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The relevance of urban agroforestry and urban remnant forest for avian diversity in a densely-populated developing country: the case of Port-au-Prince, Haiti.

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Running title: Urban green space and avian diversity in Haiti

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Summary

Urban green spaces can contribute to conserve some of the original avian diversity in urbanized landscapes. However, the relative contribution of urban agriculture vs. urban forest remnants to the preservation of the avifauna in large cities, particularly in developing countries in the tropics, remains poorly documented. We assessed bird assemblages in the densely-populated capital city of Port-au-Prince in Haiti, one of the poorest countries in the world. From early January to late April 2018, 10 censuses per site were conducted in 16 urban green sites, consisting of 8 unexploited, private forest patches and 8 agroforestry plantations, embedded within the urban landscape and ranging in size between 1 and 11 ha. Overall, 32 and 34 different bird species were recorded in forest patches and agroforestry sites, respectively, including two endemic species listed as vulnerable by IUCN, *Corvus leucognaphalus* and *Amazona ventralis*. There was a large overlap (85 %) in the composition of avian assemblages between urban agroforest sites and urban forest remnants. However, pairwise compositional dissimilarity increased with increasing distance between sites. Finally, both species richness and diversity in the 16 sites were negatively correlated with the percentage of built habitat in a 500-m radius around each site. We discuss our results in relation to the importance of small urban green spaces in countries suffering from high levels of deforestation and severe poverty.

Keywords: Agroforestry, avian assemblages, bird conservation, Caribbean, urban ecology.

1. Introduction

Although global urbanization is a major driver of biodiversity loss (McKinney 2002, McIntyre 2014, Silva-Junior et al. 2018), proper design and management of urban green spaces can contribute to the maintenance of considerable and valuable levels of biodiversity (Goddard et al. 2010, Helden et al. 2012, Kantsa et al. 2013, Kowarik and von der Lippe 2018). In particular, urban green spaces can contribute to conserve some of the original avian diversity in urbanized landscapes, as shown by studies conducted in various cities over the world (Fernández-Juricic and Jokimäki 2001, Chace and Walsh 2006, Murgui 2007, Carbó-Ramírez and Zuria 2011, Chang and Lee 2016, Lepczyk et al. 2017, Callaghan et al. 2018, Zorzal et al. 2021). However, the importance of urban green spaces for avian conservation varies with their characteristics. For instance, avian diversity inside urban parks in Italy appears to be positively related to the diversity of land covers and the presence of water bodies, and negatively related to the density of the surrounding urban matrix (Canedoli et al. 2018). On the other hand, Chaiyarat et al. (2019) found that both park size and proximity to the largest urban park positively influence bird species richness in Thailand.

The relative importance of urban green areas for bird conservation may also depend on the landscape context and the general availability of nearby suitable natural habitat (Hedblom and Söderstorm 2010). For instance, avian richness was found to increase with urbanization in a semi-arid region, in direct relation to water availability (Trammell et al. 2011). In that respect, the importance of urban green space for the conservation of numerous bird species may deserve particular attention where intense deforestation has taken place (Enedino et al. 2018). Loss of forest habitat can have drastic consequences for bird populations (Robinson et al. 1995, Mortelliti et al. 2010, Bregman et al. 2014, Karp et al. 2019), particularly in the Neotropical region (Lisboa de Carvalho et al. 2017), where human population is still growing and urban habitat expanding, particularly in the poorest countries (Dao 2002, Cohen 2004).

In this context, the value of urban green spaces for avian conservation might be particularly worth examining in Haiti. Haiti is the most densely populated country in Latin America with a current population estimated at 11.3 million and a population density of 409 persons per km² (United Nations, Department of Economic and Social Affairs,

Population Division, 2019). As of September 2020, the avifauna of Haiti includes a total of 270 species, according to Avibase (Bird Checklists of the World, <https://avibase.bsc-eoc.org/checklist.jsp?region=HT>), of which 184 can be considered as arboreal species. However, Haitian forests have largely regressed over the last century and continue to do so (Hosier 1989, Paryski et al. 1989, Gibbons 2010, Bellande 2015), with the percentage of tree cover varying between about 20 and 30% (Churches et al. 2014). According to a recent study, Haiti might have lost more than 99% of its original primary forest, with direct consequences in terms of biodiversity loss (Hedges et al. 2018; see however Wampler et al. 2019). Deforestation in Haiti is actually a complex phenomenon to which many historical and contemporary factors contribute (Bellande 2015). In particular, the incapacity of market-gardening agriculture to keep pace with increased population density directly contributes to the development of uncontrolled subsistence agriculture (Ashley, 1989). In addition, the complexity of the land tenure system (Dolisca et al. 2007) and the lack of off-farm opportunities (Pierre-Louis 1989) are important factors affecting land clearing. Finally, the high local demand for firewood and charcoal underpins much of the illegal harvesting of trees, including in supposedly protected areas (Dolisca et al 2007, Hedges et al. 2018). All of these factors combine to make Haiti one of the most deforested countries in the world.

However, if remnants of dense primary forest are scarce in Haiti, other types of tree vegetation exist in the country, including remnants of secondary forest (Bellande 2015) and multi-purpose, multi-layer agroforestry systems (Hosier 1989, Jean-Denis et al. 2014). Instead of forming large compact forested areas, such green areas generally exist under the form of a mosaic of habitats, including within the urban environment (Bellande 2015). In Haiti, the urban agroforestry practice consists in clearing part of a forested parcel to grow various plants (for fruits, rhizomes, etc.). Some of the original trees and shrubs can however be conserved, whether they are difficult to cut or are traditionally used for medicinal usages, food, firewood, providing shade, or soil and water maintenance (Bannister and Nair 2003). Cultivated tree cover can benefit avian biodiversity (Rodrigues et al. 2018), but its importance in urban environments has received less attention (see however Chong et al. 2014, Rodewald 2016, Mayorga et al. 2020), and evidence for developing countries such as Haiti is lacking (Clucas et al. 2018).

