

Lee, D. C., H. I. Mynott, J. A. Jopson, R. A. Santillan, and D. Kerhoas. 2024. Habitat-related bird community responses in northwest Panay, Philippines. *Avian Conservation and Ecology* 19(1):20. <https://doi.org/10.5751/ACE-02635-190120>
Copyright © 2024 by the author(s). Published here under license by the Resilience Alliance. Open Access. CC-BY 4.0

Research Paper

Habitat-related bird community responses in northwest Panay, Philippines

David C. Lee¹ , Holly I. Mynott² , Jack A. Jopson¹, Rhea A. Santillan³ and Daphne Kerhoas^{2,4} 

¹University of South Wales, ²Bristol Zoological Society, ³Philippine Initiative for Environmental Conservation, ⁴Wildfowl and Wetlands Trust

ABSTRACT. The Western Visayas is the most deforested region of the Philippines. The Northwest Panay Peninsula Natural Park, an Important Bird and Biodiversity Area on the island of Panay, is the largest remaining tract of lowland forest in the region supporting a number of bird species of conservation importance. The aim of this study was to quantify the diversity and community composition of bird assemblages across a gradient of habitat disturbance, identifying key species and guilds contributing to habitat-specific community differences and informing conservation management of the park. In 2020, we conducted bird surveys using distance sampling point transects with a 10-minute count period in 4 habitat types: primary forest, secondary forest, plantations, and open areas. We produced measures of taxonomic, species, and functional diversity based on ecological guilds described by diet, body size and feeding stratum, and endemism, and used generalized linear modeling to investigate the effect of habitat on bird assemblages. We used analysis of similarities to test for differences in community composition across habitats and similarity of percentages to determine what species and guilds contributed to community dissimilarities. We recorded 61 species, including 6 globally threatened species, with the highest and lowest taxonomic and species richness and endemism in primary forest and open areas, respectively. Primary forest had the largest, positive effect on bird diversity and endemism, while open habitat had a consistent negative effect. Plantations had a small positive effect on functional diversity only. Community dissimilarities across habitats were largely driven by medium general-level and upper-storey frugivores, small lower-storey insectivores, and small lower-storey and general-level omnivores. Secondary forest and plantations may play an important role in supporting at least some of the primary forest bird community. Our findings emphasize the importance of the site's primary forest for regional bird conservation, and we provide recommendations for management of this important lowland forest landscape.

Réponses des communautés d'oiseaux en fonction de l'habitat dans le nord-ouest de Panay, aux Philippines

RÉSUMÉ. Les Visayas occidentales constituent la région la plus déboisée des Philippines. Le Parc naturel du nord-ouest de la péninsule de Panay, zone importante pour la conservation des oiseaux et de la biodiversité sur l'île de Panay, représente la plus grande étendue restante de forêt de plaine dans la région. Elle abrite un certain nombre d'espèces d'oiseaux à fort enjeu de conservation. L'objectif de cette étude était de déterminer la diversité et la composition des cortèges d'oiseaux sur un gradient de perturbation de l'habitat, d'identifier les espèces clés et les guildes écologiques qui contribuent aux différences entre les communautés propres à cet habitat et d'éclairer la gestion de la conservation du parc. En 2020, nous avons réalisé des recensements avifaunistiques en utilisant la méthode de distance sampling par points-transects avec une période de comptage de 10 minutes par point, dans 4 types d'habitats : forêt primaire, forêt secondaire, plantations et zones ouvertes. Nous avons évalué les diversités taxonomique, fonctionnelle et la richesse spécifique sur la base de guildes écologiques définies par le régime alimentaire, la taille corporelle, la strate alimentaire et le caractère endémique des espèces, et nous avons exploré l'effet de l'habitat sur la composition des cortèges d'oiseaux en utilisant un modèle linéaire généralisé. Nous avons utilisé l'analyse des similitudes pour tester les différences de composition des communautés entre les habitats et la similitude des pourcentages pour déterminer quelles espèces et quelles guildes contribuaient aux dissemblances entre les communautés. Sur les 61 espèces contactées, 6 sont menacées au niveau mondial. C'est dans les forêts primaires que l'on trouve la richesse taxonomique et spécifique ainsi que l'endémisme les plus élevés tandis qu'ils sont les plus faibles dans les milieux ouverts. La forêt primaire a exercé l'effet positif le plus important sur la diversité et l'endémisme des oiseaux, tandis que l'habitat ouvert a eu un effet négatif constant. Les plantations ont eu un faible effet positif, mais uniquement sur la diversité fonctionnelle. Les dissemblances entre les communautés d'un habitat à l'autre étaient principalement dues aux guildes frugivores de niveau général moyen et de l'étage supérieur, aux petits insectivores de l'étage inférieur et aux petits omnivores de l'étage inférieur et de niveau général. Il se peut que les forêts secondaires et les plantations jouent un rôle important pour soutenir au moins une partie de la communauté d'oiseaux de la forêt primaire. Nos résultats soulignent l'importance de la forêt primaire du site pour la conservation de l'avifaune de la région, et nous formulons des recommandations pour la gestion de cet important paysage forestier de plaine.

Key Words: ANOSIM; endemism; functional diversity; GLM; guilds; Panay; SIMPER; threatened species

INTRODUCTION

The Philippines is categorized as one of the world's megadiverse countries (Mittermeier et al. 1997), which is reflected in its classification as a global biodiversity hotspot (Myers et al. 2000), and as an important centre of bird endemism (Convention on Biological Diversity 2020). It has one of the highest proportions of globally threatened bird species in the world: 13.6% of 735 species (IUCN 2023), while 29.1% of 247 country endemics are globally threatened with extinction (Brinkman et al. 2023). Despite this, conservation concerns arise from some of the world's highest deforestation rates, having lost 93% of its original forest cover (Hughes 2017), coupled with considerable pressure from unsustainable and illegal wildlife hunting (Gray et al. 2018).

The Philippines' Western Visayas faunal region (Heaney 1985) has experienced the highest rates of deforestation in the country, resulting in a reduction in old growth forest to < 6%, 4%, and 0.05% of the region's main islands of Panay, Negros, and Cebu, respectively (Lorica and Oliver 2006). It is likely that most of the remaining forest in the region is found in the west and northwest of Panay (Lorica and Oliver 2006), and particularly the Northwest Panay Peninsula Natural Park (NPPNP), which is thought to include the largest remaining area of low elevation forest in the Western Visayas (BirdLife International 2024a). Understanding how bird communities respond to habitat degradation and land-use change, and which species may be more sensitive (Gray et al. 2006), remains a conservation priority in the species-diverse and highly threatened forest landscapes of the Western Visayas.

Avian species richness, or taxonomic diversity, tends to decrease along gradients of increasing habitat modification (Schultze et al. 2004), yet species responses to the impacts of anthropogenic land-use change can differ widely (Şekercioğlu et al. 2002, Sodhi et al. 2004) across species and species groups or guilds defined by shared ecological traits (Lambert 1992, Canaday 1996, Schultze et al. 2004, Waltert et al. 2004, Tvardíková, 2010). For example, habitat degradation may differentially impact species based on dietary type and feeding strata (Flynn et al. 2009, Luck et al. 2012, Corbelli et al. 2015), while larger species may be more impacted than smaller species, which has wider ramifications relating to home range size, lifespan, population viability, and the provision of ecosystem services (Cofre et al. 2007, Flynn et al. 2009, Vandewalle et al. 2010, Luck et al. 2012, Corbelli et al. 2015).

The efficacy of taxonomic diversity as a measure of community responses to, or recovery from habitat degradation (Cianciaruso et al. 2009) or the state and resilience of a community's ecological functionality (Petchey et al. 2004, Mouillot et al. 2013) is limited by the fact that all species are considered equally (Cianciaruso et al. 2009). Incorporating into diversity measures information on the functional traits represented within a community can demonstrate how that community may respond to habitat change (Cosset and Edwards 2017). Understanding how these changes influence functional diversity, the degree of (trait-based) ecological differences among species within a specified community (Tilman 2001), has wider implications for the functionality of a system and its processes (Tilman 2000, Loreau et al. 2001). Additionally, although highest species diversity may be evidenced in non-equilibrium tropical forest communities subject to perturbations and an absence of disturbance leading to equilibrium communities of lower diversity, as posited by the

intermediate disturbance hypothesis (Connell 1978), this implies all species are equivalent, so species diversity may not represent absolute conservation value (Sheil et al. 1999). Although conservation action needs to consider how disturbance events drive species diversity in tropical forests (e.g., Sheil and Burslem 2003), which may run counter to conservation goals, functional diversity can aid in conservation planning, particularly when managing or attempting to restore functioning ecosystems (Cadotte et al. 2011).

Although previous research in the Western Visayas has focused on the responses of single bird species (Jakosalem et al. 2013, Mynott et al. 2020, 2021, Paguntalan et al. 2023) and non-avian communities (Alcala et al. 2004, Peque and Hölscher 2014) to habitat alteration, we look to address the significant knowledge gap in understanding habitat-related responses of tropical forest bird communities in this threatened faunal region. We use three measures of community diversity, taxonomic (species richness), species, and functional, to quantify patterns in bird community assemblages across a habitat gradient of primary forest, secondary forest, plantation, and open areas in NPPNP. Understanding such responses to habitat disturbance is essential for identifying priorities and applying limited conservation resources that improve habitat quality or guide habitat connectivity most effectively (Whytock et al. 2018), and this novel work provides a significant baseline for informing species and community-focused recommendations for conservation management decision making in the park. We anticipate this will provide empirical support to wider discussions on conserving forest biodiversity in the Western Visayas.

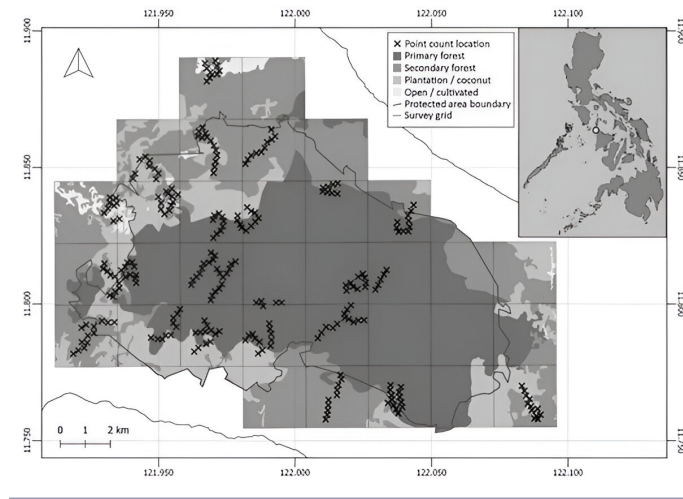
METHODS

Study site

With consent from the Protected Area Management Board, this study took place from January to April 2020 in the Northwest Panay Peninsula Natural Park (NPPNP hereafter; central coordinates 11° 49' 31" N, 121° 58' 02" E; Fig. 1), which is situated in the provinces of Aklan and Antique on the island of Panay, the Philippines. The NPPNP was designated a protected area in 2002 under the National Integrated Protected Areas System (NIPAS) Act, Philippines (UNEP-WCMC 2022) and comprises 120 km² of tall dipterocarp forest, limestone karst forest, lower montane forest, and bamboo forest across an elevational range of 27 to 915 m (BirdLife International 2024a). With up to 5000 ha of old growth forest remaining, it is one of the largest contiguous protected lowland forests in the Western Visayas (Key Biodiversity Areas Partnership 2022) and probably the most extensive lowland forest remaining in the Negros and Panay endemic bird area (EBA; Stattersfield et al. 1998). Within NPPNP, there has been some legal land conversion for farming on plots specified before the protected area was established, while illegal logging and slash-and-burn agriculture (*kaingin*) persist alongside the collection of forest products and illegal hunting (Kerhoas et al. 2019, Mynott et al. 2020, BirdLife International 2024a, Key Biodiversity Areas Partnership 2022).

