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Composition of avian assemblage in a protected forested area in Haiti: Evidence for recent decline of both forest-dependent and insectivore species

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ABSTRACT

Although past and present deforestation is regarded as a major environmental issue in Haiti, its direct impact on biodiversity remains largely undocumented. We assessed the composition of the avian assemblage at Parc National La Visite, a protected area in the Massif de La Selle, south-eastern Haiti, and compared our results to those obtained there about 15 years ago in a previous study. We combined mist-netting with visual observations and use of camera traps to document the presence and relative abundance of bird species over 13 field sessions from December 2019 to January 2022. We recorded the presence of 42 different bird species, belonging to 12 different orders, 26 families and 39 genera, during the course of our survey, including 16 (38.1 %) Hispaniola-endemics and seven North-American migrant species (16.7 %). Most species 81 % observed during our survey are currently classified as Least Concern in the IUCN red list, but eight species are of conservation concern, including one Near-Threatened species, five Vulnerable species and two Endangered. Accumulation curves and estimates of sampling completeness show that combination of mist-netting and visual observations was important in determining avian assemblage composition at Parc La Visite, whereas the contribution of camera traps was marginal. Although the relative proportions of mist-netted species according to their level of forest dependency or degree of insectivory did not differ with those recorded 15 years ago, there was a highly significant decline in the relative abundance (number of mist-netted individuals) of species with increasing level of both forest dependency and insectivory. Overall, our results indicate that Parc National La Visite remains an area of importance for avian conservation on Hispaniola. However, comparison with previous studies suggest that ongoing deforestation might be responsible for the local decline of the most sensitive bird species. We discuss the conservation implications of our results in relation to the present situation in Haiti, and make suggestions for adapted and realistic conservation strategies in order to protect avian diversity within forested ecosystems in the country.

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1. Introduction

Hispaniola, the second largest island in the Caribbean, is of high importance for avian conservation in the Neotropical Region. Divided between the Dominican Republic to the east and the Republic of Haiti to the west, it is home to over 300 bird species, of which 31 are endemics (Latta et al., 2006). Due to its central location in the Caribbean, Hispaniola is a major migration crossroads and the primary wintering site for several bird species (Keith et al., 2003; Landestoy et al., 2006). However, the island is facing severe environmental degradation due to strong anthropogenic pressure (Sambrook et al., 1999; Louis, 2018), particularly in Haiti characterized with extreme poverty, political instability, weak governance and high vulnerability to natural disasters (Felima, 2009; Pichler and Striessnig, 2013; Sutton, 2013; Gentes and Vergara-Castro, 2015; Joseph and Saffache, 2018; Pierre, 2020).

Of major conservation concern are the consequences of intense deforestation in terms of avian biodiversity loss in Haiti, where about 68 % of the extant avifauna consists of arboreal species (Latta et al., 2006; Exantus et al., 2021). In addition to habitat loss, the combination of deforestation and climate change may increase desiccation in small forest fragments (Smith et al., 2023) and thus reduce prey availability to insectivores. Forest disturbance and fragmentation may then have a stronger impact on tropical insectivorous bird species, particularly those which are unable to increase their use of the deforested countryside (Şerkercioğlu et al., 2002; Sherry, 2021), including second-growth forests (Blake and Loiselle, 2001; Stratford and Souffer, 2013) and plantations (Canaday, 1997; Barlow et al., 2007).

Deforestation has taken place in Haiti since colonial times, and is mainly driven today by agricultural expansion and charcoal production (Hosier, 1989; Paryski et al., 1989; Gibbons, 2010; Bellande, 2015). Although the current percentage of tree cover is estimated to vary between about 20 % and 30 % (Churches et al., 2014; Wampler et al., 2019; Pauleus and Aide, 2020; Rodrigues-Eklund et al., 2021), it has been estimated that Haiti has lost more than 99 % of its original primary forest (Hedges et al., 2018). In an attempt to preserve forested habitat, the Haitian authorities have established since 1983 a protected area system consisting of several terrestrial sites (Sergile, 2008). However, these areas remain exposed to intense human pressure, including ongoing deforestation, and lack effective management and conservation policies (Dolisca et al., 2007a, 2007b; Rimmer et al., 2010; Gentes and Vergara-Castro, 2015; Salomon et al., 2021a, 2021b).

Despite this critical situation, information about the current conservation status of the Haitian avifauna remains scarce due a combination of factors. First, local scientific expertize, particularly in key disciplines of conservation biology, is very limited (see however the recent contributions by Rodriguez-Silva et al., 2020; Exantus et al., 2021; Saint-Louis et al., 2022; Beaujour and Cézilly, 2022; Paul et al., 2022). Second, access to natural areas is difficult due to the poor condition of rural roads, cost of transportation, and weak local logistic support (The World Bank, 2018). Third, the country is currently plagued with increasing violence, with armed groups frequently blocking roads, kidnaping and ransoming people, making birdwatching and visits to natural areas quite risky, if not dangerous (Kovats-Bernat, 2002).

In this context, we present new and original results about avian species richness and diversity in the "Parc National La Visite" (hereafter Parc La Visite), located in the western part of the Massif de La Selle, 22 km south-southeast of the capital city of Port-au-Prince. Although this protected area was established by governmental decree in 1983, its exact boundaries were only defined 30 years later. Today, Parc La Visite officially covers 11,426 ha, with a perimeter of 106.13 km. However, the forested part of the park is limited to approximately 3 000 ha of dry forest, semi-deciduous forest, rain forest and cloud forest along an about 1800-m altitudinal gradient (Davalos and Brooks, 2001). Since June 2017, the park is part of the transnational La Selle/Jaragua-Bahoruco-Enriquillo UNESCO Man and Biosphere Reserve that extends to the Dominican Republic. Culminating at 2680 m, "Pic La Selle" is the highest point in the park, whereas three other peaks, La Visite, Cabaio and Tête Opaque, are above 2125 m. The area is famous for its rich biodiversity and high levels of animal and plant endemism (Judd, 1987; Hedges, 1999; Huber et al., 2010; Brace et al., 2012; Cano-Ortiz et al., 2016). It is home to eight globally threatened bird species, two-thirds of Hispaniola's endemic birds, and is the most important breeding site for both the endangered Black-capped Petrel, *Pterodroma hasitata*, and the vulnerable La Selle Thrush, *Turdus swalesi* (Davalos and Brooks, 2001; Rimmer et al., 2010). However, the forest ecosystem in the park is threatened by human activities such as agriculture, livestock and logging (Dolisca et al., 2007a, 2007b). In particular, forested areas are regularly damaged by slash and burn agriculture, with direct impact on the flora and fauna (Calixte, 2015). The situation recently worsened, first following the 2010 earthquake and then more recently, as a consequence of uncontrolled street violence led by gangs, both resulting in the massive displacement of populations from the densely-populated metropolitan area of Port-au-Prince, the capital city of Haiti, to the countryside (Lu et al., 2012; Kato and Lee, 2022).

