extensive local human use of the forest during the 22-year period.

Key words. avian feeding guilds, dry dipterocarp forest, long-term study, Chatthin Wildlife Sanctuary

INTRODUCTION

Seasonally dry tropical forests occur in areas of the tropics where high temperatures and seasonal rainfall result in seasonal water stress (Miles et al., 2006), and although they once covered extensive areas, tropical dry forests are more threatened but less protected than less seasonal tropical forests (Songer et al., 2009). Dry forests are also less studied than tropical rainforests (Siyum, 2020). This is especially true for dry forests of Southeast Asia relative to other areas such as the Americas (Portillo-Quintero et al., 2010; Wohlfart et al., 2014). In Southeast Asia, a prevalent forest type is the deciduous dipterocarp forest (DDF) (Nguyen & Baker, 2016). In addition to their adaptation to strongly seasonal environments, DDFs are unique among tropical forest types because of their characteristic open canopy and extensive grasslands, which support (or formerly supported) a particularly high mammalian biomass, including key browsers and grazers such as Asian elephant (*Elephas maximus*), banteng (*Bos javanicus*) and Eld's deer (*Rucervus eldii*) (Gray et al., 2011), and as such hold a relatively unique, but highly threatened fauna (Ratnam et al., 2016).

Long-term monitoring of biodiversity is critical to assess threats to forests such as DDF, link conservation action to species outcomes, and facilitate improved management. Yet, rigorous long-term monitoring of wildlife is rare (Nuttall et al., 2022). In Southeast Asia, while there has been long-term monitoring of waterbirds (Haq et al., 2018) there is little long-term wildlife monitoring of terrestrial birds. This makes trend assessments difficult not only for threatened species but also for more common species where, ideally, managers should be aware of declining trends long before

---

Accepted by: Frank E. Rheindt

[1]Natural Resource Management and Sustainability, School of Bioresources and Technology, King Mongkut's University of Technology, Thonburi, Bangkok, 10150 Thailand; Email: george.a.gale@gmail.com (*corresponding author)

[2]Conservation Ecology Program, Pilot Plant Development and Training Institute, King Mongkut's University of Technology, Thonburi, Bangkok, 10150 Thailand

ISSN 2345-7600 (electronic) | ISSN 0217-2445 (print)

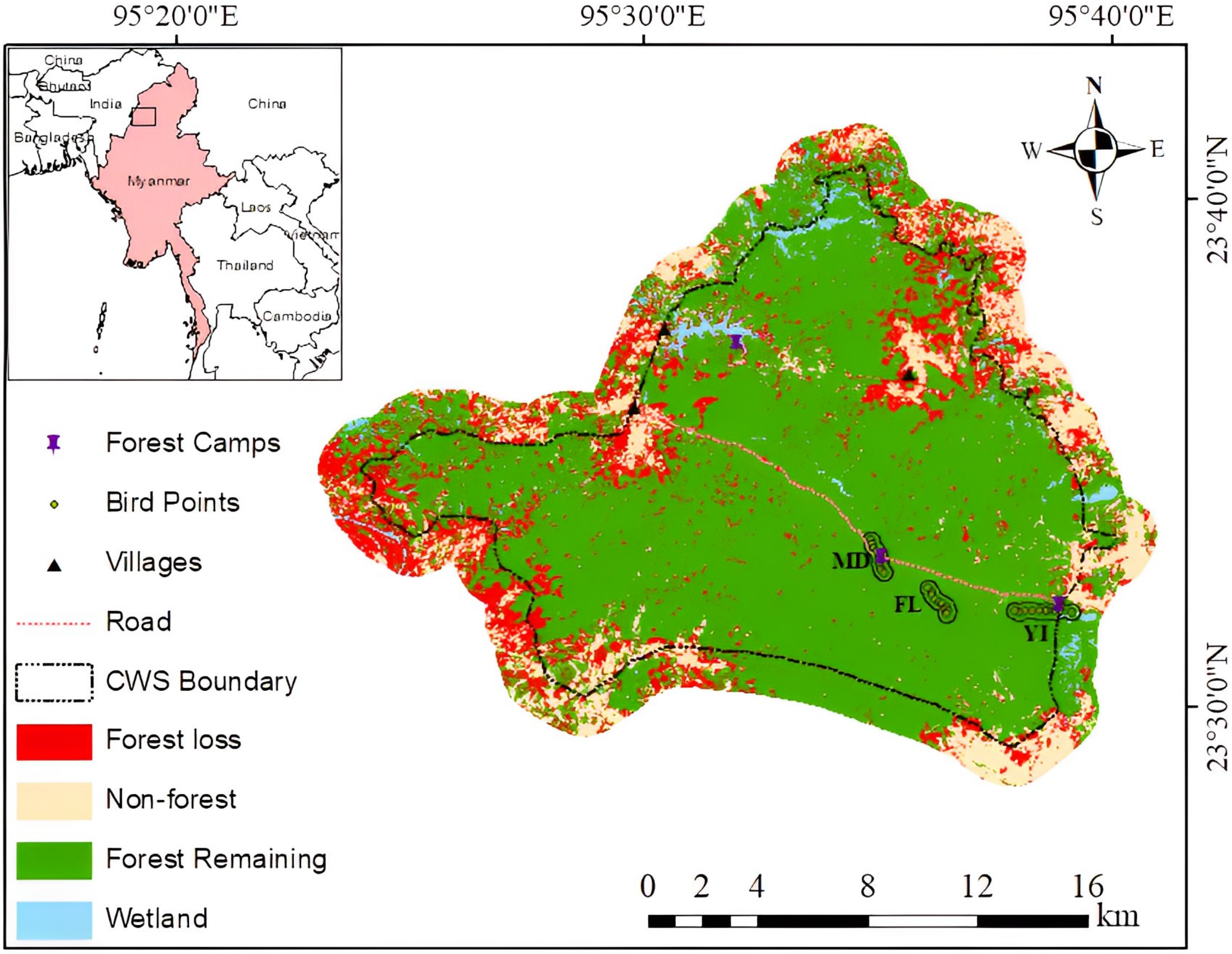

Fig. 1. Map of Chatthin Wildlife Sanctuary with land-cover changes; flooded dipterocarp forest (FL), mixed deciduous forest (MD) and young dipterocarp forest (YI) in Chatthin Wildlife Sanctuary between 1999 and 2020 and locations of bird survey points.

such species also become threatened. Here we focus on DDF in Southeast Asia where only about 156,000 km^2 of the habitat remains in mainland Southeast Asia (Wohlfart et al., 2014). Although mapping of DDF is particularly challenging, Myanmar is the country that appears to hold the largest area (~79,000 km^2) of DDF in Southeast Asia (Wohlfart et al., 2014). However, due to shifting cultivation and agricultural development projects, DDF in Myanmar suffered a 2.3% mean annual deforestation rate between 2001 and 2010, the second-largest net forest loss relative to mangroves (Wang & Myint, 2016). DDF is also important for the local human communities for food, fuel, medicine and livestock fodder (Khaing & Mitloehner, 2014). Currently, only about 2% of the remaining DDF in Myanmar is protected under existing laws (Wohlfart et al., 2014).