The NPPNP is an Important Bird and Biodiversity Area (IBA; BirdLife International 2024a) due to the presence of five globally threatened species: Negros Bleeding-heart (*Gallinolumba keayi*; CR), Writhe-billed Hornbill (*Rhabdotorrhinus waldeni*; CR), Visayan Hornbill (*Penelopides panini*; EN), White-winged

Fig. 1. Map of the Northwest Panay Peninsula Natural Park, Panay, Philippines. The park's location on Panay is highlighted in white in the thumbnail of the Philippines.



Cuckooshrike (*Analisoma ostenta*; VU), and Green-faced Parrotfinch (*Erythrura viridifacies*; VU; BirdLife International 2024b). These species also trigger the park's qualification as a key biodiversity area (KBA; Key Biodiversity Area Partnership 2022). The IBA designation pre-dates the taxonomic overhaul of del Hoyo and Collar (2014, 2016), which now includes the presence of the additional threatened species of Yellow-faced Flameback (*Chrysocolaptes xanthocephalus*; EN) and Negros Scops-owl (*Otus nigrorum*; VU), while the under-recorded Negros Jungle Flycatcher (*Vauriella albigularis*; EN) is also present (eBird 2023). The combination of the site's regionally important forest cover, existing pressures, and presence of several key species make NPPNP a conservation priority (BirdLife International 2024a).

Sampling design and bird surveys

We divided NPPNP into a grid of 2.5 km * 2.5 km cells ($n = 33$) and randomly selected 24 of these cells to survey (72.7% of the park; Fig. 1). However, once on site, the terrain of three of the selected cells was found to be inaccessible. In these instances, the nearest accessible grid cell was surveyed instead (Mynott et al. 2021). Each cell contained between one and four line transects of varying distance (0.3 to 2.5 km, $\bar{x} = 2.3 \text{ km} \pm 0.53 \text{ SD}$) that were positioned randomly, or along narrow trails where the terrain was particularly difficult. Any trails used were $< 1 \text{ m}$ wide and therefore were not considered to have affected the representativeness of the data (Cornils et al. 2015) or violated any sampling design assumptions (Bibby et al. 2000). Nine cells included transects positioned in part or entirely in adjacent habitat outside the park's boundary.

We positioned survey points $\geq 200\text{--}250 \text{ m}$ apart along each transect to minimize the chance of double-counting individual birds from adjacent points (Bibby et al. 1998, Marsden 1999). We completed 10-minute distance sampling point transect surveys with no settling down period and recorded all birds seen or heard (Lloyd et al. 2000, Lee and Marsden 2008). We used Kennedy et al. (2000) to aid with bird identification in the field and followed the taxonomy of Clements et al. (2023).

Surveys were completed between 08:00–18:00 and, although bird detectability may be (aurally) higher in early morning/late afternoon (Lynch 1995, Bibby et al. 2000), survey points were visited throughout the day in each location and habitat, minimizing any temporal detectability bias while potentially capturing fine-scale variation in species availability for detection and their detectability (Verner and Ritter 1986). We did not complete any surveys in periods of high wind or rainfall (Marsden and Pilgrim 2003). When possible, we used a laser rangefinder to record radial distances to individual birds or to the central location of a species group. However, if a clear line of sight was unobtainable, we estimated distance to a bird detection. Pre-survey training in distance estimation ensured radial distances were not consistently over- or under-estimated (Bibby et al. 1998). We recorded the group size of each detection. If we could not confirm this visually, we used a habitat-specific mean group size calculated from visual detections of the species instead (Lee and Marsden 2008). All cells were visited once, with surveys repeated in seven cells from mid-February to the start of April (29.2%; access restrictions in response to the coronavirus pandemic prevented repeats in the remaining 17 cells). In total, we surveyed 278 points, of which 80 were repeated, across an elevational range of 17 to 907 m.

We classified the habitat type at each point into one of four categories based on floristic composition and structural attributes, local knowledge of plot histories, and supported by satellite imagery (PhilGIS 2020): primary (old growth forest; $n = 111$), secondary (previously disturbed) forest ($n = 105$), plantation ($n = 41$), and open habitat ($n = 21$). Habitat categorization was supported by plot-based surveys as described in Mynott et al. (2021). Plantations had a more open canopy than forest habitat, mainly consisting of coconut palm trees with the occasional presence of regenerating forest and some newly planted areas of young mahogany. Open habitats had low canopy cover ($< 11\%$), were dominated by scrub, and generally lacked trees with $\geq 25 \text{ cm}$ diameter at breast height. In this study, we considered change between primary forest, secondary forest, plantation, and open habitat as a gradient of habitat disturbance or degradation.

Statistical analyses

For overall site diversity estimates, we used all species records, including aerial detections (e.g., *Collocalia* species), with an unlimited detection radius. For all other (habitat-focused) analyses, we excluded aerial detections and right truncated bird detections at a radial distance of 50 m to ensure species records were attributed to a habitat type spatially commensurate with the respective survey point. For each habitat type, we used species richness (S_p) as a measure of taxonomic diversity, the Shannon-Wiener (H') index for species diversity, and two indices for functional diversity (FD): functional dispersion index (FD_{Dis}), which reflects deviations in species-level trait values from a functional centroid, and Rao's quadratic entropy (Q), which considers the variance of species-species trait dissimilarities (Laliberté and Legendre 2010). We selected these two measures of FD because they allow for the inclusion of qualitative traits and are unaffected by species richness (Laliberté and Legendre 2010). Greater differences in species traits within communities are reflected in larger FD (Petchey and Gaston 2002). We also generated a simple endemism index based on the proportion of

endemic species, from the total pool of endemic species recorded in the study, detected at each point. We used Brinkman et al. (2023) for defining species' endemism status.

We estimated species richness using iNEXT software (Chao et al. 2016) as a sample size-based inter- and extrapolation for comparisons across habitat-specific communities from incomplete sample data. We produced rarefaction and extrapolation sampling curves for each habitat type using species incidence data, which are less sensitive than abundance data to aggregations of individuals (Chao et al. 2020), while also minimizing habitat-based detectability differences associated with relative abundances. We set the confidence interval (CIs) at 84% (for a Type I error rate of $p < 0.05$; Payton et al. 2003) for each curve with 999 bootstrap replications and compared diversity estimates for common sample sizes and sample coverages with \hat{C} as an estimator of sample coverage in each habitat (Chao and Jost 2012). We calculated habitat-specific species diversity indices in PRIMER 7 (Clarke and Gorley 2006).

We extracted species-level foraging attributes from Wilman et al. (2014), and references therein, to classify species into guilds based on the functional traits of body size, foraging location, and diet. For diet, we assigned species to the majority type reported (frugivorous, granivorous, insectivorous, omnivorous, or vertivorous). We allocated species to their most frequently recorded foraging strata, i.e., ground, understorey, understorey-midstorey (lower storeys), midstorey-canopy (upper storeys), general-level (all vegetation levels), or aerial. We classified body size into three categories: small (< 25 g), medium (25–125 g), and large (> 125 g). In combination, this resulted in 31 functional guilds (Appendix 1). Using these functional traits alongside a species incidence matrix, we calculated FD_{is} and Q using the FD package (Laliberté et al. 2014) in R (R Development Core Team 2017).

We used the $glm2$ package (Marschner 2011) in R to construct generalized linear models (GLMs) for comparing point values of species richness and diversity, FD , and endemism (response variables) between habitat types. We used a Poisson distribution and log link function for species richness, gaussian distribution and identity link function for species and functional diversities, and endemism. We tested overall model significance using likelihood ratio tests (LRT; $p < 0.05$) and used the $emmeans$ function, which calculates estimated marginal means among groups given a fitted model (in this case, GLMs), in the $emmeans$ R package (Lenth et al. 2022) to test for significant pairwise differences between habitats.

We used PRIMER 7 (Clarke and Gorley 2006) to analyze patterns in bird communities at the functional guild level within and between the four habitat types; Clarke (1993), and Clarke and Warwick (2001) fully describe these techniques. We weighted bird encounters by point-based survey effort, pre-treated species abundances with a square root transformation to down-weight the importance of the most abundant guilds, and factorized each point transect by habitat type (Clarke and Warwick 2001). Based on a Bray-Curtis coefficient resemblance matrix, we performed an analysis of similarity (ANOSIM) to test for spatial differences between bird communities of different habitat types. We then conducted a similarity of percentages analysis (SIMPER) to

identify which functional guilds (and member species) contributed most to any dissimilarities in community composition between habitat types. We compared the relative abundances of key contributing guilds and selected species (those that contributed $\geq 5\%$ to at least one pairwise habitat comparison) in the different habitats using Kruskal-Wallis testing with p values adjusted using a Bonferroni correction for multiple tests.

RESULTS

Overview

In total, we recorded 61 bird species, including 33 endemic to the Philippines and 7 listed on the IUCN Red List as threatened or near threatened: Negros Bleeding-heart, Writhe-billed Hornbill, Visayan Hornbill, Yellow-faced Flameback, Negros Jungle Flycatcher, White-winged Cuckooshrike, and Pink-bellied Imperial-pigeon (*Ducula poliocephala*; NT). All 7 were recorded in primary forest, although the single Writhe-billed hornbill record was outside the 50 m truncation distance. Four of these species were also recorded in secondary forest and none in plantation or open habitat. Nine species (14.8% of those recorded) were unique to primary forest, including Negros Bleeding-heart and Yellow-faced Flameback (and Writhe-billed Hornbill), and two additional Philippine endemics, Brown-breasted Kingfisher (*Halcyon gularis*) and Yellow-breasted Fruit-dove (*Ptilinopus occipitalis*; Appendix 1). Two species, Plaintive Cuckoo (*Cacomantis merulinus*) and Gray-streaked Flycatcher (*Muscicapa griseisticta*), were only recorded in plantations, and another four were only recorded in open habitat (Chestnut Munia *Lonchura atricapilla*, Common Kingfisher *Alcedo atthis*, Gray Wagtail *Motacilla cinerea*, and Lesser Coucal *Centropus bengalensis*).