We collected data on the local avifauna in Parc La Visite over a 14-months period, from December 2019 to January 2022 (Appendix A). In order to reduce bias in catchability and detectability, in particular due to some forest bird species being potentially more secretive than others, we combined mist-netting with visual observations and the use of camera traps (Remsen, 1994; Sunwarat et al., 2015; Zamora-Marin et al., 2021; Jean-Pierre et al., 2022). We compared sampling completeness based on the different methods and their combination using species accumulation curves (Willott, 2001) and assessed the influence of various factors on variation is species richness and number of mist-netted birds. In order to assess to what extent ongoing forest deforestation at Parc La Visite did affect the composition of the local avian community, we quantitatively compared our results with those obtained in 2005 by Rimmer et al. (2010). Given ongoing and severe deforestation at Park La Visite we expected a decrease in the relative number of species and their global abundance according to both their level of forest dependency and their level of insectivory. We discuss our results in relation to other observations conducted at Parc La Visite, deforestation, general decline of forest insectivorous birds, and conservation priorities.

2. Material and methods

2.1. Study area

We collected data on avian assemblages during the course of a long-term study of the population of the La Selle Thrush (LST), *Turdus swalesi*, in the locality of "Tête Opaque"(18°20.928' N, 072°14.347' W), in the central zone of Parc La Visite, at an elevation varying between 2146 and 2260 m (Fig. 1a). The site is located north of the first municipal district of Baie d'Orange, city of Belle Anse, in the South-Eastern department of Haiti. It is close to Morne Cabio, the highest peak in Parc La Visite. We used motorbikes to access the study area via the narrow mountain road from Port-au-Prince via Furcy and Ca Jacques, followed by transportation in hilly areas by mule, covering on each trip an overall distance of ca 55 km in about 6 hrs. We conducted 13 consecutive field surveys at a single field station in Parc la Visite between December 2019 and January 2022, thus allowing comparison between seasons. In Haiti, the "dry season", characterized by lower amounts of rainfall, lasts from November to March, whereas the "wet season" lasts from April to October. However, the altitudinal gradient favors the presence of a humid climate throughout the year, with maximum rainfall going from above 400 mm in October to 160 mm in February (UNEP, 2010).

The local vegetation consisted essentially of deciduous trees, coniferous trees and grasslands such as *Casearia ilicifolia* Vent., *Didymopanax tremulus, Aristolochia cordiflora Mutis ex Kunth, Arthroslylidium haitiense, Bocconia frutescens, Persea anomala, Cyathea arborea, Eucalyptus* spp., *Prunus persica, Fragaria vesca, Elymus repens L., Jacquinia aculeata* Mez., *Elymus repens*. The *Pinus* spp. was the most dominant tree and the Hispaniolan Pine (*Pinus occidentalis*) was the most widespread tree species in the study area (Fig. 1b). The main human activities consisted in agriculture and livestock. The main cultures were Asteraceae Spp, Solanacea (*Solanum tuberosum* L), Brassicaceae Spp, garden peas (*Pisum sativum sativum*), leek (*Allium porrum*), red beets (*Beta vulgaris vulgaris*), parsley (*Petroselinum crispum*), carrots (*Daucus carota sativus*), and corn (*Zea mays L.)*. Livestock included bovids, ovids, equids and pigs. To increase their land in order to grow more crops, local people, some of them being only 10-year old, are regularly clearing and burning the deciduous forest. We regularly observed local residents setting fire under *Pinus* spp. (Fig. 2a) to create space for adaptive crops such parsley, potatoes or carrots, and cutting coniferous trees to make boards and wooden poles for house construction (Fig. 2b) as well as deciduous trees for the local production of cement and lime (Fig. 2c).

2.2. Data collection

We collected data over 13 consecutive field sessions, from December 2019 to January 2022 (Appendix A). On each session, we used foot trails and pre-existing, human-created openings in vegetation to deploy mist nets (Ecotone ® 6 × 3 m, 19-mm mesh, most suitable to capture La Selle Thrushes and Ecotone ® 12 × 3 m, 30-mm mesh) along a total distance varying between 800 and 1000 m, depending on interference with agricultural activities by local people. Mist nests were then checked every 25 min, and, in a more irregular way, in response to hearing strange noises, to rapidly remove birds that were squawking loudly after being caught (mainly La Selle Thrush and Eastern Red-legged Thrush, *Turdus ardosiaceus*), or to drive away free-ranging animals (bovids, ovids, equids and pigs) that ventured to close to the mist nests. We temporarily closed or did not operate mist nets in case of rain or strong wind.

In addition, we made regular observations of the avifauna using binoculars (KITE 8 × 23) in the surroundings of our banding station, covering an area of approximately 300 ha. However, we were not able to quantify precisely the number of individuals observed with binoculars on each sighting for each species, such that we only used the information to calculate species richness. Field sessions varied in duration from 3 to 15 days (mean = 11.08; median = 12.5), partly depending on weather conditions. Consequently, both capture effort and observation effort varied between sessions (Appendix A). Total length of mist nest varied progressively from 12 m to 96 m from the first to the last session and total time spent in capturing birds in mist nets per session varied between 20 and 108 h. Overall visual capture effort, as measured by time spent in observation using binoculars varied between 30 and 120 h. From the third session to the last, we set between 5 and 17 camera traps (Moultrie M8000i) in the area. We attached each single camera to a robust tree 20–30 cm above the ground. We selected trees at locations where the vegetation was not too dense, thus allowing for an optimal range of the camera sensor in closed habitats and ensure uniformity in the radius of action of each camera. Selected parameters

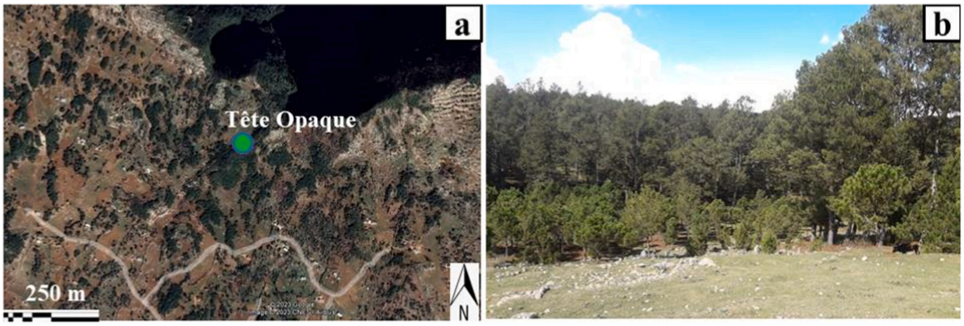

Fig. 1. Location of our study site "Tête Opaque" inside Parc La Visite (a) and view of the study area with *Pinus* spp as the most dominant and most widespread tree (b).