Bird communities are often a good indicator of the ecological status of ecosystems because they serve as important links to specific environmental changes (Bowler et al., 2019), making them a useful tool to monitor habitat over long periods of time (Şekercioğlu et al., 2019; Hendershot et al., 2020). Declines in bird communities and/or bird abundance as a consequence of anthropogenic habitat degradation can affect ecosystem integrity and decrease vital ecological, evolutionary, and economic services (BirdLife International, 2000; Inger et al., 2015). Furthermore, different feeding guilds are expected to respond differently to habitat degradation (Khamcha et al., 2018). However, to assess the impact of human disturbance on a dry forest bird community, long-term datasets are needed (Rosenberg et al., 2019). Chatthin Wildlife Sanctuary ("Chatthin") in Central Myanmar is probably the only DDF site in Myanmar (and likely the region) that has had its whole bird community repeatedly surveyed over a span of nearly 30 years. Surveys and ornithological research were initially conducted in 1994, 1996, and 1997; more systematic surveys were done in 1999, 2000, and 2001 (King & Rappole, 2001; Rappole et al., 2011). Follow-up surveys were conducted from 2015 to 2020. Although the surveys were small in scale, they allow us to investigate changes in relative avian abundance and richness in different dry forest habitat subtypes (see below) inside Chatthin during a 22-year period (1999–2020). From the data collected we predicted that insectivore (which were subdivided into four additional foraging guilds), frugivore, granivore, and woodpecker relative abundances would decrease over time as the sanctuary became increasingly used for subsistence by people in the surrounding villages. Second, because DDF is maintained by a complex balance of periodic fires and grazing (Ratnam et al., 2016), birds typically associated with DDF would probably decline because human pressures were also likely to significantly impact both fire and grazing regimes. The aim of this work was therefore to use this long-term dataset to study the changes in this bird community over time, particularly in response to increased human use inside the sanctuary.

MATERIAL AND METHODS

Study site. Chatthin covers an area of 268 km^2 in the northern part of the central dry zone (95°24′E–95°40′E, 23°30′N–23°42′N) in Myanmar (Fig. 1) and is mostly composed of DDF. It is listed as a Key Biodiversity Area (Myanmar Biodiversity Conservation Investment Vision, 2013). During our study period (1999 to 2020) the average annual rainfall was 1,645 mm, with a minimum of 623 mm and a maximum of 2,440 mm per year (Meteorology Department, Kalin Township (1999 to 2014) and Kabalu Township (2015 to 2020), unpublished data) (Fig. 2). Chatthin was established in 1941 to protect Eld's deer (currently globally endangered) and its habitat (Khant et al., 2018), and is home to one of the world's largest remaining Eld's deer populations. Overall, Chatthin appears to resemble mostly deciduous dipterocarp savannah as described by Ratnam et al. (2016) although teak (*Tectona grandis*) is also present along with mixed deciduous savannah. Regular bird surveys were conducted in three rough subtypes of habitat within the sanctuary: (1) "Seasonally flooded deciduous" areas where the most common tree species were *Dipterocarpus tuberculatus*, *Lophopetalum wallichii*, *Shorea oblongifolia*, *Dillenia parviflora*, and *Terminalia tomentosa* and *Eugenia* species. Due to extensive flooding during the rainy season, all dominant trees were stunted and gnarled. (2) "Mixed deciduous" areas included *Tectona grandis*, *Xylia xylocarpa*, *Mitragyna rotundifolia*, and *Cassia tora*. Finally, (3) "young deciduous" areas commonly included *Dipterocarpus tuberculatus*, *Shorea oblongifolia*, *Lophopetalum wallichii*, *Dillenia parviflora*, *Buchanania cochinchinensis*, *Celtis cinnamomea* and *Shorea siamensis*, where the tree size was generally small, and more than two-thirds of trees had a diameter at breast height of 4–12 cm (Khant et al., 2018). Chatthin and the area surrounding it have long been threatened by the encroachment of agricultural lands, the major cause of deforestation in Myanmar (Songer, 2006). Over time, Chatthin has become a forest island surrounded by agricultural lands and a large reservoir. Consequently, the number of villages depending on the remaining forest increased from 28 (2000) to 38 (2019) according to the available data (Forest Department, 2019). Based on sampling conducted in conjunction with annual Eld's deer surveys performed by Chatthin staff, disturbance from local people is likely widespread within the sanctuary (Nature and Wildlife Conservation Division, 2020) (see below).

Bird surveys. Using 50-m fixed-radius, 10-min point counts, with 10 points per transect per habitat subtype (see below), we conducted bird surveys monthly in the morning (between 0600 and 1000 hours) for nine years in Chatthin during two discrete periods (1999–2001 and 2015–2020). We counted all birds detected within the sample radius (Shwe, 2014; Verga et al., 2017). During both survey periods, the same 30 points were sampled across three DDF subtypes (ten per subtype) (seasonally flooded deciduous, mixed deciduous and young deciduous habitat) within the sanctuary (Shwe, 2014). These sampled subtypes were two kilometres apart (Fig. 1). Sample points were located 200 m apart to reduce the risk of double counting (Shwe, 2014). For the 2015–2020 surveys, distances between birds and observers were measured using a range finder. However, for the 1999–2001 surveys, range finders were not available and thus visual estimation was used. All point count surveys were conducted by the same person; the day of the month the surveys were conducted depended on logistical constraints and weather. The elevation and GPS coordinates were recorded at each point.

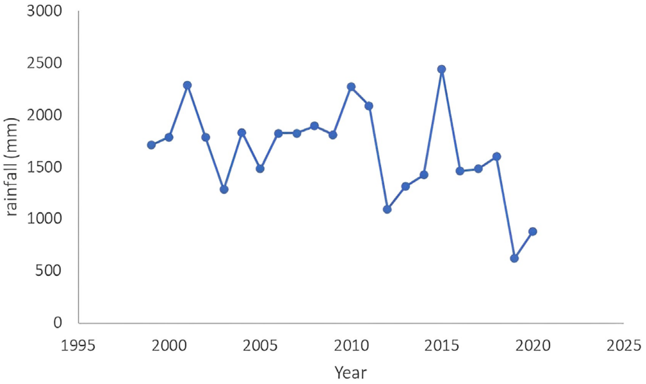

Fig. 2. Annual rainfall measured adjacent to Chatthin Wildlife Sanctuary 1999–2020.

Forest loss and forest disturbance. To assess forest loss over the course of the study period, land-use maps for 1998–2003 and 2014–2020 were created using Landsat satellite images (Landsat 5 for 1998–2001, Landsat 7 for 2002–2003 and Landsat 8 for 2014–2020). All satellite images were acquired for the month of February for each of the years when surveys were conducted (roughly corresponding to the latter half of the dry season). The images were downloaded from http://earthexplorer.usgs.gov/. We did not attempt to use more than one image per year because cloud-free images were typically limited to only the dry season (late October to early April). Supervised classification was performed using the 'Maximum Likelihood Classification' toolbox. For land-use map year 2020, 300 points were generated (from ground-truth surveys and fine-scale satellite images in Google Earth) and split into two groups: 218 were training points (70%) and 82 were assessed points (30%). Training points were used to perform the classification, while assessed points were used for assessing the map's accuracy. The overall accuracy of the land-use map for 2020 was 99%. Forest cover in 2020 was 246 km^2 (87% of the Chatthin area). For land-use maps in other years, as ground-truth data were not available, we did not assess map accuracy.

