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Effects of invasive alien plants and habitat structure on bird assemblages in a unique lowland forest in eastern Nepal

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Abstract Invasive alien species play a major role in biodiversity decline. We studied the impacts of invasive alien plant species (IAPS) on bird assemblages and their habitats in Jalthal Forest, a biogeographically unique lowland remnant forest in eastern Nepal. We surveyed birds by point counts and quantified environmental variables at 89 sites. Observations of 1,373 individuals of 86 species showed that species richness was the highest in wooded grasslands and floodplains and lowest in sal forests invaded by IAPS, whereas abundance was highest in IAPS-invaded forests. Abundance and richness decreased with increasing cover of IAPS, shrub cover and distance to water. The abundance and richness of every feeding guild except omnivores were negatively correlated with IAPS cover. Our results increase our understanding of the impacts of IAPS on birds and clearly indicate the need of controlling IAPS in Jalthal Forest, and thus, are valuable for forest management and biodiversity conservation.

Keywords: biodiversity, bird diversity, conservation, invasive plant species, tropical forest

Összefoglalás Az idegenhonos inváziós fajok jelentős szerepet játszanak a biológiai sokféleség fogyatkozásában. Jelen vizsgálatban az idegenhonos inváziós növényfajok (IIN) madáregyüttesekre és élőhelyeikre gyakorolt hatását tanulmányoztuk a Jalthal-erdőben, amely egy biogeográfiai szempontból egyedülálló alföldi maradványerdő Kelet-Nepálban. A madarakat pontszámlálással mértük fel, és a környezeti változókat számszerűsítettük 89 mintavételi helyszínen. 86 faj 1373 egyedének megfigyelése alapján a fajgazdagság a fás gyepeken és ártereken volt a legmagasabb és az IIN által előzőnlött szálafa-erdőkben volt a legalacsonyabb, míg az abundancia az IIN által előzőnlött erdőkben volt a legmagasabb. Az abundancia és a fajgazdagság csökkent az IIN borítottság, a cserjeborítás és a víztől való távolság növekedésével. A mindenevők kivételével az összes táplálkozási guild abundanciája és fajgazdagsága negatívan korrelált az IIN borítottsággal. Eredményeink hozzájárulnak az IIN fajok által a madarakra gyakorolt hatások megértéséhez, és világosan jelzik az IIN fajok gyérítésének szükségességét a Jalthal-erdőben, így módon értékesek az erdőgazdálkodás és a biológiai sokféleség megőrzése szempontjából.

Kulcsszavak: biodiverzitás, madarak sokfélesége, természetvédelem, inváziós növényfajok, trópusi erdők

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Dipendra Adhikari is deceased. The authors dedicate this paper in memory of the Late Dipendra Adhikari.

Introduction

Invasive alien species (IAS) are species established or introduced outside of their natural geographic range, which can successfully expand their range and harm biodiversity, economy, and human health (Davis *et al.* 2001). Invasive alien plant species (IAPS) are considered to have a major pressure on the current state of biodiversity globally (Butchart *et al.* 2010). IAPS destruct or alter the habitats where they are introduced (Weidlich *et al.* 2020), and often cause a cascade effect on the food web (Gan *et al.* 2009), therefore, the effects of invasion are not limited to a single trophic level.

Mikania micrantha and *Chromolaena odorata* are some of the world's most problematic IAPS (Weber 1960). Both species are native to Central and South America (Holm *et al.* 1977, Gautier 1992). In Nepal, *Ch. odorata* and *M. micrantha* was first reported in 1956 and in 1963, respectively (Shrestha & Shrestha 2021). With the latest addition of the invasive plant *Mimosa diplotricha* (Sharma *et al.* 2020), currently, there are 27 identified invasive plant species in Nepal, but the number of vascular alien plant species found in natural habitats is 183 (Shrestha 2019) and the number of alien animal species is 64 (Budha 2015). Four of these species (*Ch. odorata*, *Eichhornia crassipes*, *Lantana camara*, *M. micrantha*) are among the 100 most problematic IAPS in the world (Lowe *et al.* 2000). Invasive Alien Plant Species (IAPS) in Nepal are primarily found in the Tarai, Siwalik, and Mid-hill regions, which have similar climatic conditions to tropical and subtropical locations around the world (Shrestha 2016). Because tropical and subtropical areas are more vulnerable to IAPS invasions (Bhattarai *et al.* 2014, Shrestha 2016), it stands to reason that these parts of Nepal are also more likely to be invaded by such species.

The ability to propagate by roots, stem segments, and easily spreadable seeds helps *Ch. odorata* and *M. micrantha* to spread successfully and rapidly (Zachariades *et al.* 2009, Day *et al.* 2016). With the establishment of IAPS, the composition of the vegetation changes, and the structure of habitats also often changes in parallel, because vegetation plays a major role in defining habitat structure. For example, thickening of bushes is a change in habitat structure (Aleixo 1999), which may influence biodiversity and ecological processes (Seymour & Dean 2010). Since successfully established IAPS are known to alter the structure of habitats (Pearson 2009, Aravind *et al.* 2010), IAPS are also expected to influence the relationship between the richness of bird assemblages and habitat structure.

IAPS are expected to affect the bird diversity either positively by offering new habitat and increasing prey availability (Gan *et al.* 2009) or negatively by altering suitable habitat and decreasing prey availability (Pearson 2009, Aravind *et al.* 2010, Kessler *et al.* 2011). Nepal is known to host 892 species of birds, which accounts for 8.9% of the global species richness of birds (Grimmett *et al.* 2016, BCN 2023). Of these bird species, 39 are globally threatened and 31 are near threatened. The eastern part of Nepal shows biogeographical links with the Indo-Malayan, the Palearctic, and the Sino-Japanese regions (Kandel *et al.* 2018). From Jalthal Forest in eastern Nepal, a total of 230 bird species has been recorded by Sharma *et al.* (2021), among them eight species are globally threatened, 31 are nationally threatened and one is nationally protected. However, the number of bird species in the area is expected to be around 300 (Grimmett *et al.* 2016).

The impacts of IAPS on the abundance of birds are commonly reported. IAPS have the potential to change bird habitats, to provide non-nutritious foods, and to break functional relationships with their habitats (Flanders *et al.* 2006). Previous studies indicated that vegetation structure affects the birds' selectivity of their habitats (Karr & Roth 1971, Yahner 1982, Robinson & Holmes 1984, Dinanti *et al.* 2018). Insectivorous birds such as the Asian Green Bee-eater (*Merops orientalis*), Common Hawk-cuckoo (*Hierococcyx varius*), Large Cuckooshrike (*Coracina macei*), Black Drongo (*Dicrurus macrocercus*) and Spangled Drongo (*D. hottentottus*) are inhibited in their movement, flight, and insect capturing efficiency by the thick vegetation layer produced by many invasive species, for example, *Lantana camara* (Aravind *et al.* 2010). Frugivores are negatively affected if the fruit production of native plants is compromised due to the IAPS invasion (Mangachena & Geerts 2017), but are positively affected if the invasive species provides nutritious and abundant fruits (Ramaswami *et al.* 2019). The low abundance or absence of songbirds, water birds, and breeding birds in invaded habitats can be due to the dense vegetation of IAPS blocking the access of birds to the ground vegetation and ground surface, where birds thus are unable to forage, and hence will avoid invaded habitats (Gan *et al.* 2009). However, the IAPS in many areas can also increase habitat diversity, providing nesting and foraging sites for some bird species. Hence, the abundance of species can be high in invaded areas, but as a whole, species richness and diversity may be low (Flanders *et al.* 2006, Wilcox & Beck 2007, Gan *et al.* 2009, Aravind *et al.* 2010, Keller & Avery 2014).

Jalthal Forest is a unique forest in the southeastern part of Nepal and is highly disturbed by locals and invaded by IAPS (GGN 2021). The forest is notable as the largest remaining forest island in the Terai region of Nepal, featuring a unique mix of tropical and subtropical tree species (Sharma 2020). Hence, this forest offers a good opportunity to study the impacts of IAPS on the diversity of birds. In addition, forest management needs to know the effect of IAPS on the biodiversity of the forest to develop effective conservation management for this forest. Because birds are important indicators of biodiversity, this study aimed to (i) assess the richness, breeding status and conservation status of the birds of Jalthal, (ii) evaluate the effect of IAPS on the bird assemblages, (iii) test relationships between bird diversity or abundance and environmental variables.

Material and Methods

Study area

Jalthal Forest is in the far east lowland of Nepal (87°55' and 88°03'E, 26°27' and 26°32'N) and extends to 63 km² in area and between 60 and 120 m in elevation. The forest has a tropical monsoon climate (Figure 1). According to the nearest weather station (Chandragadhi, station No 1412), the mean annual temperature is 25 °C and total annual precipitation is 2,300 mm (DHM 2019).

This forest is surrounded by the human settlements of Haldibari Rural Municipality in the north, Barhadashi Rural Municipality in the west, Kachankawal Rural Municipality (lowest

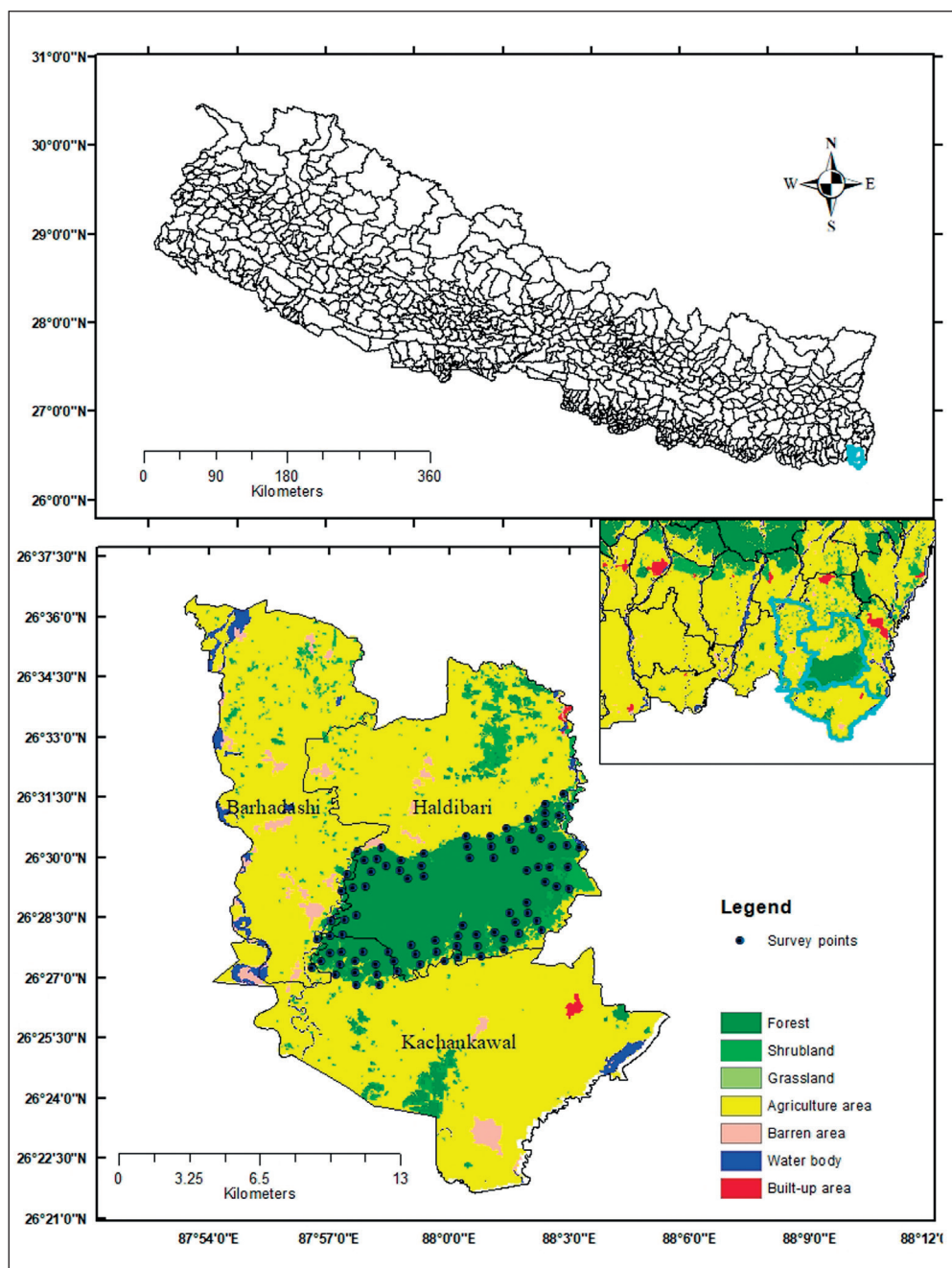


Figure 1. Map of the study area with the point count stations
 1. ábra A vizsgálati terület térképe a számlálási pontokkal

altitude of Nepal) in the south, and Bhadrapur Municipality in the east. The forest is drained by Bhuteni Khola to the western side, and Deonia Khola to the eastern side. This forest has 22 community forest user groups.

This forest contains unique and diverse plant species of tropical and subtropical regions. The forest is the one and only remnant tropical forest in the eastern Tarai region of Nepal, and sal (*Shorea robusta*) is the dominant tree species (Sharma 2018).

Data collection

We collected data in one field visit conducted in January 2020. We surveyed birds by using the point count method (Laiolo 2002). We first selected survey points on the edge of the forest with the help of Google Earth Pro. Then, we selected two survey points, one was 500 m from the edge and the second was 1,000 m from the edge towards the centre of the forest. We surveyed a total of 89 points in the 1-km-wide edge of the forest. Areas closer to the centre (interior) were not surveyed due mainly to risks of encountering wild elephants or wild boars inside the forest. The coordinates of each point count station were recorded by a GPS unit (Garmin eTrex 10).

Birds were recorded in point count stations from 7:00 A.M to 11:00 A.M and 14:00 P.M to 17:00 P.M. Usually, most of the birds are active in the morning and evening. However, because the birds were less active in early morning and late afternoon on foggy morning and cold windy evenings in winter (Robbins 1981), we sampled birds until late morning and started counting in the early afternoon. We recorded all birds that were seen within a circle of 50 m radius around the point count stations in 20 minutes (Wilcox & Beck 2007). High flying birds which crossed the point were ignored. We observed birds by using binoculars (Bushnell 20 X 50) and identified them by using a field guide (Grimmett *et al.* 2016). The calls of the birds were also recorded with a cell phone in MP3 format, and we then compared these sounds using the bird song database of Xeno-Canto (<https://www.xeno-canto.org/>). We also identified birds to the species level by comparing photographs taken in the field (Nikon D3400, 75–300 mm) with field guide images (Grimmett *et al.* 2016).

Vegetation sampling

To characterise vegetation, we recorded the visually estimated percentages of tree canopy cover (T_cc) and shrub cover (S_cc) (0–100% each), and IAPS cover class (5%, 20%, 50%, 75%, 100%) in a circle of 10 m radius around each point count station. We also recorded disturbance indicators such as distance to the nearest motorable road (Dr) and distance to the nearest water source (Dw). The combination of dominant plant species, the number of large trees (Ltn), and nearby water sources (Dw) were also noted and later used in habitat classification.

Data analysis

We first prepared Rank Abundance Curves (RAC) to show relative species abundances following Pandey *et al.* (2020) by plotting the relative abundances of species on the Y axis

Table 1. Details of the studied habitat types
 1. táblázat A vizsgált élőhelytípusok részletei

S.N.	Habitat type	Codes	Details of habitat
1	Sal forest	Sf	Dominant Sal tree and IAPS cover less than 25%
2	Sal <i>Mikania</i> forest	Sm	Dominant Sal tree with invasive <i>Mikania</i> Cover >25
3	Sal <i>Chromolaena</i>	Sc	Dominant Sal tree with invasive <i>Chromolaena</i> Cover >25
4	Invaded scrub	Is	Shrubland invaded by IAPS with very few trees
5	Flood plain	Fp	Side of flowing stream/river
6	Pastureland	Pl	Regularly grazed area, presence of cattle dung
7	Mixed woodland	Mw	Forest not dominated by any species
8	Wooded grassland	Wg	Open grassland with less than 25% shrub with trees number less than 5 per ha

and the abundance rank of species on the X axis. We used the Shannon-Wiener and the Shannon indices to characterise diversity. We built Generalized Linear Models (GLMs) to test the relationships between the diversity or abundance of birds and environmental variables. The GLMs were fitted with normal distribution and identity link function. We repeated the GLMs to assess the effects of environmental variables on the abundances of four bird feeding guilds (Herbivore, Carnivore, Insectivore, Omnivore). Rather than include all of the predictors in one model, we chose to fit each predictor separately because it allows us to perform responsible pairwise comparisons between log-ranks. This model also avoids the potential issues of multicollinearity, and shows that individual effect of a single variable on the response is actually correct. This approach also provides a clearer insight into the role of each predictor on abundance and richness separately, making sure that contributions are not masked or confounded by other factors (Burnham & Anderson 2002, Zuur *et al.* 2010).

We also studied the association of different guilds of birds to the different habitat types by using Canonical Correspondence Analysis (CCA) in Canoco v 4.5 (ter Braak & Smilauer 2002). The habitat types (*Table 1*) were classified as in Aravind *et al.* (2010) and Bergner *et al.* (2015). Pastureland, floodplains, and invaded scrub were not included in the analysis since these were present at fewer than four point count stations. The vertical strata of the forest were classified as in Rocha *et al.* (2015).

All univariate and multivariate analyses were performed by the use of different statistical software, PAST v 4.03, and MS-Excel 2016. The coordinates of the point count station were processed in ArcGIS 10.8 (ESRI 2019) to prepare the map of the study area.

Results

Richness, breeding status and conservation status

A total of 1,373 individuals of birds from 15 orders, 40 families, and 86 species were identified. The most abundant order was Passeriformes with 20 families (*Figure 2*). The most abundant families were Dicruridae and Sturnidae with five species each followed

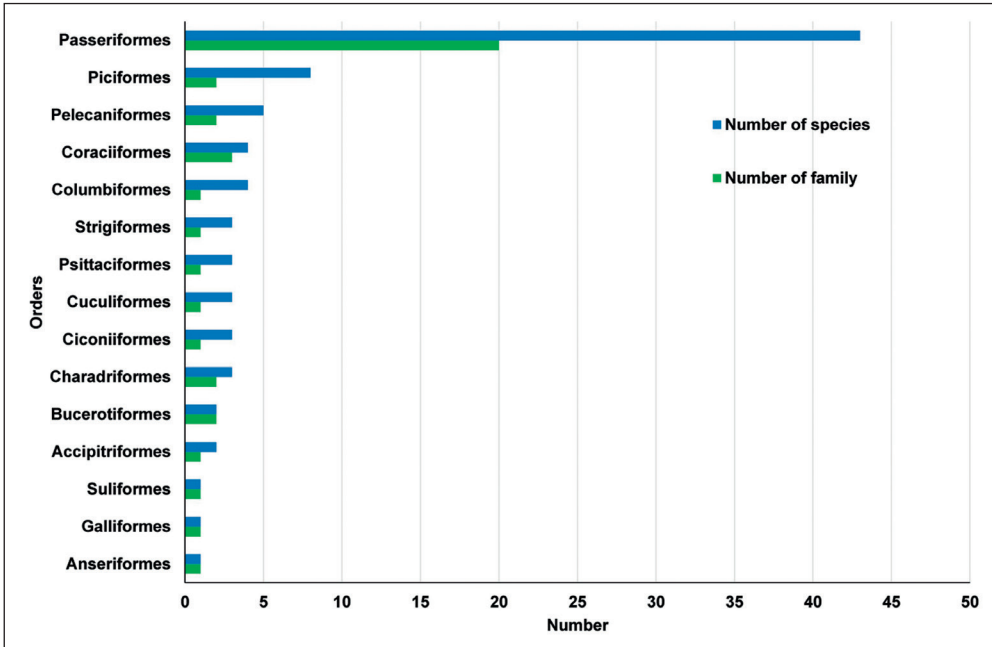


Figure 2. Numbers of families and species in each order

2. ábra Az észlelt családok és fajok száma a rendeken belül

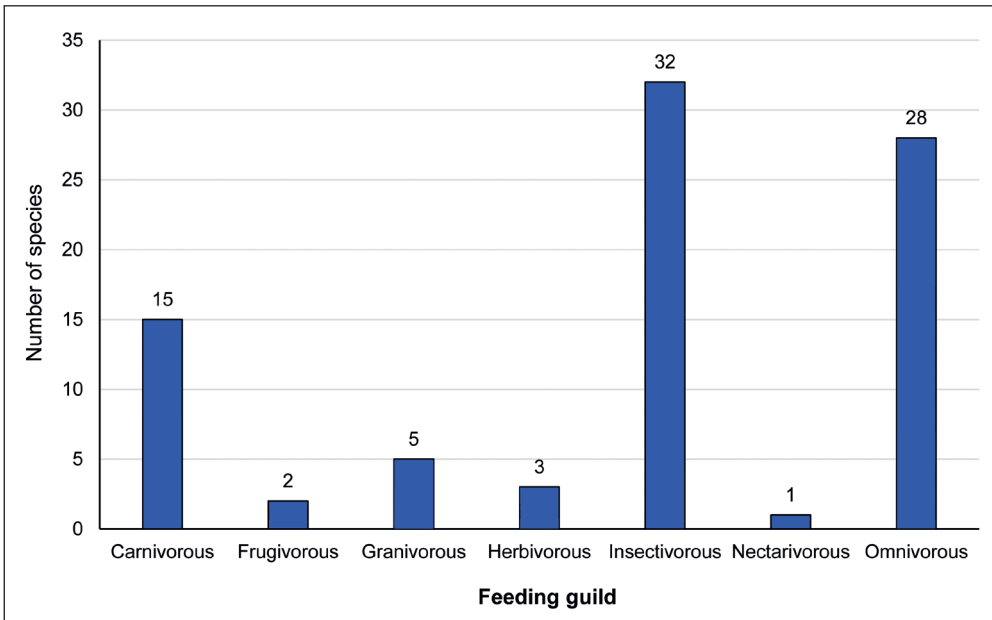


Figure 3. Number of species by feeding guild

3. ábra Fajok száma táplálkozási guildenként

by Ardeidae, Picidae, Columbidae, Motacillidae, and Muscicapidae with four species each. Wooded grasslands had the highest species richness (mean 8.2 ± 0.83) followed by floodplains (average 7.4 ± 1.67) and the lowest species richness was found in sal forests invaded by *Chromolaena* (average 3.78 ± 1.67).

Among the recorded species, 74 species were resident, 10 species were winter visitor, and one passage and summer visitor species were also recorded. Insectivorous birds were the most species-rich guild (32 species) and frugivores were the least species-rich (2 species) (Figure 3). We observed three Near Threatened (NT) birds: Lesser Adjutant (*Leptoptilos javanicus*) (Figure 4A), Asian Woollyneck (*Ciconia episcopus*) (Figure 4B) and Alexandrine Parakeet (*Psittacula eupatria*) (Figure 4C). Likewise, two nationally vulnerable species – Lesser Adjutant and Black Stork (*Ciconia nigra*) (Figure 4D) and four nationally near threatened species – Asian Woollyneck, Oriental Pied Hornbill



Figure 4. Bird rarities observed in Jalthal Forest, A. Lesser Adjutant (*Leptoptilos javanicus*) (Globally Near Threatened and Nationally Vulnerable), B. Asian Woolly-necked Stork (*Ciconia episcopus*) (Globally and nationally Near Threatened), C. Alexandrine Parakeet (*Psittacula eupatria*) (Globally and nationally Near Threatened), D. Black Stork (*Ciconia nigra*) (Globally Least Concerned and nationally Vulnerable)

4. ábra A Jalthal erdőben észlelt madár ritkaságok. A. Indiai marabu (*Leptoptilos javanicus*) (globálisan mérsékelten fenyegetett, nemzeti szinten sebezhető), B. Ázsiai gyapjasnyakú gólya (*Ciconia episcopus*) (globálisan és nemzeti szinten mérsékelten fenyegetett), C. Nagy Sándor-papagáj (*Psittacula eupatria*) (globálisan és nemzeti szinten mérsékelten fenyegetett), D. Fekete gólya (*Ciconia nigra*) (globálisan nem fenyegetett, nemzeti szinten sebezhető)

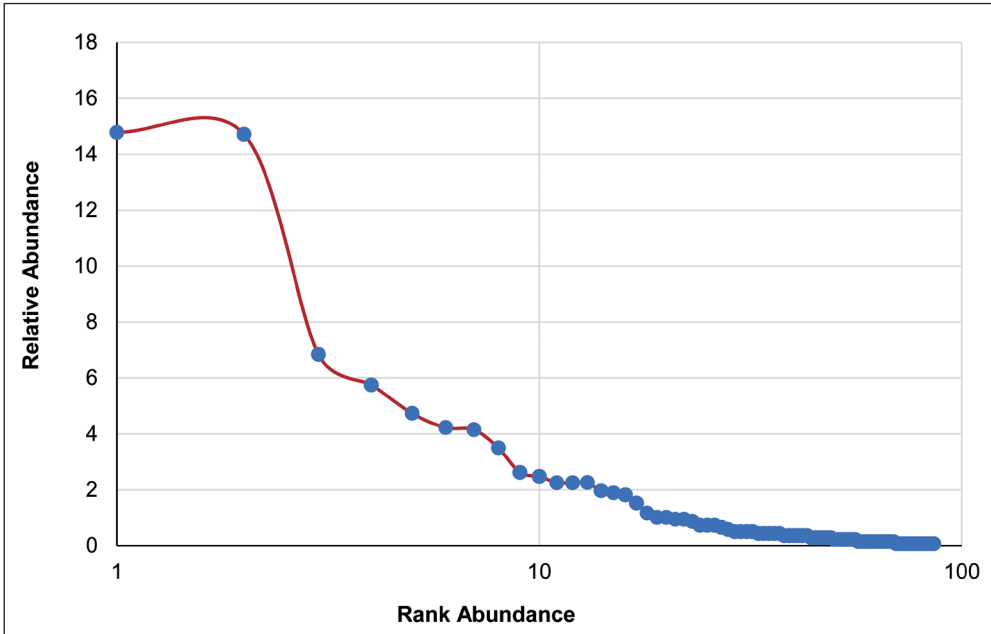


Figure 5. Rank abundance curve of bird species recorded
5. ábra Rang-abundancia görbe az észlelt fajok alapján

(*Anthracoseros albirostris*), Indian Peafowl (*Pavo cristatus*) and Alexandrine Parakeet were observed during the field study.

The RAC showed a steep decrease along the Y-axis and a gradual decline along the X-axis (Figure 5).

Bird diversity

The highest value of the Shannon-Wiener index (H) was in wooded grasslands followed by floodplains and the lowest value was in mixed woodlands. The highest evenness value was in mixed woodland and the lowest was in floodplains. The highest abundance of birds was recorded

Table 2. Bird species richness and diversity indices per habitat type
2. táblázat Madár fajgazdagság és diverzitási indexek élőhelytípusonként

Diversity measures	Habitat types							
	Wg	Sm	Sc	Sf	Mw	Fp	Is	Pl
Abundance	115.00	470.00	150.00	275.00	182.00	128.00	32.00	21.00
Species richness total	29.00±3.07	47.00±13.54	24.00±4.13	37.00±7.89	33.00±5.89	30.00±4.05	9.00±1.39	6.00±1.27
Species richness per plot	8.2±0.83	4.83±1.8	3.7±1.6	4.9±2.09	5.7±3.2	7.4±1.6	1.4±1.14	4±0.0
Shannon_H	1.99±0.11	1.40±0.35	1.18±0.50	1.41±0.44	0.12±0.07	1.78±0.11	1.52±0.18	1.35±0.02
Evenness	0.91±0.05	0.92±0.08	0.96±0.04	0.94±0.8	1.50±0.3	0.83±0.12	0.96±0.04	0.97±0.02

in the Mikania invaded sal forest with a mean abundance of 47.0 ± 13.54 birds per plot, whereas the lowest abundance was in pastureland with a mean of 10.5 ± 1.27 birds per plot (*Table 2*).

Relation of abundance and species richness with environmental variables

The Generalized Linear Model (GLM) showed that both overall abundance and species richness decreased with increasing cover of IAPS, shrub cover and distance to the nearest water source (*Table 3*). The abundance of birds also decreased with the increasing distance to the nearest motorable road (*Table 3*).

Effects of environmental variables on different feeding guilds of birds

Table 3. Summary of results of generalized linear models testing the effects of environmental variables on bird abundance and species richness (bold letters indicate significant results)

3. táblázat A környezeti változók által a madarak abundanciájára és fajgazdagságára gyakorolt hatásokat tesztelő általánosított lineáris modellek eredményeinek összefoglalása

Environmental variables	Abundance			Richness		
	Slope a	Intercept b	p-value	Slope a	Intercept b	p-value
Distance to nearest water source	-0.006	19.288	0.012	-0.001	6.437	0.014
Distance to nearest motorable road	-0.004	19.453	0.034	-0.0009	6.408	0.061
Cover of IAPS	-0.149	23.98	<0.001	-0.054	8.488	<0.001
Tree canopy cover	0.001	16.428	0.975	0.008	5.361	0.415
Shrub cover	-0.110	21.983	0.002	-0.052	8.325	<0.001
Numbers of large trees	0.204	15.322	0.525	0.110	5.103	0.177

Generalized Linear Models (GLM) carried out to assess the effects of environmental variables on four feeding guilds of birds (Herbivore, Carnivore, Insectivore and Omnivore) showed that the distance to the nearest water source positively influenced the richness of Herbivores, Insectivores and Omnivores (*Table 4*). Distance to the nearest road positively influenced the richness of Omnivores and negatively influenced the abundance of Carnivores, Insectivores and Omnivores (*Table 4*). Tree canopy cover positively influenced the richness and abundance of Herbivores and negatively influenced the richness of Carnivores (*Table 4*). Shrub cover had a positive effect on the richness and abundance of Carnivores, and a negative effect on the richness of Insectivores (*Table 4*). Other variables did not have significant effects on neither the richness nor the abundance of feeding guilds.

Relationship between the cover of invasive alien plant species and bird species

Altogether, ten abundant species were taken to test whether there is any significant difference between the abundance of different feeding guild in the different cover class of IAPS. There was a significant difference between the abundance of different feeding guilds in different cover classes ($\chi^2=54.71$, $p<0.00001$). The abundance and richness of herbivorous species were

Table 4. Results of generalized linear models with normal distribution and identity link function test showing the effects of different environmental variables on the richness and abundance of birds belonging to different feeding guilds in Jalthal forest

4. táblázat A különböző környezeti változók által a különböző táplálkozási guildekbe tartozó madarak fajgazdagságára és abundanciájára gyakorolt hatások tesztelésére alkotott általánosított lineáris modellek eredményei

Feeding Guild	Environmental variables	Response variables					
		Richness			Abundance		
		Estimate	t-value	p-value	Estimate	t-value	p-value
Herbivore	Intercept	3.13	1.54	0.120	1.5	5.65	<0.0000
	Dw	0.005	3.32	0.001	-0.0002	-0.98	0.32
	Dr	-0.001	-1.60	0.112	-0.000	-0.18	0.85
	T_cc	0.064	2.43	0.017	0.007	1.86	0.049
	S_cc	0.002	0.09	0.927	-0.0004	-0.12	0.90
	Ltn	-0.055	-0.24	0.810	-0.039	-1.24	0.21
	Cav	-0.44	-1.03	0.304	-0.039	-0.66	0.51
Carnivore	Intercept	2.077	1.069	0.296	0.693	1.17	0.252
	Dw	-0.0006	-0.311	0.758	0.0004	0.72	0.47
	Dr	0.0015	1.183	0.249	-0.0009	-2.69	0.01
	T_cc	-0.1005	-3.607	0.001	-0.004	-0.635	0.53
	S_cc	0.077	2.765	0.011	0.027	3.72	0.001
	Ltn	0.783	3.414	0.302	0.110	1.73	0.09
	Cav	-0.490	-0.869	0.394	-0.007	-0.048	0.96
Insectivore	Intercept	8.55	8.235	0.000	2.6	5.14	<0.0000
	Dw	0.003	3.83	<0.001	0.000	0.23	0.81
	Dr	0.001	1.876	0.065	-0.001	-3.13	0.002
	T_cc	-0.023	-1.397	0.167	-0.0003	-0.042	0.96
	S_cc	-0.04	-3.962	<0.001	-0.0014	-0.243	0.80
	Ltn	-0.67	-0.51	0.610	-0.019	-0.298	0.76
	Cav	0.033	0.13	0.890	-0.047	-0.408	0.68
Omnivore	Intercept	11.00	7.43	0.000	2.65	5.14	<0.0000
	Dw	0.002	1.92	0.049	0.000	0.23	0.81
	Dr	0.004	4.25	<0.001	-0.001	-3.13	0.0025
	T_cc	-0.03	-1.44	0.150	-0.0003	-0.042	0.96
	S_cc	-0.02	-1.59	0.110	-0.001	-0.24	0.80
	Ltn	0.06	0.35	0.720	-0.019	-0.29	0.76
	Cav	0.28	0.87	0.380	-0.02	-0.40	0.68

most affected by the cover of IAPS (*Table 5*). The correlation between abundance or species richness and IAPS cover class was significantly negative in every guild except for omnivorous species, although the correlation was moderate for all guilds (*Table 5, Figure 6*).