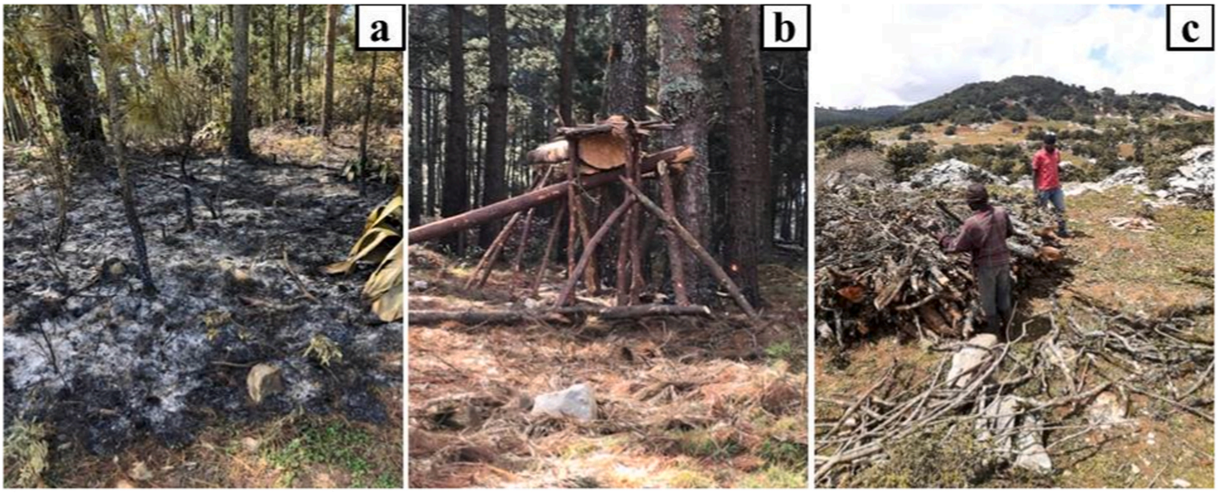

Fig. 2. Examples of local human impact on the forest in the study area: (a) setting fire under *Pinus* spp. to create space for adaptive crops; (b) fabrication of boards and wooden poles for house construction from coniferous tree cutting; (c) local people cutting deciduous trees for the local production of lime as cement for construction.

(high sensitivity of the shutter trigger, height positioning of the camera-trap, and distance of detection range) maximized the probability that ground-dwelling bird species, such as LST, present in the survey areas would be detected. We set the trigger to take one picture each time a movement was detected, with a 30-s delay between consecutive pictures to avoid multiple photographs of the same individual over short time periods. We initially intended to keep the camera-traps in activity in the field in our absence between consecutive sessions. However, after one camera-trap was stolen and because local people practiced burn agriculture around our study area, we decided to remove camera-traps from the study site between sessions, starting at the end of the 6th one. On each session, cameras were set to be operational 24 h day^{-1}. However, due to variation in battery life, all camera traps were not active for the same amount of time during each session. Given these limitations, data obtained from camera traps were only used to complement the estimation of species richness.

2.3. Analysis of incidence data

We classified bird species according to conservation status (as defined by the IUCN red list), body size, migratory status (migratory, resident), level of insectivory (on ordinal scale: non-insectivores, partial insectivores, strict insectivores), and forest dependency (on an ordinal scale: low, medium, high). Data on body size, migratory status, and diet were obtained from the Handbook of the Birds of the World (https://birdsoftheworld.org) and *Birds of The West Indies* (Kirwan et al., 2019), while information on forest dependency was obtained from BirdLife International (https://datazone.birdlife.org/home).

We first analyzed incidence data based on the simple presence or absence of species during each survey session. We drew three cumulative species richness curves to estimate the representativeness of the sampling effort, based on mist-net data only, mist-net data + visual observations, and mist-net data + visual observations + camera-trap data. Species accumulation curves were plotted using the "specaccum" function in package Vegan (v. 2.6.2; Oksanen et al., 2022), using the "collector" method to cumulate sampling sessions in chronological order. We then calculated overall species richness, i.e. total number of species observed using all combined survey methods for all surveys combined. As data on abundance were not available when combing the three survey methods, we computed Chao 2 for replicated incidence data from the 13 samples. The Chao 2 is the sum of the observed number of species and the quotient $q_1^2/2q_2$ where q_1 and q_2 are the number of unique species (i.e. detected in only a single sample) and the number of duplicate species (i.e. detected in only two samples, respectively (Gotelli and Colwell, 2011). We then examined the ratio between the total number of observed species and Chao 2 as an index of overall sampling completeness (Chao et al., 2020).

To assess to what extent differences in species richness between survey sessions was related to survey effort, we calculated capture effort for each session as the number of "net-hours" per unit time (i.e. multiplying the number of 12-m mist nets in use by the sum of the number of hours the nets were open; Ralph, 1976). We calculated observation effort simply as the total number of hours spent observing the avifauna with binoculars per session. We did not assess observation pressure from camera traps as we could not check battery duration for each camera on several sessions. As sample size (n = 13 sessions) was relatively small, conventional tests had not enough power to detect deviations from both normality and the equal variance assumption (Morgan, 2017). We therefore relied on one-tailed non-parametric tests to assess the influence of capture effort and observation effort on variation in species richness and two-tailed non-parametric test to assess differences between the dry and wet seasons. The use of one-tailed tests was justified by the expectation of increased species richness and number of capture with increasing sampling effort (Poulin et al., 2000; Melo et al., 2003).

2.4. Analysis of abundance data

The analysis of abundance data was limited to mist-netted birds. We first estimated the extent of sampling completeness from mist-netting data only using the ratio of the number of mist-netted species to Chao 1 (Chao et al., 2020). The Chao-1 index is the sum of the observed number of species and the quotient $f_1^2/2f_2$, where f_1 and f_2 are the number of singletons (i.e. species represented by only a single individual in the sample) and doubletons (i.e. species represented by only two individuals in the sample), respectively (Gotelli and Colwell, 2011). The Chao 1 estimator thus uses information on the number of rare or infrequent species in the collection to estimate the number of undetected species (Chao et al., 2009). We then assessed the influence of body size on the cumulative number of mist-netted individuals per observed species using a Spearman rank correlation coefficient. As abundance data were only available from mist-netted birds, we examined the correlation between total numbers of mist-netted birds and capture effort using a one-tailed non-parametric test and compared median abundance between the dry and wet seasons using a two-tailed non-parametric test. As capture effort varied between sessions, we did not calculate indices of diversity for each survey session as their comparison would have been meaningless.

2.5. Change in avian assemblage composition and abundance through time

In order to assess potential recent change in avian assemblage composition at Parc La Visite, we compared our results to those

Table 1
List of bird species detected at Parc La Visite during the study period, with information on family, IUCN conservation status CS (LC = least concern, NT = near threatened, VU = vulnerable, EN = endangered), distribution status DS, (E = endemic, M = migrant, R = resident), diet ID (C = carnivore, F = frugivore, G = granivore, INS-P = partial insectivore, INS-S = strict insectivore), level of forest dependency FD (L = low, M = medium, H = high), and data on relative abundance according to capture method (MN = total number of mist-netted individuals, VO = number of sessions during which the species was visually observed, and CT = number of photos obtained with camera traps).