A forest loss and forest disturbance index estimate for each year were determined for each survey point. Firstly, 300 m buffers around survey points within each habitat were generated using the 'Buffer' toolbox (Fig. 1). A distance of

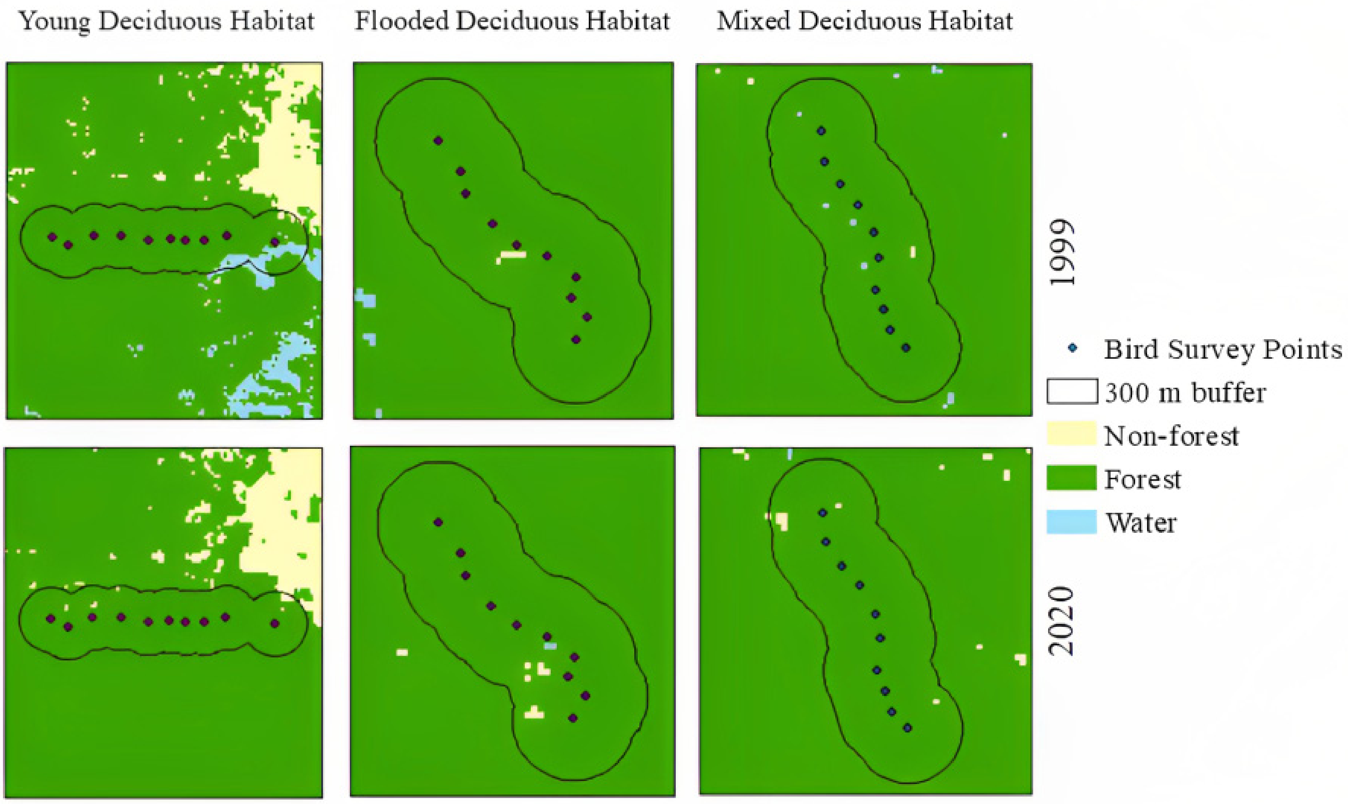

Fig. 3. Forest cover within 300 meters of bird survey points in 1999 and 2020.

300 m was chosen because a) our point counts were likely not effectively sampling habitat further than this and b) the majority of species in our study area were probably not ranging further than this. This ranging assumption was based on home range data from the black-headed woodpecker (*Picus erythropygius*) in dry dipterocarp/mixed deciduous forest in Huai Kha Khaeng Wildlife Sanctuary, Thailand (~195 m; D. Khamcha, pers. comm.), a bird with a relatively large body size (average 124 g, range 112–155 g) (A. J. Pierce, unpublished data).

The forest areas within the buffers were then calculated for each year using the 'Intersect' toolbox. To calculate forest loss each year, the forest extent of the former year was erased on the map of the assessed year using the 'Erase' toolbox, and the forest loss area within the buffer was calculated using the 'Intersect' toolbox. Because of the limitations of the spectral and spatial resolution of Landsat data, our measure of forest loss was able to quantify the abrupt transition from forest to non-forest shortly after cutting, but once scrubby regrowth occurred, our index was not able to distinguish different ages of regrowth.

FRAGSTATS 4.2.1 was used to assess forest disturbance each year for each of the three buffered habitat subtype polygons (Fig. 3). Three metrics from the software were initially used to assess forest disturbance: the landscape division index, splitting index, and number of forest patches. However, due to the significant correlation (Spearman $r > 0.5$) among these three metrics, we only used the division index to assess disturbance. The division index ranged from 0 to 1, with 0 indicating no fragmentation/disturbance (landscape is a single patch) and 1 indicating that the landscape consists of the maximum number of small, single-cell patches and is entirely fragmented (Table 1). Based on our visual inspection of the images and our ground-truthed data, the Landsat images were likely only identifying changes that were detectable for relatively short (≤ 3 years) periods. After about three years of regrowth, previously disturbed areas often could not be distinguished from areas which were classified as "forest". Therefore, the division index was assumed to provide a short-term measure of forest disturbance and we refer to it hereafter as our "disturbance" index. This index was only used to assess rough changes in vegetation cover, it was not used as a proxy to estimate poaching or other human activities on the ground. All spatial analyses were conducted in ArcGIS 10.3 (Mahmon et al., 2015).

Measures of human use of the forest. There was no specific data collection during 1999–2001 to monitor human use/disturbance in Chatthin, but from 2015–2020, we counted all human detections and evidence of human activity within 100 m of five two-kilometre transects (one transect for each of five habitat type/subtypes per month per year) (young deciduous, flooded deciduous, mixed deciduous, mature deciduous, and wetland). Surveys were conducted in the morning between 0600 and 1000 hours. For each line transect, we also counted signs of human disturbance, such as tree cutting, wildlife poaching, non-timber forest product collection, etc. In addition, we compiled secondary information (i.e., unpublished reports from the Myanmar Forest Department, literature from the study area, and questionnaire surveys) on human disturbance in the study area to supplement our direct observations. There were several socio-economic

Table 1. Climate and landscape habitat variables based on the 300 m radius around survey points.

Variable	Data type	Description
Sampling covariates		
Time	Continuous	Time of day between 0600 hrs and 1000 hrs
Climate		
Rainfall	Continuous	Total monthly rainfall (mm) from Township Meteorology Department
Abundance covariates		
Forest loss	Continuous	Forest loss area (m^2) within a 300 m radius around survey points
Forest area	Continuous	Forest area (m^2) within 300 m radius around survey points
Forest disturbance		
Disturbance	Continuous	Index of forest disturbance within 300 m radius (ranges between 0–1) using the "landscape division index" FRAGSTATS Ver. 4. The higher the value the more disturbed the landscape. The index is 0 when the landscape consists of a single patch, and 1 when every cell is a separate patch.

surveys conducted on human use during our 22-year study period. These included 1) a study published by Allendorf et al. (2012) that looked at local people's perceptions of Chatthin and its management in 1999 and 2003, 2) a 2017 assessment within the sanctuary utilising 24 camera traps to survey wildlife in the core zone of Chatthin conducted by the Myanmar Forest Department, and 3) a 2018 questionnaire survey of local people from eight villages regarding where they go and what they collect in Chatthin, conducted by the Friends of Wildlife (unpublished data).