Table 5. Spearman's rank correlation coefficients and their significance between the abundance or richness of birds belonging to different feeding guilds and IAPS cover class

5. táblázat A különböző táplálkozási guildekbe tartozó madarak abundanciája vagy fajgazdagsága és az idegenhonos inváziós növényfajok borítási kategóriái között számolt Spearman-féle rang-korrelációs együtthatók és szignifikancia-értékük

Feeding Guilds	Abundance		Richness	
	r_s -value	p-value	r_s -value	p-value
Carnivorous	-0.273	0.012	-0.200	0.067
Herbivorous	-0.366	0.0007	-0.259	0.018
Insectivorous	-0.266	0.015	-0.237	0.040
Omnivorous	-0.032	0.773	-0.036	0.740
Carnivorous	-0.273	0.012	-0.200	0.067

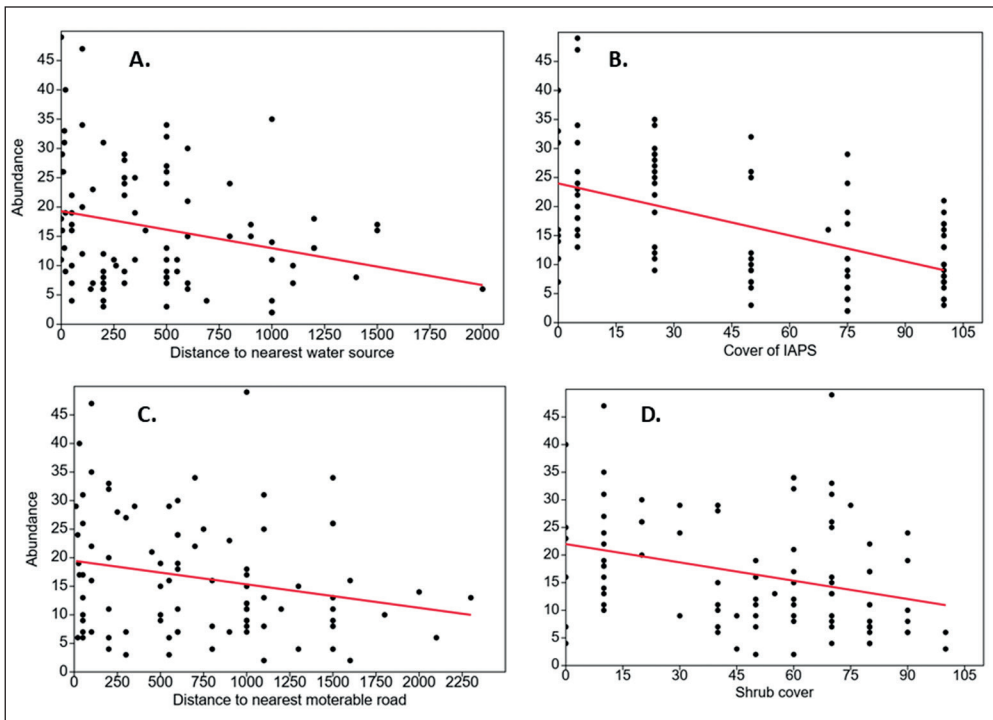


Figure 6. Relationships between bird abundance and different environmental variables: A. Distance to the nearest water source, B. Cover of IAPS, C. Distance to nearest motorable road, D. Shrub cover

6. ábra A madarak abundanciája és a környezeti változók közötti összefüggések. A. Távolság a legközelebbi vízforrástól, B. Idegenhonos inváziós növényfajok borítása, C. A legközelebbi gépjárművekkel járható út távolsága, D. Cserjeborítás

Use of vertical strata of the forest by birds

The abundance of birds was highest (32%) in the top layer of vertical strata, followed by the middle stratum (27%). The abundance of birds in the ground stratum was higher (22%) than in the lower stratum (19%). Species richness was also highest in the middle strata (33%), followed by the ground stratum (27%), and was lower in the top stratum (21%) and the lower stratum (19%).

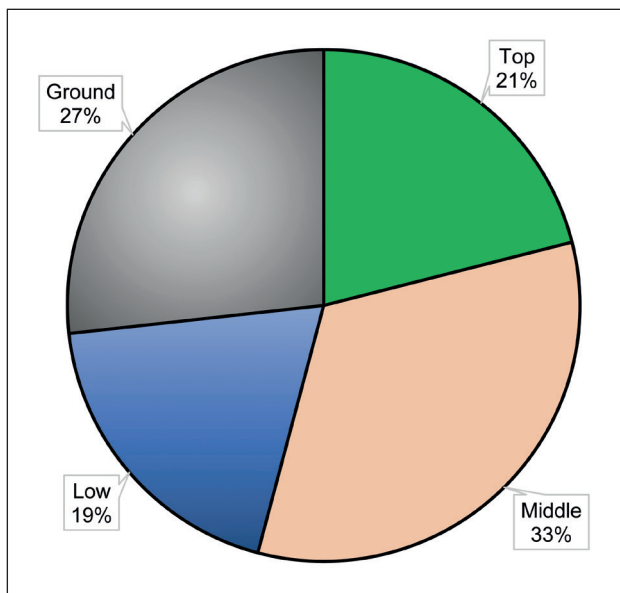


Figure 7. Species richness in each vertical stratum of trees
7. ábra Fajgazdagság vertikális eloszlása

Effects of habitat structure

The CCA followed by the Monte-Carlo permutation test of significance of all canonical axes showed significant associations with habitat type in the case of the herbivorous (Trace = 0.591, F-ratio = 1.663, $p = 0.049$) and the omnivorous guilds (Trace = 0.980, F-ratio = 1.400, $p = 0.049$), whereas there were no significant associations with habitat type in the case of the insectivorous guild (Trace = 1.871, F-ratio = 1.320, $p = 0.108$), and the carnivorous guild (Trace = 1.128, F-ratio = 1.052, $p = 0.322$) (Figure 8).

Discussion

Our study showed that Passeriformes were the most abundant and Anseriformes and Pelecaniformes had the lowest abundance, which is similar to the results found in similar habitat types in Nepal (Ghimire *et al.* 2021). Passeriformes include sparrows, tits, magpie, wagtails and other species, that usually forage in flocks and tend to remain close to agricultural fields and human settlements, where they find more foods grains. Hence, their abundance was recorded higher there. However, the observed species of Anseriformes (Lesser Whistling-duck) and Pelecaniformes (Little Cormorant) are winter visitor waterbirds mostly occurring near the stagnant water reservoir of Tarai. In Jalthal, there are no such big lakes or ponds, thus a lower abundance of waterbirds were not surprising.

There are 168 nationally threatened and 39 internationally threatened bird species in Nepal (Inskipp *et al.* 2016). We observed eight species that are globally threatened, 31 nationally threatened and one nationally protected species of birds (Shrestha 1996).

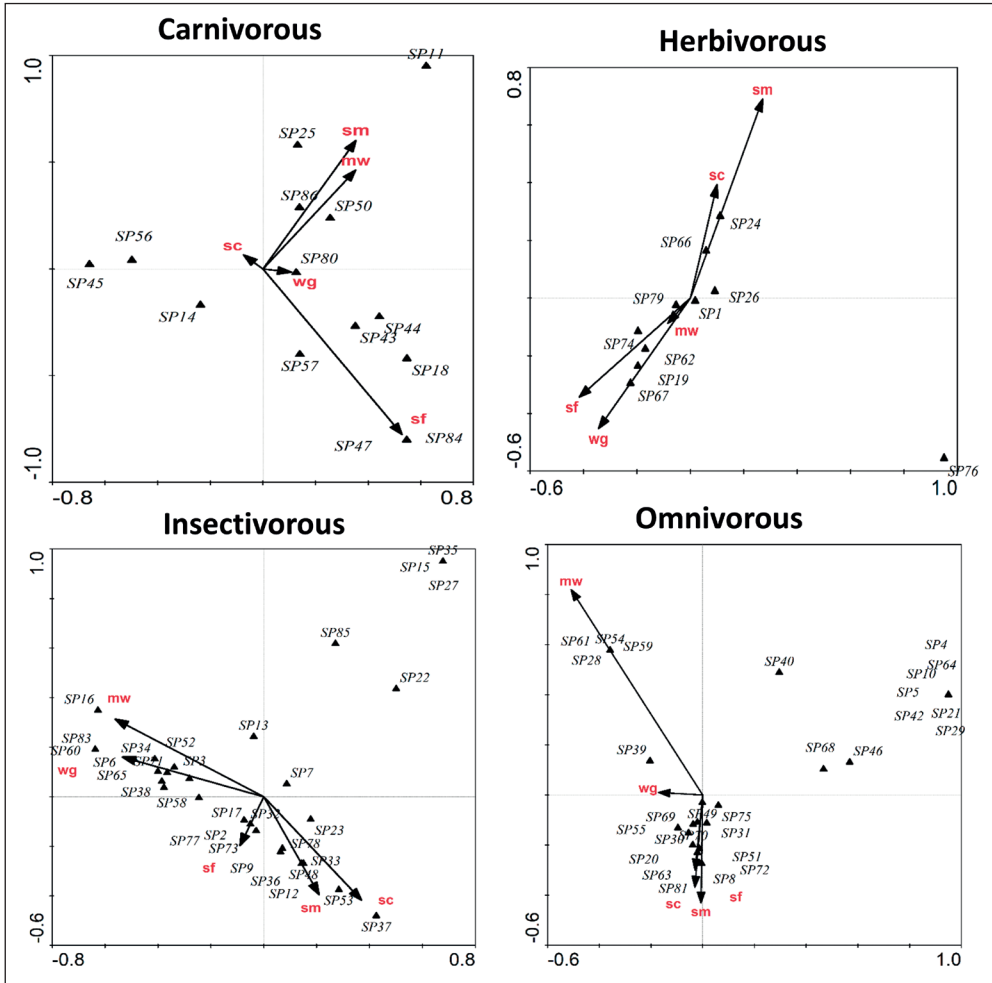


Figure 8. CCA ordination biplots showing the response of different feeding guilds to habitat types of Jalthal forest (mw: Mixed woodland, wg: Wooded grassland, sf: Sal forest, sm: Sal forest with invasive *Mikania*, sc: Sal forest with invasive *Chromolaena*). Codes used for bird species are given in Appendix 1. The first two CCA axes are displayed. The first axis and second axis account for 42% and 33% of the variance in the carnivorous guild, 51.2% and 33.6% in the herbivorous guild, 38.1% and 30.3% in the omnivorous guild, and 23.4% and 19.2% in the insectivorous guild, respectively

8. ábra Kanonikus korrelációelemzéssel (CCA) készített ordinációk diagramjai az egyes táplálkozási guildok által az élőhelytípusokra adott válaszokról (mw: vegyes fás terület, wg: fás gye, sf: szála (Shorea robusta) erdő, sm: szála-erdő inváziós *Mikania*-val, sc: szála-erdő inváziós *Chromolaena*-val. A madárfajok kódjai az 1. sz. mellékletben szerepelnek. A CCA első két tengelye került feltüntetésre. Az első és a második tengely által magyarázott variancia a ragadozók esetén rendre 42% illetve 33%, a növényevők esetén 51,2% illetve 33,6%, a mindelevők esetén 38,1% illetve 30,3%, míg a rovarvők esetén 23,4% illetve 19,2%

However, we also detected one globally vulnerable, two near threatened, two nationally vulnerable and four nationally threatened species. We expect that the number of protected species will increase if the surveys are repeated in other seasons because many protected species visit Jalthal Forest seasonally. The main limitation of this study was to collect the data from only one season, which was due to the high impacts of wild elephants in the Jalthal Forest area.

The rank-abundance curve showed that the abundance of high-ranked species declined fast, which hints at low evenness of the community. The relative abundance of lower-ranked species declined gradually, whereas the curve for medium ranked species showed a shallow decline in abundance. This type of RAC is usually common in communities which are at equilibrium state (Izsák & Pavoine 2012, Pandey *et al.* 2020).

Insectivorous birds were the most abundant, followed by omnivorous and carnivorous birds, whilst frugivorous birds were the least common. This result is consistent with the study of Ghimire *et al.* (2021) and can be explained by the high availability of insects in warm tropical-like forests. Furthermore, most of the insectivorous birds prefer the middle canopy but can also explore the top canopy of the forest as well as the ground. The low abundance of frugivorous birds might be due to the unavailability of fruits in fruiting plants during the survey season (winter). In addition, most of Jalthal Forest is dominated by the tree species sal, which does not offer quality fruits to frugivores.

Heterogeneous habitats typically have a higher bird diversity because they offer a variety of plant species which provides food and shelter for a wide range of birds. In line with this, the diversity of birds was higher in wooded grassland than in other habitat types. Higher diversity in grassland has also been reported in Tu *et al.* (2020). In contrast, the results of Pandey *et al.* (2020) showed that diversity was highest in agricultural areas and in settlements.

Disturbances from vehicles, honking, and human activities usually have a negative impact on the diversity and abundance of birds, which is probably why the richness and abundance of birds usually decrease as the distance to the road decreases (Alexander *et al.* 2019, da Silva & Silva 2020, Leveau & Leveau 2020). Our results contradicted this expectation because the diversity and abundance of birds increased with the decreasing distance to the road. Such a result may be due to the association of habitat on the edge of the forest. The forest is bisected by a motorable road and there is farmland between the road and the forest, which provides a heterogeneous habitat and is utilized by many species of birds (Moges *et al.* 2017, Callaghan *et al.* 2019). The other environmental variables did not show significant relationships with the diversity and abundance of birds. Shrub cover usually increases the bird diversity (Calladine *et al.* 2013). But, in this study, overall abundance ($p < 0.05$) and species richness decreased with shrub cover ($p < 0.001$). This unexpected relationship may be due to invasion of *M. micrantha* and *C. odorata*, because these IAPS cover the usable shrub canopy and makes it difficult to explore by birds.

Furthermore, we noted that the abundance and richness of herbivores were positively related to tree canopy cover. Canopy cover is known to provide excellent nesting sites for the herbivorous birds (Asner *et al.* 2009, Holdo & Mack 2014). Herbivorous birds such as parakeets, barbets and doves are the most abundant species of our study area, and these

species usually live in large flocks (parakeets) and prefer habitats with tall trees. Many trees of the Jalthal forest provide fruits and seeds to these birds, which can explain the higher abundance of herbivores in higher tree canopy cover. In contrast, a negative relationship between tree cover and richness of Carnivores suggested that carnivorous birds did not prefer areas with dense trees, but their abundance was positively related to high shrub cover. Carnivorous birds prefer more heterogeneous habitats and open areas to hunt fishes, snails, small mammals, and birds (Robinson 1994). Eagles and kites hover over the shrubs and hunt other birds, snakes, and small mammals, while egrets, cormorants, herons prefer to hunt on ground (Recher & Holmes 1982, Robinson 1994, Tojo 1996, Combs 2021).

Insectivorous birds such as the drongo (*Dicrurus* spp.), shrikes (*Lanius* spp.), Green Bee-eater (*Merops orientalis*), Black-hooded Orioles (*Oriolus xanthornus*) and omnivorous birds like Eurasian Tree Sparrow (*Passer montanus*), Common Myna (*Acridotheres tristis*), Jungle Babbler (*Turdoides striata*), Rufous Treepie (*Dendrocitta vagabunda*), Lesser Whistling Duck (*Dendrocygna javanica*), Red Whiskered Bulbul (*Pycnonotus jocosus*), etc., were abundant in areas near water sources. Such extensive use of habitat by these birds were also reported by Fisher *et al.* (1972) and Lee *et al.* (2017). The habitat around the water sources is usually rich in both terrestrial and aquatic insect diversity and such areas provide fruit trees, and more prey to insectivorous and omnivorous birds.

The shrub canopy has negatively affected the abundance of insectivores. The movement of insectivorous birds is limited by thick shrub cover, so they usually avoid areas with high shrub cover (Burdick 2005, Lloyd & Martin 2005, Flanders *et al.* 2006). The abundance of omnivorous birds was positively affected by the roads. This is probably a result of the heterogeneity of the habitats near the roads because the forest is usually surrounded by agricultural fields separated by a road. This heterogeneous habitat offers omnivore birds more foraging opportunities making them more abundant in such areas. The richness of carnivores, omnivores, and insectivores, however, were negatively affected by roads nearby, as expected. Many birds usually avoid regularly disturbed habitats (Laurance *et al.* 2004, Benítez-López *et al.* 2010). However, a few species such as Green Bee-eater, Black Drongo (*Dicrurus macrocercus*), Long-tailed Shrike (*Lanius schach*), etc., may prefer the road, which thus increases the abundance but not the richness of birds.

Conclusion

This study, from the one of the most unique forests in Nepal, observed high diversity of birds in the edge of forest where habitat is heterogeneous. Jalthal forest has a unique biodiversity as it is different from the forests of western Nepal regarding species distribution of trees. We found that different feeding guilds responded differently to the habitats invaded by IAPS. In areas with dense IAPS cover, both the richness and abundance of birds were low. Sal forest had the maximum diversity of birds. In general, this study provides strong evidence of the negative impact of IAPS on the diversity of the birds. Hence, this study recommends that the IAPS should be controlled in Jalthal Forest to protect the native species and the unique biodiversity of this area.

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Appendix 1. Checklist of the birds observed from Jalthal Forest. Here, MS= Migratory status, R= residential, W= winter migratory birds, LC= Least concern, VU= vulnerable, NT= Near threatened, NRDB= National Red Data Book

1. melléklet A Jalthal-erdőben megfigyelt madarak listája. MS: vonulási státusz, R: rezidens, W: téli vendég, LC: nem fenyegetett, VU: sebezhető, NT: mérsékelten fenyegetett, NRDB: nemzeti Vörös Könyv

SN	Common name	Scientific name	Feeding guild	Code	MS	NRDB	IUCN
1. Order: Accipitriformes							
Family: Accipitridae							
1	Black Kite	<i>Milvus migrans</i> (Boddaert, 1783)	Omnivorous	SP10	R	LC	LC
2	Crested Serpent-eagle	<i>Spilornis cheela</i> (Latham, 1790)	Carnivorous	SP25	R	LC	LC
2. Order: Anseriformes							
Family: Anatidae							
3	Lesser Whistling-Duck	<i>Dendrocygna javanica</i> (Horsfield, 1821)	Omnivorous	SP54	R	LC	LC
3. Order: Bucerotiformes							
Family: Bucerotidae							
4	Oriental Pied Hornbill	<i>Anthracoceros albirostris</i> (Shaw & Nodder, 1807)	Omnivorous	SP61	R	NT	LC
Family: Upupidae							
5	Common Hoopoe	<i>Upupa epops</i> Linnaeus, 1758	Insectivorous	SP16	R	LC	LC
4. Order: Charadriiformes							
Family: Charadriidae							
6	Red-wattled Lapwing	<i>Vanellus indicus</i> (Boddaert, 1783)	Insectivorous	SP71	R	LC	LC
7	River Lapwing	<i>Vanellus duvaucelii</i> (Lesson, 1826)	Insectivorous	SP73	R	LC	NT
Family: Scolopacidae							
8	Common Sandpiper	<i>Actitis hypoleucos</i> Linnaeus, 1758	Omnivorous	SP21	W	LC	LC

SN	Common name	Scientific name	Feeding guild	Code	MS	NRDB	IUCN
5. Order: Ciconiiformes							
Family: Ciconiidae							
9	Black Stork	<i>Ciconia nigra</i> (Linnaeus, 1758)	Carnivorous	SP11	W	VU	LC
10	Lesser Adjutant	<i>Leptoptilos javanicus</i> (Horsfield, 1821)	Carnivorous	SP50	R	VU	NT
11	Asian Woollyneck	<i>Ciconia episcopus</i> (Boddaert, 1783)	Carnivorous	SP86	R	NT	NT
6. Order: Columbiformes							
Family: Columbidae							
12	Eurasian Collared-dove	<i>Streptopelia decaocto</i> (Frisvaldszky, 1838)	Granivorous	SP26	R	LC	LC
13	Oriental Turtle-dove	<i>Streptopelia orientalis</i> (Latham, 1790)	Granivorous	SP62	R	LC	LC
14	Spotted Dove	<i>Spilopelia chinensis</i> (Scopoli, 1786)	Granivorous	SP79	R	LC	LC
15	Red Turtle-dove	<i>Streptopelia tranquebarica</i> (Hermann, 1804)	Granivorous	SP27	R	LC	LC
7. Order: Coraciiformes							
Family: Alcedinidae							
16	Common Kingfisher	<i>Alcedo atthis</i> (Linnaeus, 1758)	Carnivorous	SP18	R	LC	LC
17	White-breasted Kingfisher	<i>Halcyon smyrnensis</i> (Kuhl, 1758)	Carnivorous	SP84	R	LC	LC
Family: Coraciidae							
18	Indian Roller	<i>Coracias benghalensis</i> (Linnaeus, 1758)	Carnivorous	SP44	R	LC	LC
Family: Meropidae							
19	Asian Green Bee-eater	<i>Merops orientalis</i> Latham, 1802	Insectivorous	SP34	R	LC	LC
8. Order: Cuculiformes							
Family: Cuculidae							
20	Common Hawk-cuckoo	<i>Hierococcyx varius</i> (Vahl, 1797)	Insectivorous	SP17	R	LC	LC
21	Greater Coucal	<i>Centropus sinensis</i> (Stephens, 1815)	Omnivorous	SP31	R	LC	LC
22	Lesser Coucal	<i>Centropus bengalensis</i> (Gmelin, 1788)	Omnivorous	SP51	R	LC	LC
9. Order: Galliformes							
Family: Phasianidae							
23	Indian Peafowl	<i>Pavo cristatus</i> Linnaeus, 1758	Omnivorous	SP42	R	NT	LC
10. Order: Passeriformes							
Family: Aegithinidae							
24	Common lora	<i>Aegithina tiphia</i> (Linnaeus, 1758)	Frugivorous	SP19	R	LC	LC

SN	Common name	Scientific name	Feeding guild	Code	MS	NRDB	IUCN
Family: Campephagidae							
25	Black-winged Cuckooshrike	<i>Lalage melaschistos</i> (Hodgson, 1836)	Omnivorous	SP8	S	LC	LC
26	Large Cuckooshrike	<i>Coracina javensis</i> (Horsfield, 1821)	Insectivorous	SP48	R	LC	LC
27	Scarlet Minivet	<i>Pericrocotus flammeus</i> (Forster, 1781)	Insectivorous	SP77	R	LC	LC
Family: Cisticolidae							
28	Common Tailorbird	<i>Orthotomus sutorius</i> (Pennant, 1769)	Insectivorous	SP23	R	LC	LC
Family: Corvidae							
29	House Crow	<i>Corvus splendens</i> (Vieillot, 1817)	Omnivorous	SP40	R	LC	LC
30	Eastern Jungle Crow	<i>Corvus macrorhynchos</i> (Wagler, 1827)	Omnivorous	SP68	R	LC	LC
31	Rufous Treepie	<i>Dendrocitta vagabunda</i> (Latham, 1790)	Omnivorous	SP75	R	LC	LC
Family: Dicruridae							
32	Ashy Drongo	<i>Dicrurus leucophaeus</i> Vieillot, 1817	Insectivorous	SP2	W	LC	LC
33	Black Drongo	<i>Dicrurus macrocercus</i> Vieillot, 1817	Insectivorous	SP7	R	LC	LC
34	Greater Racquet-Tailed Drongo	<i>Dicrurus paradiseus</i> (Linnaeus, 1766)	Insectivorous	SP33	R	LC	LC
35	Lesser Racquet-Tailed Drongo	<i>Dicrurus remifer</i> (Temminck, 1823)	Insectivorous	SP53	R	LC	LC
36	Spangled Drongo	<i>Dicrurus bracteatus</i> (Gould, 1842)	Insectivorous	SP78	R	LC	LC
Family: Estrilidae							
37	Scaly-breasted Munia	<i>Lonchura punctulata</i> (Linnaeus, 1758)	Granivorous	SP76	R	LC	LC
Family: Hirundinidae							
38	Barn Swallow	<i>Hirundo rustica</i> Linnaeus, 1758	Insectivorous	SP6	R	LC	LC
Family: Laniidae							
39	Brown Shrike	<i>Lanius cristatus</i> Linnaeus, 1758	Insectivorous	SP13	W	LC	LC
40	Grey-backed Shrike	<i>Lanius tephronotus</i> (Vigors, 1831)	Insectivorous	SP35	W	LC	LC
41	Long-tailed Shrike	<i>Lanius schach</i> Linnaeus, 1758	Insectivorous	SP58	R	LC	LC
Family: Leiothrichidae							
42	Jungle Babbler	<i>Turdoides striata</i> (Dumont, 1823)	Omnivorous	SP49	R	LC	LC
Family: Motacillidae							
43	Olive-backed Pipit	<i>Anthus hodgsoni</i> Richmond, 1907	Omnivorous	SP59	W	LC	LC
44	Paddyfield Pipit	<i>Anthus rufulus</i> Vieillot, 1818	Omnivorous	SP64	R	LC	LC

SN	Common name	Scientific name	Feeding guild	Code	MS	NRDB	IUCN
45	White-browed Wagtail	<i>Motacilla maderaspatensis</i> (Gmelin, 1789)	Insectivorous	SP82	R	LC	LC
46	White Wagtail	<i>Motacilla alba</i> Linnaeus, 1758	Insectivorous	SP85	W	LC	LC
Family: Muscicapidae							
47	Asian brown flycatcher	<i>Muscicapa dauurica</i> (Pallas 1811)	Insectivorous	SP3	P	LC	LC
48	Common Stonechat	<i>Saxicola torquatus</i> (Linnaeus, 1766)	Insectivorous	SP22	W	LC	LC
49	Oriental Magpie-robin	<i>Copsychus saularis</i> (Linnaeus, 1758)	Insectivorous	SP60	R	LC	LC
50	Pied Bushchat	<i>Saxicola caprata</i> (Linnaeus, 1766)	Insectivorous	SP65	R	LC	LC
51	Verditer Flycatcher	<i>Eumyias thalassinus</i> Swainson, 1838	Omnivorous	SP81	W	LC	LC
Family: Nectariniidae							
52	Purple Sunbird	<i>Cinnyris asiaticus</i> (Latham, 1790)	Nectarivorous	SP67	R	LC	LC
Family: Oriolidae							
53	Black-hooded Oriole	<i>Oriolus xanthornus</i> (Linnaeus, 1758)	Insectivorous	SP9	R	LC	LC
Family: Paridae							
54	Great Tit	<i>Parus major</i> Linnaeus, 1758	Omnivorous	SP30	R	LC	LC
Family: Passeridae							
55	House Sparrow	<i>Passer domesticus</i> (Linnaeus, 1758)	Omnivorous	SP39	R	LC	LC
56	Eurasian Tree Sparrow	<i>Passer montanus</i> (Linnaeus, 1758)	Omnivorous	SP28	R	LC	LC
Family: Pycnonotidae							
57	Red-vented Bulbul	<i>Pycnonotus cafer</i> (Linnaeus, 1766)	Omnivorous	SP70	R	LC	LC
58	Red-whiskered Bulbul	<i>Pycnonotus jocosus</i> (Linnaeus, 1758)	Omnivorous	SP72	R	LC	LC
Family: Rhipiduridae							
59	White-throated Fantail	<i>Rhipidura albicollis</i> (Vieillot, 1818)	Insectivorous	SP83	R	LC	LC
Family: Stenostiridae							
60	Grey-headed Canary-flycatcher	<i>Culicicapa ceylonensis</i> (Swainson, 1820)	Insectivorous	SP37	W	LC	LC
Family: Sturnidae							
61	Indian Pied Starling	<i>Gracupica contra</i> (Linnaeus, 1758)	Omnivorous	SP4	R	LC	LC
62	Bank Myna	<i>Acridotheres ginginianus</i> (Latham, 1790)	Omnivorous	SP5	R	LC	LC
63	Chestnut-tailed Starling	<i>Sturnia malabarica</i> (Gmelin, 1789)	Insectivorous	SP15	R	LC	LC
64	Common Myna	<i>Acridotheres tristis</i> (Linnaeus, 1766)	Omnivorous	SP20	R	LC	LC

SN	Common name	Scientific name	Feeding guild	Code	MS	NRDB	IUCN
65	Jungle Myna	<i>Acridotheres fuscus</i> (Wagler, 1827)	Omnivorous	SP46	R	LC	LC
Family: Zosteropidae							
66	Indian White-eye	<i>Zosterops palpebrosus</i> (Temminck, 1824)	Omnivorous	SP63	R	LC	LC
11. Order: Pelecaniformes							
Family: Ardeidae							
67	Cattle Egret	<i>Bubulcus ibis</i> (Linnaeus, 1758)	Carnivorous	SP14	R	LC	LC
68	Great White Egret	<i>Ardea alba</i> Linnaeus, 1758	Carnivorous	SP43	R	LC	LC
69	Intermediate Egret	<i>Ardea intermedia</i> Wagler, 1829	Carnivorous	SP45	R	LC	LC
70	Little Egret	<i>Egretta garzetta</i> (Linnaeus, 1766)	Carnivorous	SP57	R	LC	LC
Family: Threskiornithidae							
71	Red-naped Ibis	<i>Pseudibis papillosa</i> (Temminck, 1824)	Omnivorous	SP69	R	LC	LC
12. Order: Piciformes							
Family: Megalaimidae							
72	Blue-throated Barbet	<i>Psilopogon asiaticus</i> (Latham, 1790)	Insectivorous	SP12	R	LC	LC
73	Coppersmith Barbet	<i>Psilopogon haemacephalus</i> (Müller, 1776)	Frugivorous	SP24	R	LC	LC
74	Lineated Barbet	<i>Psilopogon lineatus</i> (Vieillot, 1816)	Omnivorous	SP55	R	LC	LC
Family: Picidae							
75	Fulvous-breasted Woodpecker	<i>Dendrocopos macei</i> (Vieillot, 1818)	Omnivorous	SP29	R	LC	LC
76	Greater Flameback	<i>Chrysocolaptes lucidus</i> (Scopoli, 1786)	Insectivorous	SP32	R	LC	LC
77	Grey-capped Pygmy Woodpecker	<i>Picoides canicapillus</i> (Blyth, 1845)	Insectivorous	SP36	R	LC	LC
78	Himalayan Flameback	<i>Dinopium shorii</i> (Vigors, 1832)	Insectivorous	SP38	R	LC	LC
79	Black-rumped Flameback	<i>Dinopium benghalense</i> (Linnaeus, 1758)	Insectivorous	SP52	R	LC	LC
13. Order: Psittaciformes							
Family: Psittaculidae							
80	Alexandrine Parakeet	<i>Psittacula eupatria</i> (Linnaeus, 1766)	Herbivorous	SP1	R	NT	NT
81	Plum-headed Parakeet	<i>Psittacula cyanocephala</i> (Linnaeus, 1766)	Herbivorous	SP66	R	LC	LC
82	Rose-ringed Parakeet	<i>Psittacula krameri</i> (Scopoli, 1769)	Herbivorous	SP74	R	LC	LC
14. Order: Strigiformes							
Family: Strigidae							
83	Spotted Owlet	<i>Athene brama</i> (Temminck, 1821)	Carnivorous	SP80	R	LC	LC

SN	Common name	Scientific name	Feeding guild	Code	MS	NRDB	IUCN
84	Jungle Owlet	<i>Glaucidium radiatum</i> (Tickell, 1833)	Carnivorous	SP47	R	LC	LC
85	Asian Barred Owlet	<i>Glaucidium cuculoides</i> (Vigors, 1831)	Carnivorous	SP41	R	LC	LC
15. Order: Suliformes							
Family: Phalacrocoracidae							
86	Little cormorant	<i>Microcarbo niger</i> (Vieillot, 1817)	Carnivorous	SP56	R	LC	LC



Diurnal raptors of Eastern Himalayan foothills: Taxonomic and functional diversity attributes

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Abstract Raptors are crucial indicators of ecosystem health, and thus, requires research attention. Despite an upsurge in Indian ornithology, raptors remain poorly studied, particularly in Northeast India. Here, we measured the taxonomic and functional diversity of raptors in Manas National Park, Assam. The study was conducted between 2018–2022. We surveyed 39 standard transects and recorded 776 individuals of raptors belonging to 30 species in different habitats: forest, open and mosaic. The taxonomic alpha diversity was highest in mosaic habitat and lowest in the forest habitat. Crested Serpent Eagle (*Spilornis cheela*) was the most abundant raptor in open and forest habitats, while Collared Falconet (*Microhierax caerulescens*) was dominating in mosaic habitats. Of the recorded raptor species, 66.67% were classified as residents, and 33.37% were winter migrants. We also measured the functional alpha diversity of raptors using the data collected from AVONET database. We used ten functional traits and evaluated the functional alpha diversity indices. The functional richness was the highest in open habitats and functional divergence was the highest in mosaic habitats, also functional evenness was the highest in open habitats and lowest in mosaic habitats. We suggest intensified research on raptors in the region, to identify and address critical research question which facilitates the conservation of these apex predators.

Keywords: birds of prey, richness, abundance, functional traits, Assam

Összefoglalás A ragadozómadarak az ökoszisztéma egészségének kulcsfontosságú mutatói, ezért fokozott figyelmet kell fordítani kutatásukra. Az indiai ornitológia fellendülése ellenére a ragadozómadarakat továbbra is kevésbé tanulmányozzák, különösen Északkelet-Indiában. Jelen tanulmányban a ragadozómadarak taxonómiai és funkcionális diverzitását mértük fel az asszámi Manasz Nemzeti Parkban. A vizsgálatot 2018 és 2022 között végeztük. 39 transektet vizsgáltunk, és 30 faj 776 egyedét regisztráltuk különböző élőhelyeken: erdőben, nyílt és mozaikos területeken. A taxonómiai alfa-diverzitás a mozaikos élőhelyen volt a legmagasabb, az erdei élőhelyen pedig a legalacsonyabb. A nyílt és az erdei élőhelyeken a kontyos kígyász (*Spilornis cheela*) volt a leggyakoribb ragadozómadár, míg a mozaikos élőhelyeken az indiai verébsólyom (*Microhierax caerulescens*) dominált. A regisztrált ragadozómadár-fajok 66,67%-a állandó, 33,37%-a pedig téli vonuló volt. A ragadozómadarak funkcionális alfa-diverzitását is kiszámoltuk az AVONET adatbázisból gyűjtött adatok alapján. Tíz funkcionális tulajdonságot használtunk, és kiértékeljük a funkcionális alfa-diverzitás indexeit. A funkcionális fajgazdagság a nyílt élőhelyeken, a funkcionális diverzitás pedig a mozaikos élőhelyeken, valamint a funkcionális egyenletesség a nyílt élőhelyeken volt a legmagasabb, de a mozaikos élőhelyeken a legalacsonyabb. Javasoljuk a ragadozómadarak intenzívebb kutatását a régióban, hogy meghatározzuk és körüljárjuk azokat a kritikus kutatási kérdéseket, amelyek elősegítik a csúcsragadozók védelmét.