Family	Species	CS	DS	DIET	FD	MN	VO	CT
Accipitridae	Red-tailed Hawk (*Buteo jamaicensis*)	LC	R	C	L	1	13	0
Apodidae	Antillean Palm Swift (*Tachornis phoenicobia*)	LC	R	INS-S	N	0	6	0
Aramidae	Limpkin (*Aramus guarauna elucus*)	LC	R	C	L	0	6	11
Calyptophilidae	Western Chat-Tanager (*Calyptophilus tertius*)	VU	E	INS-P	L	19	8	4
Charadriidae	Killdeer (*Charadrius vociferous*)	LC	R	INS-P	N	0	9	0
Coerebidae	Bananaquit (*Coereba flaveola*)	LC	R	INS-P	L	43	12	2
Columbidae	Plain Pigeon (*Patagioenas inornata*)	NT	R	F, G	M	0	13	0
	Mourning dove (*Zenaida macroura*)	LC	R	G, F	L	20	13	0
Corvidae	Palm Crow (*Corvus palmarum palmarum*)	LC	E	INS-P	M	0	8	0
Cuculidae	Hispaniolan Lizard-Cuckoo (*Coccyzus longirostris*)	LC	E	INS-P	M	3	13	25
Emberizidae	Yellow-faced grassquit (*Tiaris olivaceus*)	LC	R	G	N	1	3	0
Falconidae	American Kestrel (*Falco sparverius*)	LC	R	INS-P	L	1	10	0
Fringillidae	Antillean Siskin (*Spinus dominicensis*)	LC	E	G	M	135	10	0
	Hispaniolan Crossbill (*Loxia megaplaga*)	EN	E	G	H	0	3	0
	Hispaniolan Spindalis (*Spindalis dominicensis*)	LC	E	INS-P	L	339	12	0
Hirundinidae	Golden Swallow (*Tachycineta euchrysea sclateri*)	VU	E	INS-S	M	2	12	0
Ictéridae	Greater Antillean Grackle (*Quiscalus niger*)	LC	R	INS-P	L	0	8	0
Mimidae	Northern mockingbird (*Mimus polyglottos*)	LC	R	INS-P	L	5	11	0
Parulidae	Black-and-white Warbler (*Mniotilta varia*)	LC	M	INS-S	M	3	3	0
	Ovenbird (*Seiurus aurocapilla*)	LC	M	INS-P	M	6	10	69
	Black-throated Blue Warbler (*Setophaga caerulescens*)	LC	M	INS-P	M	55	9	3
	Yellow-throated Warbler (*Setophaga dominica*)	LC	M	INS-P	M	3	1	0
	American Redstart (*Setophaga ruticilla*)	LC	M	INS-P	M	6	5	6
Picidae	Hispaniolan Woodpecker (*Melanerpes striatus*)	LC	E	INS-P	M	1	11	0
Procellariidae	Black-capped Petrel (*Pterodroma hasitata*)	EN	R	INS-P	H	0	9	0
Psittacidae	Hispaniolan Parakeet (*Psittacara chloropterus*)	VU	E	F, G	L	0	6	0
Scolopacidae	Wilson's Snipe (*Gallinago delicata*)	LC	M	INS-P	L	2	4	0
Thraupidae	Black-crowned Palm-Tanager (*Phaenicophilus palmarum*)	LC	E	INS-P	L	7	12	0
	Black-faced grassquit (*Melanospiza bicolor*)	LC	R	G	L	82	12	1
	Greater Antillean Bullfinch (*Melopyrrha violacea*)	LC	R	F, G	M	912	13	76
Todidae	Narrow-billed Tody (*Todus angustirostris*)	LC	E	INS-S	H	0	7	0
Trochilidae	Hispaniolan emerald (*Riccordia swainsonii*)	LC	E	INS-P	H	205	13	0
	Vervain Hummingbird (*Mellisuga minima vielotti*)	LC	E	INS-P	M	3	7	0
	Green-tailed Warbler (*Microligea palustris*)	LC	E	INS-S	M	89	13	39
	Hispaniolan Mango (*Anthracothorax dominicus*)	LC	R	INS-P	L	2	9	0
Trogonidae	Hispaniolan Trogon (*Priotelus roseigaster*)	LC	E	INS-P	H	0	3	0
Turdidae	Rufous-throated Solitaire (*Myadestes genibarbis*)	LC	R	INS-P	H	7	2	6
	Eastern Red-legged Thrush (*Turdus ardosiaceus*)	LC	R	INS-P	M	48	13	56
	La Selle Thrush (*Turdus swalesi*)	VU	R	INS-P	H	8	12	754
	Bicknell's Thrush (*Catharus bicknelli*)	VU	M	INS-P	H	1	0	3
Tyrannidae	Hispaniolan Pewee (*Contopus hispaniolensi*)	LC	E	INS-P	M	7	12	0
	Loggerhead Kingbird (*Tyrannus caudifasciatus*)	LC	R	INS-P	M	0	1	0

obtained during the last survey of avian assemblage conducted at Park La Visite in 2005 (Rimmer et al., 2010). This survey lasted for seven days (26 January–1 February) and combined mist-netting with point counts and incidental observations to assess avian assemblage. It took place at two different sites differing in elevation, "Bèrak" (18°19.73' N, 72°17.61' W; 1175–1250 m a.s.l.; 6.35 km from our study site), and "La Visite" (18°20.91'N, 12°16.88 W; 1995–2060 m a.s.l.; 4.51 km from our study site). In order to make valid comparison of avian assemblage compositions and avoid potential bias due to seasonal variation and altitude, we restricted our own data set to the 2nd (25–27 January 2020), 6th (17–31 January) and 13th session (24–31 January 2022). We pooled data from the three sessions to perform comparisons with data obtained by Rimmer et al. (2010), Table 1) at both Bèrak and La Visite. As the methodology differed between the two studies, we restricted the quantitative analysis of assemblage composition to mist-netting data. We relied on both the Jaccard index and the Morisita-Horn index to compare overlap in assemblage composition between the two studies. The Jaccard index is an incidence measure (i.e. based on the presence or absence of species) that compares the number of shared species to the total number of species in two assemblages (Jost et al., 2011). The Morisita-Horn index is an abundance index based on the squared differences of the relative abundances of each species in the two assemblages (Jost at al, 2011).

Because capture effort was not identical between the two studies, we limited further comparisons to the relative proportions of species and individuals according to forest dependency and level of insectivory using non-parametric tests for contingency tables, such that comparisons are independent of the total number of mist-netted birds in each study. We used Fisher's exact tests to compare the relative proportions of captured species according to both level of forest dependency and level of insectivory. We then used ordinal logistic regression to assess to what extent the global abundance of captured species according to level of forest dependency or level of insectivory differed between the two studies. We relied on one-tailed tests as we expected a decrease in number of species and abundance in according to their level of forest dependency and their level of insectivory. As Rimmer et al. (2010) used playbacks to attract Bicknell thrushes (BT) to mist nests and since we placed our own mist nests as to favor captures of LST, we conducted the analyzes with and without data from these two species. Because Rimmer et al. (2010) used a larger mesh size (36 mm) than during the present study, possibly missing a larger number of smaller birds, we controlled for a possible increase in species size according to level of forest dependency or insectivory using a Jonckheere test for ordered alternatives (Siegel and Castellan, 1988).