Data analysis. The seasonal status of all observed species was defined following Robson (2015). There were only sufficient data to analyse resident species and thus migrants had to be excluded. Since avian guilds respond differently to measured local habitat features and disturbance, birds were grouped into feeding guilds to streamline the assessments of the study (Atikah et al., 2021). Guilds with sufficient data for analysis included frugivores, granivores, bark-gleaning insectivores (woodpeckers), foliage-gleaning insectivores, sallying insectivores, and terrestrial insectivores (following Wilman et al., 2014; Khamcha et al., 2018); see below for a description of how we assessed whether there were sufficient data. Where possible, we further subdivided the guilds into species with small and large foraging ranges for analysis assuming that species with different-sized foraging ranges were likely to respond to habitat fragmentation differently (Watson et al., 2005; Toscano et al., 2016). Foraging range classification (small versus large) was based on our knowledge of the natural history and ecology of the species (size classifications are listed in Supplementary Table 1).

For frugivores, flowerpeckers were excluded from further analysis because the number of detections was insufficient for analysis. For guilds with relatively similar-sized species, but insufficient data to analyse separately by foraging range size, we combined foraging ranges. This was the case for granivores, woodpeckers, and terrestrial insectivores. We did not have sufficient data on foliage-gleaning insectivores with large foraging ranges for analysis. We also determined that no large-foraging-range sallying insectivores were in our dataset. Among focal species with sufficient data, some of which were specifically associated with DDF, five species could be analysed: blossom-headed parakeet (*Psittacula roseata*), common woodshrike (*Tephrodornis pondicerianus*), golden-fronted leafbird (*Chloropsis aurifrons*), rufous treepie (*Dendrocitta vagabunda*) and white-browed fantail (*Rhipidura aureola*). Overall, most species in the study area were found in all three habitat subtypes (Supplementary Table 1).

Changes in the relative abundances in the guilds and the focal species were assessed using generalised linear mixed models (GLMMs) (Zuur et al., 2009). GLMMs were appropriate for this analysis because they allow the inclusion of random effects (Gelman & Hill, 2006). In this case, we used the three habitat subtypes (seasonally flooded deciduous, mixed deciduous and young deciduous habitat) as a random intercept in all models to account for replicate surveys of each habitat over the nine survey years. In addition, rainfall may affect the abundance of birds through impacts on fruit and/or insect abundance (Chapman et al., 2005; França et al., 2020). We therefore tested for possible relationships between monthly rainfall and bird counts and found no correlation, even though based on our observed data rainfall may affect detectability. To account for variation in detectability in relation to rainfall patterns, four seasons were recognised, dry breeding (January to April), wet breeding (May to August), dry non-breeding (September to October), and wet non-breeding (October to December). Finally, we tested and then used either habitat subtype or season as random effects to best account for unexplained variation in detectability.

We performed a preliminary analysis with predictor variables (fixed effects) including survey year, forest disturbance index, rainfall, forest area, and forest loss. After this preliminary analysis, forest area and forest loss were eliminated as typically there was relatively little forest that we could detect as permanently lost within a 300-m radius of the sample points (Figs. 1, 3). Prior to data analysis, all variables were subjected

Table 2. Coefficient estimates of feeding guild models and 95% lower confidence intervals (95% LCI) and upper confidence intervals (95% UCI) for variables in the best-fitted models. The sign (+/-) of coefficient estimates refers to the influence direction of that variable on abundance, while the number refers to the magnitude of the influence.

Guild	Estimate	95% LCI	95% UCI
Frugivores [bs]			
Count model			
year	-0.60	-0.68	-0.51
forest disturbance	0.36	0.22	0.51
forest disturbance^2	-0.10	-0.19	-0.01
Granivores [ah]			
Count model			
year	-0.39	-0.45	-0.32
forest disturbance	-0.18	-0.24	-0.11
Zero model			
rain	0.29	0.26	0.48
rain^2	-0.1	-0.2	-0.04
Foliage-gleaning insectivore [bh]			
Null	-1.87	-2.08	-1.66
Sallying insectivore [ah]			
Count model			
Null	-1.98	-2.21	-1.75
Zero model			
Null	1.95	0.80	0.46
Terrestrial Insectivore [ah]			
Count model			
year	-0.17	-0.25	-0.09
forest disturbance	-0.16	-0.27	-0.04
Zero model			
Null	2.19	2.07	2.31
Woodpeckers [ah]			
Count model			
forest disturbance	-0.29	-0.44	-0.15
Zero model			
rain	0.2	0.37	0.04
rain^2	-0.09	-0.15	-0.03

[ah] zero-inflated Poisson regression models with random habitat
[bh] negative binomial regression models with random habitat
[as] zero-inflated Poisson regression models with random season
[bs] negative binomial regression models with random season

to a Spearman correlation test to check for multicollinearity among variables. The same model did not include highly correlated variables (r >0.5) (Mag & Ódor, 2015). Data were evaluated for outliers, overdispersion and zero-inflation using the DHARMa package (Hartig, 2019). We used monthly rainfall and time of day in zero-inflated models as these variables were expected to influence detectability. For each model, bird counts from monthly surveys at each survey point were used as response variables. To make the beta coefficients of the models directly comparable, all continuous variables were standardised by subtracting the mean for each variable and dividing by its standard deviation (Z-scores) (Gelman & Hill, 2006). We fitted and checked model assumptions for Poisson and negative binomial models or both distributions for the zero-inflated models (Zuur & Ieno, 2016). For model selection, we used the differences in the Akaike information criteria (ΔAIC) and AIC weights (Burnham & Anderson, 2002). If more than one model was within 2 delta AIC units, we then chose the most parsimonious model. We considered the evidence of variables influencing the relative abundances within each avian guild and focal species using 95% confidence intervals that did not overlap zero (statistically significant coefficients). We used the package "glmmTMB" (Brooks et al., 2022) to construct the models. All statistical analysis was performed in program R (R Core Team, 2022). To graphically assess changes in the general trends in relative abundances of six guilds (large and small foraging range combined) analysed above, we used "heatmap" graphical representations generated in R.

RESULTS

A total of 3,240 point counts were conducted in Chatthin during the nine survey years (1999–2001 and 2015–2020). During the study period, 8,765 detections were recorded for 144 species including 128 species of resident birds. Despite the multiple years of data, only a limited number of guilds and focal species had sufficient numbers of detections to be analysed because of the relatively small number (~30) of survey points sampled per year. Based on the distributions of the data, the analyses were limited to resident birds with >53 detections per year per guild and >10 detections per year per focal species. These included large foraging range frugivores (943 detections, of which 409 (43%) were parakeets), granivores (combining species with small and large foraging ranges; 713 detections) such as doves; small foraging range sallying insectivores (650 detections) such as drongos; terrestrial insectivores (combined foraging ranges; 476 detections) such as babblers; small foraging range foliage-gleaning insectivores (2,740 detections) such as minivets; and woodpeckers (combined foraging ranges; 556 detections), with 11 species of woodpecker included. One globally Vulnerable species was recorded, great slaty woodpecker (*Mulleripicus pulverulentus*) (Supplementary Table 1). In addition, three endemic species were recorded: Ayeyarwady bulbul (*Pycnonotus conradi*), white-throated babbler (*Turdoides gularis*), and Burmese bushlark (*Mirafra microptera*). Insectivores were the most diverse feeding guild with 85 species (Supplementary Tables 1).