Here we report new and original results on the importance of urban green spaces for bird conservation in Haiti through studying bird assemblages both in remaining patches of

secondary forest and in small agro-forestry systems located in the area of the capital city of Port-au-Prince.

2. Material and Methods

2.1. Study area

The study area, the district of Port-au-Prince (18 ° 31'32.66 N & 72 ° 21'45.66), is a highly complex and heterogeneous urban area. It lies in the Cul-de-Sac alluvial plain, extending eastwards to the River Grise, while being bordered to the west by the sea and to the south by the La Selle Mountain range. It comprises eight municipalities and is home to 2.7 million residents accounting for about 45 percent of the country's urban population and 29 percent of the total population (Lozano-Gracia and Garcia Lozano 2017). The area is characterized by the lack of institutional enforcement of land use regulations and urban planning, with the poorest part of the population (62%) living in areas of very-low or low environmental quality, whereas the middle and upper-income residents are found in the hilly southeastern part, further away from the coastline (Joseph et al. 2014, Joseph and Wang 2010).

We first identified 56 suitable agroforested and secondary forest remnants based on their accessibility, size, and site management and/or land history. From this initial set, we randomly selected eight secondary forest remnants (hereafter Fo), ranging in size between 2 ha and 11 ha, and eight agroforestry sites (hereafter Af; Fig. 1), ranging in size between 1.04 ha and 5 ha, with no significant difference in median value between the two categories (Wilcoxon-Mann-Whithney test, $z = 1.43$, $P = 0.153$). Elevation of selected sites varied between 100 and 1852m for Fo sites and between 172 and 1665 m for Af ones, with no difference between site categories ($z = 0.38$, $P = 0.713$). These sites are spread over four municipal administrations (Kenscoff, Pétion-Ville, Carrefour and Port-au-Prince), with a mean distance between two sites of 5 304 m (max = 15 727m, min = 337m). The eight Af corresponded to sites where agroforestry is practiced by local people in a traditional way, whereas the eight Fo are privately-owned, remains of secondary deciduous and shrubby forest with little exploitation, apart from occasional collection of fruits, fibers, other non-timber forest products and water. In both types of sites, the arboreal vegetation was dominated by *Hura crepitans* L., *Pinus occidentalis*, *Swietenia mahogany* L., *Delonix regia* Raf, and *Azadirachta indica*. In Af, this arboreal vegetation was associated with agricultural

crops such as *Zea mays* L., *Cajanus cajan*, *Ipomoea batatas* L., *Dioscorea* spp., *Musa* spp., *Phaseolus* spp., araceae spp. At higher altitude, coniferous trees were associated with *Daucus carota* L., *Brassica oleracea* L., *Petroselinum sativum* Hoffm., *Solanum tuberosum* L., and *Allium porrum* L.. Each site was characterized by its area size (ha), elevation (m), and the proportion of built environment in a 500m-radius around the boundary of each site, using Google earth satellite imagery at a scale of 1:1740 and QGIS 3.4. Built environment corresponded to buildings, industrial sites, roads, etc., as opposed to vegetated environment, defined as the physical areas covered with vegetation, gullies, agricultural patches, etc.

2.2. Bird census

The occurrence and the relative abundance of bird species at the different study sites were recorded by the same observer (JME) from January 4, 2018 to April 30, 2018. The period of data collection coincided with the migrating bird season in Haiti (birds generally visit Haiti from September to April; Latta *et al.*, 2006). For each site, two transect lines of equal length were plotted and treated as single sample, using a Garmin Dakota 20 GPS portable system. Transect length at each site was equal to the square root of the total surface area of the site as proportional sampling has been previously shown to be effective in characterizing species–area relationships (Schoereder *et al.* 2004, Ding *et al.* 2013). Transect length therefore varied between 59.3 and 137.3 meters, depending on the total surface area. The orientation of the transect lines followed natural paths, with a distance between the two transects varying between 100 and 200 m. Census were conducted while walking slowly along transect lines. Each contacted bird (visually and/or acoustically), on both sides of the transect line, was recorded at a perpendicular distance of up to 200 m from the transect line, using a telemeter (Leica DISTO TM D5). There was a minimum 10-min interval between the end of the first transect and the beginning of the second one, such that the probability to count the same bird twice on the two transects was low. A total of 10 surveys were made at dawn (5) and at dusk (5), on 5 different days for each site.

2.3. Statistical analysis

Species richness was computed based on the 10 census per site. Using the R software (R Development Core Team, 2020), the extrapolated cumulative diversity richness was

estimated by the random resampling method (Palmer, 1990), using bootstrap, jackknife of degree 1 and 2, and Chao method. The four estimates, that consider differently occurrences of rare species and the importance of replication (Chao, 1987; Colwell and Coddington, 1994), were calculated with the SPECPOOL function of the Vegan-R-package (Oksanen *et al.*, 2019). Cumulative diversity richness curves, which provide a good estimate of the representativeness of the sampling effort, were also drawn to estimate the number of species by extrapolation with infinite sampling and compare between Ag and Fo sites. The curves were obtained with the SPECACCUM function of the Vegan-R package.

Extrapolation of species diversity and three classical species diversity indexes, Shannon diversity index, Simpson diversity index, and Pielou's evenness index, were calculated for the 16 sites. The Shannon diversity index is the exponential of Shannon entropy (H' , which is calculated as $-\sum p_i \ln p_i$, where p_i is the proportion of counted individuals for species i). It is low under strong dominance of one single species and increases with the increasing number of species in an assemblage, especially if they are equally distributed (Magurran, 2004). In contrast, the Simpson diversity index (λ ; inverse of Simpson concentration index) is biased toward dominant species. The Pielou's evenness index, J , is a common and reliable index of relative evenness, i.e. the amount of evenness relative to the maximum and minimum possible for a given richness (Peet, 1974; Odum and Barrett, 2004; Jost, 2010). It varies between 0 and 1, and expresses how evenly individuals counted at a site are distributed among the different observed species. The less variation in communities between species, the higher J is.