All statistical tests were performed using JMP (v.10) and Statext v3.6 (https://www.statext.com/). Results were considered significant at the 0.05 level.

3. Results

3.1. Analysis of incidence data

Overall, we recorded the presence of 42 different bird species, belonging to 12 different orders, 26 families and 39 genera, during the course of survey, including 16 (38.1 %) Hispaniola-endemics and seven North-American migrant species (16.7 %), (Table 1). Out of them, 12 (28.6 %) were detected at least once with each of the three methods during the course of the study, 16 species (38.1 %) were both mist-netted and observed through binoculars, one (2.4 %) was mist-netted and photographed and one (2.4 %) was observed through binoculars and photographed, whereas 12 (28.6 %) were only observed with binoculars. Although, overall, no species was detected only with camera traps, three different species were detected only with this technique on one single (but not the same) session. Species richness, combining all three methods, varied between consecutive sessions, ranging from 21 to 34 (median value = 28), and tended to increase with both capture effort (Spearman rank correlation coefficient, $r_s = 0.4703$, n = 13, $P = 0.0525$ one-tailed) and observation effort ($r_s = 0.4480$, n = 13, $P = 0.0624$, one-tailed). There was no marked difference in assemblage composition between seasons, as 90.5 % of species were observed during both the dry and wet season.

Accumulation curves (Fig. 3) show that combination of mist-netting and visual observations was important in determining avian

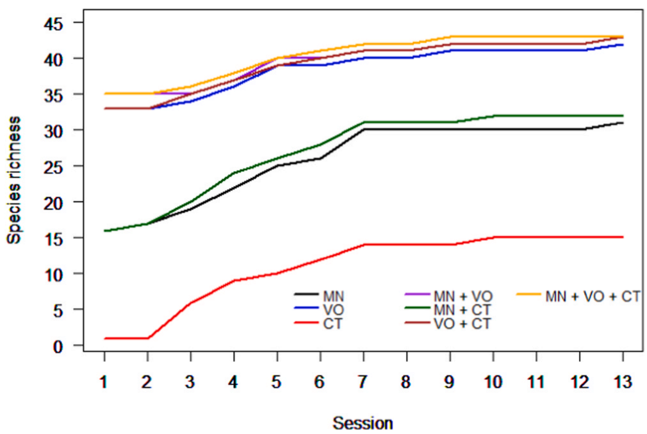

Fig. 3. Species accumulation curves over consecutive survey sessions according to different combinations of methods (MN: mist nets, VO: visual observations and CT: camera traps).

assemblage composition at Parc La Visite, whereas the contribution of camera traps was marginal (Fig. 4). Over the 13 sessions, there was only one singleton species, the Loggerhead Kingbird, *Tyrannus caudifasciatus*, and only one doubleton species, the Bicknell's Thrush, *Catharus bicknelli*. The value of the Chao 2 index was therefore 42.5, thus corresponding to an estimated completeness value of 98.8 %.

3.2. Analysis of abundance data based on mist-netted birds

Overall, 2210 captures, belonging to 30 different species were made with mist nests during the course of the study (Table 1). However, the Greater Antillean Bullfinch, *Melopyrrha violacea*, accounted on its own for 41.3 % of all captures. At the other extreme, our sample of mist-netted bird species included five singletons and one doubleton, leading to a value of the Chao 1 index equal to 42.5 and a completeness value of 70.6 %. As expected, the number of mist-netted birds per session increased with capture effort ($r_s = 0.5275$, n = 13, $P = 0.0320$, one-tailed test). Among all species observed at our study site, the cumulative number of birds captured in mist nets ranged from zero to 912 (median = 3) and decreased significantly with body size (Spearman rank-correlation coefficient, $r_s = -0.37$, n = 42, $P = 0.0162$; excluding data from LST and migrant species: $r_s = -0.40$, n = 29, $P = 0.0185$). The median number of mist-netted birds per session was significantly higher during the wet season compared to the dry season (Kruskal-Wallis test, $X^2 = 5.22$, d.f. = 1, $P = 0.0223$), although capture effort did not differ significantly between seasons ($X^2 = 0.08$, d.f. = 1, $P = 0.7751$).

3.3. Conservation status of species

From a conservation point of view, 81 % of species recorded during our survey are classified as Least Concern in the IUCN red list. However, one Near-Threatened species, five Vulnerable species and two Endangered species were captured or observed at Tête Opaque during our surveys (Table 1). The proportions of strict insectivorous species, partial insectivorous species and non-insectivorous species were 14.3 %, 61.9 % and 23.8 %, respectively. According to BirdLife International classification, 7.1 % of the observed species are not dependent on forests, 31 % have low dependency on forests, 40.5 % have medium dependency, and 21.4 % have high dependency.

3.4. Change in assemblage composition and abundance through time

Rimmer et al. (2010) mist-netted 19 and 16 different bird species at the Bèrak and La Visite station, respectively, in 2005, while we mist-netted 22 different species at Tête Opaque in 2020–2022 when pooling data from our 2nd, 6th and 13th sessions. Compositional similarity based on incidence data was higher between Tête Opaque and Bèrak (15 shared species, Jaccard Index = 0.577) than between Tête Opaque and La Visite (11 shared species, Jaccard Index = 0.407). The reverse was observed for compositional similarity based on abundance (Morisita-Horn index, Tête Opaque vs. Bèrak = 0.460, Tête Opaque vs. La Visite = 0.841). The difference between the two indices was explained by the large prevalence of the Greater Antillean Bullfinch among mist-netted individuals at both Tête Opaque and La Visite, accounting, respectively, for 46.7 % (n = 272) and 30.6 % of captures (n = 62), whereas, the same species accounted for only 10 % (n = 120) of captures at Bèrak.

Fig. 4. La Selle Thrush (a); Eastern Red-legged Thrush (b); Bicknell's Thrush (c); Limpkin (d).

We found no significant difference between the three sites in the proportions of mist-netted species according to their level of forest dependency (Table 2, Fisher exact test, $P = 0.7737$). However, the proportions of captured individuals belonging to the different categories of forest dependency differed significantly between the three sites (Ordinal logistic regression, $X^2 = 6.74$, d.f. = 2, $P = 0.0344$), even after removing BT and LST from the analysis ($X^2 = 9.28$, $P = 0.0096$). The difference was largely due to the marked reduction in the relative proportion of captured individuals belonging to species with high dependency on forests as the proportions of individuals belonging to species with low and medium dependency did not differ significantly between the three sites (Chi-square test $X^2 = 1.20$, d.f. = 1, $P = 0.5478$). Similarly, the proportions of mist-netted species according to their level of insectivory did not differ between the two studies (Table 3, Fisher exact test, $P = 0.6478$). However, the proportions of mist-netted individuals according to their level of insectivory differed significantly between the three sites (Ordinal logistic regression, $X^2 = 106.38$, d.f. = 2, $P < 0.0001$). Again, removing records from both BT and LST did not affect the results ($X^2 = 102.34$, $P < 0.0001$). The results were not confounded by difference in mesh size of mist-nets between the two studies as the body size of mist-netted species did not increase with increasing level of forest dependency (Jonckheere test for ordered alternatives, $z = 0.05$, $P = 0.4816$) or increasing level of insectivory ($z = -0.50$, P 0.6526).