Bird abundance. A set of 5 negative binomial GLMM models was generated for the large foraging range frugivores, and a set of 19 zero-inflated Poisson GLMM models was generated for all granivores combined, all bark-gleaning insectivores combined, small foraging range sallying insectivores, and terrestrial insectivores combined, and a set of four negative binomial GLMM models was generated for small foraging range foliage-gleaning insectivores. The top-ranked models

Table 3. Coefficient estimates of five focal species models and 95% lower confidence intervals (95% LCI) and upper confidence intervals (95% UCI) for variables in the best-fitted models. The sign (+/-) of coefficient estimates refers to the influence direction of that variable on abundance, while the number refers to the magnitude of the influence.

Focal Species	**Estimate**	**95% LCI**	**95% UCI**
Blossom-headed parakeet [as]			
Count model			
year	-0.36	-0.48	-0.23
Zero model			
rain	0.70	0.34	1.06
Common woodshrike [as]			
Count model			
year	0.17	0.10	0.24
Zero model			
Survey time	-0.35	-0.48	-0.22
Golden-fronted leafbird [ah]			
Count model			
year	0.15	-0.001	0.31
Zero model			
Survey time	0.17	0.03	0.31
Rufous treepie [ah]			
Count model			
forest disturbance	-0.19	-0.42	0.04
Zero model			
Survey time	0.24	0.08	0.40
White-browed fantail [bh]			
Count model			
year	-0.25	-0.36	-0.14

[ah] zero-inflated Poisson regression models with random habitat
[bh] negative binomial regression models with random habitat
[as] zero-inflated Poisson regression models with random season

for abundances for each guild are listed in Supplementary Table 2, showing the best-supported candidate models for all tested avian guilds. The estimated beta coefficients from the top models are listed in Table 2.

The models with the strongest trends for guilds were for frugivores, granivores, and terrestrial insectivores, which were affected by survey year and forest disturbance index, showing a lower abundance in areas of higher forest disturbance and in the later years of the study period compared to the earlier years. For bark-gleaning insectivores (woodpeckers) forest disturbance appeared to affect abundance: their abundances were lower with higher levels of forest disturbance, but did not show long-term declines across the study period. However, we found no evidence these factors affected foliage-gleaning insectivore and sallying insectivore abundance (Table 2).

For four of the five focal species, a set of 19 zero-inflated Poisson GLMM models was generated, while a set of four negative binomial GLMM models was generated for the remaining focal species (white-browed fantail). The top-ranked models for relative abundances of all focal species included year and/or level of forest disturbance (Supplementary Table 3). Estimated beta coefficients from the top models for all focal species indicated that blossom-headed parakeet and white-browed fantail relative abundance declined over the study years, i.e., beta coefficients were negative and 95% confidence intervals (CI) did not overlap zero (Table 3). Common woodshrike significantly increased over the course of the study period while golden-fronted leafbird also likely increased over time, although the 95% CI marginally overlapped zero (Table 3). Rufous treepie abundance did not significantly change over the study period (Table 3).

The relative abundance of frugivores fluctuated between 1999–2001 and 2015–2016. However, they appeared to be notably lower from 2017 to 2020. Granivore relative abundance increased year by year in the first three years of the study period and then declined steadily from 2015 to 2020. Woodpecker relative abundance increased in 2015 compared to the first three years of the study (1999–2001) and then fluctuated between 2015 and 2020. Sallying insectivores increased slightly in 2015 and remained steady from 2017 to 2020. The relative abundance of the terrestrial insectivores did appear to slightly decline over time. However, foliage-gleaning insectivores appeared to increase in 2015–2020 compared to the first three years of the study (Fig. 4), although our statistical analysis still indicated an overall decline for this guild.

Forest loss and forest disturbance. From 1999 to 2020, approximately 27 km^2 of the forest in Chatthin was lost (~10% of the forest cover present in 1999) (Fig. 1). The forest disturbance index increased during the first survey period from 1999 to 2001, and then fluctuated from 2015 to 2020 (Supplementary Fig. 1). During the 22-year study period several forms of human disturbance were noted. As indicated below, these included a large increase in the number of villages surrounding the sanctuary from 1999 to 2020 (Supplementary Fig. 2), frequent tree and branch cutting for fuelwood and charcoal, and non-timber forest product collection (Supplementary Fig. 3).

The 2017 camera trap survey documented that local people, livestock, and domestic dogs were accessing the area in an unregulated manner (Supplementary Fig. 4). The surveys of villages around Chatthin indicated there was a variety of collection activities for forest products depending on market demand, including *Celtis cinnamomea* (fruit), *Eugenia* species (fruit, seeds and seedlings), and various orchids (Forest Department, 2019; Friends of Wildlife, unpublished data) (Supplementary Fig. 3, Supplementary Fig. 5). According to Allendorf et al. (2012), in 2003, 87% of the households relied on the sanctuary for fuelwood, while 40% of the households used it for cattle grazing (Khant et al., 2018). Data collected from Chatthin also found that the diversity

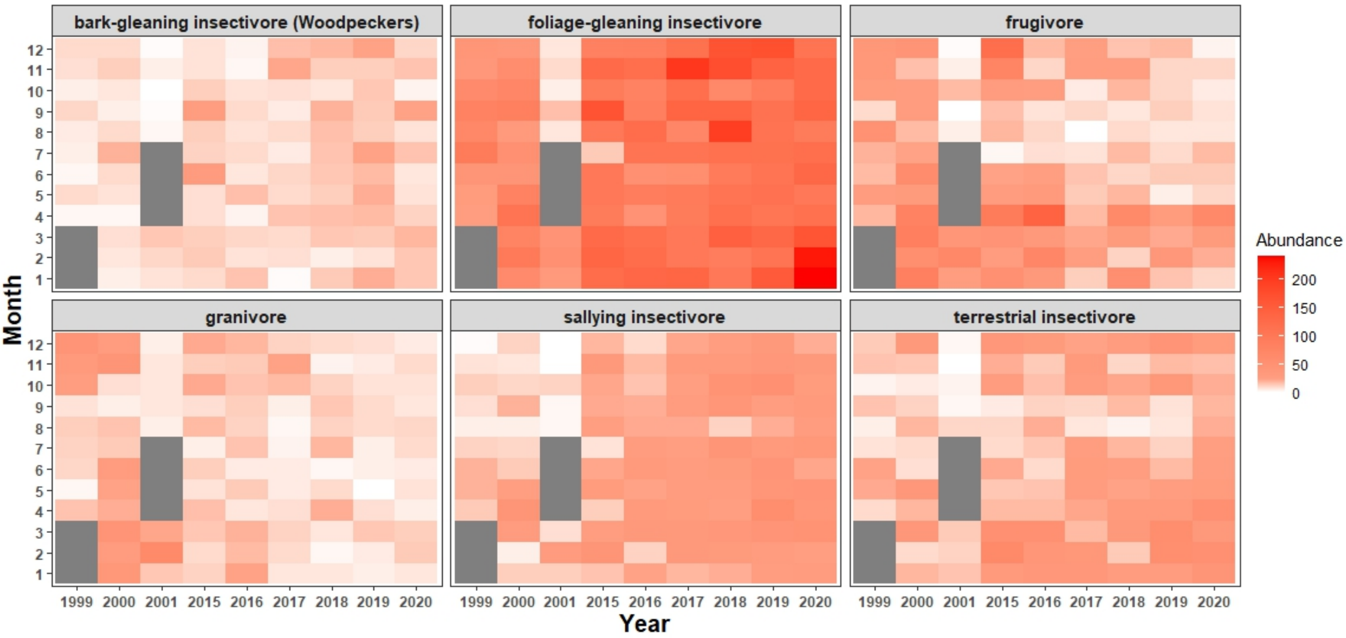

Fig 4. Seasonal (y-axis) and long-term trends of abundances of six avian guilds in Chatthin Wildlife Sanctuary from 1999 to 2020. Redder = higher abundances, white/paler = lower abundances, and grey = no data.