The compositional dissimilarity among the 16 sites was tested using both the Jaccard index and the Horn–Morisita index, using the Vegan-R-package (Oksanen *et al.*, 2019). The Jaccard dissimilarity index is used to quantify the compositional dissimilarity between different sites, based on counts at each site. The Horn-Morisita overlap index is less sensitive to hyper-abundant species, and is able to handle different sample sizes and abundance data. It is bounded between 0 and 1, where 0 means the two sites have the same composition (that is they share all the species), and 1 means the two sites do not share any species. Ordination of the 16 sites was then graphically displayed by non-metric multidimensional scaling (NMDS), in two dimensions, using Jaccard or Horn-Morisita distances (Oksanen *et al.*, 2019). The analysis of multivariate homogeneity of group (Fo and Ag) dispersions (variances) was performed using the BETADISPERSER function. Non-euclidean distances between

objects and group centroids were handled by reducing the original distances to principal coordinates (PCo), while distances between groups were tested with PERMUTEST. We assessed the influence of geographical distance on dissimilarity between bird assemblages (as measured from the Jaccard and Horn-Morisita distances) between sites using Mantel tests within the vegan R package (Oksanen *et al.*, 2019), with the Spearman correlation method and 9999 permutations.

Given the limited number of observed species (see results), functional similarity was simply assessed through comparing the overall proportions of three distinct species categories (insectivores, granivores+frugivores+nectarivores, carnivores+omnivores) between Fo and Ag sites using a Fisher exact test.

Finally, we used Spearman rank order correlations to assess the influence of site characteristics (site area size (ha), elevation (m), proportion of built environment within a 500m-radius extending from the boundary) on observed and estimated (Chao) species richness and diversity (Shanon and Simpson) at each site.

All statistical tests were considered significant at the 0.05 level.

3. Results

Overall, 2 903 birds belonging to 40 species (including 11 endemics), 21 families and 35 genera, were recorded during the study, out of which 1 505 from 32 species were observed in Fo sites and 1 398 from 34 species in Ag sites (Table 1). Species accumulation curves for the 16 sites overlapped largely (Fig. 2), irrespective of habitat type, and most of them reached a plateau, indicating good sample representativeness. This was further confirmed by the equivalence between the extrapolated number of species and the cumulative number over 10 transects observed for 75% of the sites (Table 2). The total number of all observed bird species per site ranged from 4 to 22, with a median of 14.5 (interquartile range = 12-18). There was no significant difference between Ag and Fo sites in terms of species richness or diversity (Fig. 3; see Table 2 for tests). Endemic bird species richness ranged from 1 to 5, with a median of 4 (interquartile range = 3.75-5). The total number of sites in which a species was observed was positively correlated with its overall cumulative abundance (Spearman rank-order correlation, $r_s = 0.922$, $n = 41$, $p < 0.0001$), with only one species, the Vervain Hummingbird, *Mellisuga minima*, being found in all the sites. However, the cumulative

number of birds observed over the 10 surveys at each site was independent of site area size (Spearman rank-order correlation, $r_s = 0.02$, $p = 0.94$).

The ordination of bird communities by non-metric multidimensional scaling (NMDS, displayed in two dimensions, using either Jaccard or Horn-Morisita distance), and the analysis of multivariate homogeneity of groups for the 16 sites (Fo vs. Ag; Fig. 4) indicated very weak dissimilarity between avian assemblages among the 16 sites, with a large overlap (permutation test for the homogeneity of multivariate dispersions, $F = 2.97$; $p = 0.09$; $df = 14$ and $F = 3.34$; $p = 0.1$; $df = 14$). However, the degree of dissimilarity of assemblages between pairs of sites significantly increased with increasing geographic distance between pairs of sites (Mantel test, $r = 0.4146$; $p = 0.0001$ and $r = 0.4325$; $p = 0.0004$ with Jaccard and Horn-Morisita distances, respectively).

The distribution of species according to the three functional categories (see methods) did not differ between Fo and Ag sites (Fisher exact test, $p = 0.99$).

Overall, percentage of built habitat in a 500m-radius around a site had a negative influence on both species richness (Spearman rank order correlation; observed number of species: $r_s = 0.579$, $n = 16$, $p = 0.0188$, Fig. 5; Chao index: $r_s = 0.561$, $n = 16$, $p = 0.0237$), and diversity (Shannon: $r_s = 0.559$, $n = 16$, $p = 0.0244$; Simpson: $r_s = 0.493$, $n = 16$, $p = 0.0521$). However, the proportion of insectivores among observed bird species at each site was independent of the percentage of built area in a 500m-radius around a site ($r_s = -0.235$, $n = 16$, $p = 0.38$). Finally, neither site area size nor elevation had a significant influence on species richness or diversity ($p > 0.33$).

4. Discussion

4.1. Global pattern of urban bird diversity in the Port-au-Prince area

Overall, our results show that small urban green spaces located in the densely populated metropolitan area of Port-au-Prince are of importance for various bird species, including species of conservation interest, as previously observed in other geographical areas (Mortberg 2001, Godard et al. 2010, Carbó-Ramirez and Zuria 2011, Shih 2018, Mayorga et al. 2020). We recorded 40 different bird species recorded in 16 small urban green areas in the Port-au-Prince area during our study period, what compares with the results obtained by Carbó-Ramirez and Zuria (2011) who reported 39 different bird species in 19 small green spaces (ranging in size from 0.1 to 2ha) in the city of Pachuca, Mexico. The total number of

observed bird species amounted to about 15% of the total number of bird species known for Haiti, in accordance with previous accounts showing that urban bird assemblage generally consist of a limited number of species that represent a subset of the regional species pool (Blair 1996, Chace and Walsh 2006, Dale et al. 2015). However, observed species richness slightly underestimated true species richness for 25% of the sites, as shown by species accumulation curves. Indeed, six locally common resident bird species, the Red-tailed Hawk, *Buteo jamaicensis*, the Green Heron, *Butorides striata virescens*, the Green-tailed Warbler, *Microligea palustris*, the Yellow-crowned Night-heron, *Nyctanassa violacea*, the Hispaniolan Spindalis, *Spindalis dominicensis*, and the Antillean Siskin, *Spinus dominicensis*, were not recorded during the present study, although they are known to occur in the Port-au-Prince area (J.M. Exantus unpublished data). In addition, a few other uncommon species can be occasionally observed in the Port-au Prince area as shown by data available from eBird (eBird 2021).