4. Discussion

Our results indicate that Parc National La Visite remains an area of importance for avian conservation on Hispaniola (Davalos and Brooks, 2001; Rimmer et al., 2010), and, more globally, at the insular Caribbean level (Sergile, 2008). Because of the dramatic situation in Haiti, we were unable to conduct our survey at regular time intervals and did not spend a similar amount of time in the field on each session. In addition, because of logistic constraints, we acquired progressively the different devices used for sampling the avian community during the course of the study, such that sampling effort was not constant across sessions. Therefore, our study obviously suffers from some limitations associated with the non-standardization of the protocol (Ralph et al., 2004; Gotelli and Colwell, 2011). However, cautious statistical analysis of the results allowed us to bring new and important information about avian assemblage composition at one of the major protected forested area in Haiti. In particular, comparison with previous results obtained about 15 years ago (Rimmer et al., 2010) are strongly suggestive of a decline in abundance of forest-dependent and insectivore bird species.

4.1. *Combining different methods to assess species richness*

From a methodological point of view, combining mist nests with visual observations through binoculars and camera traps increased the completeness of our survey by about 40 % compared to using mist nets only, in accordance with results from recent studies (Zhang et al., 2018; Fontúrbel et al., 2020). Surveys of forest birds using exclusively mist-nets are known to result in incomplete assessment of assemblage composition and diversity (Catry et al., 1999; Dantas et al., 2011; Mulvaney and Cherry, 2020) because of biases associated with mist-net height and variation in the behavior of bird species (MacArthur and MacArthur, 1974; Remsen and Good, 1996; Tattoni and LaBarbera, 2022). Observations with binoculars allow for a better detection of large and less mobile species that tend to be underrepresented in mist-netting data (Estades et al., 2006), while camera-traps are particularly efficient for the study of secretive ground-dwelling forest species (Jean-Pierre et al., 2022). This was largely confirmed in the present study, with larger species tending to have a lower probability of being mist-netted at least once. In addition, camera traps documented the presence of several understorey species such as the Ovenbird, *Seiurus aurocapilla*, the La Selle Thrush (Fig. 4a), the Eastern Red-legged Thrush, *Turdus ardosiaceus* (Fig. 4b), and the discrete and cryptic Bicknell Thrush (Fig. 4c) (McFarland et al., 2018). Although we made only a limited usage of camera traps during the present study, we strongly encourage their use in future studies as they allow for the collection of a large amount of information, including the presence and activity of bird species and that of their potential predators (Jean-Pierre et al., 2022). Indeed, we regularly recorded the presence of several exotic mammal species such as the Small Indian Mongoose, *Urva auropunctata*, feral domestic cats, *Felis catus,* stray dogs, *Canis lupus familiaris*, and rats, *Rattus* spp. In addition, although camera trapping has been mainly used to study ground-dwelling bird species, arboreal trapping methods have recently been developed (Gregory et al., 2014; Zhu et al., 2021). Combining arboreal and ground-level camera trapping might be of particular interest to better assess vertical habitat selection and niche breadth of tropical forest species (Marra and Remsen, 1997; Liu et al., 2020).

The combination of several different methods thus appears particularly suitable for the monitoring of avifauna in Haitian forests. In that respect, the additional use of autonomous sound recorders may nicely complement other methods and provide additional information about most elusive species (Shonfield and Bayne, 2017; Pérez-Granados and Traba, 2021). More to the point, the

Table 2
Comparison of number of mist-netted species (N) and cumulative abundance (total number of individuals and percentage) according to their level of forest dependency between "Bèrak and "La Visite" in 2005 (Rimmer et al., 2010) and "Tête Opaque" in 2019–2022 (this study).

	Level of forest dependency					
	Low		Medium		High	
	N	Cumulative abundance	N	Cumulative abundance	N	Global abundance
Bèrak (2005)	5	20 (16.7 %)	9	70 (58.3 %)	5	30 (25.0 %)
La Visite (2005)	4	9 (14.5 %)	7	41 (66.1 %)	5	12 (19.35 %)
Tête Opaque (2020–2022)	7	42 (15.4 %)	12	205 (75.4 %)	3	25 (9.2 %)

Table 3
Comparison of number of mist-netted species (N) and cumulative abundance (total number of individuals and percentage) according to their level of insectivory between "Bérak" and "La Visite" in 2005 (Rimmer et al., 2010) and "Tête Opaque" in 2019–2022 (this study).

	Level of insectivory					
	Non insectivores		Partial Insectivores		Strict insectivores	
	N	Global abundance	N	Global abundance	N	Global abundance
Bèrak (2005)	2	13 (10.8 %)	14	76 (63.3 %)	3	31 (25.8 %)
La Visite (2005)	2	4 (6.45 %)	11	43 (69.4 %)	3	15 (24.2 %)
Tête Opaque (2020–2022)	4	140 (51.7 %)	17	117 (42.4 %)	1	15 (5.9 %)

cost-effectiveness of alternative methods should be carefully evaluated when designing a survey or monitoring program (Zamora-Marin et al., 2021), particularly in a country like Haiti where only a few trained ornithologists are present and where areas of importance of avian biodiversity are often difficult to access by conventional means of transportation. The use of automated recording devices such as camera traps and sound recorders can enable researchers to optimize time spent in the field, while increasing spatial and temporal coverage. In addition, they allow for a permanent record of the data collected and reduction of observer bias. Their use can however be limited by the cost of equipment, limited storage of recordings, loss of data when units fail, and theft and vandalism (Shonfield and Bayne, 2017; Meek et al., 2019). Indeed, during the present study, we recorded some attempts at vandalizing our camera traps from traces of machete blows on the protective handcuffs and the trees to which they were attached. One camera trap, together with its security handcuff lock and protective box, was stolen, after someone cut all the clumps of shrubs to which it was firmly attached. Subsequent discussions with local people informed us that there was some fear that pictures taken camera traps could be used to identify people who destroy the forest. At the time of the camera was stolen, the place was already completely cleared and to be replaced by agricultural crops. Still, automated recording devices would be of great help to improve the monitoring of the Haitian avifauna in the future if local researchers could benefit from international financial help for the acquisition of maintenance of such refined equipment (Speaker et al., 2022).