Kulcsszavak: fajgazdagság, funkcionális jellegek, gyakoriság, ragadozómadarak, Asszam

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Introduction

Raptors or birds of prey are one such avian group which serve as excellent indicators of environmental health (Donázar *et al.* 2016) and habitat quality (Sergio *et al.* 2008), and also are key ecosystem service providers (O'Bryan *et al.* 2018). The presence of raptors in an area advocates high level of biodiversity (Sergio *et al.* 2006, 2008). These birds play a key role in maintaining ecological stability of an area (Sergio *et al.* 2008). Raptors, being apex predators in trophic levels with large home ranges and low population density, are more susceptible to anthropogenic stress compared to other birds (Newton 1979, Owens & Bennett 2000, Santangeli *et al.* 2019). Moreover, habitat alteration (Bierregaard 1998, Watson 1998) and contamination of the environment (Newton 1979) further add to their survival threats. *Gyps* vultures in the Indian subcontinent suffered a tragic decline of 99% during the 1990s, owing to the use of a veterinary drug “diclofenac” for cattle (Prakash *et al.* 2003, 2012, 2019, Green *et al.* 2004, Oaks *et al.* 2004). Other than vultures, diclofenac toxicity was also observed in a study (Sharma *et al.* 2014), which highlights the vulnerability of these raptors to contamination. Other anthropogenic threats to raptors include habitat alterations (Thiollay 1998, Goriup & Tucker 2007), killing (Symes 2012, Dalvi & Haralu 2014, Brochet *et al.* 2019), poisoning (Oaks *et al.* 2004, Galligan *et al.* 2014, Ogada *et al.* 2016), electrocutions (Lehman 2001, Angelov *et al.* 2013, Kagan 2016, Mojica *et al.* 2018), collisions (Cusa *et al.* 2015), road kills (Klippel *et al.* 2015), and climate change (Franke 2017, Iknayan & Beissinger 2018, McClure *et al.* 2018).

Species richness and abundance of raptor communities are highly influenced by habitat heterogeneity and prey availability (White 1974), as habitat types (Guisan *et al.* 2017) and their utilization (Morrison 2012) largely determine the species-habitat interactions in an area. The species-habitat relationship also determine the migration of species in different regions (Gavashelishvili & McGrady 2006, Hansson & Åkesson 2014), which again affect the distribution of species across a geographical area (Carnicer *et al.* 2012). Raptors occur in low densities and have a wide range (Newton 1979, Fuller & Mosher 1981) which makes them difficult to study. However, considering the role of the raptor species in ecosystem functioning, it is of high importance to study the functional role of these birds in ecosystem functioning and diversity. With the advancement of the different measurement tools/indices in the field of biodiversity studies, now we are able to efficiently measure such role. In this context, protected areas are important zones of interest as other ecosystem elements are less disturbed in these areas.

Although ornithology in India has flourished over time, raptors remain poorly studied in general, compared to other bird groups. The research trend is highly skewed, with 60% publications devoted to only ten species (Mahananda *et al.* 2022). Raptor studies in Northeastern India is very scarce relative to other parts of the country (Mahananda *et al.* 2022), even though the region is rich in raptors and other birds. Hence the present study emphasized on the species diversity and functional diversity of diurnal raptors in Manas National Park, Assam.

Material and Methods

Study area

The study was carried out in Manas National Park, which is located between 26.623° – 26.822° N and 90.808° – 91.251° E in the foothills of the Eastern Himalayas, in Baksa and Chirang districts of the state of Assam, India (*Figure 1*). The park covers an area of 850 km², and is situated at the junction of Indo-Gangetic, Indo-Malayan and Indo-Chinese biogeographic realms (BirdLife International 2003), and is a part of Brahmaputra Valley biogeographic province, harboring Assam valley semi-evergreen forests and Terai-duar wet alluvial savanna grasslands (Champion & Seth 1968). It is also considered as an important bird area with diverse avifauna including globally threatened species (BirdLife International 2024).

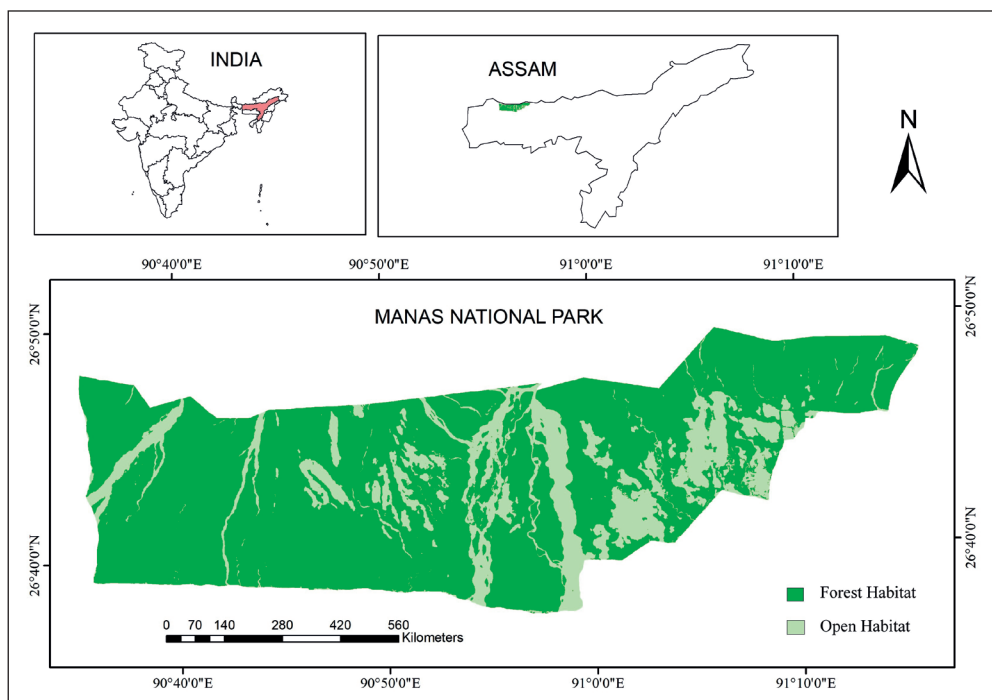


Figure 1. Map of Manas National Park showing the forest and open habitats

1. ábra A Manasz Nemzeti Park térképe az erdei és nyílt élőhelyek feltüntetésével

Sampling design

The selected study area was divided into 2×2 km grids, so that each grid can be easily covered in a day. We established 39 line transects of 2 km each in the selected area in a semi random manner based on accessibility, which were usually provided by the roads and trails. The transects were designed to be included in different grids, without overlaps (*Figure 2*). We assigned the habitat types based on field observations. During the surveys whenever

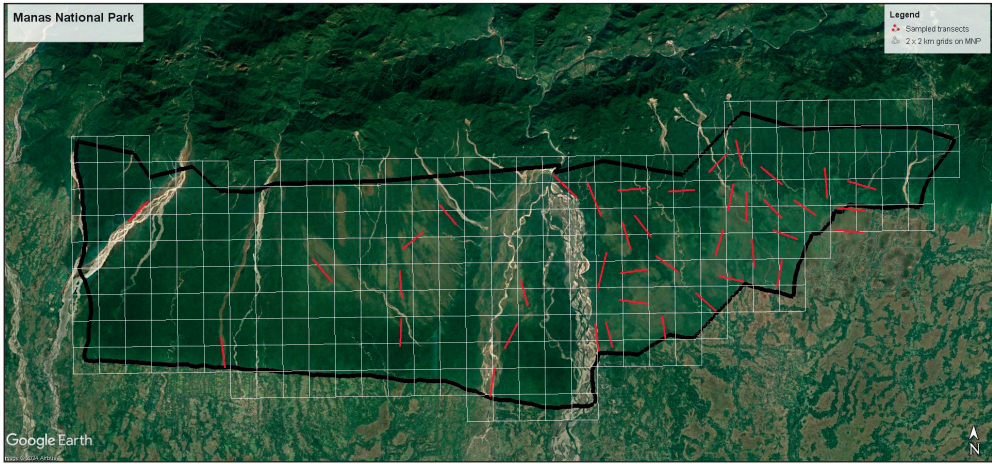


Figure 2. Map of Manas National Park showing line transects (red color) and 2×2 km grids

2. ábra A vonal transzektek (piros) és a 2×2 km háló elhelyezkedése a Manasz Nemzeti Park területén

Table 1. Geographical coordinates of the line transects and their habitat types
1. táblázat A vonal transzektek koordinátái és az élőhely típusa, amiben kijelöltük őket

Transects	Geographical Coordinates				Habitat type
	Start point		End point		
T1	26.66283	91.00211	26.68065	90.99679	Open
T2	26.70275	90.99093	26.72755	90.99740	Open
T3	26.75167	90.99344	26.77513	90.98272	Open
T4	26.78060	90.95674	26.76458	90.97502	Open
T5	26.74799	91.01010	26.73021	91.01576	Open
T6	26.72534	91.03540	26.71454	91.05259	Forest
T7	26.69552	91.00795	26.69299	91.02910	Open
T8	26.68429	91.04062	26.67032	91.04212	Mosaic
T9	26.68795	91.08163	26.70044	91.06551	Mosaic
T10	26.70822	91.10302	26.71159	91.08401	Mosaic
T11	26.72596	91.08106	26.74356	91.08708	Mosaic
T12	26.75673	91.09080	26.77494	91.09503	Mosaic
T13	26.78765	91.10153	26.80534	91.09695	Mosaic
T14	26.71789	91.11094	26.73734	91.10922	Mosaic
T15	26.70388	91.12840	26.72374	91.13232	Open
T16	26.73721	91.14433	26.74363	91.12553	Forest
T17	26.66295	90.83911	26.68202	90.84003	Mosaic
T18	26.69712	90.84017	26.71516	90.38857	Mosaic
T19	26.72992	90.84101	26.74180	90.85670	Mosaic
T20	26.76003	90.86889	26.74642	90.88217	Open

Transects	Geographical Coordinates				Habitat type
	Start point		End point		
T21	26.77056	91.00606	26.77183	91.02738	Forest
T22	26.76965	91.04493	26.77105	91.06501	Mosaic
T23	26.78294	91.07621	26.79618	91.08996	Forest
T24	26.71369	91.00808	26.71637	91.02924	Mosaic
T25	26.74961	91.10457	26.76912	91.10500	Open
T26	26.76423	91.11864	26.75105	91.13252	Open
T27	26.78638	91.11621	26.77488	91.13240	Open
T28	26.76397	91.14170	26.75344	91.15854	Mosaic
T29	26.76526	91.16718	26.78580	91.16572	Open
T30	26.77698	91.18292	26.77142	91.20505	Mosaic
T31	26.62916	90.90834	26.64841	90.91111	Open
T32	26.66182	90.91931	26.67956	90.92919	Mosaic
T33	26.69125	90.93683	26.70919	90.93155	Open
T34	26.75676	91.19618	26.75872	91.17481	Open
T35	26.74165	91.19497	26.74360	91.17382	Forest
T36	26.73888	91.03124	26.75361	91.01888	Mosaic
T37	26.74821	90.63066	26.76183	90.64536	Mosaic
T38	26.64831	90.70501	26.66867	90.70189	Mosaic
T39	26.70758	90.78655	26.72239	90.77199	Mosaic

the study encountered a raptor, it was classified into the following habitat categories: open habitats (grasslands, swamps, agricultural field and riparian areas, rivers and wetlands), forest habitats (closed canopy woodlands) and mosaic habitats (a matrix of open and forest habitats, etc.) (*Table 1*).

Data collection

The transect surveys were carried out between 2018–2022 (except the Covid 19 lockdown period). We drove on an open vehicle (Fuller & Mosher 1987) to the grid location and surveyed on foot. We recorded the species, its abundance and habitat type during the surveys. Each transect survey was repeated five times. The surveys were carried out from 06:00 to 12:00 hr during the peak soaring time of raptors (Thiollay & Meyburg 1988), in post monsoon and winters to accommodate the migratory raptors. Sampling was not performed in adverse weather conditions such as heavy rainfall and thunderstorms, due to difficulty in field survey. For functional diversity analyses, the functional traits data of raptors were obtained from the publicly available global database of AVONET (Tobias *et al.* 2022).

Data analyses

Species diversity and habitat similarity attributes

The species diversity was assessed obtaining the number of species and their abundance in all three habitat types (*Appendix A*). It was measured using the Shannon-Wiener index (H') (Shannon & Weaver 1949), which uses pooled number and abundance of each species in each habitat type, and also their evenness (J'). Species accumulation curve was plotted to evaluate the representativeness of the sampled species in the community. We also plotted individual-based rarefaction curves to compare species richness between habitats. We estimated the shared species richness between habitats using four similarity indices: Euclidean distance index, Bray-Curtis index (BC), Jaccard index (JI) and Morisita-Horn index (MH) (Magurran 2004). All the statistical analyses were executed in R 4.0.2 programming platform (R Core Team 2013). We applied the BiodiversityR (Kindt 2019) and VennDiagram (Chen & Boutros 2011) packages available within the R statistical programming framework.

Functional diversity attributes

The Functional data collected were analyzed using the R package “mFD” (Magneville *et al.* 2022), which uses three different frameworks i.e. groups of species, pairwise trait-based distances between species, and species coordinates in a multidimensional space. We used species occurrence data and raptor-traits data to measure different functional diversity indices.

Results

Species richness and diversity

The study recorded 30 species and 776 individuals of raptors belonging to 3 families within the three different habitats (*Appendix 1*). Of these, 66.67% were residents (20 species), and 33.37% were winter migrants (10 species). Species richness was observed as the highest in open habitat (28 species, $J' = 0.93$), followed by mosaic habitat (27 species, $J' = 0.94$), and lowest in forest habitat (20 species, $J' = 0.96$) (*Figure 3*). Shannon-Wiener Index was highest in mosaic habitat ($H' = 3.15$), followed by open habitat ($H' = 3.12$), and lowest in forest habitat ($H' = 2.72$). The rarefaction curve showed that raptor species reached its asymptote in all the three habitats indicating that the sampling was adequate (*Figure 4*).

Rank-abundance of raptors

A total of 776 individual raptors were recorded during the study, including 374 individuals in open habitat (48.1%), 326 in mosaic habitat (42%) and 76 individuals in forest habitat (9.7%). Collared Falconet (*Microhierax caerulescens*) was the dominating species in numbers in mosaic habitat, and Crested Serpent Eagle (*Spilornis cheela*) was the most abundant species

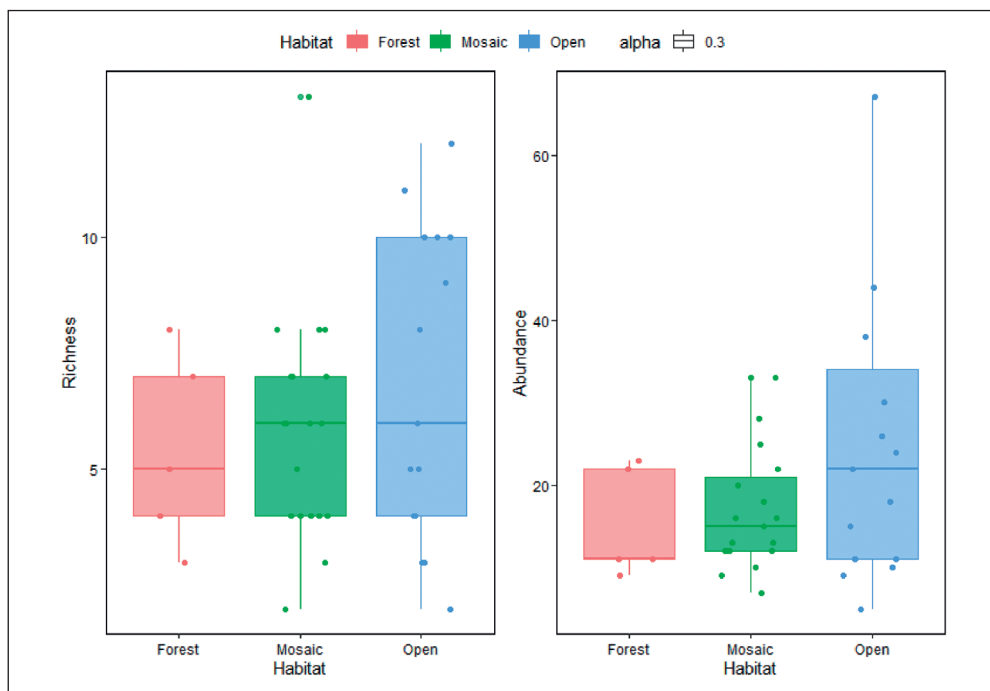


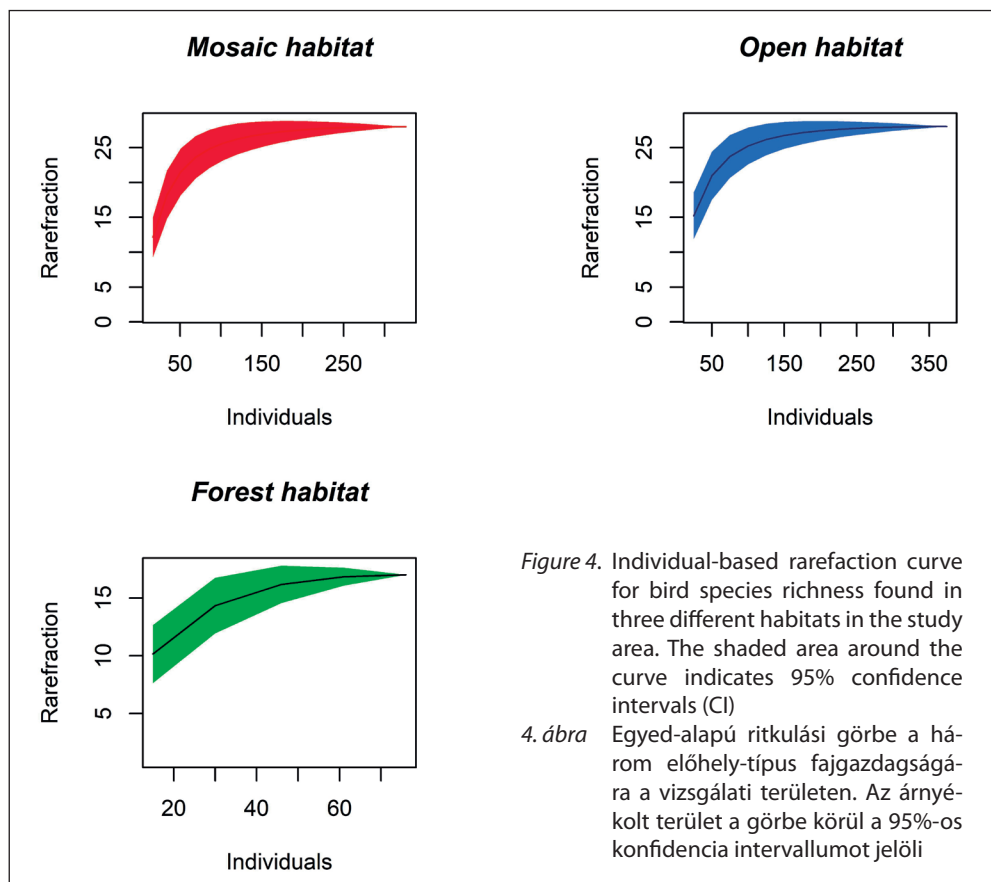
Figure 3. Richness and abundance of raptors in three habitats in the study area. The box is the interquartile range. The upper and lower edges of the box plots represent the 75% and 25% quartile of data, respectively. The horizontal line i.e. the median represents the 50% quartile. The vertical lines are the upper and lower whiskers

3. ábra A ragadozó madarak fajgazdagsága és gyakorisága a három élőhelytípusban a vizsgálati területen

in both open habitat and forest habitat. The raptors were more evenly distributed in mosaic habitat (Figure 5). Also, the Crested Serpent Eagle was the most abundant species in Manas National Park, followed by Collared Falconet (Figure 6).

Shared species richness and pairwise comparisons among habitats

During our survey, nineteen raptor species: Crested Serpent Eagle (*Spilornis cheela*), Changeable Hawk-eagle (*Nisaetus cirrhatus*) Black Eagle (*Ictinaetus malaiensis*), Booted Eagle (*Hieraaetus pennatus*), Short-toed Snake Eagle (*Circaetus gallicus*), Steppe Eagle (*Aquila nipalensis*), Greater Spotted Eagle (*Clanga clanga*), Black-winged Kite (*Elanus caeruleus*), Oriental Honey Buzzard (*Pernis ptilorhynchus*), Long-legged Buzzard (*Buteo rufinus*), Jerdon's Baza (*Aviceda jerdoni*), Shikra (*Accipiter badius*), Himalayan Vulture (*Gyps himalayensis*), Pied Harrier (*Circus melanoleucos*), Hen Harrier (*Circus cyaneus*), Peregrine Falcon (*Falco peregrinus*), Eurasian Kestrel (*Falco tinnunculus*), Red-necked Falcon (*Falco chicquera*), and Collared Falconet (*Microhierax caerulescens*) were found to occur in all the three habitats. Among the habitats, species richness was shared highest between mosaic (MO) and open (OP) habitats (27 species), followed by forest (FO) and open



(OP) habitats which shared 19 species, and forest (FO) and mosaic (MO) habitats also shared 19 species (Figure 7). Bray-Curtis index (BC) showed that there were higher dissimilarities between FO and OP, and Jaccard Index (JI) showed higher similarities between MO and OP (Table 2). The Morisita-Horn Index (MH) suggested that raptor compositional similarity was higher between FO and OP habitat, followed by FO and OP and FO and MO. Also, the Euclidean distance index (ED) showed greater similarity between FO and OP habitats, followed by FO and MO and lowest similarity between OP and MO habitats (Table 2).

Functional diversity

The functional alpha-diversity measured the variation is species composition in the three habitats based on 10 functional traits (Appendix 2, Table 3). The functional richness was highest in open habitats and lowest in forest habitats. The functional divergence was highest in mosaic habitat and lowest in forest habitat. Evenness in distribution of functional traits was highest in open habitats and lowest in mosaic habitats. A dendrogram was also plotted using linkage clustering of the 30 species based on their shared functional traits using Gower's distance which calculates the similarity between pairs of variables (Figure 7).

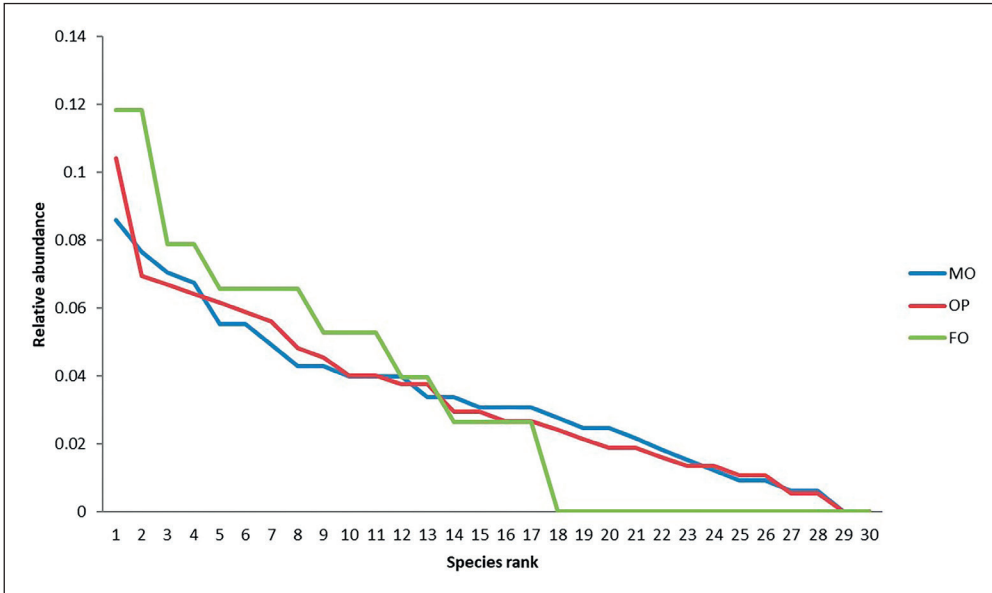


Figure 5. The rank-abundance curve showing the evenness of raptor species in each habitat type; MO = Mosaic habitat, OP = Open habitat and FO = Forest habitat

5. ábra Az egyes élőhelytípusokban a fajok egyenletességét mutató rang-abundancia görbe; MO = mozaik élőhely, OP = nyílt élőhely és FO = erdei élőhely

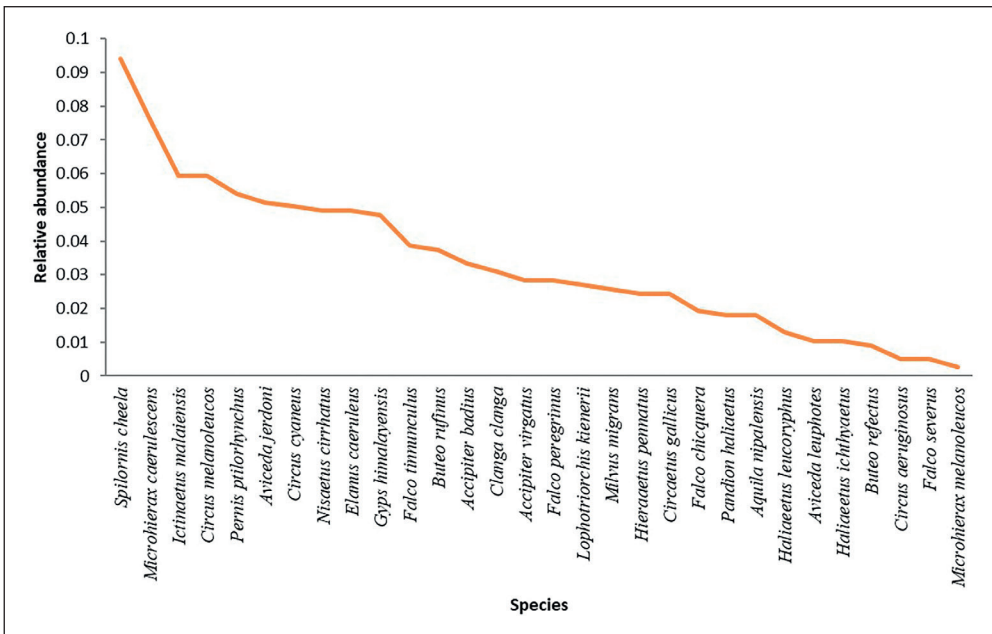


Figure 6. The relative abundance of species showing the most dominating raptor in Manas National Park

6. ábra A fajok relatív gyakorisága a legdominánsabb fajjal a Manasz Nemzeti Parkban

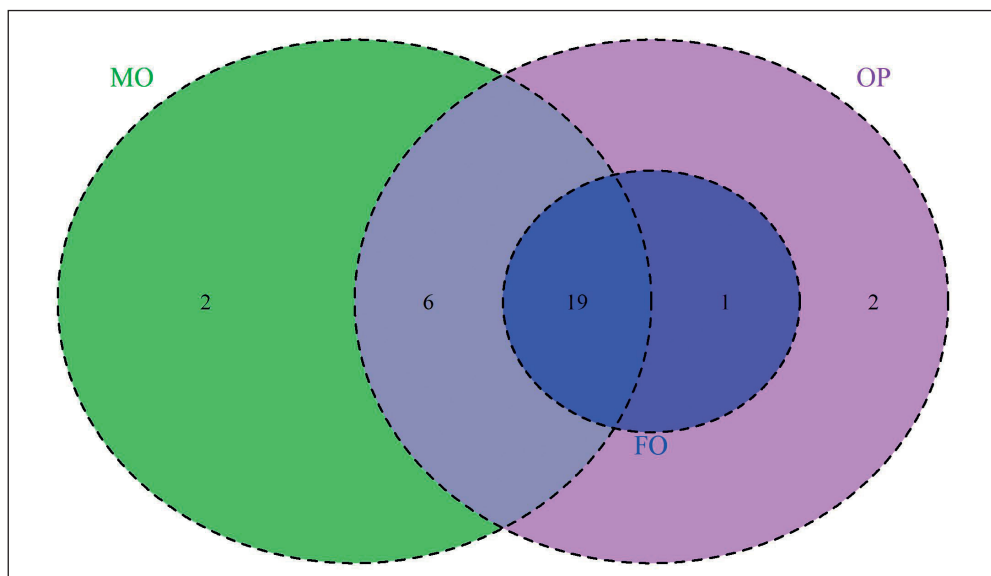


Figure 7. Venn diagram showing the number of shared species among the three different habitat types; MO: Mosaic habitat, FO: Forest habitat, OP: Open habitat

7. ábra Venn-diagram az egyes élőhely-típusok közös fajainak számával; MO: mozaik élőhely, OP: nyílt élőhely és FO: erdei élőhely

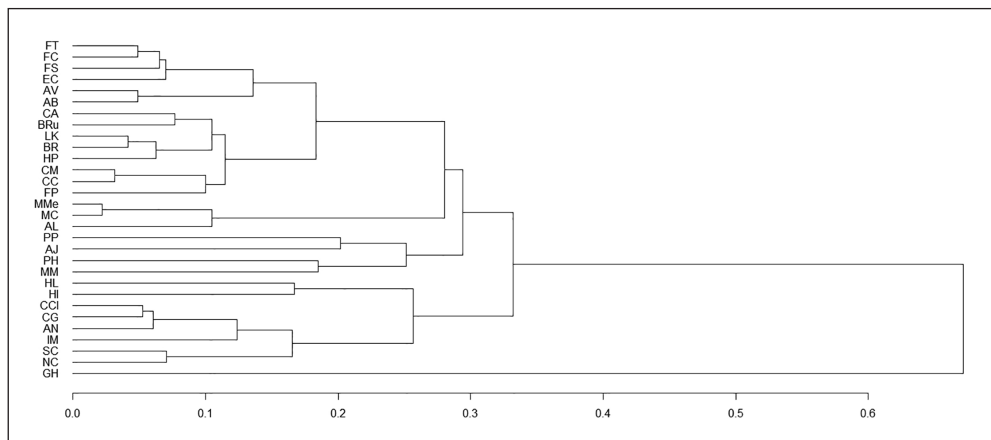


Figure 8. Cluster dendrogram showing the functional relationships shared by the 30 different species (FT: *Falco tinnunculus*, FC: *Falco chicquera*, FS: *Falco severus*, EC: *Elanus caeruleus*, AV: *Accipiter virgatus*, AB: *Accipiter badius*, CA: *Circus aeruginosus*, BRu: *Buteo rufinus*, LK: *Lophotriorchis kienerii*, BR: *Buteo reffectus*, HP: *Hieraaetus pennatus*, CM: *Circus melanoleucos*, CC: *Circus cyaneus*, FP: *Falco peregrinus*, MMe: *Microhierax melanoleucos*, *Microhierax caeruleus*, AL: *Aviceda leuphotes*, PP: *Pernis ptilorhynchus*, AJ: *Aviceda jerdoni*, PH: *Pandion haliaetus*, MM: *Milvus migrans*, HL: *Haliaeetus leucoryphus*, HI: *Haliaeetus ichthyaetus*, CCI: *Clanga clanga*, CG: *Circus gallicus*, AN: *Aquila nipalensis*, IM: *Ictinaetus malaiensis*, SC: *Spilornis cheela*, NC: *Nisaetus cirrhatus*, GH: *Gyps himalayensis*)

8. ábra A 30 különböző faj funkcionális jellegei alapján készített klaszter dendrogram. A betűk a fajok tudományos nevének rövidítései

Table 2. Pairwise comparisons of the bird communities among three different habitats (MO = Mosaic, OP = Open and FO = Forest)

2. táblázat A madárközösségek páronkénti összehasonlítása a három élőhelytípusban (MO = mozaikos, OP = nyílt, FO = erdei)

Similarity Index	Euclidean		Bray-Curtis		Morisita-Horn		Jaccard	
Habitat	MO	OP	MO	OP	MO	OP	MO	OP
OP	34.89		0.211		0.034		0.866	
FO	56.79	70.02	0.621	0.671	0.108	0.143	0.607	0.607

Discussion

Our study recorded 30 species and 776 individuals of raptors in three different habitats in Manas National Park. During the study period we observed highest number (28) of raptor species in open habitats. According to the study by Sergio *et al.* (2005), open habitats provide better hunting opportunities due to increased visibility. Also, raptors in open habitats have a higher success rate in capturing prey in open areas than in closed habitats (Eduardo *et al.* 2007). The mosaic habitats also showed almost similar number of species as open habitats. Mosaic ecosystems also have abundant foods for predatory raptors, as observed in other works (Kumar *et al.* 2022). Additionally, the physical complexity of mosaic landscapes presents varied roosting and nesting prospects for raptors, acting as suitable habitats (Jullien & Thiollay 1996). This can also be attributed to several factors such as abundance of mammalian and ground-nesting bird preys in open areas which form a primary diet in many raptors (Pedrana *et al.* 2008, Carrete *et al.* 2009, Tinajero *et al.* 2017, Kumar *et al.* 2022).

The Crested Serpent Eagle was the most abundant species in both open and forest habitats, usually seen perched in the forest and soaring in open areas closer to the forest margin. The common occurrence of the species could be attributed to its dietary items which mainly comprises of snakes and other small reptiles available in both open and forest habitats (Naorogi 2006). The Collared Falconet was dominant species in the mosaic habitat in terms of abundance, which was observed in open spaces with isolated trees for perching. Our observation on the species is supported by other studies (Naorogi 2006). We found nineteen raptor species utilizing all the three habitats, which suggested a wide food spectrum of these raptors. The raptor species shared the highest similarities between open and mosaic habitats. The open and mosaic habitats were utilized mainly by buzzards, harriers, vultures and eagles. These habitats have open areas such as wetlands, grasslands and riparian areas, which consists of different prey items for raptors such as ground-nesting birds, reptiles, small mammals, and wetland birds. Also, open habitats in Manas National Park have been known to be breeding ground of harriers (Narayan & Rosalind 1991).

We recorded 10 winter migratory raptors, compared to 20 resident species (*Appendix A*). The winter migrants were reported mainly from open habitats, which serve as an ideal foraging space. Eagles such as Greater Spotted (*Clanga clanga*) and Steppe Eagles (*Aquila nipalensis*) were often seen soaring during clear sunny mornings till late afternoon foraging for preys above the open habitats. The buzzards were seen perched on the trees present in the open habitats by

the wetlands, looking out for rodents, lizards and insects. The harriers were observed to be flying low over the grassland and marshes to capture prey with a sudden striking speed. The Booted Eagle (*Hieraaetus pennatus*) often seen perched by the wetland and marshes, flushing waterbirds and reptiles until they are preyed upon. Pallas's Fish-Eagles (*Haliaeetus leucoryphus*) were observed along the Beki river towards the northeastern side of the study area. They were seen hunting for fishes in pairs and sometimes alone. Their occurrences were mainly due to the fact that this species migrates to the region during winter to breed (Naoroji 2006). Among the migrants, two species: Pallas's Fish-Eagle and Steppe Eagle were Endangered (IUCN 2024) (Appendix 1). The Pallas's Fish-Eagle is also among the high priority species in India as per Research and Conservation Priority Index (Mahananda *et al.* 2022). The occurrence of globally threatened migratory species in the area signifies it to be a raptor conservation priority area.