Interestingly, urban agroforestry sites did not differ significantly from urban secondary forest remnants in terms of species richness, species diversity, or functional groups. In addition, there was an overall large overlap in the composition of avian assemblages between the two types of habitat, as shown by non-metric multidimensional scaling. The most abundant species tended to occur in similar numbers in both habitat types, whereas most species that were observed at one of the two site categories actually occurred in relatively low numbers, with the exception of the Black-faced Grassquit, *Melanospiza bicolor*, that was relatively abundant but only in agroforest sites. In contrast, Muhamad et al. (2013) observed that avifauna in agroforests clustered separately from that in remnant forest in West Java. However, this study investigated species richness and diversity at sites that were much larger in size and more heterogeneous than the ones we studied in the Port-au-Prince area. The high similarity in bird assemblages observed in the present study is thus probably related to similarity in site area size and arbustive vegetation between the two types of habitat. Similarity between avian assemblages between Af and Fo sites might also result from a general phenomenon of homogenization of bird communities associated with urbanization (Luck and Smallbone 2011).

The observed value of urban agroforestry for birds in the Port-au-Prince area is probably related to the way it is practiced by local people for subsistence (Bannister and Nair, 2003). Traditionally, a part of the previous vegetation is maintained and domesticated

plant species such as maize, beans, coffee trees, bananas, manioc, or sweet potatoes are planted, whereas onions, parsley, carrots, leeks tend to be planted at higher altitudes, as can be observed in the Kenscoff area. Very little or no pesticide is used (J.M. Exantus pers. obs.). The combination of plant diversity, reduced use of pesticides and presence of large trees (>10 cm DBH) may then contribute to make the agroforestry habitat suitable for a large range of bird species thus explaining the observed absence of difference with secondary forest remnants during the study period. Our study was however limited to four months during the dry season, such that further sampling during the wet season is necessary to better document the similarity in avian assemblages between forest and agroforest sites. As some bird species may exploit different food and habitats between the two seasons (see Carbó-Ramirez and Zuria 2011, Sainz-Borgo 2016), extending the sampling period to cover a whole year cycle might be of interest to better document variation in avian species richness and assemblage composition in space and time in the Port-au-Prince area.

4.2. Factors affecting avian species richness and diversity

There was a strong negative influence of the percentage of built habitat around urban green areas on both species richness and diversity. Similarly, both Carbó-Ramirez and Zuria (2011) and Amaya-Espinel et al. (2019) found a negative relationship between species richness and the percentage of area covered by buildings in Neotropical urban environments. Connectivity, even between small urban green areas, might therefore be important for maintaining the richness and diversity of avian assemblages, particularly for small green areas. The influence of percentage of built habitat in the immediate surroundings of the study sites may also contribute to explain the observed positive relationship between the degree of dissimilarity between avian assemblages and geographical distance among the 16 urban green areas. More generally, our results illustrate the importance of taking into account the heterogeneity of the urban matrix when studying the importance of urban areas for species diversity (Werner 2011).

Site area had no influence on both species richness and diversity, in contrast with previous studies reporting a positive influence of size area on resident bird species in urban green spaces (Mörtberg 2001, Chang and Lee 2016, Shih 2018, Mayorga et al. 2020). On the other hand, the absence of any effect of elevation on the same variables is in accordance with recent data on bird assemblages across both Italian (Fattorini et al. 2018) and Mexican

cities (MacGregor-Fors et al. 2021). One possibility is that area size is more important for bird abundance than for bird species richness and diversity, as observed by Morelli et al. (2017) in Beijing's urban parks. However, the cumulated number of birds observed at each site was unrelated to site area in the present study. Our results might be partly explained by the fact that the range of variation in site area size in the present study was limited (median value = 3.55 ha, interquartile range = 3 ha), as the particularly intense anthropogenic pressure drastically limits the size of urban green spaces in the Port-au-Prince area. Still, Carbó-Ramirez and Zuria (2011) reported a positive influence of area size when examining species richness in very small urban greenspaces in a Mexican city. Further investigations should therefore address the precise influence of area size in connection with other explanatory variables on avian assemblages and bird density in urban green spaces.

Reduced light pollution in the Port-au-Prince might also contribute to maintaining avian diversity. Compared to most large cities in the Northern Hemisphere, the degree of light pollution at night is low in the capital city of Haiti due to unstable and erratic supply of electric power with regular power cuts at night (Stuebi and Hatch 2018). Recent evidence shows that light pollution has significant ecological, behavioural, physiological, and health impacts on bird species (Dominioni 2015, Sun et al. 2017, Taufique et al. 2018, Jiang et al. 2020). Comparison with other urban environments with various levels of light pollution in the insular Caribbean might thus be of help to understand to what extent urban avian diversity is influenced by artificial light at night.

4.3. Importance of urban green areas for avian species of conservation interest

Most species observed in urban green spaces in the Port-au-Prince area during the study period were common residents or winter common residents in Haiti (Keith et al. 2003, Latta et al. 2006), in accordance with what has been observed in other urban green areas (Chace and Walsh 2006, Dale et al. 2015). However, we also observed three bird species of conservation interest, including two endemic species classified as vulnerable (VU) in the IUCN red list (IUCN 2016).