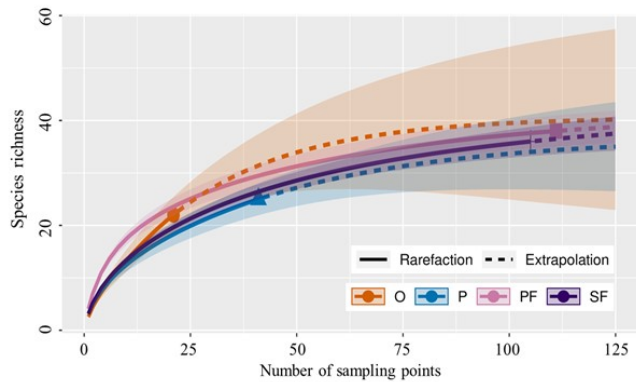
Taxonomic, species, and functional diversity, and endemism

Estimated site species richness was 62 species (62.3 +/- 84% CI [57.4, 67.2]) with 100% sample coverage obtained with 400 points (99.8% coverage with observed survey effort, $n = 278$). Observed species richness was highest in primary forest (38 species, 84% CI [30.3, 45.7]) and decreased across a gradient of increasing habitat disturbance (Table 1, Fig. 2). Shared species richness was higher between habitats least separated along a disturbance gradient, i.e. primary and secondary forest (60.9%), plantation and open habitat (46.9%), than habitats most separated, i.e., open habitat and primary (27.7%) and secondary forests (38.1%).

Table 1. Summary of iNEXT species richness (S_r with 84% CIs) rarefaction and extrapolation for the overall site and each habitat. The expected S_r and required survey effort (number of points) are based on extrapolating to 100% sample coverage.

Level	Sample coverage (%)	Observed S_r	Expected S_r	Required survey effort
Site	99.8	61 (57.7, 64.3)	62.3 (57.4, 67.2)	400
Primary forest	98.5	38 (30.3, 45.7)	41.4 (26.1, 56.6)	234
Secondary forest	97.3	36 (29.7, 42.3)	42.6 (31.8, 53.3)	304
Plantation	91.6	25 (19.5, 30.5)	36.7 (21.4, 52.0)	166
Open habitat	75.0	22 (14.2, 29.8)	40.6 (16.1, 65.1)	115

Fig. 2. Habitat-specific species richness rarefaction and extrapolation sampling curves with 84% confidence intervals (Payton et al. 2003) computed using iNEXT software (Chao et al. 2016). Habitats: open habitat (O), plantation (P), primary forest (PF), and secondary forest (SF).



Including habitat as a factor (rather than a single predictor) improved model fit for community measures except FD_{is} and Q ($LRT_{FD_{is}} = 1.498, p = 0.473$; $LRT_Q = 2.412, p = 0.299$). All habitats had a significant effect on species richness and diversity, and endemism (Table 2). Primary forest was the only habitat that had a positive effect on these measures whereas open habitat had the most negative effect. Species richness was significantly higher in primary forest than secondary forest (Tukey post-hoc pairwise contrasts; $z_{PF, SF} = 3.584, p = 0.002$) and open habitat ($z_{PF, O} = 3.177, p = 0.008$). Primary forest also had significantly higher species diversity than open habitat ($z_{PF, O} = 3.165, p = 0.009$). Endemism (E) was significantly higher in primary forest than in secondary forest ($z_{PF, SF} = 2.812, p = 0.032$) and open habitat ($z_{PF, O} = 3.834, p < 0.001$); mean $E_{PF} = 0.127 \pm 0.079$ SD, $E_{SF} = 0.103 \pm 0.051$, $E_P = 0.100 \pm 0.050$, $E_O = 0.069 \pm 0.051$.

Only primary forest had a significant (positive) effect on both FD_{is} and Q (Table 2); open habitat had a negative effect on both functional diversity measures. Highest functional diversity was described in plantations ($FD_{is} = 0.209 \pm 0.015$; $Q = 1.59 \pm 0.110$), then primary ($FD_{is} = 0.200 \pm 0.011$; $Q = 1.48 \pm 0.077$) and secondary forests ($FD_{is} = 0.206 \pm 0.011$; $Q = 1.44 \pm 0.078$), and finally open habitat ($FD_{is} = 0.174 \pm 0.027$; $Q = 1.25 \pm 0.187$), with no significant pairwise differences. Species richness and diversity, and endemism (E) were highly correlated with FD_{is} and Q across habitats ($r_{p\ SF-FD_{is}} = 0.617$, $r_{p\ H-FD_{is}} = 0.757$, $r_{p\ E-FD_{is}} = 0.610$; $r_{p\ SF-Q} = 0.732$, $r_{p\ H-Q} = 0.855$, $r_{p\ E-Q} = 0.717$; $p < 0.001$ in all cases).

Community compositions

Although overall functional diversity did not differ across habitats, community compositions defined by functional guild were significantly different across the four habitat types ($R = 0.142, p = 0.001$), with differences detected between all pairwise habitat assemblages. Medium general-level frugivores and small lower-storey insectivores were the greatest contributors to community assemblage similarities within each of the four habitats (Table 3). Small omnivores of lower storeys and small

Table 2. Results of generalized linear models for species richness (S_p) and diversity (H'), functional diversity (FD_{is} , Q), and endemism between habitats. Habitats: primary forest (intercept), secondary forest (SF), plantation (P), and open habitat (O).

Response	Factor	Estimate \pm SE	Z	p
Species richness (S_p)	Intercept	4.06 \pm 0.199	20.44	< 0.001
	SF	-0.93 \pm 0.285	-3.26	0.001
	P	-0.94 \pm 0.383	-2.46	0.015
	O	-1.49 \pm 0.499	-2.99	0.003
Species diversity (H')	Intercept	1.11 \pm 0.054	20.22	< 0.001
	SF	-0.19 \pm 0.078	-2.42	0.016
	P	-0.21 \pm 0.105	-2.01	0.045
	O	-0.43 \pm 0.137	-3.17	0.002
Functional dispersion (FD_{is})	Intercept	0.20 \pm 0.011	18.62	< 0.001
	SF	0.01 \pm 0.015	0.42	0.672
	P	0.01 \pm 0.021	1.45	0.651
	O	-0.03 \pm 0.027	-0.94	0.345
Rao's quadratic entropy (Q)	Intercept	1.48 \pm 0.076	19.63	< 0.001
	SF	-0.04 \pm 0.108	-0.36	0.717
	P	0.10 \pm 0.146	0.71	0.479
	O	-0.23 \pm 0.189	-1.24	0.216
Endemism	Intercept	0.13 \pm 0.006	20.95	< 0.001
	SF	-0.02 \pm 0.009	2.81	0.005
	P	-0.03 \pm 0.012	2.30	0.022
	O	-0.06 \pm 0.015	3.83	< 0.001

Table 3. Key contributing functional guilds (> 5% contribution) to intra-habitat community similarities. Overall intra-habitat similarities are in parentheses.

Habitat	Guild [†]	Mean similarity (\pm SD)	Overall contribution (%)
Primary forest (29.9%)	MGeF	9.9 \pm 0.75	33.1
	SLI	9.1 \pm 0.72	30.4
	SLO	4.7 \pm 0.46	15.6
	SGeI	1.8 \pm 0.34	6.0
	MGeI	1.7 \pm 0.31	5.7
Secondary forest (38.4%)	MGeF	14.5 \pm 0.93	37.7
	SLI	11.2 \pm 0.71	29.1
	SLO	9.9 \pm 0.64	25.7
Plantation (40.9%)	SGeO	17.0 \pm 0.94	41.6
	MGeF	15.1 \pm 0.85	36.9
	SLI	5.6 \pm 0.52	13.6
Open areas (25.1%)	MGeF	16.3 \pm 0.87	65.1
	SGeO	4.4 \pm 0.42	17.5
	SLI	3.6 \pm 0.35	14.5

[†] Guild membership: medium general-level frugivores (MGeF) and insectivores (MGeI), small lower-storey insectivores (SLI) and omnivores (SLO), and small general-level insectivores (SGeI) and omnivores (SGeO).

and medium general-level insectivores were important contributors to primary forest community similarities, as were small general-level omnivores to plantation and open habitat similarities.

All guild compositions were at least 67.4% dissimilar between habitats (primary and secondary forest communities), with primary forest and open habitat communities most dissimilar (79.2%; Table 4). Five guilds contributed consistently to these dissimilarities. Of these, the relative abundances of medium general-level insectivores ($H_3 = 15.70, p = 0.001$), small general-

level insectivores ($H_3 = 20.03$, $p < 0.001$), small general-level omnivores ($H_3 = 114.27$, $p < 0.001$), and small lower-storey omnivores ($H_3 = 53.46$, $p < 0.001$) were significantly different across habitats.

On average, eight key species contributed 68.8% to community composition dissimilarities between different habitats (Table 5). All eight species are endemic to the Western Visayas or the Philippines. Visayan Bulbul (*Hypsipetes guimarasensis*), Visayan Tailorbird (*Orthotomus castaneiceps*), and Magnificent Sunbird (*Aethopyga magnifica*) were the three most important species for explaining assemblage dissimilarities. The relative abundances of four species were significantly different across habitat types. Magnificent Sunbird was more frequent in plantations than all other habitats, and in open areas than forest habitats ($H_3 = 141.21$, $p < 0.001$); Maroon-naped Sunbird (*Aethopyga guimarasensis*) was more frequent in forest habitats than plantations and open habitats ($H_3 = 50.83$, $p < 0.001$); Balicassiao (*Dicrurus baliassius*) was more frequent in primary forest than secondary forest ($H_3 = 24.97$, $p < 0.001$); and Elegant Tit (*Periparus elegans*) was more frequent in primary forest than all other habitats ($H_3 = 19.97$, $p < 0.001$).

DISCUSSION

Our study is the first to quantify extensively the bird communities of the Northwest Panay Peninsula Natural Park, a KBA (Key Biodiversity Areas Partnership 2022) and IBA (BirdLife International 2024a) of the Western Visayas faunal region. Alongside the autecological work of Mynott et al. (2020, 2021), it confirms the presence of four of the IBA's five trigger species (BirdLife International 2024a); Negros Bleeding-heart, Visayan Hornbill, Writhe-billed Hornbill, and White-winged Cuckooshrike. Although the final trigger species, Green-faced Parrotfinch, was not detected and has only been recorded at one site on Panay since 2006 (de Soye 2006, eBird 2023), it is generally found > 1000 m and only irruptively in the lowlands (BirdLife International 2024b). Our records of the sparsely distributed White-winged Cuckooshrike (Birdlife International 2016a) in primary and secondary forests are important, while that of Yellow-faced Flameback in primary forest adds to the increasing records of this species on Panay, and in the park particularly (eBird 2023). We also recorded 13 of 14 diurnal species endemic to the Western Visayas (Brinkman et al. 2023) and previously recorded in NPPNP (eBird 2023).

Species richness and diversity decreased along the gradient of habitat disturbance, which is comparable to similar studies in southeast Asia (Thiollay 1995, Waltert et al. 2004), including reduced avian richness in plantation forest (Zurita et al. 2006, Mahiga et al. 2019). The taxonomic diversity we report in natural forests and plantations is similar to that recorded in equivalent elevational forests (Amoroso et al. 2018) and broadly analogous agroforestry communities (Tanalgo et al. 2015) in other faunal regions in the Philippines, although these studies do not report on sample completeness relative to survey effort, which we encourage for the evaluation and comparison of future community studies.