of tree and grass species has declined (Khant et al., 2018). We also observed bird poaching, extensive cattle grazing, and annual anthropogenic fires. In addition, there was a significant increase in flooding and flood-associated habitat (Khant et al., 2018) which may have affected as much as 60% of the sanctuary due to the construction of a dam near the sanctuary in 2001 (Aung et al., 2004; Khant et al., 2018). Earlier studies in Chatthin documented an increase in human use from the 1990s to early 2000s (Aung et al., 2004; Songer, 2006), consistent with the increase in human use that we perceived at least during the last six years of the study for which we have direct observational data (Fig. 5), suggesting that human use in the sanctuary has probably steadily increased over the past 30 years.

DISCUSSION

In our study, changes in avian communities were monitored over a 22-year period and long-term declines in relative abundances were observed in three of the six targeted feeding guilds as well as in two of five focal species (blossom-headed parakeet and white-browed fantail). Overall, some of the declines were consistent with effects of human disturbance and likely emblematic of impacts on dry forests occurring in other parts of Southeast Asia where human use is minimally regulated.

Avian feeding guilds. Among the six guilds, frugivores and granivores showed long-term declines. In the frugivores, five out of seven species were parakeets; in the granivores, doves (Columbidae) were the most dominant family (Supplementary Table 1). It is likely that the declines in parakeets were associated with capture for the local and regional cage bird trade (Khaing, 2019), whereas for the granivores, declines may be attributable to the hunting of doves, which is a common practice in Myanmar (Platt et al., 2012; Pritchard et al., 2019). One of our focal species, the blossom-headed parakeet, is also known to be declining in Shwesettaw Wildlife Area, central Myanmar (Khaing, 2019) presumably because of similar factors as observed in our study (Supplementary Fig. 3). In addition to hunting, granivore declines could be directly related to the decrease in grassy habitat recorded in the study area along with a simultaneous loss of ground cover and increased flooding (Khant et al., 2018).

Two guilds of insectivores (woodpeckers and terrestrial) were negatively associated with the forest disturbance index during the study period although only the terrestrial insectivores showed long-term declines. While it is unclear as to whether the fire regime has significantly changed in Chatthin over the study period (Khant et al., 2018), there appeared to be more evidence for increased wet season flooding, impacting as much as 60% of the sanctuary (Songer, 2006; Khant et al., 2018), combined with reduced amounts of ground cover (Khant et al., 2018). Interestingly, relative woodpecker abundances declined in response to the forest disturbance index (Supplementary Fig. 4) but did not decline across the study period (Fig. 4), suggesting a possible short-term effect of habitat disturbance on woodpeckers, whose populations may be capable of returning relatively quickly even if the habitat had been degraded. Therefore, it is possible that the disturbance captured by our forest disturbance index may have caused woodpeckers to temporarily move away from sites of disturbance (e.g., areas affected by tree cutting), but that these disturbances did not lead to permanent emigration or reduced survival of woodpeckers as a whole. In contrast, terrestrial insectivores declined over the course of our study (Table 2). We are limited in our understanding of the specific habitat requirements of terrestrial insectivores. However, it is likely that increased cattle grazing and flooding (leading to increased seasonally flooded deciduous vegetation) would change the understory structure, leading to

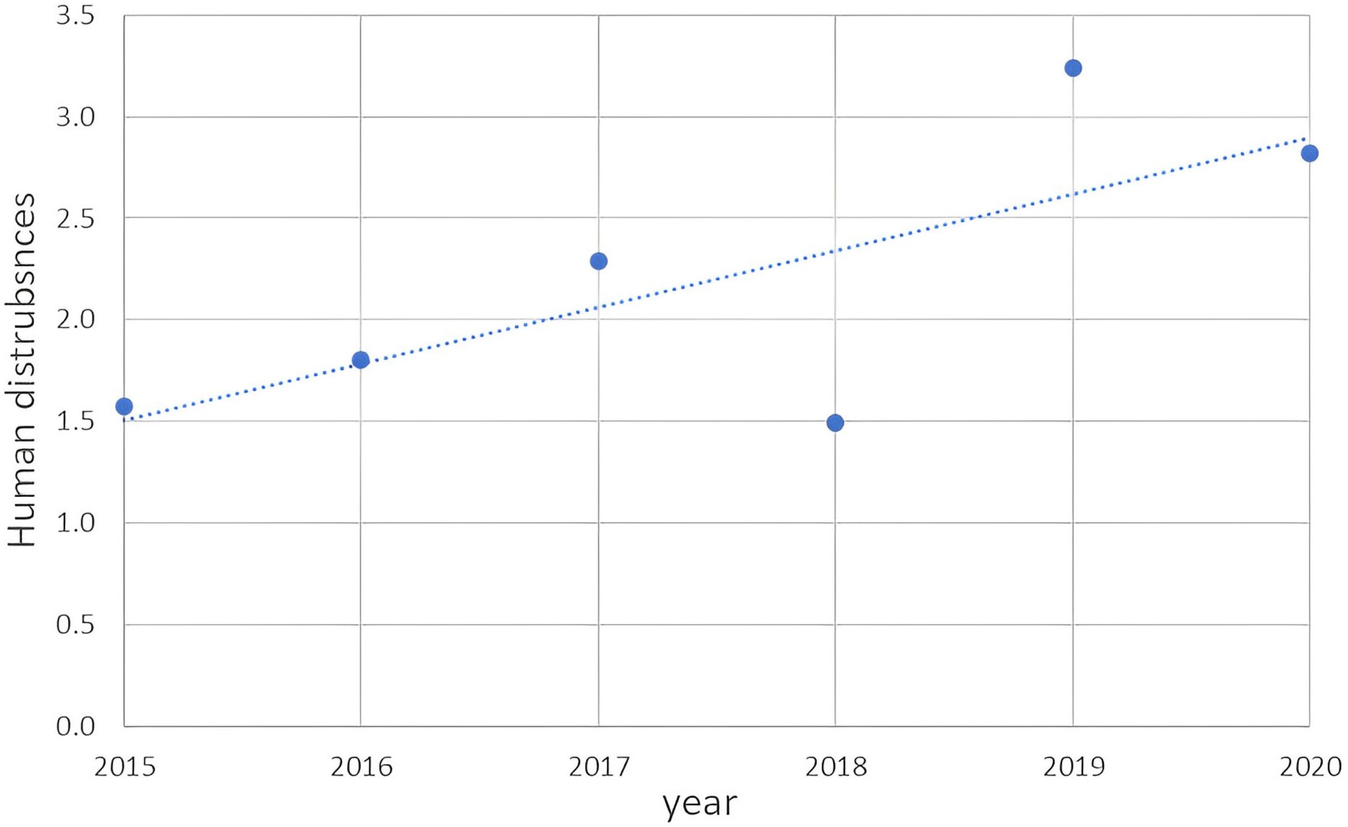

Fig. 5. Human detections along five, two-kilometre transcects, one each in five habitats/subtypes (young dipterocarp forest, flooded dipterocarp forest, mixed deciduous forest, mature dipterocarp forest, and wetlands) per month in Chatthin Wildlife Sanctuary. Effort = number of habitat subtypes * total months sampled per year * 2 km, X-axis= year, Y-axis= human disturbance index (number of human detections per one unit of effort).