The presence of the Hispaniolan Parrot, *Amazona ventralis*, was recorded only at one single forest site (Fo3), located in Pétiön-Ville. This now vulnerable species was locally abundant on Hispaniola by the turn of the 20th century (Wetmore and Swales 1931), but has dramatically declined since then due to habitat loss, hunting and poaching for the pet trade

(Keith et al. 2003, Luna et al. 2018). Several parrot species have been able to colonize the urban environment (Martens et al. 2013, Groom et al. 2014, Irumba et al. 2016, Recio et al. 2016), and Luna et al. (2018) recently observed that the relative abundance of the Hispaniolan Parrot was three times higher in cities than in natural habitats in the Dominican Republic. The origin of Hispaniolan Parrots in the Port-au-Prince area remains to be identified, as they could correspond to escapees, releases of captive birds or wild birds moving into the urban environment. Future observations will be valuable to document temporal and spatial variation in urban habitat use and resource exploitation by Hispaniolan Parrots in Haiti. An important question for the future is to what extent the urban habitat in the Port-au-Prince area can provide suitable breeding conditions for this charismatic species.

A second vulnerable species, the White-necked Crow, *Corvus leucognaphalus*, was observed at two Fo (Fo1 and Fo7) and three Ag sites (Ag2, Ag3 and Ag6). This Caribbean-endemic corvid species formerly occurred in Hispaniola, Puerto Rico, and St. Croix, but is today restricted to Hispaniola, including Île à Vache (Brooks and Davalos 2001), Île de la Gonaïve, and Isla Saona (Wiley 2006). Habitat loss, extreme climatic events, illegal hunting, shooting to protect crops, diseases and interactions with invasive vertebrate species are regarded as the main factors responsible for the general decline of the White-necked Crow, although their relative importance may differ between islands (Wiley 2006). Again, additional information is needed to better document fluctuations in numbers and habitat use by White-necked crows in the Port-au-Prince area. Interestingly, both corvids and parrots are known for advanced cognitive processing (Lambert et al. 2019) and associated features, such as large relative brains and telencephalic volumes, high levels of sociality, high longevity, slow development and a long period of parental investment (Sol et al. 2005, Iwaniuk et al. 2005, Schuck-Paim et al. 2008, Sayol et al. 2020, Uomini et al. 2020). Although commonness and ecology appear to be the main predictors of urban living in birds (Dale et al. 2015), cognitive abilities and associated features might be of importance for explaining the occurrence of relatively rare and endangered species in the urban environment.

A third species of conservation interest observed in the Port-au-Prince area during the present study is the evolutionary distinctive (Jetz et al. 2014) Limpkin, *Aramus guarauna*. This mainly solitary, medium-sized marsh bird occurs throughout the peninsular Florida, the Greater Antilles, and Central and South America. Considered as Least Concern in the IUCN red list (IUCN 2016), it is seemingly uncommon in Haiti, threatened in the Dominican

Republic (Keith et al. 2003, Latta et al. 2006), and has drastically declined in Puerto Rico (Nytych et al. 2015). Limpkins are usually tolerant of humans and their presence in relatively small urban parks (< 30ha) has previously been reported for Mexico (Molina et al. 2017) and Brazil (Hipolito and Sazima 2018). Limpkins mainly favor borders of wooded streams and open marshes with moderate vegetation cover where they can exploit apple snails (genus *Pomacea*), their favourite food source (Alexander and Hepp 2014, Marzlof et al. 2019, Sandoval & García-Renova 2019). However, they can occasionally forage under tree cover and feed on lizards, frogs, insects, crustaceans and worms (del Hoyo et al. 1996). Trees various size are used by Limpkins for roosting, sunning, and preening. Accordingly, Limpkins were regularly observed perching in trees, often calling loudly, at three Ag and three Fo sites in the Port-au-Prince area. All the six sites are situated at little distance from one or two ponds, streams or reservoirs, such as *Rivière Froide* (Ag2, Ag3, Ag6, Fo8) *Mare Bizoton* (Ag2, Ag6), *Rivière Diègue* (Fo3), and *Réservoir de la source de Bois Moquette* (Fo4). The maintenance of tree cover in the Port-au-Prince area might therefore be important as roosting sites for Limpkins foraging in nearby streams and ponds, and, possibly, for nesting.

4.4. Conclusion

Overall, our results suggest that a few hectares of urban green spaces can be beneficial to bird protection in densely-populated and highly deforested urban areas. Additional surveys would however be helpful to assess to what extent small urban green spaces in the Port-au-Prince area also benefit other taxa such as amphibians, reptiles and arthropods. Similar investigations should also be conducted in other large cities both in Haiti and elsewhere in the insular Caribbean to better understand which features of the urban landscape are most beneficial to wildlife in the region, as little information is available, except for Puerto-Rico (Ordóñez Barona et al. 2020).

The conservation value of urban patches of agroforest in the Port-au-Prince area, similar to that of patches of remnant secondary forests for bird assemblages, clearly deserves further consideration. Urban agroforestry can provide local people with several benefits associated with the sale of tree products on actively cropped fields, more secure tenure and more fertile soil (Bannister and Nair 2003, Ordóñez Barona et al. 2020). In addition to alleviating poverty, the development of urban agroforestry in Haiti could provide various ecosystem services and environmental benefits (Jose 2009, Leakey and Simons

1998). In the context of the implementation of the United Nations Sustainable Development Goals in Haiti (Monbeuil 2020), adapted management of urban agriculture in the Port-au-Prince area might thus be particularly relevant. Finally, maintaining significant levels of biodiversity within the urban landscapes may promote positive emotions and enhance well being (Cameron et al. 2020), a particularly welcomed benefit for a human population plagued with extreme poverty and post-traumatic disorders following the 2010 earthquake (Cenat et al. 2020).