Although secondary forest and plantations may help support a subset of bird diversity (Schultz et al. 2004), there are significant species losses when compared to primary forests in the park and

Table 4. SIMPER analysis showing key guild contributions (> 5%) to inter-habitat dissimilarities in community compositions and average abundances (\pm SE) of those guilds. Overall habitat dissimilarities are in parentheses.

Habitat pairing [†]	Guild [‡]	Habitat 1	Habitat 2	Overall contribution (%)
PF-SF (67.4%)		PF	SF	
	MGeF	1.04 \pm 1.335	1.20 \pm 1.170	18.1
	SLI	0.85 \pm 1.052	0.90 \pm 1.059	17.1
	SLO	0.55 \pm 0.835	1.02 \pm 1.212	17.0
	MUF	0.31 \pm 0.560	0.33 \pm 0.588	9.5
	MGeI*	0.42 \pm 0.926	0.25 \pm 1.320	8.1
PF-P (75.4%)		PF	P	
	SGeO*	0.04 \pm 0.175	1.92 \pm 1.764	21.4
	MGeF	1.04 \pm 1.335	1.17 \pm 1.408	15.6
	SLI	0.85 \pm 1.052	0.93 \pm 1.473	14.3
	SLO	0.55 \pm 0.835	0.07 \pm 0.469	8.9
	MUF	0.31 \pm 0.560	0.31 \pm 0.459	8.2
PF-O (79.2%)		PF	O	
	MGeF	1.04 \pm 1.335	0.86 \pm 0.710	16.8
	SLI	0.85 \pm 1.052	0.38 \pm 0.522	15.1
	SGeO*	0.04 \pm 0.175	0.88 \pm 1.331	10.6
	SLO	0.55 \pm 0.835	-	10.0
	MGeI	0.42 \pm 0.926	0.71 \pm 2.305	9.1
SF-P (71.8%)		SF	P	
	SGeO*	0.13 \pm 0.563	1.92 \pm 1.764	21.4
	MGeF	1.20 \pm 1.170	1.17 \pm 1.408	15.6
	SLI	0.90 \pm 1.059	0.93 \pm 1.473	14.3
	SLO	1.02 \pm 1.212	0.07 \pm 0.469	8.9
	MUF	0.33 \pm 0.588	0.31 \pm 0.459	8.2
SF-O (75.8%)		SF	O	
	MGeF	1.20 \pm 1.170	0.86 \pm 0.710	16.8
	SLI	0.90 \pm 1.059	0.38 \pm 0.522	15.1
	SGeO*	0.13 \pm 0.563	0.88 \pm 1.331	10.6
	SLO	1.02 \pm 1.212	-	10.0
	MGeI	0.25 \pm 1.320	0.71 \pm 2.305	9.1
P-O (69.0%)		P	O	
	SGeO*	1.92 \pm 1.764	0.88 \pm 1.331	24.5
	MGeF	1.17 \pm 1.408	0.86 \pm 0.710	17.9
	SLI	0.93 \pm 1.473	0.38 \pm 0.522	15.3
	MUF	0.31 \pm 0.459	0.14 \pm 0.787	8.5
	MGeI	0.29 \pm 0.806	0.71 \pm 2.305	8.0

[†] Habitat codes: primary forest (PF), secondary forest (SF), plantation (P), open habitat (O).

[‡] Guild membership: medium general-level frugivores (MGeF) and insectivores (MGeI), medium upper storey frugivores (MUF), small general-level insectivores (SGeI) and omnivores (SGeO), and small lower storey insectivores (SLI) and omnivores (SLO).

* $p < 0.008$ (α adjusted with a Bonferroni correction).

specifically for globally threatened and endemic species. This is despite the fact that these habitats are found in close proximity within the site's heterogeneous matrix (Lawton et al. 1998). Primary forest supported the highest taxonomic diversity and endemism and is essential for at least a subset of the park's bird community; eight species, including three threatened species and one additional Philippine endemic (Yellow-breasted Fruit-dove), were only recorded in undisturbed forest. The most disturbed open areas appear unsuitable for all seven threatened and near

Table 5. Ranking of key species ($\geq 5\%$ contribution to at least one pairwise comparison) contributing to community dissimilarities between habitat types. Guild membership and endemism are in parentheses.

Species [†]	Mean rank (\pm SD)	Mean contribution (\pm SD)	Habitat [‡]			
			PF	SF	P	O
Visayan Bulbul, <i>Hypsipetes guimarasensis</i> (MGeF; WV)	1.5 \pm 0.55	15.1 \pm 1.29	a	A	a	a
Visayan Tailorbird, <i>Orthotomus castaneiceps</i> (SLI; WV)	3.0 \pm 0.63	10.7 \pm 0.92	a	a	A	a
Magnificent Sunbird, <i>Aethopyga magnifica</i> (SGeO; P)*	4.5 \pm 6.25	13.8 \pm 8.11	c	c	A	b
Visayan Fantail, <i>Rhipidura albiventris</i> (SLI; WV)	5.2 \pm 0.75	5.8 \pm 1.24	A	a	a	a
Philippine Hanging-parrot, <i>Loriculus philippensis</i> (MUF; P)	6.3 \pm 1.51	5.2 \pm 0.52	A	a	a	a
Maroon-naped Sunbird, <i>Aethopyga guimarasensis</i> (SLO; WV)*	6.5 \pm 8.62	9.8 \pm 5.30	a	A	b	b
Balicassiao, <i>Dicrurus balicassius</i> (MGeI; P)*	7.3 \pm 2.73	4.3 \pm 1.39	A	b	a	a
Elegant Tit, <i>Periparus elegans</i> (SGeI; P)*	8.0 \pm 2.37	4.1 \pm 1.71	A	b	b	b

[†] For guilds: medium general-level frugivores (MGeF) and insectivores (MGeI), medium upper storey frugivores (MUF), small general-level insectivores (SGeI) and omnivores (SGeO), and small lower storey insectivores (SLI) and omnivores (SLO). For endemism: Philippines (P), and Western Visayas (WV; Brinkman et al. 2023).

[‡] Habitat codes: primary forest (PF), secondary forest (SF), plantation (P), open habitat (O). Different letters indicate significantly different relative abundances; capitalized letters indicate highest relative abundances.

* $p < 0.006$ (α adjusted with a Bonferroni correction).

threatened species detected as well as nearly 40% of endemic species recorded, which is of significant conservation concern. Any landscape management should not overlook the necessity of focusing efforts on conserving primary forest, even if some key species are at least also detected in secondary forests (Brooks et al. 1992, Kinnaird and O'Brien 2007, Mynott et al. 2021), which itself does not confirm habitat reliance or population viability.

Functional diversity may be a more sensitive metric than taxonomic diversity, which can mask changes in composition and functionality (Lawton et al. 1998), when considering the influence of habitat disturbance (Zurita and Bellocq 2010). We found important differences in both functional diversity and community composition across a gradient of habitat modification that could have consequences for ecosystem functioning (Sodhi et al. 2004) alongside conservation implications for individual species and representative endemism. For instance, most bird species described as important frugivores in the Western Visayas (Hamann and Curio 1999) were only recorded in forest habitat, including large- (e.g., both hornbill species, Pink-bellied Imperial-pigeon, Yellow-breasted Fruit-dove), medium- (e.g., Black-naped Oriole *Oriolus chinensis*, Coletto *Sarcops calvus*), and small-bodied (e.g., Yellowish White-eye *Zosterops nigrorum*) species. All these species, other than Coletto and Black-naped Oriole, are endemic to the Philippines.

As with previous work (Canaday 1996, Gray et al. 2006, Tvardiková 2010, Mahiga et al. 2019), we found that insectivorous and frugivorous guilds were particularly affected by habitat disturbance. Large general-level insectivores (e.g., Yellow-faced Flameback, White-bellied Woodpecker *Dryocopus javensis*) were only recorded in primary forest, and large frugivores (e.g., both hornbill species, Pink-bellied Imperial-pigeon, Yellow-breasted Fruit-dove) were detected considerably less frequently in non-primary forest habitat. Additionally, medium upper-storey frugivores (e.g., Philippine Hanging-parrot *Loriculus philippensis*, White-eared Brown-dove *Phapitreron leucotis*), small general-level frugivores (*Dicaeum* spp.), insectivores (Elegant Tit), and lower-storey insectivores (e.g., Visayan Fantail *Rhipidura albiventris*) appeared especially susceptible to habitat disturbance (Stouffer and Bierregaard 1995, Şekercioğlu et al. 2002, Lees and

Peres 2010, Tvardiková 2010) and the conversion of (any) forest to open habitat. Similar negative responses are exhibited by congeners of the endemic Visayan Fantail and Visayan Shama (*Copsychus saundersi*; Mallari et al. 2011, Pabico et al. 2021), classified here as small lower-level insectivores.

Although we recognize bird-habitat associations are often complex (Meents et al. 1983) and may or may not be nonlinear in nature (Radford et al. 2005, Española et al. 2016), the negative responses exhibited may result from food scarcity (Zanette et al. 2000, Mahiga et al. 2019), changes in microclimate (Karr and Freemark 1983, Canaday 1996), habitat structure and specificity (Canaday 1996, Sodhi et al. 2004, Mahiga et al. 2019), or limited dispersal (Stouffer and Bierregaard 1995, Şekercioğlu et al. 2002, Sodhi et al. 2004). In the case of insectivorous guilds, declines may be linked to changes in microhabitat characteristics or the abundance of food resources (Gray et al. 2006, Mammides et al. 2015) and the specialized nature of these requirements (Sodhi et al. 2008). The greater prevalence of large trees, indicative of primary forest in the park (Mynott et al. 2021), and their provision of suitable understorey microclimate, is an important determinant of preferred habitat for some (closely related) understorey insectivorous species in the region (e.g., White-vented Shama *Copsychus niger*; Mallari et al. 2011), while the presence of regenerating forest adjacent to primary forest in the park's heterogeneous matrix may facilitate recovery of forest insectivores in the landscape (Stouffer and Bierregaard 1995). This reduction in large trees and associated changes in floristic resources may also explain declines in large frugivores across the habitat disturbance gradient (Española et al. 2016). These larger-bodied species may be more vulnerable to habitat change due to their occurrence at naturally low densities (Sodhi et al. 2004). Consequently, there is a need for larger areas of suitable primary forest and stepwise improvements in habitat quality for their sustained conservation (Española et al. 2016).

At the species level, our findings support current understanding of some broad habitat associations. For example, Negros Jungle Flycatcher is reported in disturbed forest (Brooks et al. 1992, BirdLife International 2016a); Pink-bellied Imperial-pigeon is a species of lowland primary forest and old secondary growth

(BirdLife International 2016b); Negros Bleeding-heart is only recorded in primary forest (Brooks et al. 1992, Mynott et al. 2020); and Visayan Hornbill is recorded at significantly higher density in primary forest than in secondary forest (Mynott et al. 2021). Our records of Yellow-faced Flameback only in primary forest emphasize the need to evaluate the specific habitat requirements of this threatened, priority woodpecker (Lammertink 2014).