decreases in shrubs and grasses as documented in Chatthin (Khant et al., 2018). Our observational data and previous data from Chatthin indicate that grazing occurred in >50% of the sanctuary (Songer, 2006) and likely increased over the course of our 22-year study period. Although in dry forests in other continents grazing impacts on birds were shown to be negative (Dardanelli et al., 2022) or complex, species-specific, and scale dependent (Neilly & Schwarzkopf, 2019), these dynamics may well be different in Myanmar. In summary, while it is likely that overgrazing would have been detrimental to terrestrial insectivores in Chatthin, we were unable to effectively assess the intensity of grazing and its effects on the bird community, which remains an urgent topic for further research.

The abundances of foliage-gleaning insectivores and sallying insectivores did not change significantly over the course of our study period. Foliage-gleaning insectivores appeared to be largely unaffected or even increased, as is the case with the common woodshrike, while sallying insectivores such as drongos also appeared to be unaffected as they are relatively well adapted to a variety of open canopy habitats. Qualitative examination of some other common DDF generalist insectivores also suggested no declines. These included species such as common hoopoe, ashy drongo, Asian paradise flycatcher, black-naped monarch, common flameback, chestnut-bellied nuthatch, common iora, common tailorbird, and small minivet (see Supplementary Table 1 for scientific names).

Focal species. Two of the focal species showed decreases, blossom-headed parakeet and white-browed fantail. While the parakeet decline is most likely linked to the cage bird trade, the decline through time of the white-browed fantail, a foliage-gleaning insectivore, was not seemingly associated with any particular habitat variable. The species might have particularly narrow habitat preferences. Patterns explaining the presence and abundance of these two declining species might need to be investigated over larger time scales, perhaps utilising multiple sites with a gradient of different vegetation conditions. Finally, the rufous treepie abundance did not change significantly during our study period. This treepie was noted as being able to persist in a variety of forest types (Khan et al., 2021) which may at least partly account for its lack of a decline in Chatthin.

SUPPLEMENTARY MATERIAL

The bird species list for the study area, candidate models for predicting guilds, and candidate models for predicting focal species are provided in Supplementary Tables S1 to S3 respectively. A figure showing the amount of forest loss within 300 meters of survey points based on remote sensing, a map showing the location of villages that depend on Chatthin Wildlife Sanctuary for forest products, photos showing examples of human impacts observed in Chatthin, a map of observations of people and domestic animals in Chatthin based on a camera trap survey, and a map highlighting areas of local use within Chatthin are provided in Supplementary Figs. 1 to 5 respectively.

All supplementary material is provided as an attachment within this PDF document. They can be accessed via the Attachment panel (the 'paperclip' icon; accessed by View>Show/Hide>Navigation Pane>Attachments). For optimal compatibility, please use the Adobe Acrobat Reader (free-to-use; download here).

ACKNOWLEDGEMENTS

We thank King Mongkut's University of Technology Thonburi, Thailand, and Thailand Science Research and Innovation (TSRI) (Project ID: 42902), for supporting MMS. We would also like to thank the Smithsonian Institution as well as U Nay Myo Shwe for supporting previous fieldwork and U Myint Aung (Friends of Wildlife) for support and sharing knowledge about the study area. We are also thankful to Chatthin Wildlife Sanctuary personnel who assisted and supported the data collection. Finally, particular thanks to two anonymous reviewers and Frank Rheindt for their constructive comments on the later stages of this manuscript.

LITERATURE CITED

Allendorf TD, Aung M & Songer M (2012) Using residents' perceptions to improve park-people relationships in Chatthin Wildlife Sanctuary, Myanmar. Journal of Environmental Management, 99: 36–43.

Atikah SN, Yahya MS, Norhisham AR, Kamarudin N, Sanusi R & Azhar B (2021) Effects of vegetation structure on avian biodiversity in a selectively logged hill dipterocarp forest. Global Ecology and Conservation, 28: e01660.

Aung M, Swe KK, Oo T, Moe KK, Leimgruber P, Allendorf T & Wemmer C (2004) The environmental history of Chatthin Wildlife Sanctuary, a protected area in Myanmar (Burma). Journal of Environmental Management, 72(4): 205–216.

BirdLife International (2000) Threatened Birds of the World. Lynx Editions and Birdlife International, Barcelona and Cambridge, 852 pp.

Bowler DE, Heldbjerg H, Fox AD, de Jong M & Böhning-Gaese K (2019) Long-term declines of European insectivorous bird populations and potential causes. Conservation Biology, 33(5): 1120–1130.

Brooks ME, Kristensen K, van Benthem KJ, Magnusson A, Berg CW, Nielsen A, Skaug HJ, Maechler M & Bolker BM (2017) glmmTMB balances speed and flexibility among packages for zero-inflated generalized linear mixed modeling. The R Journal, 9(2): 378–400.

Burnham K & Anderson D (2002) Model Selection and Multimodel Inference: A Practical Information-Theoretic Approach. Springer, New York, USA, 488 pp.

Chapman CA, Chapman LJ, Struhsaker TT, Zanne AE, Clark CJ & Poulsen JR (2005). A long-term evaluation of fruiting phenology: importance of climate change. Journal of Tropical Ecology, 21(1): 31–45.

Dardanelli S, Calamari NC, Canavelli SB, Barzan FR, Goijman AP & Lezana L (2022). Vegetation structure and livestock grazing intensity affect ground-foraging birds in xerophytic forests of Central-East Argentina. Forest Ecology and Management, 521: 120439.

França LF, Figueiredo-Paixão VH, Duarte-Silva TA & Santos KBD (2020). The effects of rainfall and arthropod abundance on breeding season of insectivorous birds, in a semi-arid neotropical environment. Zoologia, 37: e37716.

Forest Department (2019) Chatthin Wildlife Sanctuary Management Plan, 2019–20 Fiscal Year to 2023–24 Fiscal Year, Myanmar, 29 pp.

Gelman A & Hill J (2006) Data analysis using regression and multilevel/hierarchical models. Cambridge University Press, Cambridge, 648 pp.

Gray TN & Phan C (2011) Habitat preferences and activity patterns of the larger mammal community in Phnom Prich Wildlife Sanctuary, Cambodia. The Raffles Bulletin of Zoology, 59(2): 311–318.

Haq RU, Eiam-Ampai K, Ngoprasert D, Sasaki N & Shrestha RP (2018). Changing landscapes and declining populations of resident waterbirds: A 12-year study in Bung Boraphet Wetland, Thailand. Tropical Conservation Science, 11: 1940082917750839.

Hartig F (2019) Package 'DHARMa': residual diagnostics for hierarchical (multi-level/mixed) regression models. R package version 0.1.5. https://cran.r-project.org/web/packages/DHARMa/index.html (Accessed 10 November 2022).

Hendershot JN, Smith JR, Anderson CB, Letten AD, Frishkoff LO, Zook JR & Daily GC (2020) Intensive farming drives long-term shifts in avian community composition. Nature, 579(7799): 393–396.