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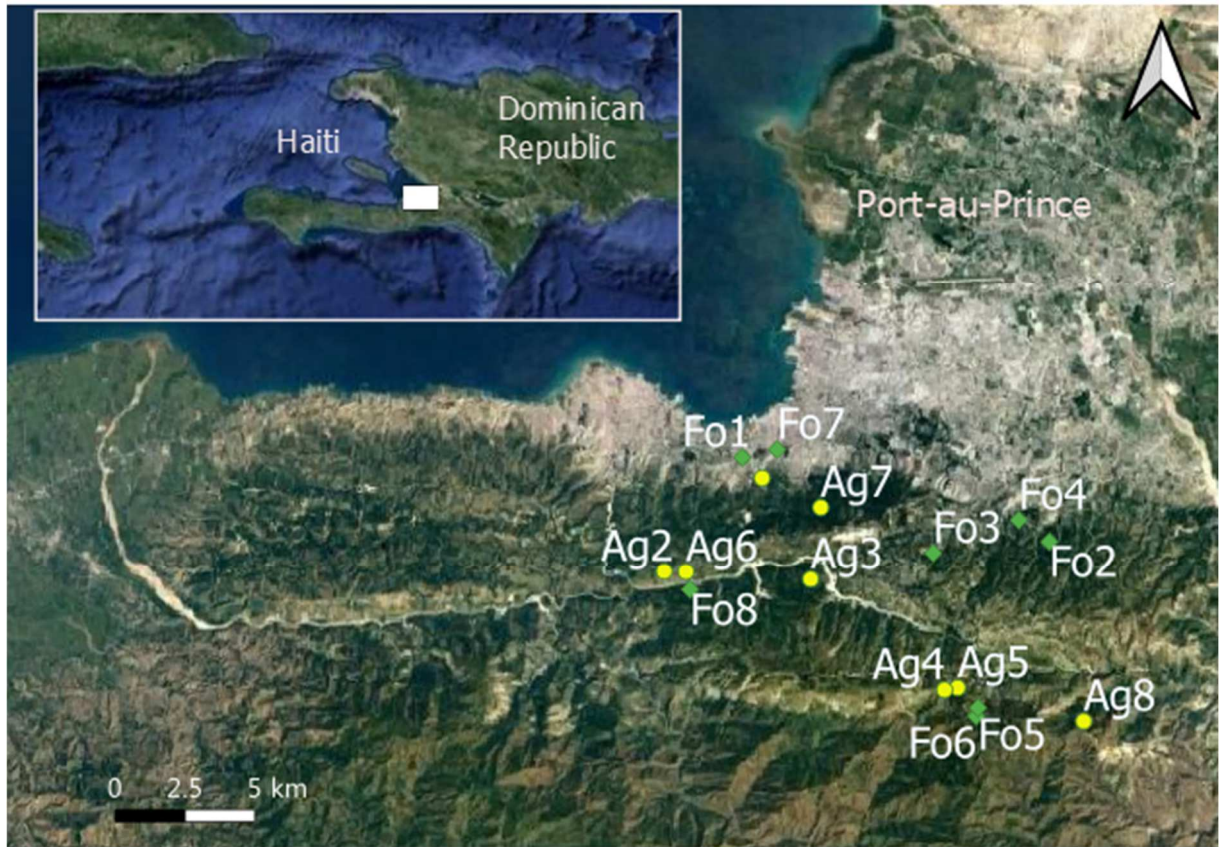


Fig 1. Map of the 16 study sites in Port-au-Princes, Haiti. Fo are forest sites and Ag are Agroforest sites.

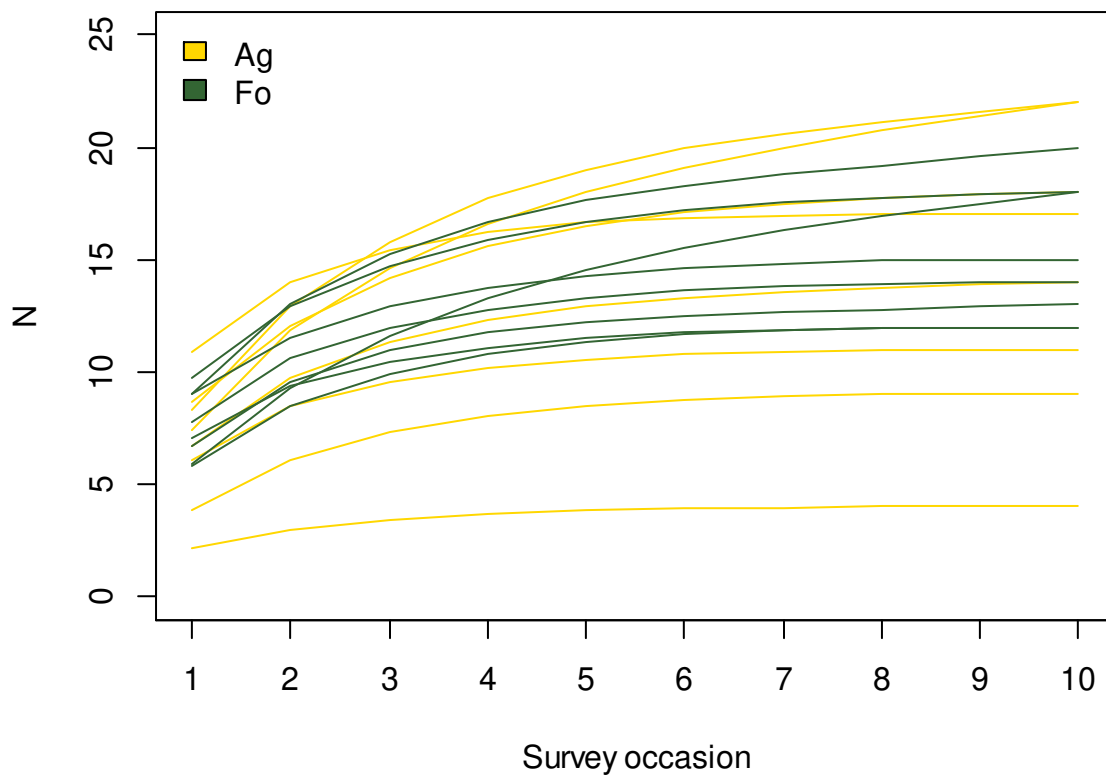


Fig. 2. Bird species accumulation curves of the 8 agroforests (Ag) and 8 Forest (Fo) sites in the Port-au-Prince area, Haiti.