Data analyses based on functional attributes are important in identifying certain trends of species composition in a habitat. Raptor species diversity based on Shannon-Wiener index showed no major differences between the three habitats, contrastingly, functional diversity analyses showed some differences in the species compositions between the three habitats. The open habitats showed higher functional richness due to the greater number of species with different morphological and functional traits such as beak length, beak width, tarsus length, wing length, body mass and trophic niche (Table 3) involved in various functional roles (Legras *et al.* 2018). The open habitat harbored species like Crested Serpent Eagle which is a vertivore in terms of trophic niche, and also aquatic predators such as Pallas's Fish-Eagle to Collared Falconet which is an invertivore (Appendix 2), thus suggesting abundance of different prey items for most raptors in the habitat. The higher functional divergence in mosaic habitats suggests that the differently sized raptors in the habitat have variety of resources to choose from without competition. Thus, habitats with higher functional divergence indicates efficient use of resources by the raptors leading to better ecosystem function (Mason *et al.* 2005).

Table 3. Functional diversity indices based on 10 traits in different habitats
3. táblázat Funkcionális diverzitás indexek 10 jelleg alapján a három élőhelytípusban

Habitat	No. of species	Functional richness (F.ric)	Functional evenness (F.eve)	Functional dispersal (F.dis)	Functional divergence (F.div)
Mosaic	27	0.84	0.60	0.45	0.75
Open	28	0.98	0.69	0.45	0.72
Forest	20	0.44	0.63	0.40	0.67

We can conclude that Manas National Park harbors a decent population of both resident and migratory raptors. Our research findings might help as a baseline data to further study the nesting ecology and behavior of the breeding raptors, including the endangered ones. Also, the study area supports open habitat raptors like Short-toed Snake Eagle, Black-winged Kite, and Red-necked Falcon, which have a declining population trend (SoIB 2023) and may face conservation challenges in future. Long-term research and monitoring are important to develop conservation strategies for these raptors.

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Appendix 1. Table showing the species and their abundance recorded in the study area, their migratory and IUCN status, and the functional traits used for functional diversity analyses (R: Resident, M: Migratory; LC: Least Concern, NT: Near Threatened, VU: Vulnerable, EN: Endangered)

1. melléklet Táblázat a vizsgált területen észlelt fajokról és gyakoriságukról, vonulási és IUCN-státuszukról, valamint a funkcionális diverzitás-elemzésekhez használt funkcionális jellemzőkről (R: rezidens, M: vonuló; LC: nem fenyegetett, NT: mérsékelten fenyegetett, VU: sebezhető, EN: veszélyeztetett)

Sl.No	Species	English name	Mosaic habitat	Open habitat	Forest habitat	Migratory status	IUCN status
1	<i>Accipiter badius</i>	Shikra	8	14	4	R	LC
2	<i>Accipiter virgatus</i>	Crested Goshawk	13	9	0	R	LC
3	<i>Aquila nipalensis</i>	Steppe Eagle	8	4	2	M	EN
4	<i>Aviceda jerdoni</i>	Jerdon's Baza	14	22	4	R	LC
5	<i>Aviceda leuphotes</i>	Black Baza	3	5	0	R	LC
6	<i>Buteo refectus</i>	Himalayan Buzzard	3	4	0	M	LC
7	<i>Buteo rufinus</i>	Long-legged Buzzard	13	7	9	M	LC
8	<i>Circaetus gallicus</i>	Short-toed Snake Eagle	6	11	2	R	LC
9	<i>Circus aeruginosus</i>	Western Marsh Eagle	4	0	0	M	LC
10	<i>Circus cyaneus</i>	Hen Harrier	18	21	0	M	LC
11	<i>Circus melanoleucos</i>	Pied Harrier	22	24	0	R	LC
12	<i>Clanga clanga</i>	Greater Spotted Eagle	14	6	4	M	VU
13	<i>Elanus caeruleus</i>	Black-winged Kite	18	17	3	R	LC
14	<i>Falco chicquera</i>	Red-necked Falcon	11	2	2	M	NT
15	<i>Falco peregrinus</i>	Peregrine Falcon	10	10	2	M	LC
16	<i>Falco severus</i>	Oriental Hobby	2	2	0	R	LC
17	<i>Falco tinnunculus</i>	Eurasian Kestrel	11	14	5	M	LC
18	<i>Gyps himalayensis</i>	Himalayan Vulture	16	15	6	M	NT
19	<i>Ichthyophaga ichthyaetus</i>	Grey-headed Fish Eagle	0	8	0	R	NT
20	<i>Haliaeetus leucoryphus</i>	Pallas's Fish Eagle	0	10	0	M	EN

Sl.No	Species	English name	Mosaic habitat	Open habitat	Forest habitat	Migratory status	IUCN status
21	<i>Hieraetus pennatus</i>	Booted Eagle	7	7	5	R	LC
22	<i>Ictinaetus malaiensis</i>	Black Eagle	23	18	5	R	LC
23	<i>Lophotriorchis kienerii</i>	Rufous-bellied Eagle	10	11	0	R	NT
24	<i>Microhierax caerulescens</i>	Collared Falconet	28	26	5	R	LC
25	<i>Microhierax melanoleucos</i>	Pied Falconet	2	0	0	R	LC
26	<i>Milvus migrans</i>	Black Kite	5	15	0	R	LC
27	<i>Nisaetus cirrhatus</i>	Changeable Hawk Eagle	10	25	3	R	LC
28	<i>Pandion haliaetus</i>	Osprey	9	5	0	M	LC
29	<i>Pernis ptilorhynchus</i>	Oriental Honey Buzzard	13	23	6	R	LC
30	<i>Spilornis cheela</i>	Crested Serpent Eagle	25	39	9	R	LC

Appendix 2. Functional traits used for the functional diversity analyses (AVONET database)
2. melléklet A funkcionális diverzitás-elemzésekhez használt funkcionális jellemzők (AVONET adatbázis)

Species	Beak. Length_ Culmen	Beak. Length_ Nares	Beak. Width	Beak. Depth	Tarsus. Length	Wing. Length	Kipps. Distance	HanWing. Index	Mass	Trophic. Niche
<i>Accipiter badius</i>	20.6	12.1	8.8	11.6	43	186.7	62.5	32.9	131.2	Vertivore
<i>Accipiter virgatus</i>	21.2	11.7	7.6	10.2	49.8	167.7	46.7	27.4	117	Vertivore
<i>Aquila nipalensis</i>	54.1	32.4	17.7	25.2	95.1	559.6	225.2	40.5	2714.3	Vertivore
<i>Aviceda jerdoni</i>	45.5	21.7	12.9	15.9	34.7	302.8	76.9	25.6	363	Omnivore
<i>Aviceda leuphotes</i>	24.6	16.4	10.2	12.5	25	230.6	82	35.3	194	Invertivore
<i>Buteo refectus</i>	33.4	20.4	11.8	16.1	64.9	346	125.7	36.3	759.1	Vertivore
<i>Buteo rufinus</i>	41.6	23.5	14	19.8	86.1	447.5	180.8	40.5	1166.2	Vertivore
<i>Circaetus gallicus</i>	47	35	15.5	24.6	90.9	537	216.8	40.5	1699.1	Vertivore
<i>Circus aeruginosus</i>	37.3	20.2	11.9	16.2	79.3	405.7	185.3	45.6	704.1	Vertivore
<i>Circus cyaneus</i>	30.1	16	9.7	13	72.1	352.4	158.9	44.7	393	Vertivore
<i>Circus melanoleucos</i>	29.9	15.5	9.4	13.1	72.5	352.6	179.8	50.5	336.3	Vertivore
<i>Clanga clanga</i>	50.2	31.2	15.6	23.2	96.8	510.6	192.2	37.9	2154.3	Vertivore
<i>Elanus caeruleus</i>	30.9	16	9.2	11.3	32.1	286.3	132.1	45.2	259.8	Vertivore
<i>Falco chicquera</i>	20	13.3	8.7	12	37.3	204.1	97.1	47.5	208	Vertivore
<i>Falco peregrinus</i>	31.1	18.4	13.7	15.4	45.1	314.7	165	52.2	760	Vertivore
<i>Falco severus</i>	23.7	13	9.3	11.3	28.5	217.4	128.8	58.6	200.6	Vertivore

Species	Beak. Length_ Culmen	Beak. Length_ Nares	Beak. Width	Beak. Depth	Tarsus. Length	Wing. Length	Kipps. Distance	HanWing. Index	Mass	Trophic. Niche
<i>Falco tinnunculus</i>	20.8	14.4	11	11.6	38.4	241.8	123.7	51	183.2	Vertivore
<i>Gyps himalayensis</i>	82.5	55.2	23.5	34	115.6	759.8	269.5	35.5	9798	Scavenger
<i>Ichthyophaga ichthyaetus</i>	66.1	36	16.8	25.7	81.7	471.2	134.6	29	1590	Aquatic. predator
<i>Haliaeetus leucoryphus</i>	63.6	39.1	19.3	28.3	95.3	566.6	241	42.8	2885.9	Aquatic. predator
<i>Hieraaetus pennatus</i>	33.8	19.1	10.1	17	65.3	361.8	151.3	42	823.8	Vertivore
<i>Ictinaetus malaiensis</i>	51.5	26.1	13.7	18.6	68.7	552.8	240.5	43.2	1028	Vertivore
<i>Lophotriorchis kienerii</i>	46.1	20.5	11.9	17.2	53.5	366.2	127.6	35.9	799	Vertivore
<i>Microhierax caerulescens</i>	14.4	9.8	8	9.1	21.7	102	39.7	39.2	38.7	Invertivore
<i>Microhierax melanoleucos</i>	15.2	10.4	8.7	10.2	24.3	111.4	45.3	40.4	64.2	Invertivore
<i>Milvus migrans</i>	40	22.4	13	17.4	51.4	457.4	219.9	48.1	734.1	Omnivore
<i>Nisaetus cirrhatus</i>	49.9	25.8	12.6	21.7	97.7	414.6	115.8	27.9	1475.1	Vertivore
<i>Pandion haliaetus</i>	47	27.7	17.3	19.6	49.8	468.4	204.8	44.9	1483.2	Aquatic. predator
<i>Pernis ptilorhynchus</i>	55.9	21.4	13.1	15.4	49	413.6	130.7	32	1141.1	Invertivore
<i>Spilornis cheela</i>	45.6	30.1	14.4	20.8	82.2	373.7	112.6	30.1	597.7	Vertivore



Avian assemblages pattern along forest-tea plantation gradients in the north bank landscape of Eastern Himalaya, Assam, India

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Abstract The study explores the patterns of avian communities along forest-tea plantation gradients in the north bank landscape of Eastern Himalayas, India, focusing on the Udalguri district of Assam. The present study aims to identify the impact of transitioning from undisturbed forest to tea plantations on bird diversity and composition. Bird surveys were conducted from August 2022 to June 2023 using point count methods at 16 sampling stations. The results revealed that forests exhibited higher species diversity and functional richness than tea plantations, which showed higher functional evenness and divergence. Additionally, forest edges supported greater species richness and abundance, emphasising their conservation significance. The study also observed significant differences in species composition at varying distances from the forest edge. Key species influencing these differences included the Yellow-footed Green Pigeon (*Treron phoenicopterus*) and various bulbul species (*Pycnonotus* spp.), indicating habitat sensitivity and edge effects. These findings suggest that while tea plantations can accommodate certain adaptable species, forests play a crucial role in providing habitats for a wider range of avian species. In addition, the study highlights the importance of conserving forest edges and comprehending avian responses to habitat mosaics for effective biodiversity conservation.

Keywords: Assam, Bornadi-Khalingduar forest complex, diversity, forest edge, species composition

Összefoglalás A tanulmány a madárközösségek mintázatát vizsgálja a Keleti-Himalája északi vidékének erdő-teaültvények gradiense mentén, Indiában, Assam Udalguri kerületére összpontosítva. A jelen tanulmány célja, hogy meghatározza az erdőből a teaültvényekbe való átmenet hatását a madárvilág diverzitására és összetételére. A madárfelméréseket 2022 augusztusától 2023 júniusáig végeztük pontszámilási módszerekkel, 16 mintavételi állomáson. Az eredmények alapján az erdők nagyobb faji diverzitást és funkcionális gazdagságot mutattak, mint a teaültvények, amelyek magasabb funkcionális egyenletességet és divergenciát mutattak. Emellett az erdőszegélyek nagyobb fajgazdagságot és egyedszámot támogattak, hangsúlyozva természetvédelmi jelentőségüket. Jelentős különbségeket figyelhetünk meg a fajösszetételben az erdőszegélytől való különböző távolságokban. A különbségeket meghatározó fajok közé tartozott a sárgalábú zöldgalamb (*Treron phoenicopterus*) és a különböző bulbúlfajok (*Pycnonotus* spp.), ami az élőhelyérzékenység és a szegélyek hatását jelzi. Ezek az eredmények azt sugallják, hogy míg a teaültvények bizonyos jól alkalmazkodó fajoknak biztosítanak élőhelyet, az erdők döntő szerepet játszanak a madárfajok szélesebb körének élőhelyeként. A tanulmány továbbá kiemeli az erdőszelek megőrzésének, és az élőhely-mozaikokra adott madárreakciók megértésének fontosságát a biodiverzitás hatékony megőrzése érdekében.

Kulcsszavak: Assam, Bornadi-Khalingduar erdő, diverzitás, erdőszegély, fajösszetétel

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Introduction

The forest edge, where forested areas meet other ecosystems, is vital for maintaining biodiversity and ecological functions. However, these edges are highly susceptible to changes in land use, mainly due to agricultural expansion and plantation development (Chaplin-Kramer *et al.* 2015). With about half of the Earth's land surface modified by human activity, landscapes now consist of a dynamic mix of native and non-native land covers varying in size, degree of separation, and land use intensity (Hansen *et al.* 2020). This includes urban developments, agricultural fields, reforested regions, and natural habitats impacted by human influence (Theobald *et al.* 2020, Ellis 2021). One prevalent form of such land-use change is the conversion of forests into tea plantations, which significantly alters habitat structure, resource availability, and microclimatic conditions (Hasan *et al.* 2020). These changes often lead to substantial shifts in the composition and diversity of bird communities (Moura *et al.* 2013, Nogueira *et al.* 2021).

Agroecosystems, including tea-forest landscapes, are human-modified areas in natural habitats formed by clearing existing forests and ecosystems (Chowdhury *et al.* 2021). Tea plantations once considered ecological wastelands, have emerged as critical habitats that can sustain significant bird diversity in the South Asian region (Kottawa-Arachchi *et al.* 2012, Sreekar *et al.* 2013, Imboma *et al.* 2020). However, the extent to which tea plantations can effectively substitute for natural forest habitats remains debatable (Chowdhury *et al.* 2021). Understanding these shifts is essential and urgent for conserving avian biodiversity and birds' ecological services, such as pollination, seed dispersal, and pest control. While tea plantations may support a significant number of bird species, the abundance and community structure of these avian assemblages can differ considerably from those found in primary forest ecosystems (Subasinghe & Sumanapala 2014, Chettri *et al.* 2018, Mola *et al.* 2021).

The Eastern Himalayan region, including the north bank landscape in Assam, India, is known for its remarkable avian diversity (Renner & Rappole 2011). While most studies primarily focus on the species diversity of forested areas, studies to understand the pattern of bird species distribution along the gradients of forest and human-modified habitats like tea plantations are lacking (Sodhi 2002, Ahmed & Dey 2014). Tea cultivation is a dominant land-use type in the Eastern Himalayan piedmont in Assam, and its expansion has significantly altered the region's once-extensive natural ecosystems (Prokop 2018). Tea plantations may serve as secondary habitat for various bird species, offering opportunities and challenges for avian conservation (Bhagwat *et al.* 2008). In the present study, we aim to investigate the patterns of bird diversity and composition along the gradient of forest and tea plantations, two prominent land cover types in the Assam region.

Material and Methods

Study area

The present study was conducted in the Udalguri district of Assam, India, a region of significant importance in the north bank landscape of the Eastern Himalayan region. The study area mainly covered two habitats: forest and tea plantation (*Figure 1*). The forest region,

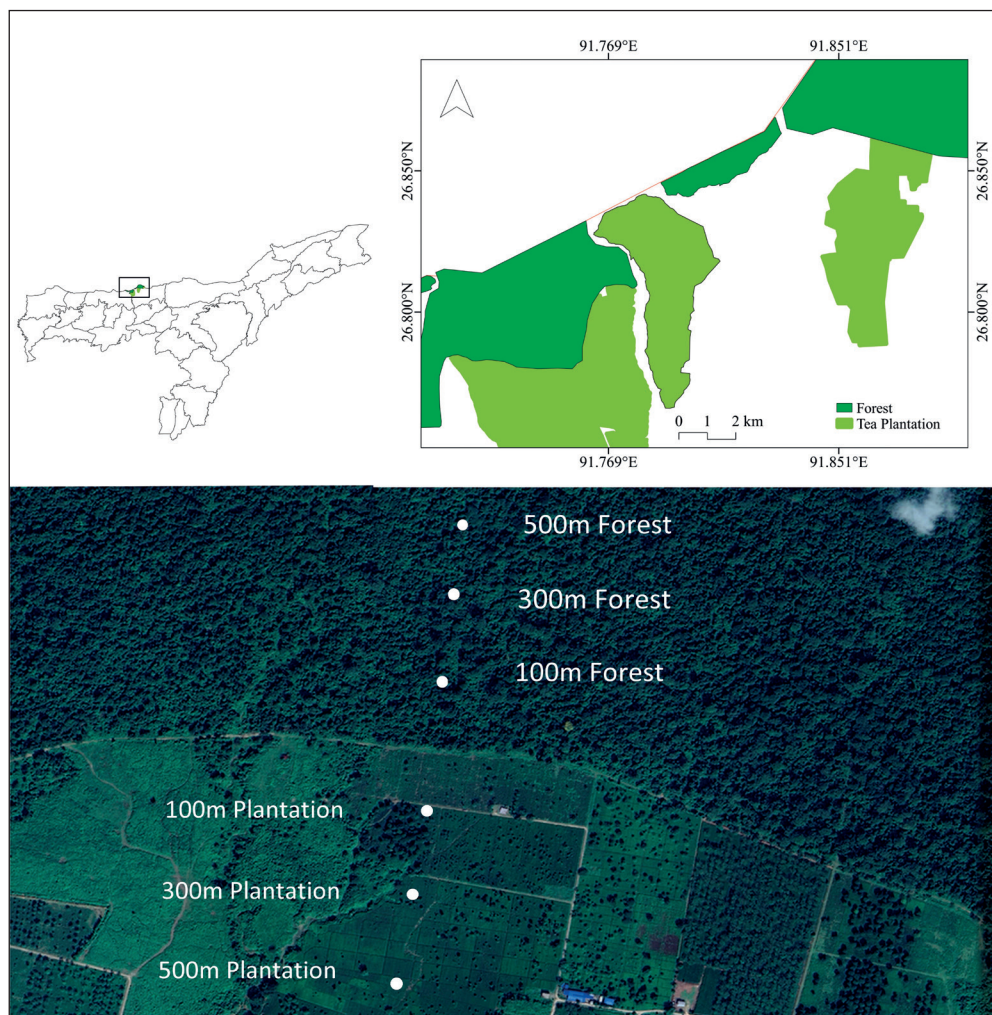


Figure 1. Map of the study area with an illustrative representation of the sampling points of the study design

1. ábra A vizsgálati terület térképe a mintavételi pontjok szemléltető ábrázolásával

known as the Bornadi-Khalingduar forest complex, mostly features Eastern Himalayan moist deciduous forest and lies at the foothill region of the Eastern Himalayas (Chakraborty *et al.* 2015). The forest complex spans an area of 108.03 km² (26.7649°N–26.9176°N, 91.7053°E–91.9635°E, 158–681 m a.s.l) and includes three forest regions: Bornadi Wildlife Sanctuary, Newlee proposed reserve forest, and Khalingduar reserve forest. The southern side of the forest complex is adjacent to human-modified habitats, mostly tea plantations. The region has a mild to moderate climate, with significantly higher rainfall in summer. The average annual temperature is 22.5 °C, and the annual rainfall is 2,621 mm (Climate-Data.org 2024).

Bird survey

Bird surveys were conducted from August 2022 to June 2023, covering all months. The study design comprised 16 sampling stations at the edge of the forest-plantation areas. Six point counts were taken at each sampling station: three inside the forest and three towards the plantation, perpendicular to the edge, each separated by a 200-metre distance. To minimise counting the same individuals from the nearest sampling station, each sampling station was separated by a distance of 1 km. Birds were observed and counted at each point for 15 minutes using the point count method considering a radius of 30 metres (Bibby *et al.* 2000). Each count was conducted between 06:00 and 08:30 hrs in fair weather conditions. Bird surveys excluded unfavourable weather conditions such as fog, wind, and human presence in tea plantations. Only birds seen and calls that could be distinctly heard and identified were considered for analysis. Species were categorised into five foraging guilds based on diet habit: FruiNect (feeding on fruits and nectar), Omnivore, Invertebrate (feeding on invertebrates), VertFishScav (birds feeding on vertebrates, fish, or scavengers) or PlantSeed (feeding on plants and seeds) (Wilman *et al.* 2014).

Data analysis

A sample coverage and species accumulation curve were plotted to evaluate whether the sampling effort was high enough for further analysis. Both analyses used iNEXT (iNterpolation and EXTrapolation) Online, an R-based interactive online version of iNEXT (Chao *et al.* 2016). Species diversity was analysed using the Shannon-Wiener Index in PAST version 4 software (Hammer *et al.* 2001), and functional diversity was obtained using the mFD package for R software (Magneville *et al.* 2022). We obtained each sampling point's Shannon-Wiener Index and functional diversity indices and considered the mean value. In the current study, three functional diversity indices, namely functional richness (FRic), functional evenness (FEve), and functional divergence (FDiv), were evaluated.

The one-way permutational analysis of variance (PERMANOVA) test (Anderson 2001) was used to determine significant differences in species composition and abundance among the various distance categories of forest and plantation from the edge. The significance level was tested at a P-value of 0.05. To further analyse the data, a pairwise comparison was conducted to observe any differences between the various distance categories in cases where the test yielded significant results.

The difference in the species assemblage composition across the distance gradient of the two habitats was observed by the analysis of similarity (ANOSIM) test, which utilises a dissimilarity matrix rather than raw data using Bray-Curtis distance (Clarke 1993). The ANOSIM statistic compares dissimilarities between and within groups. An R-value close to 1 suggests group dissimilarity, while a value close to 0 suggests even distribution within and between groups (Clarke & Warwick 2001). R values below 0 indicate greater dissimilarities within groups than between groups. Similarity percentage analysis (SIMPER) was employed to determine which species contribute to any dissimilarity between forest and tea plantation bird compositions. PERMANOVA, ANOSIM, and SIMPER analyses were performed using PAST software version 4.

Results

Species diversity

132 species of birds belonging to 13 orders and 49 families were observed during the study period. 30 species were observed exclusively from tea plantation, 35 exclusively from forest, and 67 were common in both habitats. The Shannon-Weiner Index was higher in forest ($H'=2.62$) than tea plantation ($H'=2.45$). Functional richness was higher in forest habitat ($FRic=0.11\pm0.02$) than tea plantation ($FRic=0.06\pm0.01$). However, tea plantation had higher Functional evenness ($FEve=0.59\pm0.02$) and divergence ($FDiv=0.78\pm0.02$) compared to forest habitat (Table 1).

Both forest and tea plantation reach a high sample coverage of 0.95 with a high number of individuals sampled (Figure 2). The species accumulation curve almost reached its plateau, signifying a reasonable sampling effort (Figure 2).

Table 1. Summary of species richness, abundance and diversity metrics of bird assemblages in forest and tea plantation habitats of the study area

1. táblázat A vizsgált terület erdei és teaültetvények élőhelyein élő madárállományok fajgazdagság, abundancia és diverzitás mérőszámainak összefoglalása

	Forest	Tea plantation
Species richness	102	97
Abundance	616	630
Shannon-Weiner Index (mean±SE)	2.62±0.09	2.45±0.10
Functional Richness (mean±SE)	0.11±0.02	0.06±0.01
Functional Evenness (mean±SE)	0.54±0.02	0.59±0.02
Functional Divergence (mean±SE)	0.74±0.02	0.78±0.02

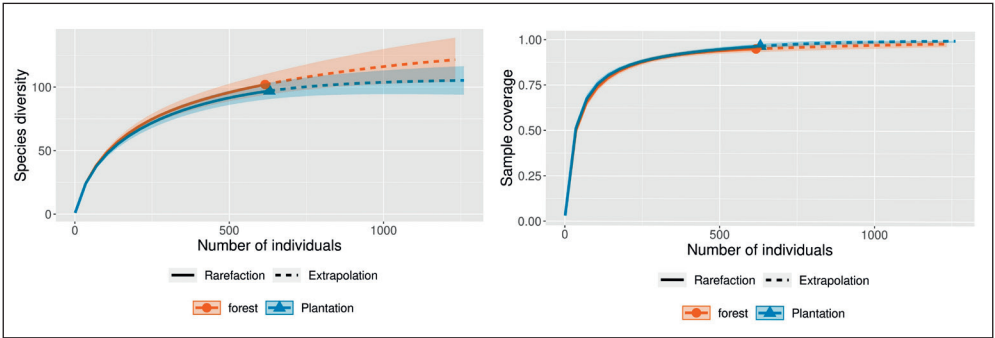


Figure 2. Species accumulation and sample coverage curve for forest and plantation sites using rarefaction and extrapolation methods, with a 95% confidence interval denoted by a shaded region

2. ábra Az erdei és ültetvényes területek fajtelítődései és mintavételi lefedettségü görbéje a rarefaction és az extrapolációs módszerekkel, a 95%-os konfidencia intervallumot árnyékolt terület jelöli

Richness and abundance

Species composition ($F = 3.02$, $P < 0.05$) significantly differed among the distance categories from the edge to forest and tea plantation (Table 2). No significant difference was observed for species abundance ($F = 1.58$, $P > 0.05$) across the distances from the edge.

Of the recorded species, 102 were observed in the forest and 97 in the tea plantation habitat. In forest habitat, Species richness ($n=79$) and abundance ($n=384$) were highest at 100 m distance from the edge. In contrast, species richness ($n=36$) and abundance ($n=97$) were lowest at 300 m forest distance from edge. In tea plantation habitat, species richness and abundance decreased as the distance from the forest edge increased. The highest species richness ($n=66$) and abundance ($n=340$) were observed at a distance of 100 m from the edge, while the lowest species richness ($n=43$) and abundance ($n=97$) were observed at a distance of 500 m (Figure 3).

Table 2. Pairwise p -values of the PERMANOVA test for species composition at various distances from the edge into forest and tea plantation. Bold numbers indicate significant values

2. táblázat A PERMANOVA-teszt páros p -értékei a fajgazdagságra vonatkozóan az erdő és a teaültetvény határától való különböző távolságokban. A félkövéren szedett számok szignifikáns értékeket jelölnek

Distance from edge	Tea plantation 100 m	Tea plantation 300 m	Tea plantation 500 m	Forest 100 m	Forest 300 m
Tea plantation 300 m	0.0293				
Tea plantation 500 m	0.0073	0.5880			
Forest 100 m	0.5795	0.0232	0.0075		
Forest 500 m	0.0148	0.8479	0.2499	0.0278	
Forest 300 m	0.0245	0.4520	0.1424	0.0136	0.1855

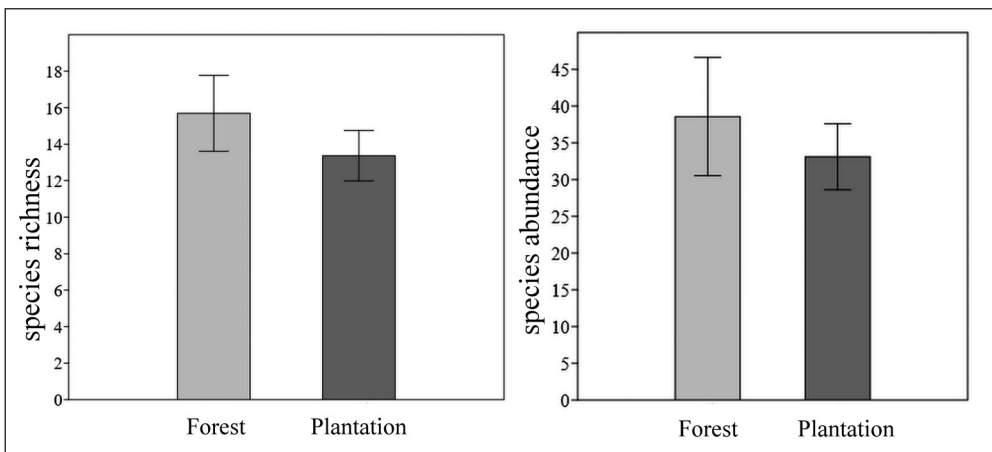


Figure 3. Species richness and abundance (mean \pm SE) in two different habitat types

3. ábra Fajgazdagság és abundancia (átlag \pm SE) két különböző élőhelytípusban

Species composition across the habitat gradients

A significant difference was observed in bird species assemblages across the forest-tea plantation distance gradient ($R=0.12$, $P<0.05$) (Figure 4). However, the moderate effect size ($R=0.12$) suggests that while the habitat gradients harbour distinct bird assemblages, there is also considerable species overlap. The pairwise comparison also showed a significant dissimilarity in the species composition at different distance categories except for the 100 m forest distance from the edge (Table 3).

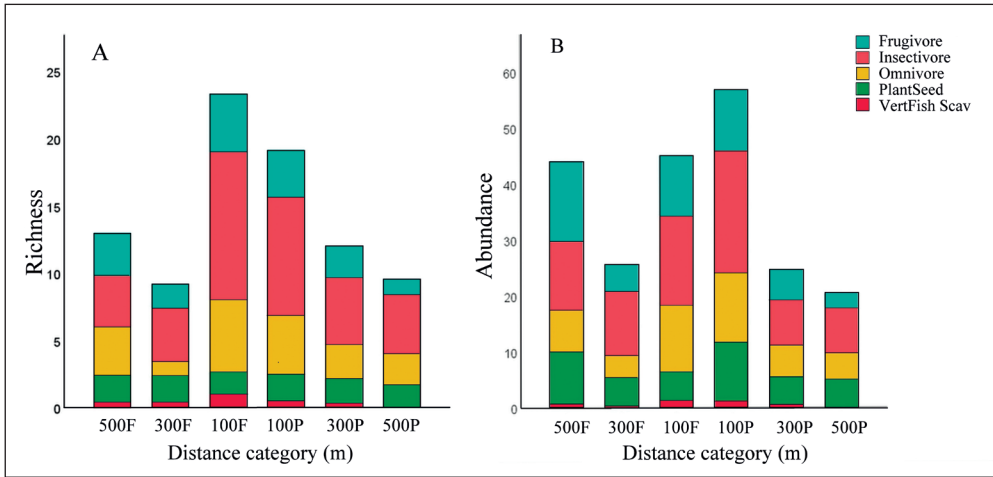


Figure 4. Mean species richness (A) and abundance (B) across distances (metres) from the edge of the forest (F) and tea plantation (P)

4. ábra Az átlagos fajgazdagság (A) és egyedszám (B) az erdő szélétől (F) és a teaültetvénytől (P) mért távolságok (méter) függvényében

Table 3. R-values and corresponding P-values of ANOSIM for comparing species composition between tea plantation (TP) and forest (F) at various distances from the edge (in metres). Bold numbers indicate significant values

3. táblázat Az ANOSIM R-értékei és a hozzátartozó P-értékek a teaültetvény (TP) és az erdő (F) fajösszetételének összehasonlítására a szegélytől való különböző távolságokban (méterben). A félkövéren szedett számok szignifikáns értékeket jelölnek

Distance from edge	500TP	300TP	100TP	100F	300F
300TP	-0.10 (0.79)				
100TP	0.26 (0.02)	0.11 (0.14)			
100F	0.21(0.04)	0.21 (0.01)	0.14 (0.10)		
300F	0.08 (0.24)	0.09 (0.23)	0.34 (0.01)	0.16 (0.06)	
500F	0.06 (0.29)	0.16 (0.11)	0.21 (0.05)	0.03 (0.33)	-0.16 (0.87)

SIMPER analysis reveals that among the various foraging guilds, FruitNect ($n=2$) and Omnivore-feeding birds ($n=3$) were the top five contributors to the dissimilarity of bird assemblages, with higher occurrence in forest habitats. The Yellow-footed Green Pigeon

Table 4. SIMPER test results for the top five contributing species to the dissimilarity between tea plantation and forest habitat [Co: contributing, Cu: cumulative, TP: tea plantation, F: forest]

4. táblázat A SIMPER teszt eredményei a teaültetvény és az erdei élőhely közötti különbségekhez leginkább hozzájáruló öt fajra vonatkozóan [Co: hozzájáruló, Cu: kumulatív, TP: teaültetvény, F: erdő]

Common Name	Foraging Guild	Average dissimilarity	Co %	Cu %	Mean abundance (TP)	Mean abundance (F)
Yellow-footed Green Pigeon	FruiNect	6.79	7.94	7.94	3.79	4.63
Black-crested Bulbul	FruiNect	4.17	4.88	12.83	1.47	2.44
Red-whiskered Bulbul	Omnivore	3.20	3.75	16.58	1.26	1.81
Blue-throated Barbet	Omnivore	2.93	3.43	20.01	1.95	0.43
Red-vented Bulbul	Omnivore	2.55	2.98	23.00	1.11	1.38

(*Treron phoenicopterus*) contributed the highest dissimilarity, accounting for 7.94%. Three species of bulbul, namely, Black-crested Bulbul (*Rubigula flaviventris*), Red-whiskered Bulbul (*Pycnonotus jocosus*), and Red-vented Bulbul (*Pycnonotus cafer*), were among the top five contributors to the dissimilarity of bird assemblages (Table 4).

Discussion

High species diversity was observed in forests compared to tea plantations, suggesting that forests provide a more heterogeneous and complex environment supporting a wider variety of bird species. Forests typically offer diverse food resources, nesting sites, and microhabitats, contributing to higher biodiversity (Cavard *et al.* 2011, Tinya *et al.* 2021). Additionally, the simplified vegetation structure of tea plantations, with limited undergrowth and complex canopy layers, may not offer the same nesting and foraging opportunities to birds that prefer more diverse forest habitats. This contrast in habitat complexity and resource availability between tea plantations and nearby forests could make the tea plantations less attractive to these bird species, reducing their presence.