Inger R, Gregory R, Duffy JP, Stott I, Voříšek P & Gaston KJ (2015) Common European birds are declining rapidly while less abundant species' numbers are rising. Ecology Letters, 18(1): 28–36.

Khaing N & Mitloehner R (2014) Structure and composition of dry deciduous forests in Central Myanmar. Annual Conference of Forest Research Institute, Nay Pyi Taw, Myanmar, 11 pp.

Khaing TT (2019) Parakeet Trade in Shwesettaw Wildlife Area, Minbu Township, Magway Region, during Pagoda Festival. International Journal of Innovative Science and Research Technology, 4(8): 252–255.

Khamcha D, Corlett RT, Powell LA, Savini T, Lynam AJ & Gale GA (2018) Road induced edge effects on a forest bird community in tropical Asia. Avian Research, 9(1): 1–13.

Khan RA, Ullah Z, Zaman IU, Khan MS, Mahmood S, Akhtar N & Hussain SS (2021). Population distribution and habitat analysis of Rufous treepie (*Dendrocitta vagabunda*) in Abbottabad, Pakistan. Brazilian Journal of Biology, 83: e247018.

Khant HE, Bhumpakphan N & Kutintra U (2018) Change in the habitat of indaing and the future of Eld's deer in Chatthin Wildlife Sanctuary, Myanmar. Journal of Tropical Forest Research, 2(2): 1–15.

King DI & Rappole JH (2001). Mixed-species bird flocks in dipterocarp forest of north central Burma (Myanmar). Ibis, 143(4): 380–390.

Mag Z & Ódor P (2015) The effect of stand-level habitat characteristics on breeding bird assemblages in Hungarian temperate mixed forests. Community Ecology, 16(2): 156–166.

Mahmon NA, Ya'acob N & Yusof AL (2015) Differences of image classification techniques for land use and land cover classification. Proceedings: IEEE 11th International Colloquium on Signal Processing & its Applications (CSPA), pp. 90–94.

Miles L, Newton AC, DeFries RS, Ravilious C, May I, Blyth S & Gordon JE (2006) A global overview of the conservation status of tropical dry forests. Journal of Biogeography, 33(3): 491–505.

Nature and Wildlife Conservation Division (2020) Annual Eld's Deer Census, Chatthin Wildlife Sanctuary, Forest Department, Myanmar, pp. 10–11.

Neilly H & Schwarzkopf L (2019) The impact of cattle grazing regimes on tropical savanna bird assemblages. Austral Ecology, 44(2): 187–198.

Nguyen TT & Baker PJ (2016) Structure and composition of deciduous dipterocarp forest in Central Vietnam: patterns of species dominance and regeneration failure. Plant Ecology & Diversity, 9(5–6), 589–601.

Nuttall MN, Griffin O, Fewster RM, McGowan PJ, Abernethy K, O'Kelly H & Bunnefeld N (2022) Long term monitoring of wildlife populations for protected area management in Southeast Asia. Conservation Science and Practice, 4(2): e614.

Platt SG, Platt K, Naing TZ, Meng H, Ko WK, Lin N & Rainwater TR (2012). Birdlime in Western Myanmar: Preparation, use, and conservation implications for an endemic bird. Ethnobiology Letters, 3: 68–75.

Portillo-Quintero CA & Sánchez-Azofeifa GA (2010). Extent and conservation of tropical dry forests in the Americas. Biological Conservation, 143(1): 144–155.

Pritchard B, Vicol M, Rammohan A & Welch E (2019). Studying home gardens as if people mattered: Why don't food-insecure households in rural Myanmar cultivate home gardens? The Journal of Peasant Studies, 46(5): 1047–1067.

R Core Team (2020) R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/ (Accessed 10 November 2022).

Rappole JH, Shwe NM & Aung M (2011) Seasonality in avian communities of a dipterocarp monsoon forest and related habitats in Myanmar's central dry zone. In: McShea WJ, Davies SJ & Bhumpakphan N (eds.) Ecology and Conservation of Dry Tropical Forests in Asia. Smithsonian Scholarly Press, Washington, DC, pp. 131–146.

Ratnam J, Tomlinson KW, Rasquinha DN & Sankaran M (2016). Savannahs of Asia: antiquity, biogeography, and an uncertain future. Philosophical Transactions of the Royal Society B: Biological Sciences, 371(1703): 20150305.

Robson C (2015) Birds of Southeast Asia: Concise Edition. Bloomsbury Publications, London, 304 pp.

Rosenberg KV, Dokter AM, Blancher PJ, Sauer JR, Smith AC, Smith PA & Marra PP (2019) Decline of the North American avifauna. Science, 366(6461): 120–124.

Şekercioğlu ÇH, Mendenhall CD, Oviedo-Brenes F, Horns JJ, Ehrlich PR & Daily GC (2019) Long-term declines in bird populations in tropical agricultural countryside. Proceedings of the National Academy of Sciences, 116(20): 9903–9912.

Shwe NM (2014) Temporal changes in habitat use by avifauna in seasonally dry tropical forest. Unpublished Master's Thesis. King Mongkut's University of Technology Thonburi, Thailand, 12 pp.

Siyum ZG (2020) Tropical dry forest dynamics in the context of climate change: syntheses of drivers, gaps, and management perspectives. Ecological Processes, 9(1): 1–16.

Songer M, Aung M, Senior B, DeFries R & Leimgruber P (2009) Spatial and temporal deforestation dynamics in protected and unprotected dry forests: a case study from Myanmar (Burma). Biodiversity and Conservation, 18(4): 1001–1018.

Songer MA (2006) Endangered dry deciduous forests of Upper Myanmar (Burma): a multi-scale approach for research and conservation. Unpublished PhD Thesis. University of Maryland, College Park, USA, 143 pp.

Toscano BJ, Gownaris NJ, Heerhartz SM & Monaco CJ (2016) Personality, foraging behavior and specialization: integrating behavioral and food web ecology at the individual level. Oecologia, 182(1): 55–69.

Verga EG, Sánchez Hümöller, HL, Peluc SI, & Galetto L (2017). Forest fragmentation negatively affects common bird species in subtropical fragmented forests. Emu - Austral Ornithology, 117(4): 359–369.

Wang C & Myint SW (2016) Environmental concerns of deforestation in Myanmar 2001–2010. Remote Sensing, 8(9): 728.

Watson JE, Whittaker RJ & Freudenberger, D (2005). Bird community responses to habitat fragmentation: how consistent are they across landscapes? Journal of Biogeography, 32(8): 1353–1370.

Wildlife Conservation Society (2013) Myanmar Biodiversity: Conservation, Investment, Vision. Wildlife Conservation Society, Yangon, Myanmar, 16 pp.

Wilman H, Belmaker J, Simpson J, de la Rosa C, Rivadeneira MM & Jetz W (2014) Elton Traits 1.0: Species level foraging attributes of the world's birds and mammals. Ecology, 95(7): 2027–2027.

Wohlfart C, Wegmann M & Leimgruber P (2014) Mapping threatened dry deciduous dipterocarp forest in South-east Asia for conservation management. Tropical Conservation Science, 7(4): 597–613.

Zuur AF & Ieno EN (2016) Beginner's guide to zero-inflated models with R. Highland Statistics Limited, Newburgh, 411 pp.

Zuur AF, Ieno EN, Walker NJ, Saveliev AA & Smith GM (2009) Mixed effects models and extensions in ecology with R. Springer, NY 10013: USA.