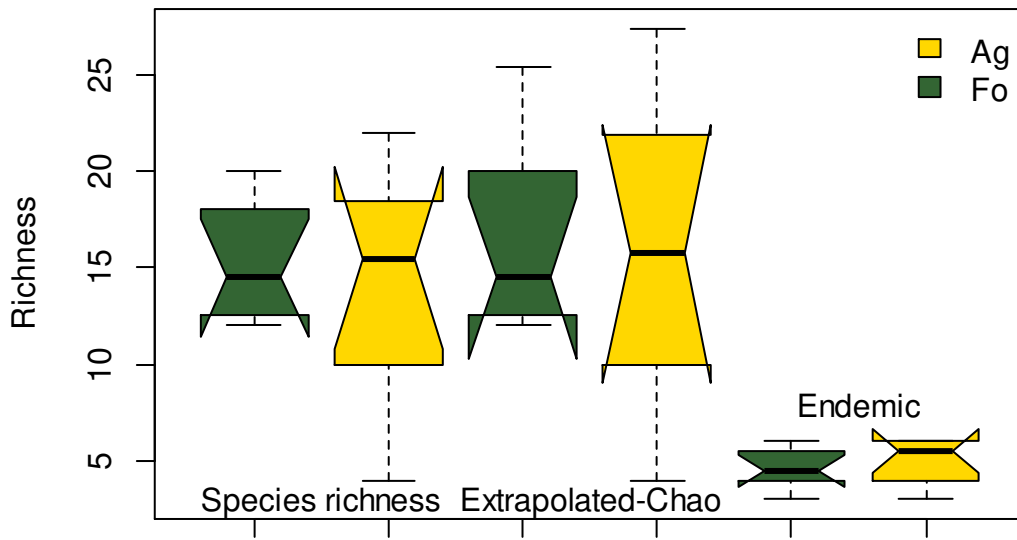


Fig. 3. Notched box plots for observed species richness, extrapolated-Chao richness, and number of endemic bird species for the 8 forest (Fo) and 8 agroforest (Ag) sites in the Port-au-Prince area, Haiti.

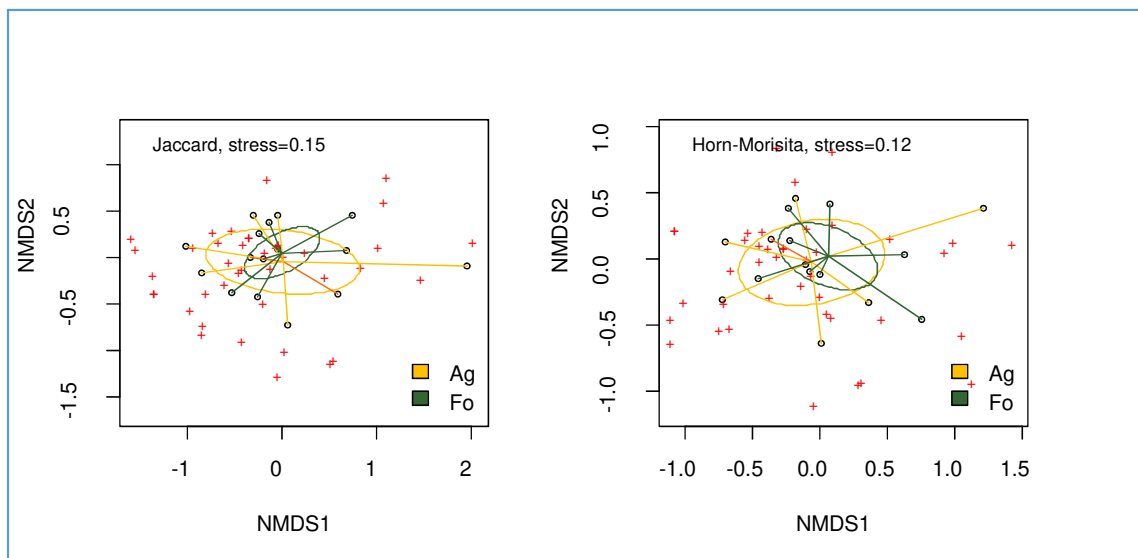


Fig. 4. Bird community ordination by non-metric multidimensional scaling (NMDS, in two dimensions display; using Jaccard (left) and Horn-Morisita (right) distances. The two groups (8 Forests vs. 8 Agroforests) are overlapping with both 'spider' diagrams where each point is connected to the group centroids and both standard errors of the average of scores (ellipses) overlapping. Each dot represents a site coordinate while each cross represents a factor bird species.