Both hornbill species (large frugivores) are considered at high risk of extinction based on the fragmented forest habitat configuration within their limited geographic ranges (Kinnaird and O'Brien 2007). Despite the recent suggestion that Writhe-billed Hornbill is locally extirpated in NPPNP (Reintar et al. 2022), our single record of the species (in primary forest) further confirms the importance of this protected area. Although we do not evidence breeding in the park (and there are no breeding records since 1997; BirdLife International 2018) or the park as a source population, it serves to emphasize the potential importance of a site containing considerable primary forest cover characterized by ~3x more trees of nest cavity size (Klop et al. 2000) than in secondary forests (Mynott et al. 2021). It also highlights the importance of ensuring site connectivity across the landscape for a species that may be nomadic in response to spatio-temporal variation in fruit availability (BirdLife International 2018), increasing their vulnerability to habitat change (Sodhi et al. 2004), and for which the current protected area system remains inadequate (Kinnaird and O'Brien 2007). Enhancing ecological connectivity with the Central Panay Mountain Range, where there appears to be a healthy breeding population of the Writhe-billed Hornbill (A. Alabado, R. Lestino, J. Venus, M. Ibabao, T. Kuenzel, and E. Curio 2009, *unpublished report*), and ensuring the park is not isolated by > 10 km for the Visayan Hornbill and > 20 km for the Writhe-billed Hornbill, enabling dispersal between forest patches in the wider landscape, will contribute to future proofing both hornbill species and the ecosystem services they provide as seed dispersers (Kinnaird and O'Brien 2007), providing hunting pressures are also addressed synchronously (E. Klop, 1998, *unpublished manuscript*). This concern may contribute to national discussions on the Philippines' commitment to the targets of the High Ambition Coalition for Nature and People's 30 x 30 initiative, especially when only 15% of Philippine forests are currently protected (Wolf et al. 2021).

Our results indicate an expected richness of 62 species in the park and includes detecting 85–92% of forest species, which suggests there may be 3–6 diurnal forest species that remained undetected in the specific survey season. Despite this, eBird records report 84 diurnal species at the same time of year in the park (eBird 2023). Although such bird lists may result from a spatially targeted approach to recording specific species, including the use of playback, and maximizing species detection (Watson et al. 2018), this discrepancy in species richness highlights some considerations regarding survey implementation.

Some of our surveys were restricted by the severity of the local terrain where point transects could not be accessed safely and/or surveyed around dawn (e.g., Lynch 1995, Bibby et al. 2000, Lee and Marsden 2008). Temporally then, some species may have been under recorded, e.g., Negros Jungle Flycatcher, or gone undetected. For example, we did not detect Gray-breasted Brown-dove (*Phapitreron amethystinus maculipectus*), a near threatened

Philippine endemic generally found above 1000 m, although recorded below 500 m (del Hoyo et al. 2020a). Although reported as “numerous in March/April” in the park in 2014 (BirdLife International 2023), there is only one eBird record since 2016 (eBird 2023). Similarly, we failed to detect Visayan Rhabdornis (*Rhabdornis rabori*), which is a scarce and local resident of submontane and montane forests above 800 m (del Hoyo et al. 2020b), although our sampling effort included only one point in this elevational range.

Implementing complementary survey approaches that can capture peaks in all species' detectability and without spatial restrictions will help circumvent local challenges with accessing specific locations for conducting point transect surveys immediately following sunrise (e.g., Lynch 1995, Bibby et al. 2000). Consequently, we recommend the systematic inclusion of passive bioacoustic recording to increase the likelihood of detecting all species (e.g., Metcalf et al. 2022) and support robust, constant-effort surveys necessary for monitoring population trends and conservation planning (Thompson 2002). Caveating this is the understanding that aural detection and spatial coverage for some species may be comparatively small, e.g., Negros Jungle Flycatcher, which is only audible “at close range” (Clement 2020), which may contribute to it being under recorded (eBird 2023).

Our survey effort was sufficient in capturing most species of the avian assemblage in primary forests, but less so in more disturbed habitats. Although this may relate to inter-habitat differences in daily patterns in bird activity and, hence, detectability (Fontúrbel et al. 2021), our sampling design minimized any temporal detectability differences across habitats (Verner and Ritter 1986). Instead, the higher proportion of unique species (those only detected from a single point count) recorded in open habitat (63.6%) compared to primary forest (18.4%) may contribute to lower estimated sample coverage and more variable measures of observed and expected species richness for this habitat (Chao and Jost 2012, Chao et al. 2020). To obtain similar sample completeness in plantations and open habitats requires 2.0 and 3.1 times more survey effort, respectively. This is unlikely in the park in which our lower survey effort in these habitats proportionally reflected the smaller areas of these habitats in the landscape. Including surveys of representative plantations and open habitats adjacent to the park may then contribute to fully capturing species richness with improved precision and provide a more robust temporal dataset for monitoring bird communities in these anthropic habitats. Although reducing survey effort in natural forests in the park can support more cost-effective, long-term monitoring (Thompson 2002), we are cautious in making such a recommendation if the objective is to monitor all species and entire communities because not all forest species were detected. However, 75 and 83 points in primary and secondary forests, respectively, still captured 90% of predicted diurnal species present, while reducing survey effort by 22–36 points, or 2–4 days surveying, in each forest type, and this may be a sufficient subset of species richness for conservation monitoring objectives (Vellend et al. 2007).

Since 2001, the park has lost 2.2% of its overall tree cover (249 ha; Global Forest Watch 2023a, b) including 1.3% (49 ha) of remaining primary forest (Hansen et al. 2013). This rate of primary forest loss is similar to, but still higher than the

surrounding areas (mean = $1.1 \pm 0.41\%$), whereas overall tree cover loss is noticeably higher outside the park (mean = $4.6 \pm 1.93\%$; Global Forest Watch 2023a, b), which is comparable to the national trend of higher forest loss outside protected areas (Wolf et al. 2021). This suggests that despite its protected status, primary forest in the park is as vulnerable to clearance as unprotected primary forest remnants outside the park's boundaries. Specifically, the southwest edge of the park is a sporadic hotspot of primary forest loss (defined as on-off annual activity, with < 12 of the last 14 years showing significant forest loss; Harris et al. 2017). Although overall tree cover loss is lower in the park, our findings demonstrate the importance of primary forest over secondary habitats in maintaining avian endemism and supporting globally threatened species in the heavily impacted Western Visayas faunal region (Lorica and Oliver 2006). To this end, it is essential to prevent further primary forest losses and conversion through the main drivers of commercially driven and, depending on permanence and intensity, shifting agriculture (*kaingin*; Curtis et al. 2018, BirdLife International 2024a) while also considering empirically the potentially synergistic impacts of illegal hunting (e.g., Sathya Chandra Sagar et al. 2023). Although there is no current forest restoration in place in the park or environs, accompanying passive (e.g., natural regeneration of secondary forest) and/or active (e.g., tree planting strategies in degraded areas) restoration approaches (e.g., Chazdon 2019) will improve the condition and area of crucial forest habitat for key bird species and support wider ecosystem co-benefits in this conservation priority landscape (BirdLife International 2024a).

CONCLUSIONS

The Northwest Panay Peninsula Natural Park is one of the few protected areas in the highly deforested Western Visayas with intact lowland primary forest cover remaining (Lorica and Oliver 2006), and in a region where most primary forest is now at submontane and montane elevations (Hamann et al. 1999). Our findings highlight that declines in the extent of primary forest (or forest quality) will lead to reductions in community diversity and regional endemism, including those species of most conservation concern, within the park's heterogeneous habitat matrix. This is potentially indicative of the wider region and emphasizes the high conservation value of primary forest for avian diversity, however defined, and endemism, paralleling that of other faunal regions in the Philippines (Mallari et al. 2011). The different responses of functioning guilds to habitat disturbance suggests this affects the trophic organization of bird communities and, consequently, their ecosystem functioning (Gray et al. 2006). We also demonstrated the need for robust ecological understanding and consideration of different species groups when designing conservation strategies that ensure intact, functioning communities are maintained (Canaday 1996) and support protected area management strategies (Sodhi et al. 2004, Mallari et al. 2011). Although the park remains threatened by small-scale selective logging, conversion to agriculture (e.g., coconut groves) and hunting (Kerhoas et al. 2019, Mynott et al. 2020), adjacent local communities have a high level of awareness of the importance of local endangered species, but little monetary capacity to help actively conserve them (Subade and Jugado 2010). Based on our study and previous recent work (Mynott et al. 2020, 2021), we believe that an effective future-proofed conservation strategy for bird populations of this protected area, and a key representation of the wider Western Visayas, requires strong collaboration

between city and provincial Environment and Natural Resources offices and local communities to protect primary forest areas from anthropogenic threats, to enable secondary forest areas to recover, and to support measures that retain and enhance connectivity with forest in the wider landscape.

Author Contributions:

DL, HM, and DK wrote the main manuscript, with JJ completing a first draft. JJ undertook preliminary PRIMER analyses, with DL completing the diversity, GLM, and community analyses. DK organized the project design with input from DL, managed the project, and secured project funding. RS facilitated field logistics and survey permissions within the park. HM completed the bird surveys alongside others (see acknowledgements). All authors read and approved the final manuscript.

Acknowledgments:

We thank the Protected Area Management Board members for their support and consent for completing this research project in the protected area. We also thank the Philippine Department of Environment of Natural Resources, and especially Mr. Andres Untal (PENRO), Mme Cynthia Blanca (CENRO), and their staff for their support. Special thanks for the data collection to Benjamin (Jhun) Tacud, Arcel (Putput) Dryden Fernandez, and Ilke Geladi, and to PhilinCon for the logistics and permissions to survey within the protected area. We thank Emma Higgins (University of South Wales, UK) for assistance with the R coding and three anonymous reviewers for helping improve the manuscript. This study was funded by the Mohamed bin Zayed Species Conservation Fund (project 192514132, awardee: Daphne Kerhoas) and Bristol Zoological Society. No samples were collected during the course of this research project.