The bird species found in the forest have a wide range of functional traits, indicating a broader diversity of roles within the ecosystem compared to bird species found in tea plantations. Forests provide a variety of ecological roles for birds, including insectivores, frugivores, canopy specialists, and ground foragers due to their structural complexity and abundant resources. This results in a highly diverse community of bird species in the forest (Pigot *et al.* 2016). In contrast, tea plantations are highly managed habitats where resources are more evenly distributed, leading to higher functional evenness. This means that bird communities in tea plantations have a more uniform species distribution across different ecological roles. The higher functional divergence in tea plantations suggests that species adapt and exploit resources more variedly. As a result, plantations support species that

can thrive in novel or disturbed environments, leading to a more functionally divergent community. The higher functional evenness and divergence suggest that plantations might serve as important habitats for certain adaptable or generalist species, contributing to landscape-level biodiversity.

The significant difference in bird species richness in the varying distance categories of forest and tea plantations signifies that landscape-scale metrics influence species richness. At the same time, abundance is affected by habitat quality (Wu *et al.* 2023). Both species richness and abundance were higher at distances closer to the edge. Such findings sharply contrast with previous studies in tropical regions, challenging our current understanding (Watson *et al.* 2004, Lindell *et al.* 2007, Zakaria *et al.* 2014). Although edge habitats in the tropical zone tend to have less avian diversity compared to the temperate zone (Báldi 1996), the possibility of such analogous findings might be due to the high abundance of frugivores and insectivores in the forest edge habitat of the tropical zone (Menke *et al.* 2012). Also, the composition of the feeding guild might be a determining factor for high species richness at the forest edge. High-contrast edges, such as those between forest and tea plantation habitats, are known to exhibit higher species richness across various latitudinal regions (Willmer *et al.* 2022). This is attributed to historical disturbances that favour generalist and resilient bird species, facilitate cross-habitat spillover, and are influenced by local factors such as the adjacent habitat matrix (Boesing *et al.* 2018, Willmer *et al.* 2022). In forests and tea plantation habitats, insectivores were the most prominent among the different foraging guilds, a finding consistent with other researches (Githiru *et al.* 2009, Ahmed & Dey 2014). Studies suggest that edge habitats ensure high prey availability and nesting choices due to abundant tree cavities, undergrowth, and shrubs (McCollin 1998, Keyel *et al.* 2013).

The ANOSIM results demonstrate the variation in bird species composition between forest edge and interior habitats across different distance categories. While the overall difference in species composition between the two habitats is evident, it is not substantial, suggesting some degree of species overlap. This suggests that spatial variation in habitat characteristics influences bird assemblages but not in a major way. It is, however, noticeable across habitats that a few bird species, like the Yellow-footed Green-Pigeon, Black-crested Bulbul, Red-whiskered Bulbul, Blue-throated Barbet, and Red-vented Bulbul, contributed the most to the observed differences. These species are apparently sensitive to habitat changes and thus may indicate the habitat edge effect on bird communities. Of these five species, birds from the FuitNect foraging guild, Yellow-footed Green Pigeon, and Black-crested Bulbul had higher mean abundances in the forest than in tea plantation. It is evident from previous studies that these birds are found less frequently in tea plantations adjacent to forests (Li *et al.* 2020). One possible explanation for the lower abundance of fruit and nectar-feeding birds in tea plantations near forests is the difference in the availability of their preferred food sources. Forest habitat offers a wide variety of fruits ranging from small berries to large fruits and nectar from flowering plants providing various food resources, whereas shaded tea plantations typically feature a monoculture of tea plants, with very few trees, which may not provide the same level of resources for these specialised bird species (Singh & Bhagwat 2013). A higher abundance of FruitNect birds also indicates a resilient forest, as they contribute to the dispersal of seeds and the pollination of plants (Blendinger 1999).

The present study provides evidence that forest – tea plantation edges are essential habitats that can hold high species assemblages. It highlights the intricate relationships between distance to edge and bird community assemblages. Conservation efforts should also focus on forest edges while understanding the role of plantation habitats in supporting avian biodiversity. Further study is needed to understand how birds respond to mosaic habitats of forests with tea plantations at a landscape level and whether such habitats can limit the distribution of some birds.

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Appendix 1. List of bird species recorded during the study period [TP: Tea Plantation, F: Forest, IUCN: International Union for Conservation of Nature, '+' : species present, '-' : species absent]

1. melléklet A vizsgálati időszak alatt észlelt madárfajok listája [TP: teaültetvény, F: erdő, IUCN: Természetvédelmi Világszövetség, "+" : jelenlévő faj, "-" : hiányzó faj]

Sl. No.	Family	Common Name	Scientific Name	TP	F	IUCN Status
Order – Galliformes						
1	Phasianidae	Indian Peafowl	<i>Pavo cristatus</i>	+	+	LC
2	Phasianidae	Kalij Pheasant	<i>Lophura leucomelanos</i>	-	+	LC
3	Phasianidae	Red Junglefowl	<i>Gallus gallus</i>	+	+	LC
Order – Pelecaniformes						
4	Ardeidae	Cattle Egret	<i>Bubulcus ibis</i>	+	-	LC
5	Ardeidae	Indian Pond Heron	<i>Ardeola grayii</i>	+	-	LC
Order – Accipitriformes						
6	Accipitridae	Besra	<i>Accipiter virgatus</i>	-	+	LC
7	Accipitridae	Crested Goshawk	<i>Accipiter trivirgatus</i>	-	+	LC
8	Accipitridae	Shikra	<i>Accipiter badius</i>	-	+	LC

Sl. No.	Family	Common Name	Scientific Name	TP	F	IUCN Status
9	Accipitridae	Crested Serpent Eagle	<i>Spilornis cheela</i>		+	LC
10	Accipitridae	Oriental Honey Buzzard	<i>Pernis ptilorhynchus</i>	+	-	LC
Order – Strigiiiformes						
11	Strigidae	Oriental Scops Owl	<i>Otus sunia</i>	+	-	LC
Order – Columbiformes						
12	Columbidae	Asian Emerald Dove	<i>Chalcophaps indica</i>	-	+	LC
13	Columbidae	Eurasian Collared Dove	<i>Streptopelia decaocto</i>	+	-	LC
14	Columbidae	Green Imperial Pigeon	<i>Ducula aenea</i>	+	-	NT
15	Columbidae	Oriental Turtle Dove	<i>Streptopelia orientalis</i>	-	+	LC
16	Columbidae	Red Collared Dove	<i>Streptopelia tranquebarica</i>	+	+	LC
17	Columbidae	Rock Pigeon	<i>Columba livia</i>	+	+	LC
18	Columbidae	Spotted Dove	<i>Streptopelia chinensis</i>	+	+	LC
19	Columbidae	Wedge-tailed Green Pigeon	<i>Treron apicauda</i>	-	+	LC
20	Columbidae	Yellow-footed Green Pigeon	<i>Treron phoenicoptera</i>	-	+	LC
Order – Psittaciformes						
21	Psittacidae	Red-breasted Parakeet	<i>Psittacula alexandri</i>	+	-	NT
Order – Cuculiformes						
22	Cuculidae	Asian Koel	<i>Eudynamys scolopaceus</i>	-	+	LC
23	Cuculidae	Common Hawk Cuckoo	<i>Hierococcyx varius</i>	+	+	LC
24	Cuculidae	Greater Coucal	<i>Centropus sinensis</i>	+	-	LC
25	Cuculidae	Green-billed Malkoha	<i>Phaenicophaeus tristis</i>	+	-	LC
26	Cuculidae	Indian Cuckoo	<i>Cuculus micropterus</i>	+	-	LC
27	Cuculidae	Square-tailed Drongo Cuckoo	<i>Surniculus lugubris</i>	-	+	LC
Order – Coraciiformes						
28	Coraciidae	Indochinese Roller	<i>Coracias affinis</i>	-	+	LC
29	Meropidae	Blue-bearded Bee-eater	<i>Nyctornis athertoni</i>	+	+	LC
30	Meropidae	Blue-tailed Bee-eater	<i>Merops philippinus</i>	-	+	LC
31	Meropidae	Chestnut-headed Bee-eater	<i>Merops leschenaulti</i>	+	+	LC
32	Meropidae	Green Bee-eater	<i>Merops orientalis</i>	+	+	LC
33	Upupidae	Eurasian Hoopoe	<i>Upupa epops</i>	+	+	LC
Order – Bucerotiformes						
34	Bucerotidae	Oriental Pied Hornbill	<i>Anthracoceros albirostris</i>	+	-	LC
35	Bucerotidae	Wreathed Hornbill	<i>Rhyticeros undulatus</i>	+	+	VU
Order – Piciformes						
36	Picidae	Black-rumped Flameback	<i>Dinopium benghalense</i>	+	+	LC
37	Picidae	Fulvous-breasted Woodpecker	<i>Dendrocopos macei</i>	+	+	LC

Sl. No.	Family	Common Name	Scientific Name	TP	F	IUCN Status
38	Picidae	Greater Flameback	<i>Chrysocolaptes guttacristatus</i>	+	+	LC
39	Picidae	Greater Yellownape	<i>Chrysophlegma flavinucha</i>	+	-	LC
40	Picidae	Grey-headed Woodpecker	<i>Picus canus</i>	+	-	LC
41	Picidae	Lesser Yellownape	<i>Picus chlorolophus</i>	-	+	LC
42	Picidae	Rufous Woodpecker	<i>Micropternus brachyurus</i>	+	-	LC
43	Picidae	Speckled Piculet	<i>Picumnus innominatus</i>	+	+	LC
44	Megalaimidae	Blue-eared Barbet	<i>Psilopogon duvaucelii</i>	+	+	LC
45	Megalaimidae	Blue-throated Barbet	<i>Psilopogon asiaticus</i>	+	+	LC
46	Megalaimidae	Coppersmith Barbet	<i>Psilopogon haemacephalus</i>	+	+	LC
47	Megalaimidae	Great Barbet	<i>Psilopogon virens</i>	-	+	LC
48	Megalaimidae	Lineated Barbet	<i>Psilopogon lineata</i>	+	-	LC
Order – Caprimulgiformes						
49	Caprimulgidae	Large-tailed Nightjar	<i>Caprimulgus climacurus</i>	-	+	LC
50	Caprimulgidae	Savanna Nightjar	<i>Caprimulgus affinis</i>	-	+	LC
Order – Apodiformes						
51	Apodidae	Asian Palm Swift	<i>Cypsiurus balaisiensis</i>	-	+	LC
52	Apodidae	House Swift	<i>Apus nipalensis</i>	-	+	LC
Order – Passeriformes						
53	Acrocephalidae	Blyth's Reed Warbler	<i>Acrocephalus dumetorum</i>	+	+	LC
54	Aegithinidae	Common Iora	<i>Aegithina tiphia</i>	+	-	LC
55	Alaudinae	Oriental Skylark	<i>Alauda gulgula</i>	-	+	LC
56	Alcippeidae	Brown-cheeked Fulvetta	<i>Alcippe poioicephala</i>	+	+	LC
57	Artamidae	Ashy Woodswallow	<i>Artamus fuscus</i>	+	+	LC
58	Campephagidae	Black-winged Cuckooshrike	<i>Lalage melaschistos</i>	+	+	LC
59	Campephagidae	Large Cuckooshrike	<i>Coracina macei</i>	-	+	LC
60	Campephagidae	Long-tailed Minivet	<i>Pericrocotus ethologus</i>	+	+	LC
61	Campephagidae	Scarlet Minivet	<i>Pericrocotus speciosus</i>	+	+	LC
62	Campephagidae	Small Minivet	<i>Pericrocotus cinnamomeus</i>	+	+	LC
63	Chloropseidae	Blue-winged Leafbird	<i>Chloropsis moluccensis</i>	+	+	LC
64	Chloropseidae	Golden-fronted Leafbird	<i>Chloropsis aurifrons</i>	+	+	LC
65	Chloropseidae	Orange-bellied Leafbird	<i>Chloropsis hardwickii</i>	+	+	LC
66	Cisticolidae	Common Tailorbird	<i>Orthotomus sutorius</i>	+	-	LC
67	Cisticolidae	Zitting Cisticola	<i>Cisticola juncidis</i>	+		LC
68	Corvidae	Grey Treepie	<i>Dendrocitta formosae</i>	-	+	LC
69	Corvidae	House Crow	<i>Corvus splendens</i>	+	-	LC
70	Corvidae	Large-billed Crow	<i>Corvus macrorhynchos</i>	+	-	LC

Sl. No.	Family	Common Name	Scientific Name	TP	F	IUCN Status
71	Corvidae	Rufous Treepie	<i>Dendrocitta vagabunda</i>	+	+	LC
72	Dicaeidae	Plain Flowerpecker	<i>Dicaeum minullum</i>	+	+	LC
73	Dicaeidae	Scarlet-backed Flowerpecker	<i>Dicaeum cruentatum</i>	+	+	LC
74	Dicruridae	Ashy Drongo	<i>Dicrurus leucophaeus</i>	+	+	LC
75	Dicruridae	Black Drongo	<i>Dicrurus macrocercus</i>	+	+	LC
76	Dicruridae	Bronzed Drongo	<i>Dicrurus aeneus</i>	+	+	LC
77	Dicruridae	Greater Racket-tailed Drongo	<i>Dicrurus paradiseus</i>	+	+	LC
78	Dicruridae	Hair-crested Drongo	<i>Dicrurus hottentottus</i>	+	+	LC
79	Dicruridae	Lesser Racket-tailed Drongo	<i>Dicrurus remifer</i>	-	+	LC
80	Estrildidae	Chesnut Munia	<i>Lonchura atricapilla</i>	+		LC
81	Irenidae	Asian Fairy Bluebird	<i>Irena puella</i>	+	+	LC
82	Laniidae	Brown Shrike	<i>Lanius cristatus</i>	+	-	LC
83	Laniidae	Grey-backed Shrike	<i>Lanius tephronotus</i>	+	+	LC
84	Laniidae	Long-tailed Shrike	<i>Lanius schach</i>	+	+	LC
85	Leiothrichidae	Greater Necklaced Laughingthrush	<i>Pterorhinus pectoralis</i>	-	+	LC
86	Leiothrichidae	Rufous-necked Laughingthrush	<i>Pterorhinus ruficollis</i>	+	+	LC
87	Locustellidae	Striated Grassbird	<i>Megalurus palustris</i>	+	+	LC
88	Monarchidae	Black-naped Monarch	<i>Hypothymis azurea</i>		+	LC
89	Muscicapidae	Blue Rock Thrush	<i>Monticola solitarius</i>	+		LC
90	Muscicapidae	Grey-headed Canary Flycatcher	<i>Culicicapa ceylonensis</i>	+	+	LC
91	Muscicapidae	Oriental Magpie Robin	<i>Copsychus saularis</i>	+		LC
92	Muscicapidae	Pale Blue Flycatcher	<i>Cyornis unicolor</i>		+	LC
93	Muscicapidae	Rufous-gorgeted Flycatcher	<i>Ficedula strophilata</i>	+	+	LC
94	Muscicapidae	Siberian Stonechat	<i>Saxicola maurus</i>	+	-	LC
95	Muscicapidae	Small Niltava	<i>Niltava macgrigoriae</i>	+	+	LC
96	Muscicapidae	Taiga Flycatcher	<i>Ficedula albicilla</i>	+	+	LC
97	Muscicapidae	Verditer Flycatcher	<i>Eumyias thalassinus</i>	-	+	LC
98	Muscicapidae	White-rumped Shama	<i>Copsychus malabaricus</i>	-	+	LC
99	Nectariniidae	Crimson Sunbird	<i>Aethopyga siparaja</i>	+	-	LC
100	Nectariniidae	Purple Sunbird	<i>Cinnyris asiaticus</i>	+	+	LC
101	Nectariniidae	Ruby-cheeked Sunbird	<i>Chalcoparia singalensis</i>	+	+	LC
102	Oriolidae	Black-hooded Oriole	<i>Oriolus xanthornus</i>	+	+	LC
103	Oriolidae	Maroon Oriole	<i>Oriolus traillii</i>		+	LC
104	Passeridae	Eurasian Tree Sparrow	<i>Passer montanus</i>	+	+	LC
105	Pellorneidae	Abbott's Babbler	<i>Malacocincla abbotti</i>	-	+	LC
106	Phylloscopidae	Blyth's Leaf Warbler	<i>Phylloscopus reguloides</i>	+	+	LC

Sl. No.	Family	Common Name	Scientific Name	TP	F	IUCN Status
107	Phylloscopidae	Green-crowned Warbler	<i>Phylloscopus burkii</i>	+	+	LC
108	Phylloscopidae	Greenish Warbler	<i>Phylloscopus trochiloides</i>	+	+	LC
109	Phylloscopidae	Tickell's Leaf Warbler	<i>Phylloscopus affinis</i>	+	+	LC
110	Ploceidae	Baya Weaver	<i>Ploceus philippinus</i>	+		LC
111	Pycnonotidae	Black Bulbul	<i>Hypsipetes leucocephalus</i>	-	+	LC
112	Pycnonotidae	Black-crested Bulbul	<i>Pycnonotus flaviventris</i>	+	+	LC
113	Pycnonotidae	Red-vented Bulbul	<i>Pycnonotus cafer</i>	+	+	LC
114	Pycnonotidae	Red-whiskered Bulbul	<i>Pycnonotus jocosus</i>	+	+	LC
115	Pycnonotidae	White-throated Bulbul	<i>Alophoixus flaveolus</i>	+	+	LC
116	Rhipiduridae	White-throated Fantail	<i>Rhipidura albicollis</i>	+	+	LC
117	Sturnidae	Asian Pied Starling	<i>Gracupica contra</i>	+	+	LC
118	Sturnidae	Chestnut-tailed Starling	<i>Sturnia malabarica</i>	+	+	LC
119	Sturnidae	Common Hill Myna	<i>Gracula religiosa</i>	+	-	LC
120	Sturnidae	Common Myna	<i>Acridotheres tristis</i>	+	+	LC
121	Sturnidae	Great Myna	<i>Acridotheres grandis</i>	+	-	LC
122	Sturnidae	Jungle Myna	<i>Acridotheres fuscus</i>	+	+	LC
123	Sturnidae	Spot-winged Starling	<i>Saroglossa spilopterus</i>	-	+	LC
124	Timaliidae	Jungle Babbler	<i>Turdoides striata</i>	+	+	LC
125	Timaliidae	Pin-striped Tit Babbler	<i>Mixornis gularis</i>	-	+	LC
126	Turdinae	Blue Whistling Thrush	<i>Myophonus caeruleus</i>	+	+	LC
127	Turdinae	Orange-headed Thrush	<i>Geokichla citrina</i>	+	+	LC
128	Vangidae	Large Woodshrike	<i>Tephrodornis virgatus</i>	-	+	LC
129	Zosteropidae	Indian White-eye	<i>Zosterops palpebrosus</i>	+	-	LC
130	Phylloscopidae	Dusky Warbler	<i>Phylloscopus fuscatus</i>	+	+	LC
131	Pittidae	Hooded Pitta	<i>Pitta sordida</i>	+	+	LC
132	Scotocercidae	Mountain Tailorbird	<i>Phyllergates cucullatus</i>	-	+	LC



Species composition, diversity and foraging guilds of avifauna in two freshwater lakes of Mysore district, Karnataka, India

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Abstract We conducted the present study in two freshwater lakes in the Mysore district of Karnataka, India, from December 2023 to May 2024, with the aim of evaluating the diversity, relative abundance, and feeding guilds of avifauna in Giribetta and Hadinaru lakes. The “point count” method was used to evaluate the species composition of birds. A total of 5,790 individuals, representing 106 species from 18 orders and 48 families, were recorded. Four species were in the near-threatened category: Painted Stork (*Mycteria leucocephala*), Black-headed Ibis (*Threskiornis melanocephalus*), Oriental Darter (*Anhinga melanogaster*), and Spot-billed Pelican (*Pelecanus philippensis*), while the others were of least concern. In terms of species richness, Giribetta Lake had 87 species belonging to 18 orders and 42 families, as compared to Hadinaru Lake, which had 86 species from 16 orders and 40 families. The two areas had similar species composition, with a Sorenson index of 76.3% and a Jaccard index of 61.7%, which indicates that the species diversity is fairly similar across the habitats. Based on the diet and foraging habitat, the recorded bird species were classified into eight major feeding guilds, which revealed a higher abundance of carnivorous and insectivorous birds, followed by omnivorous, granivorous, frugivorous, nectarivorous, herbivorous, and mixed guild types. It is expected that this study will provide a preliminary database for the waterbirds of this area, useful for further research and assessment.

Keywords: species diversity, relative abundance, diversity indices, feeding guild, point count method

Összefoglalás A vizsgálatot az indiai Karnataka Mysore körzetében található Giribetta- és Hadinaru-tó madárvilágának diverzitása, a fajok relatív abundanciája és a táplálkozási guilddek felmérésére végeztük 2023 decembere és 2024 májusa között. A madarak fajösszetételét pontszámlálási eljárással határoztuk meg. A két tónál összesen 5790 egyedet jegyeztünk fel, amelyek 18 rendből és 48 családból 106 fajt képviseltek. Négy faj esett a „mérsékelten veszélyeztetett” kategóriába: a hindu gólya (*Mycteria leucocephala*), a feketefejű gödény (*Threskiornis melanocephalus*), a feketehasú kígyónyakúmadár (*Anhinga melanogaster*) és a foltoscsőrű gödény (*Pelecanus philippensis*), míg a többiek a „nem veszélyeztetett” csoportba tartoztak. A Giribetta-tónál 18 rendbe, 42 családba tartozó 87 faj, a Hadinaru-tónál 16 rendbe, 40 családba sorolható 86 faj volt jelen. A két terület összehasonlításában a Sorenson-index 76,3%, míg a Jaccard-index 61,7% értéke azt jelzi, hogy a fajok diverzitása meglehetősen hasonló az élőhelyeken. A táplálék és a táplálkozásra használt élőhely alapján a feljegyzett madárfajokat nyolc fő táplálkozó guildbe soroltuk, amelyek a húsevő és rovarevő madarak nagyobb egyedszámát mutatták, ezt követték a mindenevő, magevő, gyümölcsevő, nektárevő, növényevő és vegyes guilddek. A tanulmány egy előzetes adatbázist nyújt a terület vízimadaráiról, amely hasznos lehet a további kutatásokhoz és értékelésekhez.

Kulcsszavak: fajdiverzitás, relatív abundancia, diverzitási indexek, táplálkozási guild, pontszámlálási módszer

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Introduction

More than 10,000 species of birds have been identified worldwide out of which 13% of the species are known to reside in the Indian subcontinent (Grimmett *et al.* 2016). Birds perform various ecological roles as scavengers, pollinators, seed dispersal agents, insect pest predators and bioindicators (Padmavathy *et al.* 1970). Due to their high mobility and habitat selectivity, they are often used as indicators of ecosystem health (Gregory *et al.* 2003). Furthermore, the study of bird diversity is an integral part of the biodiversity assessment of a particular geographical region, and biodiversity surveys cannot be complete by ignoring avifaunal diversity.

India has over 58.2 million hectares of wetland cover and is known to support more than 310 bird species (Prasad *et al.* 2002, Kumar *et al.* 2005). More than half of the wetland birds that are found in India are migratory, migrating from the colder regions of central Asia, Russia, China, and the lower Himalayas (Harisha & Hosetti 2018). Wetlands provide key habitats for both resident and migratory birds for breeding, drinking, feeding, resting, and for social interactions (Kaur & Braich 2021). These freshwater bodies often succumb to changes in land use in their catchment areas, resulting in reduced inflows and a declining quality of the “runoff” that passes through agricultural fields and urban areas (Verma *et al.* 2001). The depletion of wetlands as a result of numerous human activities and/or climate change poses a serious threat to the diversity and number of waterbird species worldwide, with some species becoming extinct across several wetlands (Wetlands International 2012).

Mysore city, the administrative headquarters of Mysore district, is the second fastest growing city in the state of Karnataka and is known for its palaces, pilgrimage centres, gardens, lakes and rivers. Due to the steady growth of population, there is a continuous expansion of urban areas across all landscapes to create residential layouts, which has led to widespread destruction of natural habitats and pollution of several waterbodies in and around the district headquarters. The wetlands of the study area and their surroundings support a variety of resident as well as migratory bird species, such as the Bar-headed Goose (*Anser indicus*), Northern Shoveler (*Spatula clypeata*), Garganey (*Spatula querquedula*), Whiskered Tern (*Chlidonias hybrida*), Black-winged Stilt (*Himantopus himantopus*), Glossy Ibis (*Plegadis falcinellus*) and Barn Swallow (*Hirundo rustica*).

Understanding biodiversity and the composition of bird species in relation to habitat variation is critical to assess the condition of the local ecosystem and to implement successful conservation strategies. In this regard, an attempt has been made to understand the species composition, diversity, relative abundance and feeding guilds of birds in Giribetta Lake and Hadinaru Lake of Mysore district both experiencing varying levels of human interference.

Material and Methods

Study area

The study was conducted in two freshwater perennial lakes in the Mysore district of Karnataka state. The two lakes are located at a distance of 9 kilometres apart (*Figure 1*). Giribetta Lake (Site G) is 17 km away from district headquarters, located adjacent to the Mysore-Trichy national highway. It lies at $12^{\circ}25'93''\text{N}$ and $76^{\circ}77'30''\text{E}$, at an altitude of 688 metres above mean sea level. The lake has an approximate water-spread area of 55.54 acres. The lake has a fair portion of submerged, floating, and emergent vegetation that attracts a lot of waterbirds. The major sources of lake water are precipitation, surface runoff, and Varuna canal water. The lake is surrounded by a diverse amount of vegetation, both cultivated (coconut *Cocos nucifera*, arecanut *Areca catechu*, mango *Mangifera indica*, banana *Musa* sp., jack fruit *Artocarpus heterophyllus*) and wild plants: *Lantana*, *Tecoma*, *Agave*, *Calotropis*, *Pterolobium indicum*, *Ficus* sp., *Albizia* sp., *Acacia* sp., *Eucalyptus* sp., etc.

Hadinaru Lake (Site H) is situated in the Nanjangud Taluk of Mysore district, which is 33 km away from district headquarters. It lies at $12^{\circ}17'32''\text{N}$ and $76^{\circ}75'42''\text{E}$, at an altitude of 653 metres above mean sea level. The water-spread area of the lake is around 88.68 acres, with an independent catchment area of 8.57 km². The lake's water is replenished by canal water from

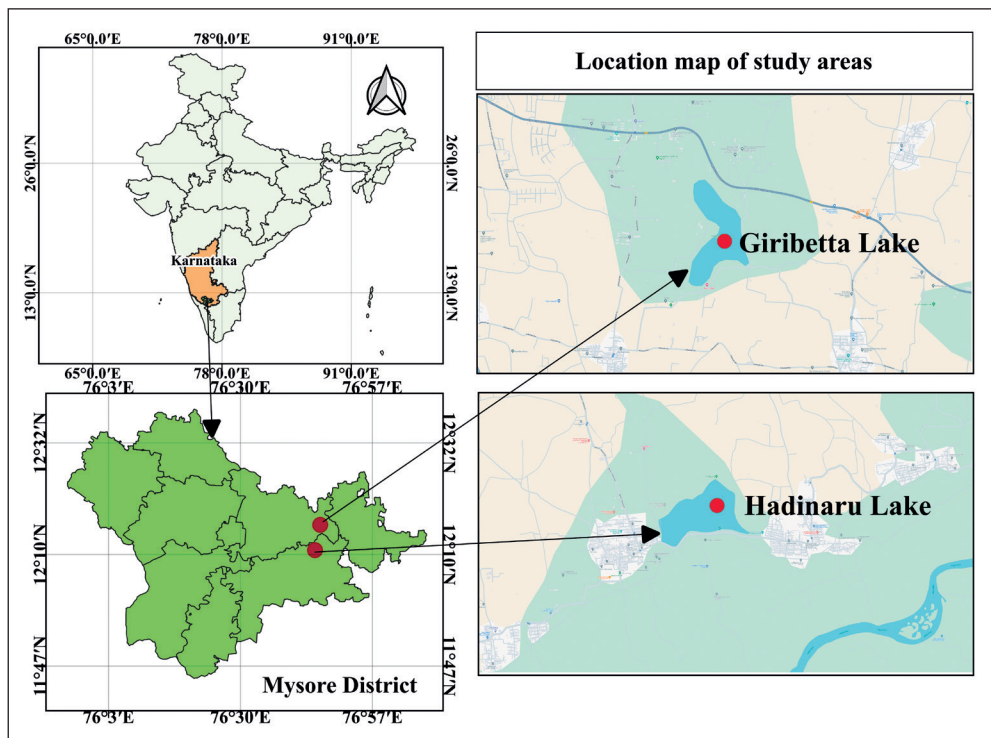


Figure 1. Geographic location of the study area

1. ábra A vizsgált terület földrajzi elhelyezkedése

the River Cauvery, rainfall, and runoff from the catchment area. The lake is teeming with aquatic hydrophytes such as water lily (*Nymphaea pubescens*), horn wort (*Ceratophyllum demersum*), cattail (*Typha latifolia*), lotus (*Nelumbo*), tape grass (*Vallisneria*), and duck lettuce (*Ottelia alismoides*). The lake's water catchment area is covered with *Acacia* trees, and the trees are annually flooded, providing an important roosting spot for water birds. There is an island in the centre of the lake that is physically cut off from the shore and provides an ideal nesting spot for several birds. The lake's shoreline was covered with a variety of grass species; therefore, livestock grazing was frequently observed.

Bird survey

Survey of birds was conducted in the late afternoon from 4.00 pm to 6.00 pm during the last week of every month from December 2023 to May 2024. Point count method was employed to study and record the birds by choosing an appropriate vantage point for a fixed amount of time (10–15 minutes) (Bibby *et al.* 2002, Lambert *et al.* 2009, Girma *et al.* 2017, Kumbhar & Mhaske 2021, Awash & Tekalign 2023, Jangral & Vashishat 2023). Observation of birds was done by using Olympus binocular (OLYMPUS 8X40 DPS I, Field 8.2°) and Nikon D3200 camera was used for photography. The birds were identified using field guides (Ali & Ripley 1995, Grewal *et al.* 2016, Grimmett *et al.* 2016). The common and scientific names of birds were retrieved from (Grimmett *et al.* 2016). The conservation status of birds was adopted from IUCN Red List of Threatened Species (IUCN 2023). The recorded birds were classified into different feeding guilds depending upon their food ecological similarities, such as carnivore, insectivore, omnivore, granivore, frugivore, mixed guild, nectarivore, and herbivore (Basnet *et al.* 2016, Grimmett *et al.* 2016, Jangral & Vashishat 2022).

Data analysis

The Paleontological Statistics (PAST) 4.03 educational software (Hammer *et al.* 2001) was used to measure various α -diversity indices such as Fisher's alpha, Shannon-Weiner index, Simpson diversity and Evenness index for the summarised data for each habitat type. Sorensen's similarity index was performed to measure the similarity between the two water bodies (Magurran 2004) using the formula below:

$$C_s = 2c/a + b$$

where 'a' is the number of species of birds found in Giribetta Lake, 'b' is the number of species found in Hadinaru Lake, and 'c' is the number of species found in both sites.

The Jaccard similarity index was calculated according to the equation below (Chao *et al.* 2006):

$$C_j = j/(a + b - j)$$

where 'j' is the number of species common to both sites, 'a' is the number of species of birds found in Giribetta Lake and 'b' is the number of species found in Hadinaru Lake.