4.2. Factors influencing interspecific variation in presence and abundance

We could only estimate relative abundance of bird species from mist-netting data. Although assessing abundance from mist-net captures can be misleading (Remsen and Good, 1996), it remains valid here as all sessions took place in the same location and same mist-net height. Two factors may contribute to explain observed interspecific variation in number of mist-netted individuals. First, larger avian species typically occur at lower abundances because of their higher energy and resource requirements compared with smaller species (Jetz et al., 2012). Accordingly, we observed a significant negative relationship between number of mist-netted individuals and species' body size. However, this relationship may be partly influence with differences in flight height and behavior associated with size differences between species. On the other hand, we mainly used mist nets with a 19-mm mesh size, particularly suitable for medium size passerines, such that smaller passerines may have escaped more frequently than larger species (Jenni-Eiermann and Jenni, 1996). Second, we observed some significant seasonal variation in global abundance, independently of differences in capture effort between sessions, with more birds being mist-netted during the wet season than during the dry one. This pattern is similar to what has been observed in other forest bird assemblages of some tropical forest mountains (Malizia, 2001; Dinesen et al., 2022; but see Serra Thomas et al., 2020), possibly reflecting seasonal variation in food abundance (Blake and Loiselle, 1991; Poulin et al., 1993).

4.3. Species of conservation interest

Several species of conservation interest according to the IUCN red list were present during our surveys. The endangered Hispaniolan Crossbill, *Loxia megaplaga*, was observed only on three different sessions, as single individuals or in pairs. The species seems to have declined steadily since the 1930s when it was reported to be abundant at La Visite (Wetmore and Swales, 1931) and has been only observed in sporadic small flocks since then (Woods and Ottenwalder, 1986; Davalos and Brooks, 2001; Rimmer et al., 2010). The Hispaniolan Crossbill coevolved with and is highly dependent on the Hispaniolan Pine, the only native cone-bearing conifer (Latta et al., 2000; Parchman et al., 2007). Its local abundance appears to vary with annual fluctuations in cone pine productivity (Latta et al., 2000), such that a multiyear survey would be necessary to better ascertain demographic trends at Parc La Visite. However, the ongoing and intense deforestation observed at Tête Opaque, particularly affecting Hispaniolan Pines through burning and logging, may put the species at risk of severe population decline, if not local extinction.

The endangered Black-capped Petrel, *Pterodroma hasitata*, was opportunistically heard or observed only at night on nine different sessions. The species has been known to nest at Tête Opaque for several years (Woods and Ottenwalder, 1992) and is threatened by habitat destruction and predation of adults and juveniles at nest sites by introduced mammalian predators (Satgé et al., 2021). However, the species was observed to occupy burrows within cultivated patches, especially those adjacent to step cliffs.

The vulnerable Hispaniolan Parakeet, *Psittacara chloropterus*, was observed with binoculars on six different sessions, but only in small groups of up to 13 individuals.The species was considered to be abundant in the area in the 1930s (Wetmore and Swales, 1931), and still common about 40 years ago, as gathered from observations of flocks of up to 30 individuals (Woods and Ottenwalder, 1986).

Since then, the species appeared to have declined, and Davalos and Brooks (2001) did not observed it during a four-day survey in January 2020 while Rimmer et al. (2010) observed only two pairs on a single occasion in January 2005. The species is threatened by deforestation, shooting, illegal trade (Keith et al., 2003; Luna et al., 2018) and, possibly, competition with the Olive-throated Parakeet, *Eupsittula nana* recently introduced on Hispaniola (Latta et al., 2012). In the Dominican Republic, its relative abundance appears to be negligible in rural areas today and very low in natural habitats (Luna et al., 2018). The largest/densest known population of the Hispaniolan Parakeet is actually located in the Santo Domingo urban area where the species forms breeding colonies in old buildings (Geary et al., 2021). However, the role of this urban population as seed dispersers is very limited, and the persistence of populations in natural habitats is important to conserve ecological functions (Luna et al., 2018). Hispaniolan Parakeets are obligate cavity nesters and seem to rely on cavities produced by the Hispaniolan Woodpecker, *Melanerpes striatus* (Latta et al., 2006). We regularly observed the Hispaniolan Woodpecker (on 11out of 13 sessions) during our study, potentially explaining the presence of the Hispaniolan Parakeet. However, additional information is needed to better document population trends and breeding success of the species at Parc La Visite.

The vulnerable Western Chat-Tanager, *Calyptophilus tertius*, was regularly observed through binoculars and/or captured in mist nets (on 11 out of 13 sessions) on during our survey, and one individual was photographed while feeding on the ground on one session. The species then appears to remain locally common, confirming earlier observations (Rimmer et al., 2010).

The vulnerable Golden Swallow, *Tachycineta euchrysea sclateri*, was detected on all sessions but one, in numbers ranging from 1 to 3 individuals, although only three individuals were mist-netted. In accordance with previous reports (Woods and Ottenwalder, 1986; Rimmer et al., 2010), the species appears to be fairly common at Parc La Visite.

The vulnerable LST, which as the initial target species of our research program and whose area distribution is limited to the La Selle ridge on both sides of the border, was regularly mist-netted, observed and camera-trapped. In contrast, another turdid the migratory BT, was captured on only one occasion, but several individuals were camera-trapped while foraging on the ground on the sixth session (January 2021). This is in accordance with observations made by Townsend et al. (2010) in the Dominican Republic, where wintering individuals observed at high elevation (1600–1800 m) mainly feed on arthropods collected on the ground, whereas at lower elevation (350–600 m) the species is more arboreal and primarily consumes a fruit-based diet.

Interestingly, the near-threatened Plain Pigeon, *Patagioenas inornata*, was recorded on all sessions, in group size varying between 2 and 3 individuals, and was always observed in patches of broadleaf forest. The species was deemed as "uncommon almost everywhere" on Hispaniola by Keith et al. (2003), while Kirwan et al. (2019), reported its distribution to be limited to pine and broadleaf forests up to 1200 m. Accordingly, Davalos and Brooks (2001) did not observe the Plain Pigeon at Parc La Visite by and Rimmer et al. (2010) observed only three individuals at lower elevation than our study site. Our results show that the species distribution now extends to higher elevations.

Some species of least-concern observed during our study also deserve some attention. The Green-tailed Warbler, *Microligea palustris*, is relatively widespread but possibly declining in the Dominican Republic (Lloyd et al., 2016) and is considered rare and local in Haiti (McDonald, 1987; Kirwan et al., 2019). However, our data confirm, together with that of Rimmer et al. (2010), that the species is relatively abundant at Parc La Visite. Similarly, the Greater Antillean Bullfinch appears to be increasingly abundant, although it was formerly considered as uncommon to rare above 1500 m on Hispaniola (Keith et al., 2003) and has been recently declared as declining in the Dominican Republic (Lloyd et al., 2016). Another least-concern species, the Limpkin, *Aramus guarauna*, was observed in the broadleaf vegetation on six sessions and camera-trapped as well on one of them (Fig. 4d). Although the distribution of the species is generally associated with open wetlands with low vegetation and the availability of its preferred food, apple snails (*Pomacea* spp.; Dobbs et al., 2019; Marzolf et al., 2019), it can also forage in abandoned plantations or under tree cover and feed upon other prey types, including small vertebrates (del Hoyo et al., 1996; Currie et al., 2005). The species is reported to be increasingly local and rare on Hispaniola (Woods and Ottenwalder, 1986; Kirwan et al., 2019), although it was recently found to be relatively frequent in small urban green spaces close to water bodies in the metropolitan area of Port-au-Prince in Haiti (Exantus et al., 2021). It is generally present below 1500 m (Rimmer et al., 2010), with a single observation at 1675 m in moist broadleaf forest in Sierra de Bahoruco, Dominican Republic (Keith et al., 2003). Our results indicate that the species can actually exploit forested habitats at higher elevation and not necessarily close to water bodies. More intensive camera trapping at Parc La Visite may help identifying prey consumption by Limpkins there.