LITERATURE CITED

- Alcala, E. L., A. C. Alcala, and C. N. Dolino. 2004. Amphibians and reptiles in tropical rainforest fragments on Negros Island, the Philippines. *Environmental Conservation* 31:254–261. <https://doi.org/10.1017/S0376892904001407>
- Amoroso, V. B., A. B. Mohagan, F. P. Coritico, N. E. Lagunday, A. P. Yorong, R. D. Colong, and R. G. Ponce. 2018. Avifaunal assemblage in the expansion sites of the Mt. Hamiguitan Range Wildlife Sanctuary, Mindanao, Philippines. *Nature Conservation Research* 3:103–109. <https://doi.org/10.24189/ncr.2018.069>
- Bibby, C. J., N. D. Burgess, and D. A. Hill. 2000. Bird census techniques. Revised edition. Academic, London, UK.
- Bibby, C., S. J. Marsden, and A. Fielding. 1998. Bird-habitat studies. Pages 99–114 in C. Bibby, M. Jones, and S. J. Marsden, editors. *Expedition field techniques: bird surveys*. Royal Geographical Society, London, UK.
- BirdLife International. 2016a. *Edolisoma ostentum*. The IUCN red list of threatened species 2016:e.T22706629A94080485. BirdLife International, Cambridge, UK. <https://dx.doi.org/10.2305/IUCN.UK.2016-3.RLTS.T22706629A94080485.en>

- BirdLife International. 2016b. *Vauriella albigularis*. The IUCN red list of threatened species 2016:e.T22709182A9419538. Birdlife International, Cambridge, UK. <https://dx.doi.org/10.2305/IUCN.UK.2016-3.RLTS.T22709182A94195238.en>
- BirdLife International. 2016c. *Ducula poliocephala*. The IUCN red list of threatened species 2016:e.T22691611A93318553. Birdlife International, Cambridge, UK. <https://dx.doi.org/10.2305/IUCN.UK.2016-3.RLTS.T22691611A93318553.en>
- BirdLife International. 2018. *Rhabdotorrhinus waldeni* (amended version of 2016 assessment). The IUCN red list of threatened species 2018:e.T22682517A125519634. Birdlife International, Cambridge, UK. <http://dx.doi.org/10.2305/IUCN.UK.2016-3.RLTS.T22682517A125519634.en>
- BirdLife International. 2023. Species factsheet: Grey-breasted Brown-dove *Phapitreron maculipectus*. Birdlife International, Cambridge, UK. <http://datazone.birdlife.org/species/factsheet/grey-breasted-brown-dove-phapitreron-maculipectus>
- BirdLife International. 2024a. Important bird areas factsheet: North-west Panay peninsula (Pandan). Birdlife International, Cambridge, UK. [https://datazone.birdlife.org/site/factsheet/north-west-panay-peninsula-\(pandan\)-iba-philippines](https://datazone.birdlife.org/site/factsheet/north-west-panay-peninsula-(pandan)-iba-philippines)
- BirdLife International. 2024b. Species factsheet: *Erythrura viridifacies*. Birdlife International, Cambridge, UK. <https://datazone.birdlife.org/species/factsheet/green-faced-parrotfinch-erythrura-viridifacies>
- Brinkman, J. J., D. Allen, R. Hutchinson, A. E. Jensen, and C. Perez. 2023. Checklist of birds of the Philippines. Wild Bird Club of the Philippines, Candaba, Philippines. <https://birdwatch.ph/philippine-checklist/>
- Brooks, T. M., T. D. Evans, G. C. L. Dutson, G. Q. A. Anderson, D. C. Asane, R. J. Timmins, and A. G. Toledo. 1992. The conservation status of the birds of Negros, Philippines. Bird Conservation International 2:273–302. <https://doi.org/10.1017/S0959270900002501>
- Cadotte, M. W., K. Carscadden, and N. Mirotchnick. 2011. Beyond species: functional diversity and the maintenance of ecological processes and services. Journal of Applied Ecology 48:1079–1087. <https://doi.org/10.1111/j.1365-2664.2011.02048.x>
- Canaday, C. 1996. Loss of insectivorous birds along a gradient of human impact in Amazonia. Biological Conservation 77:63–77. [https://doi.org/10.1016/0006-3207\(95\)00115-8](https://doi.org/10.1016/0006-3207(95)00115-8)
- Chao, A., and L. Jost. 2012. Coverage-based rarefaction and extrapolation: standardizing samples by completeness rather than size. Ecology 93:2533–2547. <https://doi.org/10.1890/11-1952.1>
- Chao, A., Y. Kubota, D. Zelený, C.-H. Chiu, C.-F. Li, B. Kusumoto, M. Yasuhara, S. Thorn, C.-L. Wei, M. J. Costello, and R. K. Colwell. 2020. Quantifying sample completeness of a biological survey and comparing diversities among assemblages based on incomplete surveys. Ecological Research 35:292–314. <https://doi.org/10.1111/1440-1703.12102>
- Chao A., K. H. Ma, and T. C. Hsieh. 2016. iNEXT (INterpolation and EXTrapolation). <https://chao.shinyapps.io/iNEXTOnline/>
- Chazdon, R. L. 2019. Towards more effective integration of tropical forest restoration and conservation. Biotropica 51:463–472. <https://doi.org/10.1111/btp.12678>
- Cianciaruso, M. V., I. A. Silva, and M. A. Batalha. 2009. Phylogenetic and functional diversities: new approaches to community ecology. Biota Neotropica 9:93–103. <https://doi.org/10.1590/S1676-06032009000300008>
- Clarke, K. R. 1993. Non-parametric multivariate analyses of changes in community structure. Australian Journal of Ecology 18:117–143. <https://doi.org/10.1111/j.1442-9993.1993.tb00438.x>
- Clarke, K. R., and R. N. Gorley. 2006. Primer V6: user manual/tutorial. Primer-E, Plymouth, UK.
- Clarke, K. R., and R. M. Warwick. 2001. Change in marine communities: an approach to statistical analysis and interpretation. Second edition. PRIMER-E, Plymouth, UK.
- Clement, P. 2020. Negros Jungle Flycatcher (*Vauriella albigularis*). Version 1.0. In J. del Hoyo, A. Elliott, J. Sargatal, D. A. Christie, and E. de Juana, editors. Birds of the world. Cornell Lab of Ornithology, Ithaca, New York, USA. <https://doi.org/10.2173/bow.negjuf1.01>
- Clements, J. F., P. C. Rasmussen, T. S. Schulenberg, M. J. Iliff, T. A. Fredericks, J. A. Gerbracht, D. Lepage, A. Spencer, S. M. Billerman, B. L. Sullivan, and C. L. Wood. 2023. The eBird/Clements checklist of Birds of the World: v2023. Cornell Lab of Ornithology, Ithaca, New York, USA. <https://doi.org/10.1111/j.1472-4642.2006.00312.x>
- Connell, J. H. 1978. Diversity in tropical rain forests and coral reefs: high diversity of trees and corals is maintained only in a non-equilibrium state. Science 199:1302–1310. <https://doi.org/10.1126/science.199.4335.1302>
- Convention on Biological Diversity. 2020. Country profiles: Philippines - country profile. Convention on Biological Diversity, Montreal, Quebec, Canada. <https://www.cbd.int/countries/profile/?country=ph>
- Corbelli, J. M., G. A. Zurita, J. Filloy, J. P. Galvis, N. I. Vespa, and I. Bellocq. 2015. Integrating taxonomic, functional and phylogenetic beta diversities: interactive effects with the biome and land use across taxa. PLoS One 10:e0126854. <https://doi.org/10.1371/journal.pone.0126854>
- Cornils, J. S., I. Riedl, J. Fricke, M. Katz, and C. H. Schulze. 2015. Population density and habitat preferences of the Black-cheeked Ant-tanager *Habia atrimaxillaris*. Bird Conservation International 25:306–321. <https://doi.org/10.1017/S0959270914000215>
- Cosset, C. C. P., and D. P. Edwards. 2017. The effects of restoring logged tropical forests on avian phylogenetic and functional diversity. Ecological Applications 27:1932–1945. <https://doi.org/10.1002/eap.1578>
- Curtis, P. G., C. M. Slay, N. L. Harris, A. Tyukavina, and M. C. Hansen. 2018. Classifying drivers of global forest loss. Science 361:1108–1111. <https://doi.org/10.1126/science.aau3445>
- del Hoyo, J., L. F. Baptista, N. Collar, P. W. Trail, G. M. Kirwan, H. M. Horblit, E. F. J. Garcia, and C. J. Sharpe. 2020a. Amethyst

- Brown-Dove (*Phapitreron amethystinus*). Version 1.0. In S. M. Billerman, B. K. Keeney, P. G. Rodewald, and T. S. Schulenberg, editors. Birds of the world. Cornell Lab of Ornithology, Ithaca, New York, USA. <https://doi.org/10.2173/bow.amedov1.01>
- del Hoyo, J., and N. J. Collar. 2014. HBW and BirdLife International: illustrated checklist of the birds of the world. Volume 1: non-passerines. Lynx Edicions, Barcelona, Spain.
- del Hoyo, J., and N. J. Collar. 2016. HBW and BirdLife International: illustrated checklist of the birds of the world. Volume 2: passerines. Lynx Edicions, Barcelona, Spain.
- del Hoyo, J., N. J. Collar, and D. A. Christie. 2020b. Visayan Rhabdornis (*Rhabdornis rabori*). Version 1.0. In J. del Hoyo, A. Elliott, J. Sargatal, D. A. Christie, and E. de Juana, editors. Birds of the world. Cornell Lab of Ornithology, Ithaca, New York, USA. <https://doi.org/10.2173/bow.visrha1.01>
- de Soye, Y. 2006. eBird checklist. eBird, Ithaca, New York, USA. <https://ebird.org/checklist/S65009105>
- eBird. 2023. eBird: an online database of bird distribution and abundance. eBird, Cornell Lab of Ornithology, Ithaca, New York, USA. <http://www.ebird.org>
- Española, C. P., N. J. Collar, N. A. D. Mallari, and S. J. Marsden. 2016. Large avian frugivores in the Philippines show linear responses to improvements in forest quality. *Forest Ecology and Management* 375:127–133. <https://doi.org/10.1016/j.foreco.2016.05.003>
- Flynn, D. F. B., M. Gogol-Prokurat, T. Nogeire, N. Molinari, B. T. Richers, B. B. Lin, N. Simpson, M. M. Mayfield, and F. DeClerck. 2009. Loss of functional diversity under land use intensification across multiple taxa. *Ecology Letters* 12:22–33. <https://doi.org/10.1111/j.1461-0248.2008.01255.x>
- Fontúrbel, F. E., J. I. Orellana, G. B. Rodríguez-Gómez, C. A. Tabilao, and G. J. Castaño-Villae. 2021. Habitat disturbance can alter forest understory bird activity patterns: A regional-scale assessment with camera-traps. *Forest Ecology and Management* 479:118618. <https://doi.org/10.1016/j.foreco.2020.118618>
- Global Forest Watch. 2023a. Philippines: Aklan. Global Forest Watch, Washington, D.C., USA. <https://www.globalforestwatch.org/dashboards/country/PHL>
- Global Forest Watch. 2023b. Philippines: Antique. Global Forest Watch, Washington, D.C., USA. <https://www.globalforestwatch.org/dashboards/country/PHL>
- Gray, M. A., S. L. Baldauf, P. J. Mayhew, and J. K. Hill. 2006. The response of avian feeding guilds to tropical forest disturbance. *Conservation Biology* 21:133–141. <https://doi.org/10.1111/j.1523-1739.2006.00557.x>
- Gray, T. N. E., A. C. Hughes, W. F. Laurance, B. Long, A. J. Lynam, H. O’Kelly, W. J. Ripple, T. Seng, L. Scotson, and N. M. Wilkinson. 2018. The wildlife snaring crisis: an insidious and pervasive threat to biodiversity in Southeast Asia. *Biodiversity and Conservation* 27:1031–1037. <https://doi.org/10.1007/s10531-017-1450-5>
- Hamann, A., E. B. Barbon, E. Curio, and D. A. Madulid. 1999. A botanical inventory of a submontane tropical rainforest on Negros Island, Philippines. *Biodiversity and Conservation* 8 (8):1017–1031. <https://doi.org/10.1023/A:1008847704539>
- Hamann, A., and E. Curio. 1999. Interactions among frugivores and fleshy fruit trees in a Philippine submontane rainforest. *Conservation Biology* 13:766–773. <https://doi.org/10.1046/j.1523-1739.1999.97420.x>
- Hansen, M. C., P. V. Potapov, R. Moore, M. Hancher, S. A. Turubanova, A. Tyukavina, D. Thau, S. V. Stehman, S. J. Goetz, T. R. Loveland, A. Kommareddy, A. Egorov, L. Chini, C. O. Justice, and J. R. G. Townshend. 2013. High-resolution global maps of 21st-century forest cover change. *Science* 342:850–853. <https://doi.org/10.1126/science.1244693>. Data available from: <https://glad.earthengine.app/view/global-forest-change>
- Harris, N. L., E. Goldman, C. Gabris, J. Nordling, S. Minnemeyer, S. Ansari, M. Lippmann, L. Bennett, M. Raad, M. dazHansen, and P. Potapov. 2017. Using spatial statistics to identify emerging hot spots of forest loss. *Environmental Research Letters* 12:024012. <https://doi.org/10.1088/1748-9326/aa5a2f>
- Heaney, L. R. 1985. Zoogeographic evidence for Middle and Late Pleistocene land bridges to the Philippine islands. *Modern Quaternary Research in Southeast Asia* 9:127–143. https://www.academia.edu/24812790/Zoogeographic_evidence_for_Middle_and_Late_Pleistocene_landbridges_to_the_Philippine_Islands
- Hughes, A. C. 2017. Understanding the drivers of Southeast Asian biodiversity loss. *Ecosphere* 8:e01624. <https://doi.org/10.1002/ecs2.1624>
- International Union for the Conservation of Nature (IUCN). 2023. The IUCN red list of threatened species. Version 2023-1. IUCN, Gland, Switzerland. <https://www.iucnredlist.org>
- Jakosalem, P. G. C., N. J. Collar, and J. A. Gill. 2013. Habitat selection and conservation status of the endemic Ninox Hawk-owl on Cebu, Philippines. *Bird Conservation International* 23:360–370. <https://doi.org/10.1017/S0959270912000317>
- Karr, J. R., and K. E. Freemark. 1983. Habitat selection and environmental gradients: dynamics in the “stable” tropics. *Ecology* 64:1481–1494. <https://doi.org/10.2307/1937503>
- Kennedy, R. S., P. C. Gonzales, E. C. Dickinson, H. C. Miranda, Jr., and T. H. Fisher. 2000. A guide to the birds of the Philippines. Oxford University Press, Oxford, UK.
- Kerhoas, D., J. Poole, and M. Abrahams. 2019. A systematic survey for the Critically Endangered Negros Bleeding-heart *Gallicolumba keayi* in the Northwest Panay Peninsula Natural Park, Philippines. *BirdingAsia* 32:50–56.
- Key Biodiversity Areas Partnership. 2022. Key Biodiversity Areas factsheet: North-west Panay Peninsula (Pandan). Extracted from the World Database of Key Biodiversity Areas. Key Biodiversity Areas Partnership: BirdLife International, IUCN, American Bird Conservancy, Amphibian Survival Alliance, Conservation International, Critical Ecosystem Partnership Fund, Global Environment Facility, Global Wildlife Conservation, NatureServe, Rainforest Trust, Royal Society for the Protection of Birds, World Wildlife Fund, and Wildlife Conservation Society. <http://www.keybiodiversityareas.org/>
- Kinnaid, M. F., and T. G. O’Brien. 2007. The ecology and conservation of Asian hornbills: farmers of the forest. University of Chicago Press, Chicago, Illinois, USA.