Results

We recorded a total of 5,790 individuals (Site G = 1,702, Site H = 4,088) belonging to 106 species from 18 orders and 48 families during the six-month study period from the selected lakes of Mysore district (*Appendix 1*). The most abundant species were from order Passeriformes (35%) followed by Pelecaniformes (12%), Charadriiformes (9%), Coraciiformes (7%), Anseriformes (6%), Cuculiformes (5%), Accipetriformes (4%), Columbiformes (4%), Gruiformes (4%), Suliformes (4%), Piciformes (3%), Galliformes (2%), Ciconiiformes (2%). Four species of global conservation concern were also recorded: Black-headed Ibis (*Threskiornis melanocephalus*), Spot-billed Pelican (*Pelecanus philippensis*), Oriental Darter (*Anhinga melanogaster*), and Painted Stork (*Mycteria leucocephala*), all of which are under Near-Threatened category according to the IUCN Red List of Threatened Species. In terms of species richness, Site G had 87 species belonging to 18 orders and 42 families (*Figure 2*), compared to Site H, which had 86 species from 16 orders and 40 families (*Figure 3*). The two areas had similar species composition, with 76.3% ($C_s = 0.763$) reflecting a very high similarity, which indicates that the species diversity is fairly similar across the habitats (Ratliff 1993). This is also evident with the Jaccard similarity index ($C_j = 0.617$), which enables the comparison of two communities by taking into account the number of species common to both habitats and the number of species that are present exclusively in each of them (Jaccard 1901). It was observed that the abundance of bird species was larger in Site H than Site G. Different diversity indices of the bird species were evaluated to gain insight into the stability of the ecosystem. Site G had the highest values of the Shannon diversity

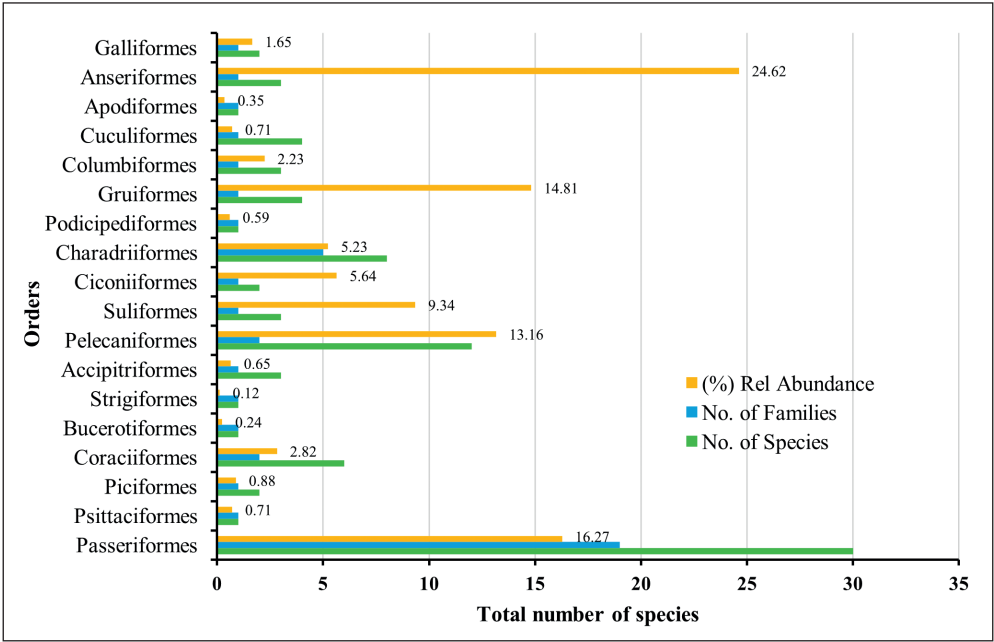


Figure 2. Bird species richness by taxonomic order in Giribetta Lake area of Mysore, India
2. ábra Madárfajgazdagság rendszertani sorrend szerint az indiai Mysore Giribetta-tónál

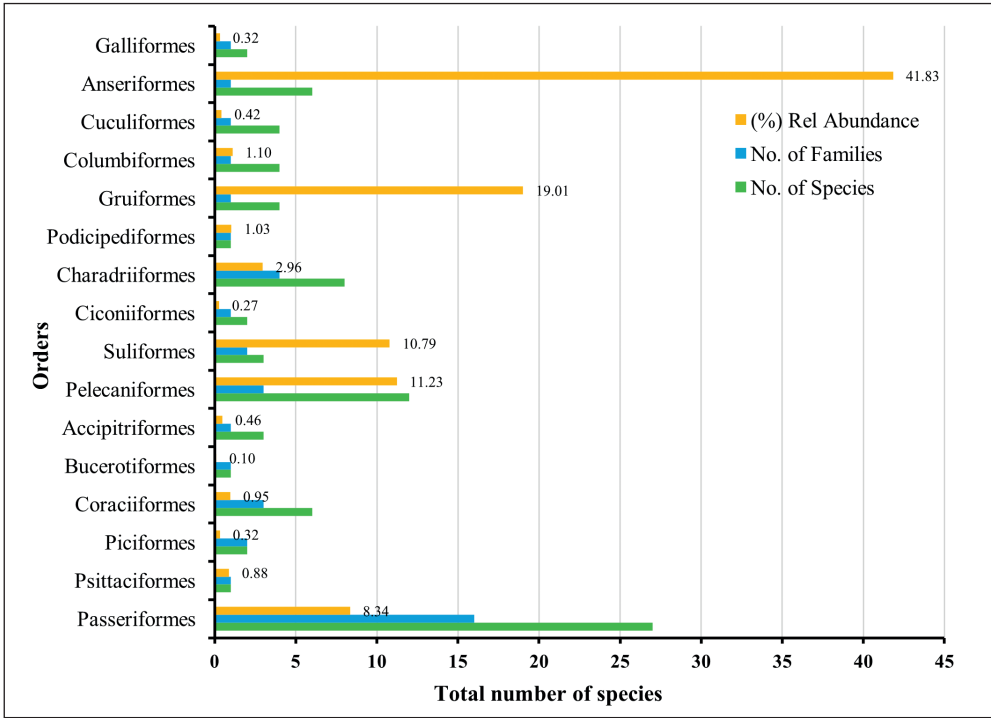


Figure 3. Bird species richness by taxonomic order in Hadinaru Lake area of Mysore, India

3. ábra Madárfajgazdagság rendszertani sorrend szerint az indiai Mysore Hadinaru-tó területén

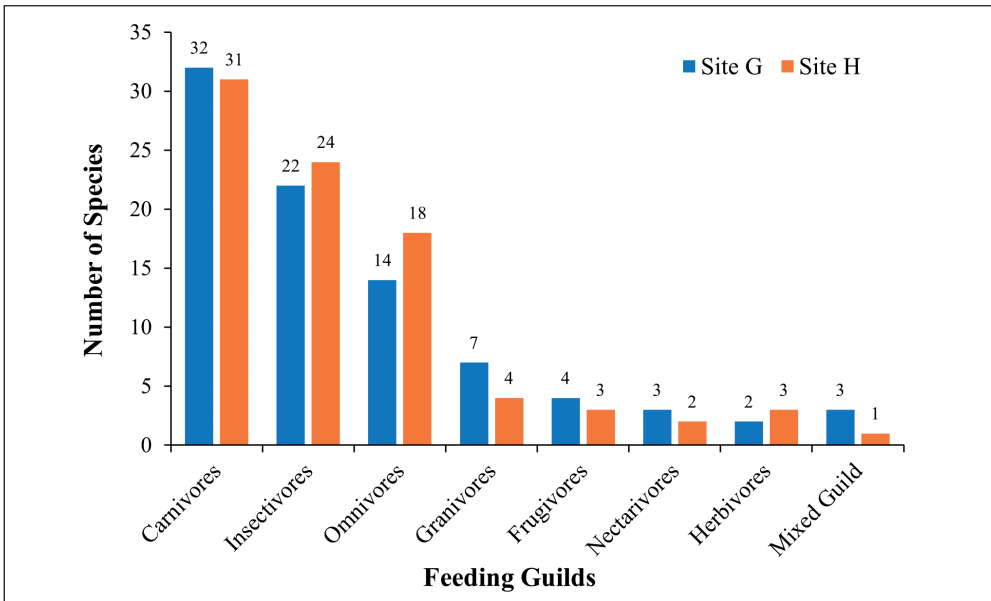


Figure 4. Feeding guilds of bird species in Giribetta Lake (Site G) and Hadinaru Lake (Site H)

4. ábra Madárfajok táplálkozási guildjei a Giribetta- és a Hadinaru-tónál

index ($H = 3.495$) and Simpson's index ($D = 0.947$) than Site H ($H = 3.106$) and ($D = 0.923$). The same trend was observed with evenness and the Fisher's α -diversity index. Site G had the highest evenness index ($J = 0.378$) and Fisher's α -diversity index (19.39) as compared to Site H ($J = 0.259$) and Fisher's α -diversity index (15.40). According to the foraging guild, carnivorous bird species were highest in both lakes, followed by insectivorous, omnivorous, granivorous, frugivorous, nectarivorous, herbivorous, and mixed guild (Figure 4).

Discussion

The study revealed that the lakes were significantly homogeneous in terms of species diversity. However, we did detect a significant difference in the abundance of birds in the two sites. This could be attributed to the differences in size of the studied lakes. Site H was comparatively larger in area than Site G. There are several reports indicating an increase in the abundance of waterbirds with the size of the wetlands (Hoyer & Canfield Jr. 1990, Celada & Bogliani 1993, Riffell *et al.* 2001). Larger wetlands have the potential to provide more microhabitats, which in turn can attract more species (Paszkowski & Tonn 2000). Further, it was observed that Site H was visited by a large number of winter migratory birds, such as Bar-headed Goose, Garganey, Northern Shoveler, Little Ringed Plover (*Charadrius dubius*), Black-winged Stilt, and Western Yellow Wagtail (*Motacilla flava*). This is possibly due to the large catchment area of the lake and the fact that the lake is also surrounded by irrigated agricultural fields, providing suitable foraging grounds for both resident and migratory birds, which ultimately adds to the high abundance of birds in Site H. According to Smith *et al.* (1998), one of the most important variables in influencing the species diversity in a given location is the availability of food (Smith *et al.* 1998). Both the abundance and distribution of many bird species are influenced by the vegetation that makes up a significant portion of their habitats. As vegetation shifts along intricate geographical and environmental gradients, specific bird species might show up, increase or decrease in population, and eventually disappear as the habitat changes (Lee & Rotenberry 2005). The dominance of Passeriformes in Site G could be explained by that the lake is surrounded by a variety of vegetation, especially farmed trees such as mango, coconut, arecanut, and *Eucalyptus* sp. Further, the boundary of these farmlands is surrounded by variety of shrubs like *Lantana*, *Tecoma*, *Agave*, *Calotropis*, *Pterolobium indicum*, which provide suitable nesting sites for these birds. In a similar study by Awash and Tekalign (2023) on bird assemblages in two Ethiopian wetlands, it was reported that more than half of the species and families were from Passeriformes, which could be attributed to the fact that this order is the biggest and most diversified group of avian organisms (Awash & Tekalign 2023).

The diversity indices provide an overview of the relative abundance of bird species and their communities by examining the relationship between the number of bird species in the study area (Latumahina *et al.* 2020). The result of the diversity analysis shows that higher bird diversity and evenness were recorded at Site G as compared to Site H. This could be attributed to differences in the richness and diversity of vegetation, such as trees, shrubs, grasses, emergent and submerged plants. In comparison to Site H, there was more

vegetation cover at Site G. According to Kiros *et al.* (2018), a shift in bird species diversity, richness, and abundance is linked to vegetation composition, which affects the availability of food, nesting, and protection depending on the preferences and feeding habits of the birds (Kiros *et al.* 2018).

The trophic composition of birds in the current study revealed that carnivorous birds dominated the feeding guild, followed by insectivorous, omnivorous, granivorous, frugivorous, nectarivorous, herbivorous, and mixed guilds from both lakes. The prevalence of carnivorous birds in wetland ecosystems is a sign of the abundance of food supplies, including fish, amphibians, molluscs, and insects (Prajapati & Prajapati 2013). Since both study areas were adjacent to agricultural lands, insectivorous birds were also abundant. Azman *et al.* (2011) have reported that agricultural lands are rich in insect diversity, making them an ideal foraging ground for insectivorous birds (Azman *et al.* 2011). Omnivorous birds like the Eurasian Moorhen (*Gallinula chloropus*), Grey-headed Swamphe (Porphyrion poliocephalus), White-breasted Waterhen (Amaurornis phoenicurus), Eurasian Coot (Fulica atra), Lesser Whistling-duck (Dendrocygna javanica), and Garganey were spotted in large numbers in both lakes, which could be related to the greater richness of macroinvertebrates and fish in shallow water zones (Meerhoff *et al.* 2003). In addition, a significant proportion of the bird population was documented around the lake margins, owing to the diverse terrestrial vegetation. The trees and shrubs produce a variety of flowers and fruits, attracting frugivorous and nectarivorous birds (Chettri *et al.* 2005). Studies on bird feeding guilds shed light on species ecology and are particularly helpful in identifying the specific ecological causes of community change (Jangral & Vashishat 2022). The recording of eight feeding guilds from both lakes indicates a well-defined trophic segregation within the community, comprising species that exploit diverse food sources using a variety of behavioural tactics and morphological adaptations (Davis & Smith 2001, Rajpar *et al.* 2018).

Throughout the study period, a variety of anthropogenic activities were observed in both the lakes, including urban encroachment, agricultural expansion, agricultural runoff, boating, washing clothes, washing livestock, and tourism. All of these activities are either directly or indirectly responsible for the process of eutrophication of lakes. Similar conditions were reported by Suryakanth (2017) in his study on avifauna diversity in wetlands of Kolhapur, Maharashtra, India. Despite being used for farming, fishing, and grazing, both wetlands provide shelter and foraging grounds for a variety of bird species. However, the high level of human encroachment has caused the wetlands to shrink, depriving a variety of resident and migratory birds of their homes. Therefore, a suitable management plan should be put into effect by law in order to preserve the lakes and the bird population in the study area.

Conclusions

The study on the diversity and composition of birds in Giribetta Lake and Hadinaru Lake shows that the sites are productive avian habitats as they harbour a variety of bird species, including resident, migratory, and species with conservation value (Painted Stork, Black-headed Ibis, Oriental Darter, and Spot-billed Pelican). The current study emphasizes

the significance of semi-urban and rural water bodies as key feeding grounds for both migratory and resident bird species. Even though the lakes support diverse bird populations, anthropogenic activities near the waterbodies are reducing habitat accessibility for birds, which may eventually have a negative impact on bird diversity and the overall health of the lakes. The present study suggests that proper and regular maintenance of the lakes is needed to increase and conserve the avifaunal diversity.

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Appendix 1. Checklist of bird species recorded during the study period, with their common name, scientific name, order, family, IUCN status, and feeding guild

1. melléklet A vizsgálati időszak során feljegyzett madárfajok listája IUCN-státuszukkal és táplálkozási besorolásukkal

Sl. No.	Order/ Family	Common Name	Scientific Name	IUCN Status	Food Habit
1. Order – Galliformes					
1	Phasianidae	Gray Francolin	<i>Francolinus pondicerianus</i> (Gmelin, 1789)	LC	I
2	Phasianidae	Indian Peafowl	<i>Pavo cristatus</i> (Linnaeus, 1758)	LC	O
2. Order – Anseriformes					
3	Anatidae	Indian Spot-billed Duck	<i>Anas poecilorhyncha</i> (Forster, 1781)	LC	H
4	Anatidae	Lesser Whistling-duck	<i>Dendrocygna javanica</i> (Horsfield, 1821)	LC	O
5	Anatidae	Cotton Pygmy-goose	<i>Nettapus coromandelianus</i> (Gmelin, 1789)	LC	H
6	Anatidae	Garganey	<i>Spatula querquedula</i> (Linnaeus, 1758)	LC	O
7	Anatidae	Northern Shoveler	<i>Spatula clypeata</i> (Linnaeus, 1758)	LC	O
8	Anatidae	Bar-headed Goose	<i>Anser indicus</i> (Latham, 1790)	LC	H
3. Order – Apodiformes					
9	Apodidae	Little Swift	<i>Apus affinis</i> (Gray, 1830)	LC	I
4. Order – Cuculiformes					
10	Cuculidae	Common Hawk Cuckoo	<i>Hierococcyx varius</i> (Vahl, 1797)	LC	I
11	Cuculidae	Blue-faced Malkoha	<i>Phaenicophaeus viridirostris</i> (Jerdon, 1840)	LC	I
12	Cuculidae	Grey-bellied Cuckoo	<i>Cacomantis passerines</i> (Vahl, 1797)	LC	I
13	Cuculidae	Asian Koel	<i>Eudynamys scolopaceus</i> (Linnaeus, 1758)	LC	I
14	Cuculidae	Greater Coucal	<i>Centropus sinensis</i> (Stephens, 1815)	LC	C
5. Order – Columbiformes					
15	Columbidae	Eurasian Collared Dove	<i>Streptopelia decaocto</i> (Frivaldszky, 1838)	LC	G
16	Columbidae	Spotted Dove	<i>Spilopelia chinensis</i> (Scopoli, 1786)	LC	G
17	Columbidae	Laughing Dove	<i>Spilopelia senegalensis</i> (Linnaeus, 1766)	LC	G
18	Columbidae	Rock Pigeon	<i>Columba livia</i> (Gmelin, 1789)	LC	O
6. Order – Gruiformes					
19	Rallidae	Eurasian Moorhen	<i>Gallinula chloropus</i> (Linnaeus, 1758)	LC	O
20	Rallidae	Grey-headed Swampphen	<i>Porphyrio poliocephalus</i> (Latham, 1801)	LC	O
21	Rallidae	White-breasted Waterhen	<i>Amaurornis phoenicurus</i> (Pennant, 1769)	LC	O
22	Rallidae	Eurasian Coot	<i>Fulica atra</i> (Linnaeus, 1758)	LC	O
7. Order – Podicipediformes					
23	Podicipedidae	Little Grebe	<i>Tachybaptus ruficollis</i> (Pallas, 1764)	LC	O

Sl. No.	Order/ Family	Common Name	Scientific Name	IUCN Status	Food Habit
8. Order – Charadriiformes					
24	Charadriidae	Red-wattled Lapwing	<i>Vanellus indicus</i> (Boddaert, 1783)	LC	MG
25	Charadriidae	Yellow-wattled Lapwing	<i>Vanellus malabaricus</i> (Boddaert, 1783)	LC	I
26	Charadriidae	Little Ringed Plover	<i>Charadrius dubius</i> (Scopoli, 1786)	LC	I
27	Recurvirostridae	Black-winged Stilt	<i>Himantopus himantopus</i> (Linnaeus, 1758)	LC	C
28	Jacanidae	Bronze-winged Jacana	<i>Metopidius indicus</i> (Latham, 1790)	LC	C
29	Jacanidae	Pheasant-tailed Jacana	<i>Hydrophasianus chirurgus</i> (Scopoli, 1786)	LC	C
30	Scolopacidae	Wood Sandpiper	<i>Tringa glareola</i> (Linnaeus, 1758)	LC	C
31	Scolopacidae	Common Sandpiper	<i>Actitis hypoleucos</i> (Linnaeus, 1758)	LC	C
32	Scolopacidae	Green Sandpiper	<i>Tringa ochropus</i> (Linnaeus, 1758)	LC	C
33	Laridae	Whiskered Tern	<i>Chlidonias hybrida</i> (Pallas, 1811)	LC	C
9. Order – Ciconiiformes					
34	Ciconiidae	Asian Openbill	<i>Anastomus oscitans</i> (Boddaert, 1783)	LC	C
35	Ciconiidae	Painted Stork	<i>Mycteria leucocephala</i> (Pennant, 1769)	NT	C
10. Order – Suliformes					
36	Phalacrocoracidae	Little Cormorant	<i>Microcarbo niger</i> (Vieillot, 1817)	LC	C
37	Phalacrocoracidae	Great Cormorant	<i>Phalacrocorax carbo</i> (Linnaeus, 1758)	LC	C
38	Phalacrocoracidae	Indian Cormorant	<i>Phalacrocorax fuscicollis</i> (Stephens, 1826)	LC	C
39	Anhingidae	Oriental Darter	<i>Anhinga melanogaster</i> (Pennant, 1769)	NT	C
11. Order – Pelecaniformes					
40	Pelecanidae	Spot-billed Pelican	<i>Pelecanus philippensis</i> (Gmelin, 1789)	NT	O
41	Ardeidae	Little Egret	<i>Egretta garzetta</i> (Linnaeus, 1766)	LC	C
42	Ardeidae	Eastern Cattle Egret	<i>Bubulcus coromandus</i> (Boddaert, 1783)	LC	C
43	Ardeidae	Black-crowned Night Heron	<i>Nycticorax nycticorax</i> (Linnaeus, 1758)	LC	C
44	Ardeidae	Indian Pond Heron	<i>Ardeola grayii</i> (Sykes, 1832)	LC	C
45	Ardeidae	Great Egret	<i>Ardea alba</i> (Linnaeus, 1758)	LC	C
46	Ardeidae	Medium Egret	<i>Ardea intermedia</i> (Wagler, 1829)	LC	C
47	Ardeidae	Grey Heron	<i>Ardea cinerea</i> (Linnaeus, 1758)	LC	C
48	Ardeidae	Purple Heron	<i>Ardea purpurea</i> (Linnaeus, 1766)	LC	C
49	Threskiornithidae	Black-headed Ibis	<i>Threskiornis melanocephalus</i> (Latham, 1790)	NT	C
50	Threskiornithidae	Red-naped Ibis	<i>Pseudibis papillosa</i> (Temminck, 1824)	LC	C
51	Threskiornithidae	Glossy Ibis	<i>Plegadis falcinellus</i> (Linnaeus, 1766)	LC	C
52	Threskiornithidae	Eurasian Spoonbill	<i>Platalea leucorodia</i> (Linnaeus, 1758)	LC	C
12. Order – Accipitriformes					
53	Accipitridae	Black Kite	<i>Milvus migrans</i> (Boddaert, 1783)	LC	C

Sl. No.	Order/ Family	Common Name	Scientific Name	IUCN Status	Food Habit
54	Accipitridae	Brahminy Kite	<i>Haliastur indus</i> (Boddaert, 1783)	LC	C
55	Accipitridae	Western Marsh Harrier	<i>Circus aeruginosus</i> (Linnaeus, 1758)	LC	C
56	Accipitridae	Shikra	<i>Accipiter badius</i> (Gmelin, 1788)	LC	C
13. Order – Strigiformes					
57	Strigidae	Spotted Owlet	<i>Athene brama</i> (Temminck, 1821)	LC	C
14. Order – Bucerotiformes					
58	Bucerotidae	Indian Grey Hornbill	<i>Ocyrceros birostris</i> (Scopoli, 1786)	LC	F
15. Order – Coraciiformes					
59	Alcedinidae	Common Kingfisher	<i>Alcedo atthis</i> (Linnaeus, 1758)	LC	C
60	Alcedinidae	White-throated Kingfisher	<i>Halcyon smyrnensis</i> (Linnaeus, 1758)	LC	C
61	Alcedinidae	Pied Kingfisher	<i>Ceryle rudis</i> (Linnaeus, 1758)	LC	C
62	Alcedinidae	Stork-billed Kingfisher	<i>Pelargopsis capensis</i> (Linnaeus, 1766)	LC	MG
63	Coraciidae	Indian Roller	<i>Coracias benghalensis</i> (Linnaeus, 1758)	LC	C
64	Meropidae	Green Bee-eater	<i>Merops orientalis</i> (Latham, 1801)	LC	I
65	Meropidae	Blue-tailed Bee-eater	<i>Merops philippinus</i> (Linnaeus, 1767)	LC	I
16. Order – Piciformes					
66	Megalaimidae	Coppersmith Barbet	<i>Psilopogon haemacephalus</i> (Muller, 1776)	LC	F
67	Megalaimidae	White-cheeked Barbet	<i>Psilopogon viridis</i> (Boddaert, 1783)	LC	F
68	Picidae	Lesser Goldenback	<i>Dinopium benghalense</i> (Linnaeus, 1758)	LC	I
17. Order – Psittaciformes					
69	Psittaculidae	Rose-ringed Parakeet	<i>Psittacula kramera</i> (Scopoli, 1769)	LC	F
18. Order – Passeriformes					
70	Aegithinidae	Common Iora	<i>Aegithina tiphia</i> (Linnaeus, 1758)	LC	I
71	Alaudidae	Jerdon's Bush Lark	<i>Mirafra affinis</i> (Blyth, 1845)	LC	I
72	Acrocephalidae	Eurasian Reed Warbler	<i>Acrocephalus scirpaceus</i> (Hermann, 1804)	LC	I
73	Campephagidae	Small Minivet	<i>Pericrocotus cinnamomeus</i> (Linnaeus, 1776)	LC	I
74	Dicruridae	Black Drongo	<i>Dicrurus macrocercus</i> (Vieillot, 1817)	LC	I
75	Dicaeidae	Pale-billed Flowerpecker	<i>Dicaeum erythrorhynchos</i> (Latham, 1790)	LC	N
76	Cisticolidae	Ashy Prinia	<i>Prinia socialis</i> (Sykes, 1832)	LC	I
77	Cisticolidae	Common Tailorbird	<i>Orthotomus sutorius</i> (Pennant, 1769)	LC	I
78	Pycnonotidae	White-browed Bulbul	<i>Pycnonotus luteolus</i> (Lesson, 1841)	LC	O
79	Pycnonotidae	Red-vented Bulbul	<i>Pycnonotus cafer</i> (Linnaeus, 1766)	LC	O
80	Pycnonotidae	Red-whiskered Bulbul	<i>Pycnonotus jocosus</i> (Linnaeus, 1758)	LC	O
81	Sturnidae	Common Myna	<i>Acridotheres tristis</i> (Linnaeus, 1766)	LC	O
82	Sturnidae	Jungle Myna	<i>Acridotheres fuscus</i> (Wagler, 1827)	LC	O

Sl. No.	Order/ Family	Common Name	Scientific Name	IUCN Status	Food Habit
83	Sturnidae	Brahminy Starling	<i>Sturnia pagodarum</i> (Gmelin, 1789)	LC	O
84	Muscicapidae	Pied Bushchat	<i>Saxicola caprata</i> (Linnaeus, 1766)	LC	I
85	Muscicapidae	Indian Robin	<i>Copsychus fulicatus</i> (Linnaeus, 1766)	LC	I
86	Muscicapidae	Oriental Magpie Robin	<i>Copsychus saularis</i> (Linnaeus, 1758)	LC	I
87	Nectariniidae	Purple-rumped Sunbird	<i>Leptocoma zeylonica</i> (Linnaeus, 1766)	LC	N
88	Nectariniidae	Purple Sunbird	<i>Cinnyris asiaticus</i> (Latham, 1790)	LC	N
89	Motacillidae	White-browed Wagtail	<i>Motacilla maderaspatensis</i> (Gmelin, 1789)	LC	I
90	Motacillidae	Grey Wagtail	<i>Motacilla cinerea</i> (Tunstall, 1771)	LC	I
91	Motacillidae	Western Yellow Wagtail	<i>Motacilla flava</i> (Linnaeus, 1758)	LC	I
92	Motacillidae	Paddyfield Pipit	<i>Anthus rufulus</i> (Vieillot, 1818)	LC	I
93	Oriolidae	Indian Golden Oriole	<i>Oriolus kundoo</i> (Sykes, 1832)	LC	I
94	Laniidae	Brown Shrike	<i>Lanius cristatus</i> (Linnaeus, 1758)	LC	C
95	Laniidae	Long-tailed Shrike	<i>Lanius schach</i> (Linnaeus, 1758)	LC	C
96	Leiothrichidae	Yellow-billed Babbler	<i>Argya affinis</i> (Jerdon, 1845)	LC	O
97	Corvidae	House Crow	<i>Corvus splendens</i> (Vieillot, 1817)	LC	O
98	Corvidae	Large-billed Crow	<i>Corvus macrorhynchos</i> (Wagler, 1827)	LC	O
99	Paridae	Cinereous Tit	<i>Parus cinereus</i> (Vieillot, 1818)	LC	I
100	Ploceidae	Baya Weaver	<i>Ploceus philippinus</i> (Linnaeus, 1766)	LC	G
101	Hirundinidae	Barn Swallow	<i>Hirundo rustica</i> (Linnaeus, 1766)	LC	I
102	Hirundinidae	Red-rumped Swallow	<i>Cecropis daurica</i> (Laxmann, 1769)	LC	I
103	Estrildidae	Tricoloured Munia	<i>Lonchura malacca</i> (Linnaeus, 1766)	LC	G
104	Estrildidae	Scaly-breasted Munia	<i>Lonchura punctulate</i> (Linnaeus, 1758)	LC	G
105	Estrildidae	Red Avadavat	<i>Amandava amandava</i> (Linnaeus, 1758)	LC	G
106	Rhipiduridae	Spot-breasted Fantail	<i>Rhipidura albogularis</i> (Lesson, 1831)	LC	I

Abbreviations: IUCN – International Union for Conservation of Nature Status, LC – Least Concern, NT – Near Threatened, VU – Vulnerable. **Key:** C (Carnivore), F (Frugivore), G (Granivore), I (Insectivore), MG (Mixed Guild), N (Nectarivore), O (Omnivore), H (Herbivore)



A preliminary study on avifaunal diversity of Tehsil Karsog in Mandi district, Himachal Pradesh, India

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Abstract The avifaunal diversity assessment is a valuable ecological tool, providing a significant measure for assessing the quality and quantity of various habitats. Karsog is located in Mandi district of Himachal Pradesh, India, in the Pir Panjal Himalayan range. Avifaunal study was conducted on a monthly basis in the Tehsil Karsog from December 2023 to June 2024. The study revealed the presence of 91 bird species, spanning over 63 genera, 38 families, 12 orders. The Order Passeriformes was the most diverse with 64 species, while Bucerotiformes and Strigiformes were represented by one species each. The Muscicapidae, with 12 species, was the dominant family in the study area. Analysis of data on relative abundance showed that of the 91 species recorded, 23 species (25.27%) were very common, 27 species (29.67%) were common, 26 species (28.57%) were uncommon, and 15 species (16.48%) were rare. Among the documented avifauna, 2 species, the Alexandrine Parakeet and the Himalayan Griffon are Near Threatened species as categorized by the IUCN Red List (2024). Assessment of data on feeding guild shows that, out of 91 species, 2 species (2.19%) are granivores, 3 species (3.29%) are carnivores, 11 species (12.08%) are omnivores, 17 species (18.68%) are insectivores and 58 species (63.73%) have varied type of feeding habits. It shows that 22 species (24.17%) have a narrow feeding guild meaning that they rely on limited type of food resources. Further analysis of data on the population status reveals that 17 species (18.68%) have declining population trend as per the IUCN global population trend assessment. This emphasizes the importance of implementing conservation strategies in the area under investigation.

Keywords: biodiversity, avifauna, Himalayas, threat

Összefoglalás A madarak sokféleségének mérése fontos ökológiai eszköz, amely jelentős adalékokat adhat a különböző élőhelyek minőségi és mennyiségi felmérésének hatékonyságához. Karsog az indiai Himachal Pradesh Mandi kerületében található, a Pir Panjal Himalája tartományban. A Tehsil Karsogban 2023 decembere és 2024 júniusa között havi rendszerességgel végeztek felméréseket. A vizsgálat 91 madárfaj jelenlétét tárta fel, több mint 63 nemzetségre, 38 családra és 12 rendre vonatkoztatva. Az énekesmadarak rendje volt a legfajgazdagabb 64 fajjal, míg a Bucerotiformes és Strigiformes egy-egy fajjal volt képviselve. A Muscicapidae 12 fajjal volt a legdominánsabb család a vizsgált területen. A relatív abundanciára vonatkozó adatok elemzése azt mutatta, hogy a feljegyzett 91 fajból 23 faj (25,27%) volt nagyon gyakori, 27 faj (29,67%) gyakori, 26 faj (28,57%) nem gyakori, és 15 faj (16,48%) ritka. A dokumentált madárfajok közül 2, a Nagy Sándor-papagáj és a havasi fakókese-lyű az IUCN Vörös Listáján (2024) a mérsékelten fenyegetett fajok közé tartoznak. A táplálkozási guildek adatainak értékelése azt mutatja, hogy a vizsgált területen található 91 fajból 2 faj (2,19%) a magevő, 3 faj (3,29%) a ragadozó, 11 faj (12,08%) a mindenevő, 17 faj (18,68%) a rovarevők és 58 faj (63,73%) vegyes táplálkozási stratégiákkal rendelkezik. Ezek alapján 22 faj (24,17%) szűk táplálkozási spektrummal rendelkezik, így erősen korlátozott táplálékforrásokra támaszkodnak. A populáció státuszára vonatkozó adatok további elemzése azt mutatja, hogy az IUCN globális populációtrend értékelése szerint 17 faj (18,68%) populációja csökkenő tendenciát mutat, ami alátámasztja a természetvédelmi kezelések fontosságát a vizsgált területen.

Kulcsszavak: biodiverzitás, madárfauna, Himalája, veszélyeztetettség

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Introduction

The Himalayas is a young and massive mountain range spanning approximately 2,400 km from North-West to South-East, across 6 countries viz. Afghanistan, Pakistan, India, Nepal, Bhutan, and China. They are 230–320 km wide from North-East to South-West (Gansser 1974). Approximately 27% of the land surface of our planet and 18% of the land surface of India is covered by them (Sharma & Sidhu 2016). The Himalayas have a wide range of climatic conditions based on temperature, altitude and latitude, contributing to its rich biodiversity (Jodha 2000, Pun & Mares 2000, Rautela & Lakhera 2000, Mishra *et al.* 2006). In the Himalayan ranges, the Indian state of Himachal Pradesh is situated in the northwest of India at a latitude of 30°22' – 33°13' north, and a longitude of 75°36' – 79°02' east. In the east, the state serves as the international boundary between India and Tibet. It is surrounded by the states of Jammu and Kashmir to the north, Uttarakhand to the southeast, Haryana to the south, and Punjab to the west. (Narwade *et al.* 2006). The region contains around 10.54% of the Himalayan land mass and accounts for 17% of the north-western Himalayan landscape, with an altitude range from 450 m to 7,000 m above sea level (Paliwal *et al.* 2019). The region is characterized by its intricate topography and habitats, and is home to diverse collection of temperate flora and fauna (Islam & Rahmani 2004). According to ISFR (2021), the state has a total geographical area of 55,673 km². Forests can be broadly classified into alpine, subalpine, temperate, or subtropical vegetation types (Rodgers & Panwar 1988, Sathyakumar & Bashir 2010). The state of Himachal Pradesh from South to North has been geographically divided into three regions: 1) The Outer Himalayas or Shivaliks, 2) the Inner Himalayas or Mid Mountains, and 3) the Greater Himalayas or Alpine zone. The lower Himalayas and Shivalik hills are largely covered by mixed and deciduous forests. The Inner Himalayas or Mid Mountains contain wet and Himalayan temperate vegetation like Deodar, Spruce and Kail. The Greater Himalayas comprise subalpine, and dry alpine vegetation like Kharsu, Blue Pine, etc. (Kumar 2015).

Investigating avifaunal diversity is a crucial ecological instrument that is a significant indicator for both the qualitative and quantitative assessment of various habitats (Bilgrami 1995). Birds are warm-blooded and egg-laying vertebrates, each with a unique appearance and habits. They perform several roles, including as pest management, scavenging, seed dispersion, and serve as pollinators of crop plants (Kumar *et al.* 2023). There are more than 11,000 species of birds reported globally (Birdlife International 2024). India is rich in avian diversity, as evidenced by the 1,211 bird species reported by BirdLife International (2024). Out of these species, 970 (about 80%) are found only in the Himalayas (Naithani & Bhatt 2010, Praveen *et al.* 2016). The eastern Himalayas have twice as many bird species as the western Himalayas (Price *et al.* 2003). The majority of Himachal Pradesh's avifauna has been reported from the lower Himalayas; however, a sizeable number has also been reported from the Greater and Trans Himalayan regions of the state (Singh &

Banyal 2013, Negi *et al.* 2015). Under British colonial administration, many extensive surveys were conducted to study the birds of Himachal Pradesh like Theobald (1862), Tytler (1868), Stoliczka (1868), Hume and Marshall (1879), Dodsworth (1910 to 1914) and Koelz (1937) to mention a few. Sir Norman Frederick Frome (1946) and Ganguli (1967) wrote on birds in Shimla and neighboring places whereas Gaston and Singh (1980) investigated pheasant distribution in Himachal Pradesh. Ali and Ripley (1983) described the avifauna of Himachal Pradesh, highlighting regions like Shimla, Dalhousie, and Dharamshala. Mahabal and Sharma (1992), Thakur *et al.* (2002), Besten (2004), Sharma *et al.* (2009) and Singh *et al.* (2014) examined avifauna in various areas of Himachal Pradesh. Kumar (2015) documented 618 bird species from the state, spanning 20 orders and 91 families. Even though avifaunal research has a long history in the state dating back to the British colonial era, most of these studies were restricted to specific regions within the state because of its challenging topography and adverse weather conditions. The vast majority of the state still remains unexplored by field biologists.