4.4. Change in species presence and abundance through time

We surveyed birds at Parc La Visite about 15 years after the last survey conducted by Rimmer et al. (2010) and found subtle but significant changes in assemblage composition. Contrary to our expectations based on similarity in elevation, assemblage at Tête Opaque was more similar to that recorded at Bèrak than at La Visite when considering species incidence (Jaccard index). The reverse was true when considering species abundance. However, the Horn-Morisita index is largely dominated by the most abundant species (such as the Greater Antillean Bullfinch in the present case) and poorly accounts for relatively rare species (Jost et al., 2011). More importantly, although the relative proportions of species according to their level of forest dependency or their level of insectivory did not differ between the two surveys, we found significant evidence for a difference in the relative proportions of captured individuals according to both their specific level of forest dependency and their specific level of insectivory. Overall, species highly dependent on forests and mainly insectivorous species appear to have declined in relative abundance over the last 15 years, suggesting that continuous deforestation does have an impact on the most sensitive avian species.

4.5. Habitat alteration and change in avian assemblage composition: conservation implications

We consider that the observed avian assemblage composition and differences with results from Rimmer et al. (2010), are the consequence of the ongoing deforestation and increasing agricultural activity at Parc La Visite. On the one hand, deforestation directly affects species which are highly dependent on tree species for their food and/or nest sites (Şerkercioğlu et al., 2002; Sherry, 2021). On the other hand, the replacement of forest habitat with cropland may favor some species over others, depending on their ability to exploit cultivated areas. For instance, the Greater Antillean Bullfinch was quite regularly observed feeding on garden pea crops, and two other abundant species, the Antillean Siskin and the Hispaniolan Spindalis, were also frequently observed feeding on various crops. The relative abundance of the La Selle Thrush might be explained by its ability to adapt to pastoral practices, as we regularly observed the species on patches of pasture, following cattle (sheeps and cows) to prey on seeds and invertebrates they stir up from the ground. In contrast, the related Red-legged Thrush was more wary of cattle and did not venture outside of the thick vegetation. The regular presence of the Red-tailed hawk was most probably related to the abundance of small passerines attracted to crops. Indeed, local peasants increasingly keep poultry in cages to prevent predation by hawks, probably leading this avian predator to switch to other preys. We captured a Red-tailed Hawk attempting to prey upon captured passerines in the mist-nests only once, during the 11th session. However, two mist-netted birds, one Eastern Red-legged Thrush and one Greater Antillean Bullfinch, were killed by this raptor on two different sessions. Predation by raptors on mist-netted birds is well known in the Northern Hemisphere (Spotswood et al., 2012; Clewley et al., 2018), but has been little documented so far in tropical forests. Mortality due to predation in mist-nets accounted for 0.09 % of all captures in the present study, a lower value than what has been previously reported for tropical forests (Ruiz-Esparza et al., 2012; Melo et al., 2018; Guimarães et al., 2020). Relatively short intervals between consecutive mist-net inspections may have contributed to limit predation by the Red-tailed Hawk in the present study.

4.6. Conclusion

Although Parc La Visite is officially protected, agriculture practices are still increasing. Local people cultivate small patches of crops within larger forested patches, at good distance from the main paths taken by occasional surveillance patrols from the Haitian authorities, thus remaining largely unnoticed. The ability to halt and reverse deforestation at Parc La Visite should be a priority, in accordance with recent recommendations of the 26th UN Climate Change Conference (COP 26) that took place in November 2021 in Glasgow. However, the situation is particularly complex due to land tenure regime, extreme poverty, reduced access to education, and local people's perception of forest conservation and biodiversity (Dolisca et al., 2007a, 2007b, 2009). The situation worsened following the 2010 earthquake, with massive immigration from the metropolitan area of Port-au-Prince. Although several people that settled at Tête Opaque at that time no longer live permanently there, they come back regularly to exploit forest patches that they cleared for agriculture and now regard as their property. In addition, small groups of local children aged 10 yrs old or less, those do not have access to school and education, commonly engage in clearing the forest to grow vegetables for their own subsistence (J.M. Exantus pers. obs.). The reality of the situation seems to have eluded most international organizations that pretend to contribute to environmental conservation in Haiti. Based on our observations and regular discussions with local people, we strongly advise international funding agencies to provide financial resources for the implementation of a new and more appropriate conservation program (see Chambers et al., 2020). This program should include the massive employment of local people with decent salaries to work on reforestation program and assist local scientists in wildlife monitoring, contingent on the abandonment of their farming activities. It should also include the development of an ambitious educational program for local children. In that respect local graduate students with a basic training in conservation biology could be recruited to both contribute to wildlife research and monitoring and children education. We see such a program as the only valid solution to save one the last natural forest of Haiti and a large part of its natural heritage.

Ethics approval

Access to the protected area, "Parc National La Visite", as well as the capture and handling of animals were carried out with the agreement of the Direction of Biodiversity in the Haitian Ministry of the Environment. The study complied with relevant laws and guidelines for capture and banding of birds.

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CRediT authorship contribution statement

Jean-Marry Exantus: was involved in study design and methodology, data collection, part of statistical analysis, and writing up of the article. **Frank Cézilly:** was involved in study design, supervision, project administration, statistical analysis and writing up of the article.

Declaration of Competing Interest

The authors report no competing of interest or personal relationships that could have appeared to influence.

Data Availability

Data will be made available on request.

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Appendix A. Mist-netting session dates, capture effort, observation effort, and number of captures at La Visite Park. For each session, capture effort is calculated as the number of "net-hours" per unit time while observation effort corresponds to the total number of hours spent in observation using binoculars (see Section 2)

Survey number	Year	Dates	Season	Capture effort (mh^{-1})	Observation effort (h)	Number of captures
1	2019	December 1–15	Dry	1092	104	78
2	2020	January 25–27	Dry	240	30	33
3	2020	February 22–26	Dry	432	38	29
4	2020	September 26–October 10	Wet	2592	120	149
5	2020	November 15–19	Dry	6732	120	173
6	2021	January 17–31	Dry	7128	120	129
7	2021	March 28–April 11	Wet	5346	120	332
8	2021	May 22–31	Wet	4896	82	304
9	2021	July 6–13	Wet	3456	62	118
10	2021	August 2–13	Wet	5472	98	230
11	2021	September 5–18	Wet	6048	100	309
12	2021	November 8–17	Dry	6912	82	216
13	2022	January 24–31	Dry	5952	73	110

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