- Klop, E., E. Curio, and L. L. Lastimoza. 2000. Breeding biology, nest site characteristics and nest spacing of the Visayan Tarictic Hornbill *Penelopides panini* on Panay, Philippines. *Bird Conservation International* 10:17–27. <https://doi.org/10.1017/S0959270900000022>
- Laliberté, E., and P. Legendre. 2010. A distance-based framework for measuring functional diversity from multiple traits. *Ecology* 91:299–305. = <https://doi.org/10.1890/08-2244.1>
- Laliberté, E., P. Legendre, and B. Shipley. 2014. FD: measuring functional diversity from multiple traits, and other tools for functional ecology. R Foundation for Statistical Computing, Vienna, Austria. <http://cran.r-project.org/web/packages/FD/>
- Lambert, F. R. 1992. The consequences of selective logging for Bornean lowland forest birds. *Philosophical Transactions of the Royal Society B: Biological Sciences* 335:443–457. <https://doi.org/10.1098/rstb.1992.0036>
- Lammertink, M. 2014. Trends in threat status and priorities in conservation of the woodpeckers of the World. *Acta Ornithologica* 49:207–219. <https://doi.org/10.3161/173484714X687109>
- Lawton, J. H., D. E. Bignell, B. Bolton, G. F. Bloemers, P. Eggleton, P. M. Hammond, M. Hodda, R. D. Holt, T. B. Larsen, N. A. Mawdsley, N. E. Stork, D. S. Srivastava, and A. D. Watt. 1998. Biodiversity inventories, indicator taxa and effects of habitat modification in tropical forest. *Nature* 391:72–76. <https://doi.org/10.1038/34166>
- Lee, D. C., and S. J. Marsden. 2008. Adjusting count period strategies to improve the accuracy of forest bird abundance estimates from point transect distance sampling surveys. *Ibis* 150:315–325. <https://doi.org/10.1111/j.1474-919X.2007.00790.x>
- Lees, A. C., and C. A. Peres. 2010. Habitat and life history determinants of antbird occurrence in variable sized Amazonian forest fragments. *Biotropica* 42:614–621. <https://doi.org/10.1111/j.1744-7429.2010.00625.x>
- Lenth, R., P. Buerkner, M. Herve, J. Love, H. Riebl, and H. Singmann. 2022. R package emmeans: estimated marginal means, aka least-squares means. R Foundation for Statistical Computing, Vienna, Austria. <https://github.com/rvnlenth/emmeans>
- Lloyd, H., A. Cahill, M. Jones, and S. J. Marsden. 2000. Estimating bird densities using distance sampling. Pages 35–52 in C. Bibby, M. Jones, and S. J. Marsden, editors. *Expedition field techniques: bird surveys*. Royal Geographic Society, London, UK.
- Loreau, M., S. Naeem, P. Inchausti, J. Bengtsson, J. P. Grime, A. Hector, D. U. Hooper, M. A. Huston, D. Raffaelli, B. Schmid, D. Tilman, and D. A. Wardle. 2001. Biodiversity and ecosystem functioning: current knowledge and future challenges. *Science* 294:804–808. <https://doi.org/10.1126/science.1064088>
- Lorica, R., and W. Oliver. 2006. Distribution and habitat utilization of the Visayan leopard cat *Prionailurus bengalensis rabori*. *Banwa* 3(1&2):117–129.
- Luck, G. W., S. Lavorel, S. McIntyre, and K. Lumb. 2012. Improving the application of vertebrate trait-based frameworks to the study of ecosystem services. *Journal of Animal Ecology* 81:1065–1076. <https://doi.org/10.1111/j.1365-2656.2012.01974.x>
- Lynch, J. F. 1995. Effects of point count duration, time-of-day, and aural stimuli on detectability of migratory and resident bird species in Quintana Roo, Mexico. Pages 1–6 in C. J. Ralph, J. R. Sauer, and S. Droege, editors. *Monitoring bird populations by point counts*. General technical report PSW-GTR-149. U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, California. https://www.fs.usda.gov/psw/publications/documents/psw_gtr149/psw_gtr149_pg1_6.pdf
- Mahiga, S. N., P. Webala, M. J. Mware, and P. K. Ndong'ang'a. 2019. Influence of land-use type on forest bird community composition in Mount Kenya Forest. *International Journal of Ecology* 2019:8248270. <https://doi.org/10.1155/2019/8248270>
- Mallari, N. A. D., N. J. Collar, D. C. Lee, P. J. K. McGowan, R. Wilkinson, and S. J. Marsden. 2011. Population densities of understorey birds across a habitat gradient in Palawan, Philippines: Implications for conservation. *Oryx* 45:234–242. <https://doi.org/10.1017/S0030605310001031>
- Mammides, C., M. Schleuning, K. Böhning-Gaese, G. Schaab, N. Farwig, C. Kadis, and T. Coulson. 2015. The indirect effects of habitat disturbance on the bird communities in a tropical African forest. *Biodiversity Conservation* 24:3083–3107. <https://doi.org/10.1007/s10531-015-1001-x>
- Marschner, I. C. 2011. glm2: fitting generalized linear models. 2011. R package version 1.0. R Foundation for Statistical Computing, Vienna, Austria. R package version 1.0.
- Marsden, S. J. 1999. Estimation of parrot and hornbill densities using a point count distance sampling method. *Ibis* 141:377–390. <https://doi.org/10.1111/j.1474-919X.1999.tb04405.x>
- Marsden, S. J., and J. D. Pilgrim. 2003. Factors influencing the abundance of parrots and hornbills in pristine and disturbed forests on New Britain, PNG. *Ibis* 145:45–53. <https://doi.org/10.1046/j.1474-919X.2003.00107.x>
- Meents, J. K., J. Rice, B. W. Anderson, and R. D. Ohmart. 1983. Non-linear relationships between birds and vegetation. *Ecology* 64:1022–1027. <https://doi.org/10.2307/1937809>
- Metcalf, O. C., J. Barlow, S. Marsden, N. Gomes de Moura, E. Berenguer, J. Ferreira, and A. C. Lees. 2022. Optimizing tropical forest bird surveys using passive acoustic monitoring and high temporal resolution sampling. *Remote Sensing in Ecology and Conservation* 8:45–56. <https://doi.org/10.1002/rse2.227>
- Mittermeier, R. A., P. Robles Gil, and C. G. Mittermeier. 1997. Megadiversity: Earth's biologically wealthiest nations. CEMEX and Agrupacion Sierra Madre, Monterrey and Mexico City, Mexico.
- Mouillot, D., N. A. J. Graham, S. Villéger, N. W. H. Mason, and D. R. Bellwood. 2013. A functional approach reveals community responses to disturbances. *Trends in Ecology and Evolution* 28:167–177. <https://doi.org/10.1016/j.tree.2012.10.004>
- Myers, N., R. A. Mittermeier, C. G. Mittermeier, A. B. G. da Fonseca, and J. Kent. 2000. Biodiversity hotspots for conservation priorities. *Nature* 403:853–858. <https://doi.org/10.1038/35002501>
- Mynott, H. I., M. Abrahams, and D. Kerhoas. 2020. Negros Bleeding-heart *Gallicolumba keayi* prefers dense understorey vegetation and dense canopy cover, and species distribution