The avifauna is perpetually threatened by various factors including habitat loss, forest degradation, hunting, invasive species, diseases and is highly sensitive to pollution levels in both terrestrial and aquatic ecosystems (Gaston 1975, Sodhi *et al.* 2011, Mohapatra *et al.* 2019). Currently, there are approximately 1,159 species of birds in the threatened category as per IUCN (2024).

The area of Tehsil Karsog of district Mandi, Himachal Pradesh was selected for the avifaunal investigations, as this pristine area is rich in floral and faunal diversity attributable to its climatic conditions. Further, the avifauna in the region is jeopardized due to the anthropogenic activities; hence it is necessary to record the avifauna in order to implement appropriate conservation measures to the area.

Material and Methods

Study area

The Karsog Tehsil of Mandi District is an alluring valley nestled in the lap of Himalayas in Himachal Pradesh, lying between 31.3835 °N and 77.2028 °E, with an area of 524 km² (Figure 1). It is a picturesque and serene destination, located in the Pir Panjal Himalayan range and shares its boundaries with the districts of Kullu and Shimla (Kumar *et al.* 2021). It encompasses a wide spectrum of altitudes, ranging from 800 to 3,100 meters above sea level with moderate to difficult topography, having 15–65% slope (Bharti & Sharma 2022). It is situated within the agro-climatic ‘North Western Himalayan Hill Zone’ of India, which is classified as a humid sub-tropical climate. The valley experiences pleasant weather year-round with average annual temperature range of 12.4 °C to 26.4 °C and average annual precipitation of 1,010 mm, both of which contribute to its mild and temperate climate (CHP 2010). The valley is teeming with lush green landscapes and verdant vegetation, adorned with an abundance of mixed-variety trees such as Himalayan blue pine (*Pinus wallichiana*), chir pine (*Pinus wallichiana*), morinda spruce (*Picea*

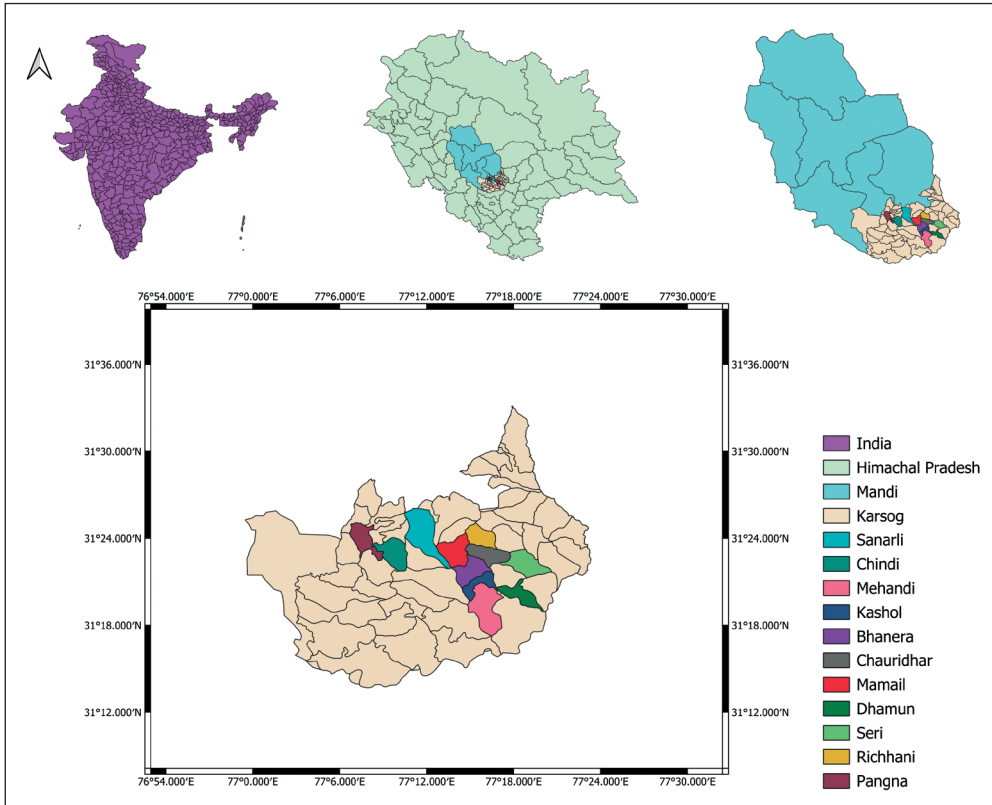


Figure 1. Map of Karsog, the study area of District Mandi, Himachal Pradesh, India
 1. ábra Karsog vizsgálati terület térképe (Mandi, Himachal Pradesh, India)

smithiana), West Himalayan fir (*Abies pindrow*), banjh (*Quercus leucotrichophora*) and apple (*Malus domestica*). The months of winter are chilly due to frost and fog (Singh *et al.* 2022). The Karsog area is divided into four forest ranges: Karsog, Pangna, Seri, and Magroo (Bharti *et al.* 2021). Some portions of the adjacent Shikari Devi Wildlife sanctuary lie in the Karsog range. The important wild fauna of the area includes Indian Leopard (*Panthera pardus*), Asiatic Black Bear (*Ursus thibetanus*), Himalayan Goral (*Naemorhedus goral*), Barking Deer (*Muntiacus muntjak*), Langur (*Semnopithecus entellus*), Rhesus Macaque (*Macaca mulatta*), Himalayan Weasel (*Mustela sibirica*), Himalayan Griffon (*Gyps himalayensis*), Alexandrine Parakeet (*Psittacula eupatria*) and Indian Peafowl (*Pavo cristatus*).

Methodology

The study area was surveyed for ornithological studies in several locations on a monthly basis from December 2023 to June 2024 spanning three seasons: winter (December 2023 to March 2024), spring (April 2024), and summer (May to June 2024). The area was primarily explored by walking. The Tehsil Karsog is divided into twelve administrative

blocks with four to six units in each block. During this study, the field surveys were conducted in 6 blocks namely Karsog, Mamail, Pangna, Chindi, Seri, and Richhani. A total of 11 units from among these blocks were selected for the survey. Four of the eleven units studied were in the Mamail block, they are Mamail, Bhanera, Mehendi, and Kashol. In the Richhani block, two units studied were Chauridhar and Dhamoon, while in the Seri block, Richhani and Seri units were explored. Sanarli, Pangna, and Chindi were the sole units evaluated from the blocks Karsog, Pangna, and Chindi respectively. Different sites in the selected units were visited during each survey. Due to the large size of the study area, a typical sample of each habitat was chosen using the stratified random sampling approach (Snedecore & Cochran 1993) to include all the diverse habitats. Throughout the survey, the line transect approach was used for estimating the abundance of the birds which is based on the principle of walking along a specified route to record the objects on or near the line (Burnham *et al.* 1980). To account for the uneven and harsh topography of the study area, piecewise linear line transects were marked and monitored monthly at various experimental sites. Moreover, these transects were observed as open width transects, meaning that birds were noted along the transect regardless of how far away they were from it (Mehta *et al.* 2002). Since, birds are most active in the few hours just after sunrise or before sunset, bird observations were made in the early morning and evening (Negi *et al.* 2015). Nikon (10x40) binoculars were used to observe the birds. To verify the identifications, pictures were captured using a Nikon DSLR D90 equipped with a 55–300 mm telephoto lens. Different field guides were employed to identify birds in the field (Ali & Ripley 1983, Grimmett *et al.* 1999) and reverified by the bird database application named Merlin Bird ID by Cornell Lab version 3.0.3 (800-2023.0) (Cornell University 2024). Taxonomical classification of birds was done with the help of Avibase – The World Bird Database (Lepage 2024). Relative abundance was computed by considering the frequency of occurrence and data collected from different habitats in the Karsog region (McKinnon & Philips 1993). Based on the data sheets, four categories were determined: VC: Very Common (Abundant number of sightings are 80–100% in field), C: Common (Regular sightings but less than VC, Sightings is 50–80% in the field), UC: Uncommon (Occasional sightings, Sightings are 6–50% in field), R: Rare (Hardly visible, Sightings are 1–5% of field visit).

The feeding guild of the birds was also assessed with the help of available literature (Ali & Ripley 1983). Based on their feeding guild, birds are categorized into five distinct groups namely Insectivores, Carnivores, Granivores, Varied type and Omnivores. Insectivores solely consume insects, Carnivores feeds exclusively on flesh, Granivores consume only seeds. The birds included in the varied feeding guild exhibit a varied but limited dietary preferences, like some may be insectivores and nectarivores, some may be granivores and frugivores and other may be carnivores and insectivores. In contrast, omnivores can consume all types of food without showing any specific preferences. They are opportunistic feeders and can consume any type of food. The IUCN Red Data list was used to examine the conservation status as well as population trends of the reported bird species (IUCN 2024).

Results

This study on the avifaunal diversity of the Tehsil Karsog in the Mandi district of Himachal Pradesh has shown the presence of 91 species of birds in the area, distributed over 63 genera, 38 families, and 12 orders. Passeriformes was the dominant order in the study area with 64 species under the 25 families. Charadriiformes and Piciformes have two families each. The remaining orders namely Galliformes, Cuculiformes, Columbiformes, Gruiformes, Accipitriformes, Strigiformes, Bucerotiformes, Coraciiformes and Psittaciformes are represented by single families each (Table 1). The Muscicapidae family, which consists of 12 species, was the dominant family of birds found in the study area, which is also the dominant family in Himachal Pradesh and India. The second largest family in the study area was represented by Motacillidae, with six species, closely followed by Corvidae, with five species. The families Columbidae and Leiotrichidae contain four species each, while the families Phasianidae, Cuculidae, Accipitridae, Picidae, Paridae, Pycnonotidae, Phylloscopidae, Cisticolidae and Fringillidae represent three species each. The families Alcedinidae, Ramphastidae, Psittaculidae, Laniidae, Nectariniidae, Passeridae, Sturnidae, Prunellidae and Emberizidae represented two species each. However, a large number of families (15 in total), viz., Strigidae, Rallidae, Charadriidae, Scolopacidae, Upupidae, Campephagidae, Dicruridae, Rhipiduridae, Oriolidae, Monarchidae, Hirundinidae, Aegithalidae, Zosteropidae, Timaliidae and Sittidae are present in the area with single species each (Table 1).

Analysis of data on relative abundance showed that of the 91 species recorded, 23 species (25.27%) were very common, 27 species (29.67%) were common, 26 species (28.57%) were uncommon, and 15 (16.48%) species were rare to the study area. The species in the rare group includes the Common Hawk Cuckoo (*Hierococcyx varius varius*), Green Sandpiper (*Tringa ochropus*), Blue-throated Barbet (*Psilopogon asiaticus asiaticus*), Rufous-bellied Woodpecker (*Dendrocopos hyperythrus*), Indian Golden Oriole (*Oriolus kundoo*), Bay-backed Shrike (*Lanius vittatus*), Wallcreeper (*Tichodroma muraria muraria*), Brahminy Starling (*Sturnia pagodarum*), Blue-fronted Redstart (*Phoenicurus frontalis*), Altai Accentor/Himalayan Accentor (*Prunella himalayana*), Rufous-breasted Accentor (*Prunella strophiatea jerdoni*), Citrine Wagtail (*Motacilla citreola*), Rosy Pipit (*Anthus roseatus*), Black-and-yellow Grosbeak (*Mycerobas icteroides*) and Chestnut-eared Bunting (*Emberiza fucata arcuata*) as they were observed only once during the survey (Table 1, Figure 2).

Assessment of data on feeding guild shows that, out of 91 species in the study area, 2 species (2.19%) are Granivores, 3 species (3.29%) are Carnivores, 11 species (12.08%) are Omnivores, 17 species (18.68%) are Insectivores and 58 species (63.73%) are classified into Varied type of feeding guild (Table 1, Figure 3). The further analysis of data shows that 22 species (24.17%) have a narrow feeding guild meaning that they rely on limited type of food resources.

The analysis of data on population trend reveals that, out of 91 species, 9 species (10%) have increasing population trend, 57 species (62.63%) have stable, 8 species (8.79%) have unknown and 17 species (18.68%) have declining population trend as provided by the IUCN population trend assessment done globally (Table 1, Figure 4). The species with decreasing population trend globally reported in area are Common Cuckoo (*Cuculus canorus*), Rock Pigeon (Rock Dove) (*Columba livia*), Himalayan Griffon (*Gyps himalayensis*) Eurasian

Table 1. List of bird species recorded in Tehsil Karsog, District Mandi, Himachal Pradesh, India
1. táblázat A Tehsil Karsog, District Mandi, Himachal Pradesh, India térségében felmért madárfajok

S. no.	Common Name	Scientific Name	Abundance Status	Feeding Guild	IUCN Status	Population Trend
Order Galliformes						
Family Phasianidae						
1	Indian Peafowl	<i>Pavo cristatus</i>	C	O	LC	STABLE
2	Black Francolin	<i>Francolinus francolinus asiae</i>	C	O	LC	STABLE
3	Hill Partridge	<i>Arborophila torqueola</i>	UC	G, I	LC	STABLE
Order Cuculiformes						
Family Cuculidae						
4	Asian Koel	<i>Eudynamys scolopaceus</i>	C	O	LC	STABLE
5	Common Hawk Cuckoo	<i>Hierococyx varius varius</i>	R	I, C	LC	STABLE
6	Common Cuckoo	<i>Cuculus canorus</i>	VC	I	LC	DECREASING
Order Columbiformes						
Family Columbidae						
7	Rock Pigeon (Rock Dove)	<i>Columba livia</i>	VC	G, H	LC	DECREASING
8	Oriental Turtle Dove	<i>Streptopelia orientalis</i>	C	G	LC	STABLE
9	Eurasian Collared Dove	<i>Streptopelia decaocto</i>	UC	G	LC	INCREASING
10	Spotted Dove	<i>Spilopelia chinensis</i>	C	G, H	LC	INCREASING
Order Gruiformes						
Family Rallidae						
11	Brown Crake	<i>Zapornia akool akool</i>	UC	I, C, G	LC	UNKNOWN
Order Charadriiformes						
Family Charadriidae						
12	Red-wattled Lapwing	<i>Vanellus indicus indicus</i>	UC	I, C	LC	UNKNOWN
Family Scolopacidae						
13	Green Sandpiper	<i>Tringa ochropus</i>	R	I, C	LC	INCREASING
Order Accipitriformes						
Family Accipitridae						
14	Black-Winged Kite	<i>Elanus caeruleus vociferus</i>	UC	I, C	LC	STABLE
15	Himalayan Griffon	<i>Gyps himalayensis</i>	C	C	NT	DECREASING
16	Black Kite	<i>Milvus migrans</i>	C	C	LC	STABLE
Order Strigiformes						
Family Strigidae						
17	Asian Barred Owlet	<i>Glaucidium cuculoides</i>	VC	C, I	LC	INCREASING
Order Bucerotiformes						
Family Upupidae						
18	Eurasian Hoopoe	<i>Upupa epops</i>	UC	I	LC	DECREASING

S. no.	Common Name	Scientific Name	Abundance Status	Feeding Guild	IUCN Status	Population Trend
Order Coraciiformes						
Family Alcedinidae						
19	Common Kingfisher	<i>Alcedo atthis atthis</i>	UC	C	LC	UNKNOWN
20	Crested Kingfisher	<i>Megaceryle lugubris guttulate</i>	UC	Aq, C	LC	DECREASING
Order Piciformes						
Family Ramphastidae						
21	Great Barbet	<i>Psilopogon virens</i>	VC	F, I	LC	DECREASING
22	Blue-throated Barbet	<i>Psilopogon asiaticus asiaticus</i>	R	F, I	LC	STABLE
Family Picidae						
23	Rufous-bellied Woodpecker	<i>Dendrocopos hyperythrus</i>	R	I, F, G	LC	STABLE
24	Fulvous-breasted Woodpecker	<i>Dendrocopos macei</i>	UC	I, F, G	LC	DECREASING
25	Scaly-bellied Woodpecker	<i>Picus squamatus</i>	C	I, F, G	LC	STABLE
Order Psittaciformes						
Family Psittaculidae						
26	Plum-headed Parakeet	<i>Psittacula cyanocephala cyanocephala</i>	VC	F, G, H	LC	DECREASING
27	Alexandrine Parakeet	<i>Psittacula eupatria</i>	C	F, G, H	NT	DECREASING
Order Passeriformes						
Family Campephagidae						
28	Long-tailed Minivet	<i>Pericrocotus ethologus</i>	C	I	LC	DECREASING
Family Oriolidae						
29	Indian Golden Oriole	<i>Oriolus kundoo</i>	R	O	LC	UNKNOWN
Family Dicruridae						
30	Ashy Drongo	<i>Dicrurus leucophaeus</i>	C	I	LC	UNKNOWN
Family Rhipiduridae						
31	White-throated Fantail	<i>Rhipidura albicollis albicollis</i>	C	I	LC	STABLE
Family Monarchidae						
32	Indian Paradise-flycatcher	<i>Terpsiphone paradisi leucogaster</i>	UC	I, C	LC	STABLE
Family Laniidae						
33	Bay-backed Shrike	<i>Lanius vittatus</i>	R	I, C, F	LC	STABLE
34	Long-tailed Shrike	<i>Lanius schach</i>	UC	I, C, F	LC	UNKNOWN
Family Corvidae						
35	Yellow-billed Blue Magpie	<i>Urocissa flavirostris</i>	C	C, I	LC	STABLE
36	Red-billed Blue Magpie	<i>Urocissa erythroryncha</i>	VC	C, I	LC	STABLE
37	Rufous Treepie	<i>Dendrocitta vagabunda</i>	UC	O	LC	DECREASING

S. no.	Common Name	Scientific Name	Abundance Status	Feeding Guild	IUCN Status	Population Trend
38	Grey Treepie	<i>Dendrocitta formosae</i>	VC	O	LC	DECREASING
39	Large-billed Crow	<i>Corvus macrorhynchos</i>	VC	O	LC	STABLE
Family Paridae						
40	Coal Tit	<i>Parus ater</i>	C	I	LC	DECREASING
41	Cinereous Tit (Great Tit)	<i>Parus cinereus</i>	VC	I	LC	INCREASING
42	Green-backed Tit	<i>Parus monticolus</i>	C	I, F, H	LC	STABLE
Family Pycnonotidae						
43	Himalayan Black Bulbul	<i>Hypsipetes leucocephalus</i>	C	G, F, I	LC	STABLE
44	Red-vented Bulbul	<i>Pycnonotus cafer</i>	VC	O	LC	STABLE
45	Himalayan Bulbul	<i>Pycnonotus leucogenys</i>	VC	G, F, I	LC	INCREASING
Family Hirundinidae						
46	Red-rumped Swallow	<i>Cecropis daurica erythropygia</i>	C	I	LC	STABLE
Family Aegithalidae						
47	Black-throated Tit	<i>Aegithalos concinnus</i>	VC	G, F, I	LC	STABLE
Family Phylloscopidae						
48	Hume's Leaf Warbler	<i>Phylloscopus humei humei</i>	UC	I	LC	STABLE
49	Lemon-rumped Warbler or Pale-rumped Warbler	<i>Phylloscopus chloronotus</i>	UC	I	LC	STABLE
50	Grey-hooded Warbler	<i>Phylloscopus xanthoschistos</i>	VC	I	LC	STABLE
Family Cisticolidae						
51	Himalayan Prinia	<i>Prinia crinigera</i>	C	I, C, G	LC	STABLE
52	Grey-breasted Prinia	<i>Prinia hodgsonii hodgsonii</i>	C	I	LC	STABLE
53	Common Tailorbird	<i>Orthotomus sutorius</i>	C	I, F, G	LC	STABLE
Family Zosteropidae						
54	Indian White-eye/ Oriental White-eye	<i>Zosterops palpebrosus occidentis</i>	C	I, H	LC	DECREASING
Family Timaliidae						
55	Rusty-cheeked Scimitar-babbler	<i>Erythrogenys erythrogenys erythrogenys</i>	C	I, G, F	LC	STABLE
Family Leiothrichidae						
56	Streaked Laughingthrush	<i>Trochalopteron lineatum</i>	VC	I, F, G	LC	STABLE
57	Variegated Laughingthrush	<i>Trochalopteron variegatum</i>	UC	I, F, G	LC	DECREASING
58	Rufous Sibia (capistrata)	<i>Heterophasia capistrata capistrata</i>	C	F, I	LC	UNKNOWN
59	Jungle Babbler	<i>Argya striata</i>	VC	O	LC	STABLE
Family Sittidae						
60	Wallcreeper	<i>Tichodroma muraria muraria</i>	R	I, C	LC	STABLE

S. no.	Common Name	Scientific Name	Abundance Status	Feeding Guild	IUCN Status	Population Trend
Family Sturnidae						
61	Common Myna	<i>Acridotheres tristis</i>	VC	G, F, I	LC	INCREASING
62	Brahminy Starling	<i>Sturnia pagodarum</i>	R	F, N	LC	UNKNOWN
Family Muscicapidae						
63	Oriental Magpie-Robin	<i>Copsychus saularis</i>	VC	I, C, N	LC	STABLE
64	Indian Robin	<i>Copsychus fulicatus fulicatus</i>	UC	I	LC	STABLE
65	Verditer Flycatcher	<i>Eumyias thalassinus</i>	C	I	LC	STABLE
66	Spotted Forktail	<i>Enicurus maculatus</i>	UC	I, C	LC	STABLE
67	Blue Whistling-Thrush	<i>Myophonus caeruleus</i>	VC	O	LC	STABLE
68	Blue-fronted Redstart	<i>Phoenicurus frontalis</i>	R	I, F, G	LC	STABLE
69	Blue-capped Redstart	<i>Phoenicurus coeruleocephala</i>	UC	I	LC	STABLE
70	Plumbeous Water Redstart	<i>Phoenicurus fuliginosus</i>	VC	AqI	LC	STABLE
71	White-capped Redstart	<i>Phoenicurus leucocephalus</i>	UC	O	LC	STABLE
72	Grey Bushchat	<i>Saxicola ferreus</i>	VC	I, G	LC	STABLE
73	Pied Bushchat	<i>Saxicola caprata bicolor</i>	C	I, H, C	LC	STABLE
74	Siberian Stonechat	<i>Saxicola maurus</i>	UC	I	LC	STABLE
Family Nectariniidae						
75	Purple Sunbird	<i>Cinnyris asiaticus asiaticus</i>	UC	N, I	LC	STABLE
76	Crimson Sunbird	<i>Aethopyga siparaja</i>	UC	N, I	LC	STABLE
Family Passeridae						
77	Russet Sparrow	<i>Passer cinnamomeus</i>	VC	I, G, F	LC	STABLE
78	House Sparrow	<i>Passer domesticus</i>	VC	G, I	LC	DECREASING
Family Prunellidae						
79	Altai Accentor/Himalayan accentor	<i>Prunella himalayana</i>	R	I, G	LC	STABLE
80	Rufous-breasted Accentor	<i>Prunella strophciata jerdoni</i>	R	I, G	LC	STABLE
Family Motacillidae						
81	Citrine Wagtail	<i>Motacilla citreola</i>	R	I, C	LC	INCREASING
82	Grey Wagtail	<i>Motacilla cinerea</i>	C	I, C	LC	STABLE
83	White Wagtail	<i>Motacilla alba</i>	UC	I, C, G	LC	STABLE
84	Tree Pipit	<i>Anthus trivialis</i>	UC	I, H, G	LC	STABLE
85	Rosy Pipit	<i>Anthus roseatus</i>	R	I, G	LC	STABLE
86	Water Pipit	<i>Anthus rubescens japonicus</i>	UC	I	LC	STABLE
Family Fringillidae						
87	Black-and-yellow Grosbeak	<i>Mycerobas icteroides</i>	R	F, G	LC	STABLE

S. no.	Common Name	Scientific Name	Abundance Status	Feeding Guild	IUCN Status	Population Trend
88	Common Rosefinch	<i>Carpodacus erythrinus</i>	C	F, G	LC	DECREASING
89	Pink-browed Rosefinch	<i>Carpodacus rodochroa</i>	UC	F, G	LC	STABLE
Family Emberizidae						
90	Rock Bunting	<i>Emberiza cia</i>	VC	G, I	LC	INCREASING
91	Chestnut-eared Bunting	<i>Emberiza fucata arcuate</i>	R	G, I	LC	STABLE

Abbreviations: F – Frugivore; G – Granivores; O – Omnivore; N – Nectarivore; H – Herbivore; I – Insectivores; Aql – Aquatic Insectivores; C – Carnivores; LC – Least Concern; NT – Near Threatened

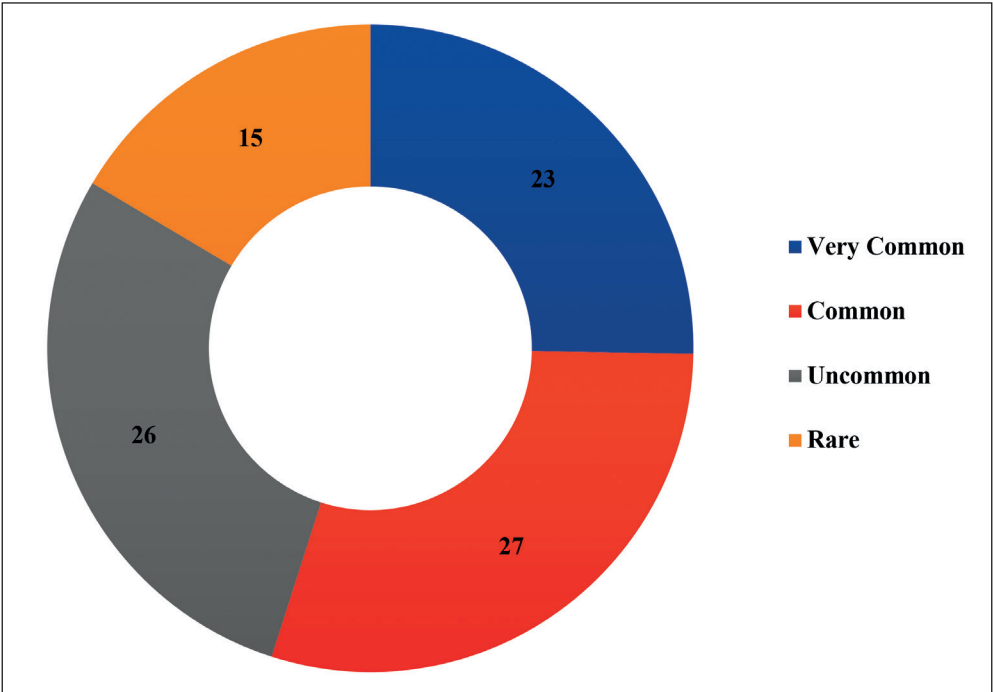


Figure 2. The abundance status of recorded bird species
2. ábra A megfigyelt madárfajok abundancia szerinti megoszlása

Hoopoe (*Upupa epops*), Crested Kingfisher (*Megaceryle lugubris guttulate*), Great Barbet (*Psilopogon virens*), Fulvous-breasted Woodpecker (*Dendrocopos macei*), Plum-headed Parakeet (*Psittacula cyanocephala cyanocephala*) Alexandrine Parakeet (*Psittacula eupatria*), Long-tailed Minivet (*Pericrocotus ethologus*), Rufous Treepie (*Dendrocitta vagabunda*), Grey Treepie (*Dendrocitta formosae*), Coal Tit (*Periparus ater*), Indian White-eye/Oriental White-eye (*Zosterops palpebrosus occidentis*), Variegated Laughingthrush (*Trochalopteron variegatum*), House Sparrow (*Passer domesticus*), Common Rosefinch (*Carpodacus erythrinus*).

The study area also harbours species of conservation importance like Himalayan Griffon and Alexandrine Parakeet which have been classified as Near Threatened in IUCN Red List (2024).

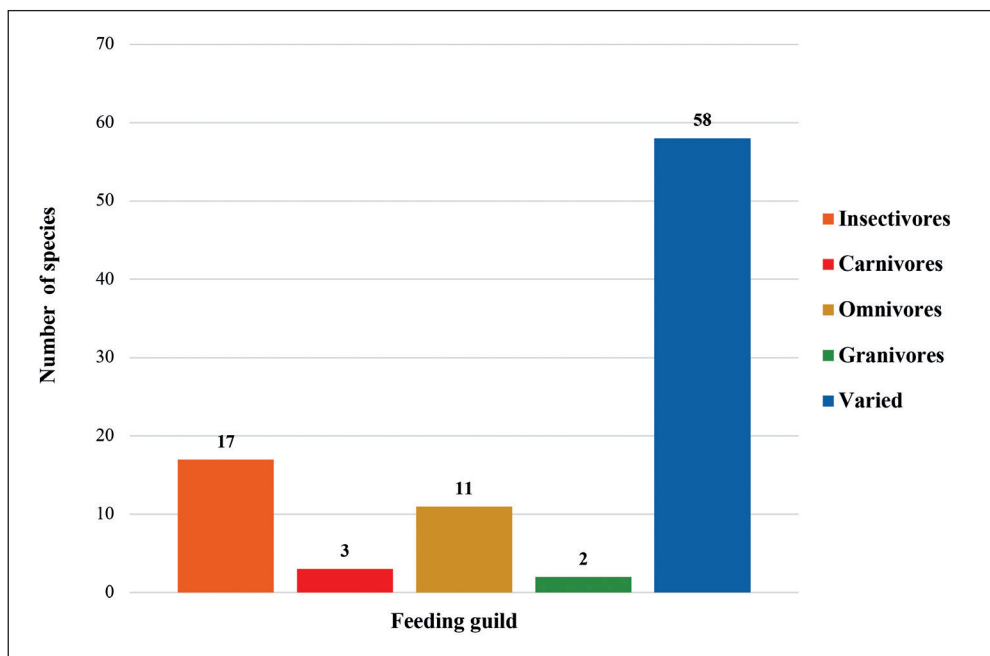


Figure 3. The distribution of the recorded bird species in the feeding guilds
 3. ábra A megfigyelt madárfajok táplálkozási guildenkénti megoszlása

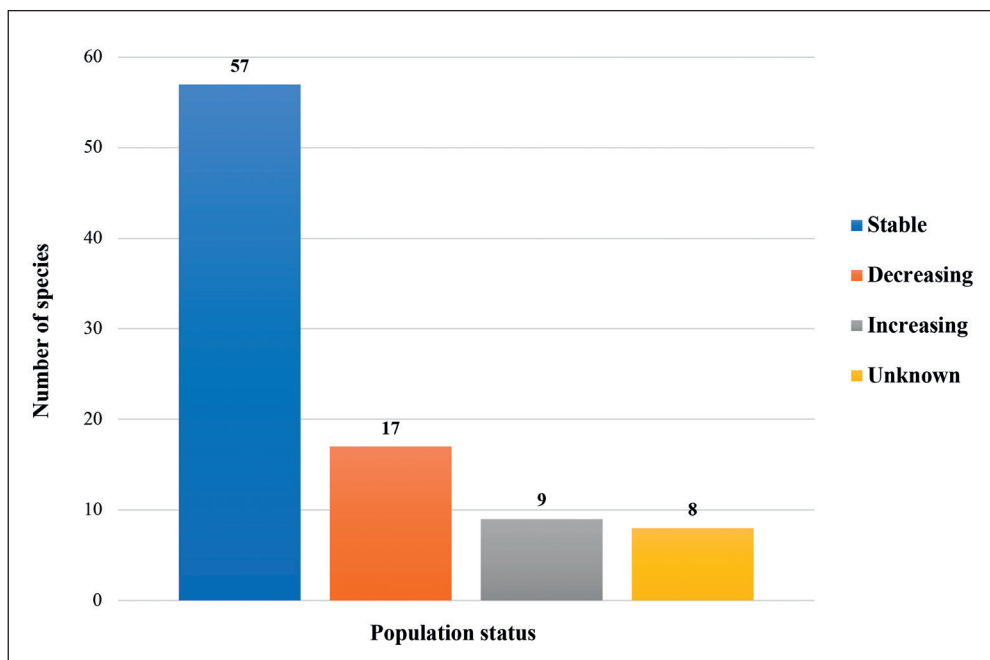


Figure 4. Population trends for recorded bird species
 4. ábra A megfigyelt madárfajok állománytrendjei

Discussion

The study represents the avifaunal diversity of tehsil Karsog, which is situated in the agro-climatic zone of the Pir Panjal Himalayan Range, Himachal Pradesh, India. The study area is characterized by a mix of forests and grasslands, distributed across an altitudinal gradient of 800-3100 m, which contributes its avian diversity. The study revealed the presence of 91 bird species, spanning over 63 genera, 38 families, and 12 orders. The present investigations align with earlier observations made by various researchers in different regions of Himachal Pradesh. Negi *et al.* (2015) unveiled the avian diversity within the Rakchham-Chhitkul Wildlife Sanctuary, encompassing 73 species. These species were spread across 52 genera, categorized into 24 families and 9 orders. Shah *et al.* (2016) studied the avian diversity within Chamba District, specifically focusing on the Kalatop-Khajjiar Wildlife Sanctuary. A total of 95 bird species, dispersed over 12 orders and 40 families, were recognized. Paliwal *et al.* (2019) conducted a comprehensive survey of avifauna at Bandali Wildlife Sanctuary, located in Mandi District, and identified 70 bird species, classified into 9 orders and 32 families. Dutta *et al.* (2022) enumerated the bird species in trans-Himalayan Lahaul and Spiti, Himachal Pradesh, and reported 278 bird species, representing 20 orders and 59 families. The avifaunal diversity study conducted by Tiwari (2023) in the unprotected Kanetiya area in Shimla, Himachal Pradesh, revealed the presence of 124 bird species, belonging to 13 orders and 43 families.

Passeriformes was the most dominant order, representing 70.3% of all recorded species. Numerous other studies have similarly highlighted the prevalence of the Passeriformes in various regions of the state of Himachal Pradesh. Singh (2015) studied the avifaunal diversity of the Mandi district. He found 94 species of birds belonging to 12 orders and 30 families. The highest number of species were found in the Passeriformes. Mohapatra *et al.* (2019) documented a total of 156 species of birds from YSPUHF campus, Solan, belonging to 16 orders, and 52 families. In their study, Passeriformes was found to be dominant (66.03% occurrence) with 33 families of birds. Kumar *et al.* (2023) prepared a checklist of avian diversity and composition in Kalatop-Khajjiar Wildlife Sanctuary, Himachal Pradesh, India. A list of 137 species was prepared pertaining to 15 orders and 46 families. Passeriformes was found to be the most dominant. Kaur and Banyal (2024) reported 216 species of birds belonging to 16 orders and 56 families from the Solan district of Himachal Pradesh. Out of the 16 orders, they also reported that Passeriformes was the most dominant. Passerine species exhibit high diversity in the study area because of their adaptability to a broad range of habitats and their consumption of numerous food items, such as invertebrates, fruits, grains, nuts, nectar, and floral buds (Beresford *et al.* 2005).