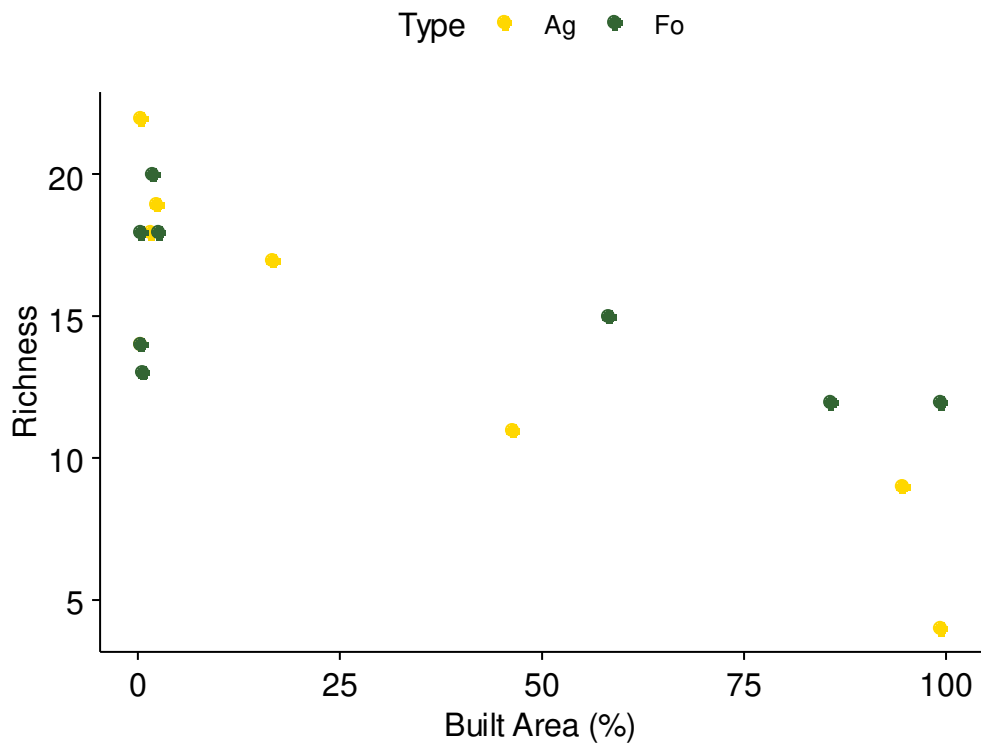


Fig. 5. Species richness according to the percentage of built area number in a 500-m radius around the sites.

Table 1

Cumulative number of observed birds per species (listed in alphabetical order) in the agroforest and forest sites in the Port-au-Prince area, Haiti during the study period; * denotes Hispaniola-endemic species.

Species	Agroforest	Forest	Total
<i>Amazona ventralis</i> *	0	8	8
<i>Anthracothorax dominicus</i>	14	19	33
<i>Aramus guarauna</i>	15	20	35
<i>Bubulcus ibis</i>	0	4	4
<i>Charadrius vociferus</i>	0	11	11
<i>Chlorostilbon swainsonii</i> *	0	4	4
<i>Coccyzus longirostris</i> *	4	8	12
<i>Coereba flaveola</i>	76	117	193
<i>Columbina passerina</i>	73	38	111
<i>Contopus hispaniolensis</i> *	3	16	19
<i>Corvus leucognaphalus</i> *	20	8	28
<i>Crotophaga ani</i>	34	7	41
<i>Dulus dominicus</i> *	295	439	734
<i>Falco sparverius</i>	14	0	14
<i>Icterus dominicensis dominicensis</i> *	7	0	7
<i>Melanerpes striatus</i> *	156	165	321
<i>Melanospiza bicolor</i>	63	0	63
<i>Mellisuga minima</i>	45	55	100
<i>Melopyrrha violacea</i>	18	28	46
<i>Mimus polyglottos</i>	36	13	49
<i>Mniotilta varia</i>	8	12	20
<i>Parkesia noveboracensis</i>	15	0	15
<i>Passer domesticus</i>	33	18	51
<i>Patagioenas inornata</i>	1	0	1
<i>Phaenicophilus palmarum</i> *	49	41	90
<i>Quiscalus niger</i>	15	0	15

<i>Seiurus aurocapilla</i>	7	26	33
<i>Setophaga caerulea</i>	19	14	33
<i>Setophaga discolor</i>	2	0	2
<i>Setophaga ruticilla</i>	42	81	123
<i>Setophaga tigrina</i>	12	4	16
<i>Tachornis phoenicobia</i>	2	1	3
<i>Todus angustirostris</i> *	0	20	20
<i>Todus subulatus</i> *	28	11	39
<i>Turdus plumbeus</i>	27	42	69
<i>Tyrannus dominicensis</i>	44	42	86
<i>Vireo altiloquus</i>	0	17	17
<i>Zenaida asiatica</i>	1	0	1
<i>Zenaida macroura</i>	199	214	413

Table 2

Comparison of average bird diversity between agroforest (Ag) and forest sites (Fo), in the Port-au-Prince district, Haiti. *W* = Wilcoxon-Mann-Whitney test (see methods for details).

<i>Sites</i>	<i>Species</i>	<i>Chao</i>	<i>Jack1</i>	<i>Jack2</i>	<i>Bootstrap</i>	<i>Shannon</i>	<i>Simpson</i>	<i>Pielou Evenness</i>	<i>Endemics</i>
	<i>W</i> = 29.5 <i>P</i> = 0.833	<i>W</i> = 30.5 <i>P</i> = 0.916	<i>W</i> = 30.5 <i>P</i> = 0.916	<i>W</i> = 31.5 <i>P</i> = 1	<i>W</i> = 31 <i>P</i> = 0.959	<i>W</i> = 34 <i>P</i> = 0.879	<i>W</i> = 34 <i>P</i> = 0.873	<i>W</i> = 16.5 <i>P</i> = 0.113	<i>W</i> = 38.5 <i>P</i> = 0.509
<i>Ag1</i>	11	11	11	10.3	11.2	2.25	0.88	0.33	6
<i>Ag2</i>	19	25.6	25.6	27.4	23.8	2.68	0.91	0.35	5
<i>Ag3</i>	18	18.225	18.9	18.3	18.7	2.48	0.88	0.34	3
<i>Ag4</i>	4	4	4	4	4.11	0.98	0.52	0.25	6
<i>Ag5</i>	9	9	9	8.3	9.2	2.11	0.85	0.33	6
<i>Ag6</i>	22	27.4	27.4	30.1	24.5	2.88	0.93	0.36	4
<i>Ag7</i>	14	14.45	14.9	15	14.6	2.42	0.89	0.43	4
<i>Ag8</i>	17	17	17	17	17.1	2.62	0.91	0.35	6
<i>Fo1</i>	12	12	12	11.3	12.2	2.22	0.86	0.41	5
<i>Fo2</i>	15	15	15	13.6	15.3	2.46	0.9	0.39	6
<i>Fo3</i>	18	21.75	22.5	24.4	20.2	2.38	0.87	0.37	4
<i>Fo4</i>	18	18.225	18.9	18.3	18.7	2.56	0.9	0.37	3
<i>Fo5</i>	12	12	12	10.6	12.3	2.31	0.88	0.33	5
<i>Fo6</i>	13	13	13.9	14.7	13.4	2.36	0.88	0.42	4
<i>Fo7</i>	14	14	14	12.6	14.3	2.34	0.88	0.34	6
<i>Fo8</i>	20	25.4	23.6	26.8	21.5	2.77	0.92	0.38	4