- modelling shows little remaining suitable habitat. *Bird Conservation International* 31(4):640–655. <https://doi.org/10.1017/S0959270920000611>
- Mynott, H. I., D. C. Lee, R. A. Santillan, C. J. Schwarz, B. Tacud, A. D. Fernandez, and D. Kerhoas. 2021. Population assessment and habitat associations of the Visayan Hornbill (*Penelopides panini*) in Northwest Panay, Philippines. *Avian Research* 12:67. <https://doi.org/10.1186/s40657-021-00303-3>
- Pabico, L. A., M. V. Duya, J. S. Fidelino, P. S. Ong, and M. R. M. Duya. 2021. Bird feeding guild assemblage along a disturbance gradient in the Pantabangan-Carranglan Watershed and Forest Reserve, Central Luzon Island, Philippines. *Philippine Journal of Science* 150(S1):237–255. <https://doi.org/10.56899/150.S1.16>
- Paguntalan, L. J., A. R. T. Reintar, G. C. Jakosalem, G. Peña, and A. Villarojo. 2023. Population density and distribution of the Endangered Black Shama *Kittacincla cebuensis*. *Bird Conservation International* 33:e33:1–8. <https://doi.org/10.1017/S095927092200034X>
- Payton, M. E., M. H. Greenstone, and N. Schenkerk. 2003. Overlapping confidence intervals or standard error intervals: What do they mean in terms of statistical significance? *Journal of Insect Science* 3:34. <https://doi.org/10.1673/031.003.3401>
- Petchey, O. L., and K. J. Gaston. 2002. Functional diversity (FD), species richness and community composition. *Ecology Letters* 5:402–411. <https://doi.org/10.1046/j.1461-0248.2002.00339.x>
- Petchey, O. L., A. Hector, and K. J. Gaston. 2004. How do different measures of functional diversity perform? *Ecology* 85:847–857. <https://doi.org/10.1890/03-0226>
- Peque, D., and D. Hölscher. 2014. The abundance of rare tree species in remnant forests across the Visayas, Philippines. *Biodiversity and Conservation* 23:2183–2200. <https://doi.org/10.1007/s10531-014-0714-6>
- PhilGIS. Land cover in the Philippines, 2010. 2020. <https://www.geoportal.gov.ph/>
- R Development Core Team. 2017. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. <https://www.r-project.org/>
- Radford, J. Q., A. F. Bennett, and G. J. Cheers. 2005. Landscape-level thresholds of habitat cover for woodland-dependent birds. *Biological Conservation* 124:317–337. <https://doi.org/10.1016/j.biocon.2005.01.039>
- Reintar, A. R. T., L. J. Paguntalan, P. G. C. Jakosalem, A. C. D. Quidet, D. A. Warguez, and E. Peñaranda. 2022. Habitat preference and population density of threatened Visayan hornbills *Penelopides panini* and *Rhabdotorrhinus waldeni* in the Philippines. *Journal of Threatened Taxa* 14:20713–20720. <https://doi.org/10.11609/jott.7261.14.3.20713-20720>
- Sathya Chandra Sagar, H. S., J. J. Gilroy, T. Swinfield, Z. Burivalova, D. L. Yong, E. Gemita, N. Novriyanti, D. C. Lee, M. N. Janra, A. Balmford, and F. Hua. 2023. Avifauna recovers faster in areas less accessible to trapping in regenerating tropical forests. *Biological Conservation* 279:109901. <https://doi.org/10.1016/j.biocon.2023.109901>
- Schultze, C. H., M. Waltert, P. J. A. Kessler, R. Pitopang, D. Veddele, M. Mühlenberg, S. R. Gradstein, C. Leuschner, I. Steffan-Dewenter, and T. Tschardt. 2004. Biodiversity indicator groups of tropical land-use systems: comparing plants, birds, and insects. *Ecological Applications* 14:1321–1333. <https://doi.org/10.1890/02-5409>
- Şekercioğlu, C. H., P. R. Ehrlich, G. C. Daily, D. Aygen, D. Goehring, and R. F. Sandi. 2002. Disappearance of insectivorous birds from tropical forest fragments. *Proceedings of the National Academy of Sciences* 99:263–267. <https://doi.org/10.1073/pnas.012616199>
- Sheil, D., and D. F. R. P. Burslem. 2003. Disturbing hypotheses in tropical forests. *Trends in Ecology and Evolution* 18:18–26. [https://doi.org/10.1016/S0169-5347\(02\)00005-8](https://doi.org/10.1016/S0169-5347(02)00005-8)
- Sheil, D., J. A. Sayer, and T. O'Brien. 1999. Tree diversity and conservation in logged rainforest. *Science* 284:1587. <https://doi.org/10.1126/science.284.5420.1587a>
- Sodhi, N. S., L. H. Liow, and F. A. Bazzaz. 2004. Avian extinctions from tropical and subtropical forests. *Annual Review of Ecology and Systematics* 35:323–345. <https://doi.org/10.1146/annurev.ecolsys.35.112202.130209>
- Sodhi, N. S., M. R. C. Posa, T. M. Lee, and I. G. Warkentin. 2008. Perspectives in ornithology: effects of disturbance or loss of tropical rainforest on birds. *Auk* 125(3):511–519. <https://doi.org/10.1525/auk.2008.1708>
- Stattersfield, A. J., M. J. Crosby, A. J. Long, D. C. Wege, and A. P. Rayner. 1998. Endemic bird areas of the world: priorities for biodiversity conservation. BirdLife International, Cambridge, UK.
- Stouffer, P. C., and R. O. Bierregaard, Jr. 1995. Use of Amazonian forest fragments by understory insectivorous birds. *Ecology* 76(8):2429–2445. <https://doi.org/10.2307/2265818>
- Subade, R., and E. Jugado. 2010. Economic valuation of conserving endangered species and their habitats in the northwest Panay Peninsula, Central Philippines. EEPSEA Special and Technical Paper tp201010t1. Revised Oct 2010. Economy and Environment Program for Southeast Asia (EEPSEA), Ho Chi Minh City, China. <https://core.ac.uk/download/pdf/6725235.pdf>
- Tanalgo, K. C., J. A. F. Pineda, M. E. Agravante, and Z. M. Amerol. 2015. Bird diversity and structure in different land-use types in lowland south-central Mindanao, Philippines. *Tropical Life Sciences Research* 26(2):85–103.
- Thiollay, J.-M. 1995. The role of traditional agroforests in the conservation of rain forest bird diversity in Sumatra. *Conservation Biology* 9:335–353. <https://doi.org/10.1046/j.1523-1739.1995.9020335.x>
- Thompson, S. K. 2002. Sampling. Wiley, New York, New York, USA.
- Tilman, D. 2000. Causes, consequences and ethics of biodiversity. *Nature* 405:208–211. <https://doi.org/10.1038/35012217>
- Tilman, D. 2001. Functional diversity. Pages 109–120 in S. A. Levin, editor. *Encyclopaedia of biodiversity*. Academic, San Diego, California, USA.

Tvardíková, K. 2010. Bird abundances in primary and secondary growths in Papua New Guinea: a preliminary assessment. *Tropical Conservation Science* 3:373–388. <https://doi.org/10.1177/194008291000300403>

United Nations Environment Programme-World Conservation Monitoring Centre (UNEP-WCMC). 2022. Protected area profile for Northwest Panay Peninsula Natural Park. World database of protected areas. UNEP-WCMC, Cambridge, UK. www.protectedplanet.net

Vandewalle, M., F. de Bello, M. P. Berg, T. Bolger, S. Dolédec, F. Dubs, C. K. Feld, R. Harrington, P. A. Harrison, S. Lavorel, P. M. da Silva, M. Moretti, J. Niemelä, P. Santos, T. Sattler, J. P. Sousa, M. T. Sykes, A. J. Vanbergen, and B. A. Woodcock. 2010. Functional traits as indicators of biodiversity response to land use changes across ecosystems and organisms. *Biodiversity and Conservation* 19:2921–2947. <https://doi.org/10.1007/s10531-010-9798-9>

Vellend, M., P. L. Lilley, and B. M. Starzomski. 2007. Using subsets of species in biodiversity surveys. *Journal of Applied Ecology* 45:161–169. <https://doi.org/10.1111/j.1365-2664.2007.01413.x>

Verner, J., and L. V. Ritter. 1986. Hourly variation in morning point counts of birds. *Auk* 103:117–124. <https://doi.org/10.1093/auk/103.1.117>

Waltert, M., A. Mardiasuti, and M. Mühlenberg. 2004. Effects of land use on bird species richness in Sulawesi, Indonesia. *Conservation Biology* 18(5):1339–1346. <https://doi.org/10.1111/j.1523-1739.2004.00127.x>

Watson, D. M., E. Znidersic, and M. D. Craig. 2018. Ethical birding call playback and conservation. *Conservation Biology* 33(2):469–471. <https://doi.org/10.1111/cobi.13199>

Whytock, R. C., E. Fuentes-Montemayor, K. Watts, P. Barbosa De Andrade, R. T. Whytock, P. French, N. A. Macgregor, and K. J. Park. 2018. Bird-community responses to habitat creation in a long-term, large-scale natural experiment. *Conservation Biology* 32(2):345–354. <https://doi.org/10.1111/cobi.12983>

Wilman, H., J. Belmaker, J. Simpson, C. De la Rosa, M. M. Rivadeneira, and W. Jetz. 2014. EltonTraits 1.0: species-level foraging attributes of the world's birds and mammals. *Ecology* 95:2027. <https://doi.org/10.1890/13-1917.1>

Wolf, C., T. Levi, W. J. Ripple, D. A. Zérrate-Charry, and M. G. Betts. 2021. A forest loss report card for the world's protected areas. *Nature Ecology and Evolution* 5:520–529. <https://doi.org/10.1038/s41559-021-01389-0>

Zanette, L., P. Doyle, and S. M. Trémont. 2000. Food shortage in small fragments: evidence from an area-sensitive passerine. *Ecology* 81:1654–1666. [https://doi.org/10.1890/0012-9658\(2000\)081\[1654:FSISFE\]2.0.CO;2](https://doi.org/10.1890/0012-9658(2000)081[1654:FSISFE]2.0.CO;2)

Zurita, G. A., and M. I. Bellocq. 2010. Spatial patterns of bird community similarity: bird responses to landscape composition and configuration in the Atlantic Forest. *Landscape Ecology* 25(1):147–158. <https://doi.org/10.1007/s10980-009-9410-4>

Zurita, G. A., N. Rey, D. M. Varela, M. Villagra, and M. I. Bellocq. 2006. Conversion of the Atlantic Forest into native and exotic tree plantations: effects on bird communities from the local and regional perspectives. *Forest Ecology and Management* 235(1-3):164–173. <https://doi.org/10.1016/j.foreco.2006.08.009>



Appendix 1. Number and percentage (in parentheses) of point transects bird species were recorded at, overall and in each habitat type (primary forest (PF), secondary forest (SF), plantation (P), open habitat (O)). Total numbers include all species records, regardless of distance from a point or whether they were aerial-only detections. Habitat-specific numbers only include birds recorded within 50m of points and no aerial-only detections (e.g., Ameline Swiftlet, Pygmy Swiftlet, and Brahminy Kite were removed. If a species was recorded >50m away or aurally this is indicated by '-'). Globally threatened and near threatened species are listed first; unless otherwise stated, species are considered as least concern. Species endemic to the Philippines are listed in bold, with endemism status coded as: Negros, Panay, Masbate and Ticao (NPMT); Philippines (P); and Western Visayas (WV; Brinkman et al. 2023).

Please click here to download file 'appendix1.rtf'.