Among the 54 documented families, the avifauna in the study area was dominated by the Muscicapidae, comprising 12 species. It is noteworthy that the family Muscicapidae is the largest family of birds in India, with 370 species (Manakadan & Pittie 2001). A study conducted by Thakur *et al.* (2010) on the Arki Hills revealed similar results, with the presence of 85 species of birds belonging to 66 genera spread over 30 families and 12 orders. Their investigations also revealed that Muscicapidae dominated the avifauna, with 27 species. Singh *et al.* (2014) documented a total of 95 species of birds from Parashar Lake

and its surrounding area in Mandi district, Himachal Pradesh belonging to 11 orders, and 30 families. They found that Muscicapidae was the dominant family with a total of 24 species. Similarly, Shah *et al.* (2016) observed 95 species of birds belonging to 12 orders and 40 families in The Chamba District, Himachal Pradesh, India. Their analysis also showed that the Muscicapidae (15 species) was most dominant family. Paliwal *et al.* (2019) studied avifaunal diversity of Bandli Wildlife Sanctuary, Mandi, Himachal Pradesh and recorded a total of 70 bird species. They found Muscicapidae to be the dominant family, comprising 6 species. Kaur and Banyal (2024) studied the avifaunal diversity of Shimla region of Himachal Pradesh, and recorded a total of 52 bird species. They also found Muscicapidae to be the dominant family in their study.

During the present study, the relative abundance status of bird species was documented, considering their relative frequency of sightings in the area. Relative abundance provides insights about the increasing or decreasing populations of the bird species, which can be helpful in biodiversity monitoring and conservation (Callaghan *et al.* 2024). Several factors affect the relative number of birds in a given location. These include the size of the wetlands within the study area, the distribution and quality of food supplies, and the presence of appropriate roosting and nursery sites within the study area (Terborgh 1985). The analysis of the frequency of sightings showed that, out of the total 91 species recorded, 23 species (25.27%) were very common, 27 species (29.67%) were common, 26 species (28.57%) were uncommon, and 15 (16.48%) species were rare to the study area. Present studies on relative abundance support the earlier findings by Chopra and Sharma (2014) in the lower Shivalik foothills of the Himalayas. They categorized the observed 156 bird species as abundant (60 species, 38.46%), common (48 species, 30.77%), uncommon (41 species, 26.28%), and rare (7 species, 4.49%). Similarly, Singh (2018) studied the avifauna of Bhattiyat and Dalhousie areas of Chamba district, Himachal Pradesh. He found that out of 185 species of birds, 25 were Very Common, 28 were Common, 38 were Fairly Common, 65 were Uncommon and 29 were Rare. Recently, Quyoom *et al.* (2024) studied the avifaunal diversity of the Bani Wildlife Sanctuary, Jammu and Kashmir, India. They recorded a total of 135 bird species belonging to 45 families during their study. Further analysis of the relative abundance revealed that, among the recorded species, 36 were abundant, 40 were common, 33 were occasional, and 26 were rare.

The birds fulfill their nutritional demands by utilizing different kinds of microhabitats and using different foraging techniques (Wiens 1989). In order to maximize their chances of survival, birds use a wide variety of foraging strategies. Their habitat selection is contingent upon the proximity of their grazing grounds or reproductive areas (Raeside *et al.* 2007). Assessment of data on feeding guild shows that, out of 91 species in the study area, 2 species (2.19%) are granivores, 3 species (3.29%) are carnivores, 11 species (12.08%) are omnivores, 17 species (18.68%) are insectivores, and 58 species (63.73%) have varied types of feeding guild. Aggarwal *et al.* (2008) reported similar results, wherein they documented 75 species of birds from the Nandini Wildlife Sanctuary, Jammu and Kashmir. They observed five major feeding guilds, namely, insectivorous, granivorous, carnivorous, frugivorous, and omnivorous. Of the total 75 species reported, 23 were insectivorous, 10 were carnivorous, 6 were granivorous, 3 were omnivorous, and 6 were frugivorous, and the rest of the 27

species used more than one feeding guild. A study was conducted by Mukhopadhyay & Mazumdar (2017) on composition and foraging guilds of birds of suburban area of southern West Bengal, India. They recorded 119 species and categorized them into seven groups as, insectivorous (41.2%), carnivorous (24.4%), omnivorous (18.5%), granivorous (7.6%), frugivorous (3.4%), nectarivorous (3.4%) and herbivorous species (1.7%). In a study on avifaunal diversity in the Tilyar Lake, Rohtak, Haryana, India, Singh *et al.* (2020) recorded a total of 73 avian species belonging to 62 genera and 31 families under 15 orders. The feeding habits of the recorded birds showed that the maximum number of species (45 species) were omnivorous, followed by carnivorous (20), insectivorous (4), herbivorous (2), frugivorous (1), and granivorous (1).

The majority of birds in the study area (63.73%) have a varied type of feeding guild, which suggests the presence of a very heterogeneous habitat in terms of the availability of food in the study area. Further analysis of the data shows that more than 24% of the bird species present in the area show a restrictive feeding guild, which means they rely on very specific kinds of food for their survival. Species with a broad range of food choices have a higher degree of availability of resources. They may shift their feeding pattern from one kind of diet to another. However, the birds with restrictive food ranges have a lesser degree of resource availability. The exiguous alterations in the environment are leading to the depletion of their food resources, which may lead to their ultimate extinction.

The study area harbors species of conservation importance like the Himalayan Griffon and Alexandrine Parakeet, which have been placed in the Near Threatened category as per the IUCN Red List (2024). The study area provides habitat to threatened wild fauna as well as it includes 17 bird species with a decreasing population trend worldwide as per IUCN (2024). This raises concerns about the essential conservation efforts that need to be implemented in the study area.

The area is witnessing anthropogenic activities causing habitat destruction, modification, fragmentation, degradation over-exploitation of natural resources etc., which have devastating effect on wildlife and biodiversity of the area. Various factors, including construction projects, fast urbanization, variations in land cover, forest fires, modern farming methods, a spike in the development of roads and industries, and continued excessive resource extraction from the forest may alter the composition and structure of the vegetation, resulting in landslides and the formation of barren regions. This will therefore have an impact on avian occupancy and resource use patterns (Chettri *et al.* 2005). Frequent forest fires were seen in the study area during the study period, causing loss of habitat and food resources. Various developmental activities are ongoing in the study area such as construction of new roads and buildings, posing threat to the avifauna of the study area. The fragmentation of habitats due to the installation of cell phone towers in and around agricultural landscapes have been reported as source of disturbance to avifauna (Afrifa *et al.* 2023). Similarly, in the study area, new cell phone towers are being installed in the hill regions, which could potentially harm the local avifauna.

The study area harbors ample number of wetlands. Due to the intrinsic fragility of the ecosystems that they contain, wetlands are vulnerable to deterioration and modification due to the anthropogenic stresses such as agricultural runoff of fertilizers (Gupta & Singh

2003). The increased use of chemical fertilizers and pesticides is causing degradation of the wetlands in the study area which may affect the avian diversity of the area. All these facts underscore the importance of the study area for the conservation of wildlife. Therefore, further scientific studies and conservation measures are required to be implemented. The present study provides important preliminary data of the avian diversity of the Tehsil Karsog, which will provide a baseline data for future research and conservation efforts in this area. Moreover, regular bird surveys, systematic monitoring and assessment of the habitat, involving consideration of multiple drivers of change including climate change, should be conducted to monitor population trends, threats to local bird species and understanding their effects on bird species.

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Avifaunal diversity of Daranghati Wildlife Sanctuary, India

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Abstract Birds are among the most charismatic animals on our planet. Worldwide, there are about 11,000 different avian species. BirdLife International (2024) has listed 1,211 bird species that occur in India, demonstrating its immense diversity. Of these species, the majority (about 80%) are restricted to the Himalayas. To contribute to our knowledge on avian diversity in India, we conducted avifaunal surveys in the Daranghati Wildlife Sanctuary, located in Shimla district of Himachal Pradesh, India. A total of 76 species of birds, representing 11 orders, 32 families, and 59 genera, were recorded. Passeriformes dominated the avifauna with 56 species. Muscicapidae was the most abundant family enumerating 12 species. The second largest family was Fringillidae including 6 species, closely followed by Paridae and Phasianidae, with 5 species each. Of the 76 species recorded, 22 (28.95%) were classified as very common, 25 (32.90%) as common, 21 (27.63%) as uncommon, and 8 species (10.52%) as rare. The Western Tragopan (*Tragopan melanocephalus*) is a vulnerable species and the Himalayan Griffon (*Gyps himalayensis*) is listed as a near-threatened species as per IUCN Red List. Around 30% of the species belong to very narrow and restricted feeding guilds. The study also revealed the presence of 21 bird species with a decreasing population trends as provided by the global IUCN population assessments, thereby highlighting the importance and imperative of conservation measures in this Wildlife Sanctuary.

Keywords: biodiversity, Indian Himalayan Region, avifauna, threat

Összefoglalás A madarak változatos tollazatukkal és viselkedésükkel bolygónk leglenyűgözőbb állatai közé tartoznak. Világszerte körülbelül 11 000 különböző madárfaj él. A BirdLife International (2024) szerint Indiában 1 211 faj fordul elő. Ezek többsége (mintegy 80%) a Himalájára korlátozódik. A Daranghati Wildlife Sanctuary-ban madárállomány-felmérést végeztek. A terület az indiai Himachal Pradesh állam Shimla kerületében található. Összesen 76 madárfajt jegyeztek fel, amelyek 11 rendet, 32 családot és 59 nemzetséget képviselnek. A verébalakúak 56 fajjal domináltak. A légykapófélék (Muscicapidae) volt a legmagasabb abundancia-értékeket mutató család 12 fajjal. A második legnagyobb pintyfélék (Fringillidae) volt, 6 fajjal, őket követte a cinegefélék (Paridae) és a fácánfélék (Phasianidae), 5-5 fajjal. A feljegyzett 76 fajból 22 (28,95%) a nagyon gyakori, 25 (32,90%) a gyakori, 21 (27,63%) a rendszeres, 8 faj (10,52%) pedig a ritka kategóriába esett. A feljegyzett madárfajok közül a nyugati tragopán (*Tragopan melanocephalus*) a sérülékeny fajok, a himalájai keselyű (*Gyps himalayensis*) pedig a közel-veszélyeztetett fajok közé tartozik az IUCN Vörös Listáján. A fajok körülbelül 30%-a tartozik nagyon szűk és korlátozott táplálkozási guilddekhez. A tanulmány 21 olyan madárfaj jelenlétét tárta fel, amelyek populációja csökkenő tendenciát mutat az IUCN világszerte végzett állomány-felmérése szerint, ezzel is hangsúlyozva a védelmi intézkedések fontosságát és elengedhetetlenségét ebben a vadrezervátumban.

Kulcsszavak: biodiverzitás, Indiai Himalája régió, madárfauna, veszélyeztetettség

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Introduction

The Indian Himalayan Region serves as a massively diverse biological hotspot (Myers *et al.* 2000). The Himalayas, which stretch over 2,400 km, have a wide range of climatic conditions, spanning subtropical to the boreal zone (Singh & Singh 1987, Rana *et al.* 2021). The Indian Himalayan Region (IHR) encompasses thirteen states and half a million square kilometers of mountainous terrain in India (Uppendra *et al.* 2002). The Himalayan ecology is distinctive, with an incredibly rich variety of ecological resources and biological abundance (Thakur *et al.* 2010). The Himalayas are categorized into Eastern Himalayas, Central or Nepal Himalayas, and Western Himalayas depending on the environmental conditions and geographic location (Sati 2016). With elevations ranging from 300 meters to 6,000 meters, the Western Himalayas cover the Indian states of Himachal Pradesh and Uttarakhand, as well as Jammu and Kashmir (Tewari *et al.* 2017). The Western Himalayas comprise approximately 17% of the total land area of Himachal Pradesh (Kumar *et al.* 2023). The state of Himachal Pradesh is located in the northwest of India. The eastern part of Himachal Pradesh defines the international border between Tibet and India. The state is surrounded by Jammu and Kashmir in the north, Uttarakhand in the southeast, Haryana in the south, and Punjab in the west (Narwade *et al.* 2006). According to Indian State of Forest Report (2021), the state has a total geographical area of 55,673 km². About 68.16% of the state is covered by forested lands amounting to 37,948 km² total area, which is further divided into Reserved Forests, containing 1,883 km² area, Protected Forests (with 12,853 km² area under demarcated protected forests and 16,035 km² area under un-demarcated protected forests) contain about 28,887 km² area in total and other forests covering altogether an area of 7,160 km², including unclassified forests, Shamlats, co-operative forests, charagah and strip forests. The diverse climate of Himachal Pradesh, which ranges between tropical in the lower Himachal to cold in the trans-Himalayan area, is largely responsible for the extreme biodiversity of Himachal Pradesh (Negi *et al.* 2015).

Worldwide, there are about 11,000 different species of birds. BirdLife International (2024) has listed 1,211 bird species that exist in India, demonstrating its great avian diversity. Of the 1,211 bird species, the majority (about 80%) are restricted to the Himalayas (Naithani & Bhatt 2010). In Himachal Pradesh, the Shivalik hill area has the highest avian diversity values, followed by Greater and trans-Himalayan areas (Praveen *et al.* 2016). Avifaunal studies in Himachal Pradesh has a long historical background. Due to the establishment of numerous well-known hill towns, like Dalhousie, Dharamsala, and Kullu, as well as the installation of the British summer capital Shimla, earlier research on the avian fauna of Himachal Pradesh began under the British rule. The oldest reference of the observation on bird species of Himachal Pradesh is attributed to Theobald (1862). Later, a number of field studies were performed by various scientists such as Tytler (1868), Anderson (1889) and Dodsworth (1910, 1913). Field based biological research on the diversity, distribution, abundance and presence of birds in the state has been conducted in recent decades. Mahabal and Mukherjee (1991) studied the bird diversity in Mandi district, Mahabal and Sharma (1992) studied the distribution of birds in Kangra Valley, Thakur *et al.* (2002, 2003, 2006) studied the bird diversity in Chamba, Mandi, and Shimla, Jan Willem den Besten (2004)

reported avifaunal diversity of district Kangra, Mattu and Thakur (2006), Singh *et al.* 2014, Negi *et al.* 2015 studied birds in different region of Himachal Pradesh. Though it has a long history, most of the studies were confined only to certain specific regions of the state. Majority of the state region especially the Greater Himalayan part including the present study area of Daranghati Wildlife Sanctuary has not received the attention of field biologists due to its tough terrain, less communication network, and harsh climate. Hence the present study was undertaken with an aim to elucidate the biodiversity, unique species assemblage with special reference to avian diversity of the area

Materials and Methods

Present research has been conducted in Daranghati Wildlife Sanctuary, situated between 31.3917° N, 77.803° E, in Shimla district of Himachal Pradesh, India. The sanctuary is surrounded by the Rampur, Rohru, and Kinnaur Territorial Forest Divisions. The area was used as a hunting reserve by the Rampur Bushahr royal family. It was declared a sanctuary in March 1962 and then re-notified on March 27, 1974 (*Figure 1*). Daranghati Wildlife Sanctuary I and II are the two non-contiguous parts that comprise the sanctuary. Together,

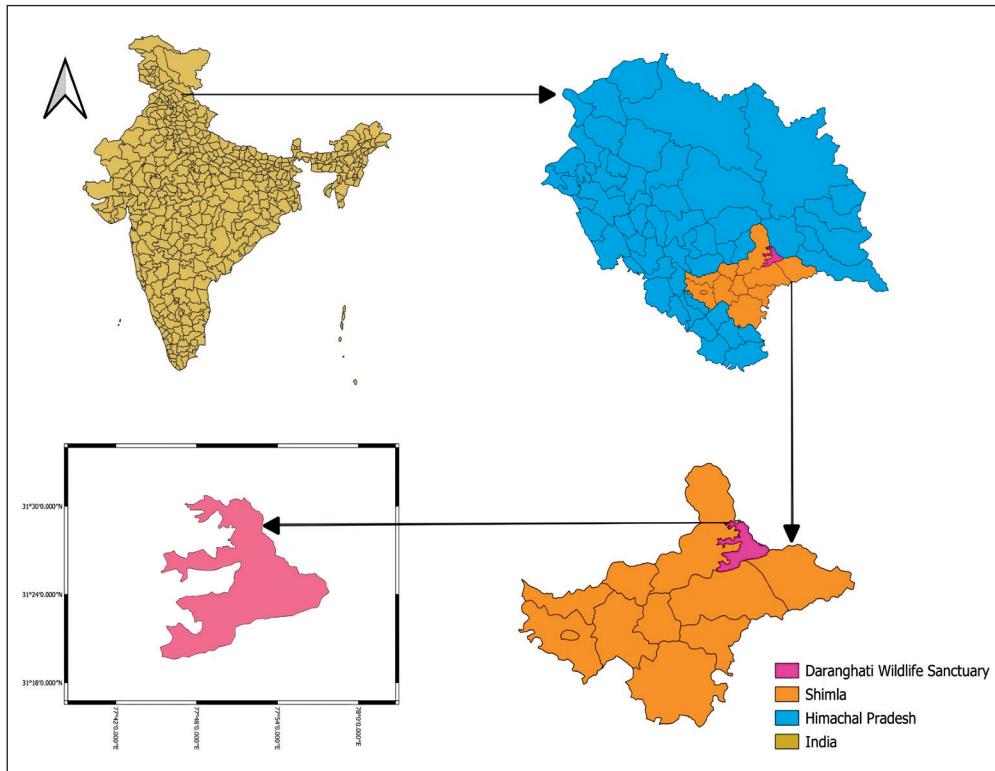


Figure 1. A map showing Daranghati Wildlife Sanctuary, Shimla, Himachal Pradesh, India
 1. ábra A Daranghati Wildlife Sanctuary térképe (indiai Himachal Pradesh állam Shimla kerület)

the two sections cover a 171.50 km² area. The two sections of the sanctuary are located on either side of the Dhauladhar Range. The altitude varies from 2,155 m (Nogli Gad upstream point) to 5,400 m (Hans Beshan peak follows the district boundary of Shimla and Kinnaur). The sanctuary is dominated by cool summers and harsh winters, temperatures varying from –8 °C in winter to 28 °C in summer. The average annual rainfall ranges from 625 to 900 mm, with intense monsoon rains from July to September and regular snowfall from December to March (Pandey 1990). The Daranghati Wildlife Sanctuary is divided into nine administrative units (Department of Forest, Govt. of Himachal Pradesh). In terms of vegetation, the Sanctuary is classified into a diverse range of forest and alpine meadows. The sanctuary has five kinds of forests: moist cedar forest, western mixed coniferous forest, moist temperate deciduous forest, oak forest and West Himalayan sub-alpine forests (Pandey 1990). The lower altitudes are dominated by mixed oak and coniferous trees, while the higher elevations feature alpine scrubs, dwarf junipers and grasslands.

The present study area of Daranghati Wildlife Sanctuary has been explored from December 2023 to May 2024 in various locations of the sanctuary. Six out of a total of nine administrative units, namely Sarahan, Runpu, Bajwa, Munish, Kasha, and Pat were explored in the present study. The surveys were conducted at the altitudinal gradient between 2,155 and 3,100 m during 3 seasons, winter (December 2023–March 2024), spring (April 2024), and summer (May 2024). Because the research area is situated in tough terrain and difficult to reach, most of the areas were explored by walking. The stratified random sample approach was used to study birds (Snedecore & Cochran 1967). During the survey, the line transect method was also used for the estimation of avian diversity which is based on the principle of walking along the specified path to record the birds on and near the line (Burnham *et al.* 1980). Also, these transect were observed as open width transect, meaning that birds were observed along the transect regardless of how far away they were from it (Mehta *et al.* 2002). During the survey, the line transect of 500 m to 1 km were selected in the aforementioned 6 areas. The transect was covered by walking slowly along the length periodically standing at random points to observe the birds all around as far as visibility permits. The line transects were also set along different altitudinal gradients extending from 2,150 m to 3,100 m above sea level. During the survey, five to six random transects were drawn in each covered area.

The study focused on various habitat types, such as woods and streams, located throughout the sanctuary at different altitudes. Birds were observed in the early morning and late evening, as the birds reach their peak activity within a couple of hours after sunrise or before nightfall (Negi *et al.* 2015). Nikon (10x40) fieldwork binoculars were used for observation. In order to verify the identifications, images of the birds were captured using a Nikon DSLR D90 equipped with a 70–300 mm telephoto lens. Birds were identified in the field using various field guides (Ali & Ripley 1983, Grimmett *et al.* 1999) and re-verified by bird database application (Merlin Bird ID by Cornell Lab version 3.0.3 (800–2023.0)). Birds were classified by the following Avibase – The World Bird Database (Lepage 2024). Feeding guild was assessed as per Ali and Ripley (1968 to 1999). The IUCN Red List was referred to examine the conservation status as well as population trends of the reported species (IUCN 2024). Data collected from several habitat categories in the study area was analyzed for

relative abundance based on frequency of encounters (McKinnon & Philips 1993). Different categories assigned based on data were Very Common (80–100% sightings), Common (50–80% sightings), Uncommon (6–50% sightings), and Scarce/Rare (1–5% sightings).

Results and Discussion

This study on the avifaunal diversity of the Daranghati Wildlife Sanctuary in the Shimla districts of Himachal Pradesh revealed the presence of 76 species of birds, distributed over 59 genera, 32 families, and 11 orders. The Passeriformes, with 21 families was the most species-rich order in this sanctuary, followed by Piciformes containing two families. The rest of nine orders namely Galliformes, Apodiformes, Cuculiformes, Columbiformes, Accipitriformes, Strigiformes, Bucerotiformes, Falconiformes and Psittaciformes were present with a single family each (Table 1).

The Muscicapidae family, including 12 species, was the largest family in this sanctuary. The results substantiate the fact that the family Muscicapidae, is the largest bird family in India and Himachal Pradesh (Singh *et al.* 2014, Negi *et al.* 2015). The second largest family was Fringillidae harbouring six species, closely followed by Phasianidae and Paridae with five species each. The families Corvidae, Phylloscopidae and Leiotrichidae, contain four species each, while the families Accipitridae, Picidae, Turdidae and Motacillidae were represented by three species each. The families Columbidae, Pycnonotidae and Passeridae were represented by two species each. The rest of eighteen families, such as, Apodidae, Cuculidae, Strigidae, Upupidae, Ramphastidae, Falconidae, Psittaculidae, Campephagidae, Dicuridae, Stenostiridae, Aegithalidae, Cisticolidae, Certhiidae, Sturnidae, Cinclidae, Nectariniidae, Prunellidae, and Emberizidae were present in area with single species each (Table 1).

Analysis of data on relative abundance showed that of the 76 species recorded, 22 species (28.95%) were very common, 25 species (32.90%) were common, 21 species (27.63%) were uncommon, and 8 species (10.52%) were rare to the sanctuary (Table 1, Figure 2). Among 76 bird species, eight species were found to be rare and observed only once during survey. These rare species included Common Swift (*Apus apus*), Oriental Honey-buzzard (*Pernis ptilorhynchus*), Rufous-bellied Woodpecker (*Dendrocopos hyperythrus*), Rufous-naped Tit (*Periparus rufonuchalis*), Dusky Warbler (*Phylloscopus fuscatus*), Alpine Accentor (*Prunella collaris*), Tree Pipit (*Anthus trivialis*), Himalayan Beautiful Rosefinch (*Carpodacus pulcherrimus*).

The feeding guild assessment shows a wide variety of diet strategies among the species present in the study area. Analysis of data showed that of the 76 species recorded, 12 species (15.78%) were omnivores, 19 species (25%) were insectivores, 2 (2.63%) species were carnivores, 2 species (2.63%) were granivores and 41 species (53.94%) showed a mixed feeding guild (Table 1, Figure 3). Analysis of data shows that about 70% of species have wide feeding guild while remaining 30% have restricted and narrow feeding guild.

Worldwide, IUCN assessed population trend of reported avifauna in Daranghati Wildlife Sanctuary was increasing for 4 bird species, decreasing for 21 species, stable for

Table 1. List of avian fauna recorded in Daranghati Wildlife Sanctuary, Shimla, Himachal Pradesh (India)

1. táblázat A Daranghati Wildlife Sanctuary-ban felmért madárfajok listája (India, Himachal Pradesh állam Shimla kerület)

S. no.	Common Name	Scientific Name	Abundance Status	Feeding Guild	IUCN Status	Population Trend
Order Galliformes						
Family Phasianidae						
1	Hill Partridge	<i>Arborophila torqueola</i>	UC	G,I	LC	STABLE
2	Western Tragopan	<i>Tragopan melanocephalus</i>	UC	O	VU, LD	DECREASING
3	Himalayan Monal	<i>Lophophorus impejanus</i>	UC	O	LC, MD	DECREASING
4	Koklass Pheasant	<i>Pucrasia macrolopha</i>	C	O	LC	DECREASING
5	Kalij Pheasant	<i>Lophura leucomelanos</i>	VC	O	LC	DECREASING
Order Apodiformes						
Family Apodidae						
6	Common Swift	<i>Apus apus</i>	R	I	LC	STABLE
Order Cuculiformes						
Family Cuculidae						
7	Common Cuckoo	<i>Cuculus canorus</i>	VC	I	LC	DECREASING
Order Columbiformes						
Family Columbidae						
8	Snow Pigeon	<i>Columba leuconota</i>	UC	G,F	LC	STABLE
9	Oriental Turtle Dove	<i>Streptopelia orientalis</i>	C	G	LC	STABLE
Order Accipitriformes						
Family Accipitridae						
10	Oriental Honey Buzzard	<i>Pernis ptilorhynchus</i>	R	I	LC	DECREASING
11	Himalayan Griffon	<i>Gyps himalayensis</i>	C	C	NT	DECREASING
12	Black Kite	<i>Milvus migrans</i>	UC	C	LC	STABLE
Order Strigiformes						
Family Strigidae						
13	Asian Barred Owlet	<i>Glaucidium cuculoides</i>	VC	C, I	LC	INCREASING
Order Bucerotiformes						
Family Upupidae						
14	Eurasian Hoopoe	<i>Upupa epops</i>	C	I	LC	DECREASING
Order Piciformes						
Family Ramphastidae						
15	Great Barbet	<i>Psilopogon virens</i>	VC	F, I	LC	DECREASING
Family Picidae						
16	Rufous-bellied Woodpecker	<i>Dendrocopos hyperythrus</i>	R	I,F, G	LC	DECREASING

S. no.	Common Name	Scientific Name	Abundance Status	Feeding Guild	IUCN Status	Population Trend
17	Himalayan Woodpecker	<i>Dendrocopos himalayensis</i>	VC	I, F,G	LC	STABLE
18	Scaly-bellied Woodpecker	<i>Picus squamatus</i>	C	I,F,G	LC	STABLE
Order Falconiformes						
Family Falconidae						
19	Eurasian Kestrel	<i>Falco tinnunculus</i>	UC	I,C	LC	DECREASING
Order Psittaciformes						
Family Psittaculidae						
20	Slaty-headed Parakeet	<i>Psittacula himalayana</i>	VC	F, H, G	LC	STABLE
Order Passeriformes						
Family Campephagidae						
21	Long-tailed Minivet	<i>Pericrocotus ethologus</i>	C	I	LC	DECREASING
Family Dicruridae						
22	Ashy Drongo	<i>Dicrurus leucophaeus</i>	C	I	LC	UNKNOWN
Family Corvidae						
23	Yellow-billed Blue Magpie	<i>Urocissa flavirostris</i>	VC	C,I	LC	STABLE
24	Red-billed Blue Magpie	<i>Urocissa erythroryncha</i>	VC	C,I	LC	STABLE
25	Northern Nutcracker	<i>Nucifraga caryocatactes</i>	C	O	LC	DECREASING
26	Large-billed Crow	<i>Corvus macrorhynchos</i>	VC	O	LC	STABLE
Family Stenostiridae						
27	Grey-headed Canary Flycatcher	<i>Culicicapa ceylonensis</i>	C	I	LC	STABLE
Family Paridae						
28	Rufous-naped Tit	<i>Periparus rufonuchalis</i>	R	O	LC	STABLE
29	Rufous-vented Tit	<i>Periparus rubidiventris beavani</i>	UC	I, C, S	LC	STABLE
30	Coal Tit	<i>Periparus ater</i>	VC	I	LC	DECREASING
31	Green-backed Tit	<i>Parus monticolus</i>	VC	I,F,H	LC	STABLE
32	Indian Yellow Tit	<i>Machlolophus aplonotus</i>	UC	I,F,C	LC	STABLE
Family Pycnonotidae						
33	Black Bulbul	<i>Hypsipetes leucocephalus</i>	C	G,F,I	LC	STABLE
34	Himalayan Bulbul	<i>Pycnonotus leucogenys</i>	VC	G,F,I	LC	INCREASING
Family Aegithalidae						
35	Black-throated Tit	<i>Aegithalos concinnus</i>	VC	G,F,I	LC	STABLE
Family Phylloscopidae						
36	Buff-barred Warbler	<i>Phylloscopus pulcher</i>	UC	I	LC	STABLE
37	Dusky Warbler	<i>Phylloscopus fuscatus</i>	R	I	LC	STABLE
38	Lemon-rumped Warbler	<i>Phylloscopus chloronotus</i>	C	I	LC	STABLE
39	Grey-hooded Warbler	<i>Phylloscopus xanthoschistos</i>	C	I	LC	STABLE

S. no.	Common Name	Scientific Name	Abundance Status	Feeding Guild	IUCN Status	Population Trend
Family Cisticolidae						
40	Himalayan Prinia	<i>Prinia crinigera</i>	UC	I,C,G	LC	STABLE
Family Leiothrichidae						
41	Striated Laughingthrush	<i>Grammatoptila striata</i>	UC	I,F,G	LC	DECREASING
42	Streaked Laughingthrush	<i>Trochalopteron lineatum</i>	VC	I,F,G	LC	STABLE
43	Variegated Laughingthrush	<i>Trochalopteron variegatum</i>	VC	I,F,G	LC	DECREASING
44	Rufous Sibia	<i>Heterophasia capistrata capistrata</i>	C	F,I	LC	UNKNOWN
Family Certhiidae						
45	Bar-tailed Treecreeper	<i>Certhia himalayana</i>	C	I	LC	DECREASING
Family Sturnidae						
46	Common Myna	<i>Acridotheres tristis</i>	VC	G, F, I	LC	INCREASING
Family Turdidae						
47	Mistle Thrush	<i>Turdus viscivorus</i>	UC	I,F	LC	DECREASING
48	Tickell's Thrush	<i>Turdus unicolor</i>	C	I, C	LC	UNKNOWN
49	White-collared Blackbird	<i>Turdus albocinctus</i>	C	O	LC	UNKNOWN
Family Muscicapidae						
50	Oriental Magpie Robin	<i>Copsychus saularis</i>	UC	O	LC	STABLE
51	Rufous-bellied Niltava	<i>Niltava sundara</i>	UC	I,F	LC	STABLE
52	Verditer Flycatcher	<i>Eumyias thalassinus</i>	C	I	LC	STABLE
53	Little Forktail	<i>Enicurus scouleri</i>	UC	AqI	LC	STABLE
54	Spotted Forktail	<i>Enicurus maculatus</i>	C	I,C	LC	STABLE
55	Blue WhistlingThrush	<i>Myophonus caeruleus</i>	VC	O	LC	UNKNOWN
56	Ultramarine Flycatcher	<i>Ficedula superciliaris</i>	UC	I	LC	STABLE
57	Blue-capped Redstart	<i>Phoenicurus coeruleocephala</i>	UC	I	LC	STABLE
58	Plumbeous Water Redstart	<i>Phoenicurus fuliginosus</i>	C	AqI	LC	STABLE
59	White-capped Redstart	<i>Phoenicurus leucocephalus</i>	C	O	LC	STABLE
60	Blue-capped Rock Thrush	<i>Monticola cinclorhyncha</i>	C	O	LC	STABLE
61	Grey Bushchat	<i>Saxicola ferreus</i>	VC	I,C	LC	STABLE
Family Cinclidae						
62	Brown Dipper	<i>Cinclus pallasii pallasii</i>	UC	I,C	LC	STABLE
Family Nectariniidae						
63	Gould's Sunbird	<i>Aethopyga gouldiae</i>	UC	N,I,C	LC	STABLE
Family Passeridae						
64	Russet Sparrow	<i>Passer cinnamomeus</i>	VC	I,G,F	LC	STABLE
65	House Sparrow	<i>Passer domesticus</i>	VC	G,I	LC	DECREASING

S. no.	Common Name	Scientific Name	Abundance Status	Feeding Guild	IUCN Status	Population Trend
Family Prunellidae						
66	Alpine Accentor	<i>Prunella collaris</i>	R	I,G	LC	STABLE
Family Motacillidae						
67	Gray Wagtail	<i>Motacilla cinerea</i>	C	I,C	LC	STABLE
68	Tree Pipit	<i>Anthus trivialis</i>	R	I,G	LC	DECREASING
69	Upland Pipit	<i>Anthus sylvanus</i>	UC	I	LC	STABLE
Family Fringillidae						
70	Black-and-yellow Grosbeak	<i>Mycerobas icteroides</i>	UC	F,G	LC	STABLE
71	Plain Mountain Finch	<i>Leucosticte nemoricola</i>	C	G	LC	STABLE
72	Red-headed Bullfinch	<i>Pyrrhula erythrocephala</i>	VC	H,N,F	LC	STABLE
73	Common Rosefinch	<i>Carpodacus erythrinus</i>	C	F,G	LC	DECREASING
74	Himalayan beautiful Rosefinch	<i>Carpodacus pulcherrimus</i>	R	F,G	LC	STABLE
75	Pink-browed Rosefinch	<i>Carpodacus rodochroa</i>	C	F,G	LC	STABLE
Family Emberizidae						
76	Rock Bunting	<i>Emberiza cia</i>	VC	G,I	LC	INCREASING

Abbreviations

F – Frugivore; G – Granivore; O – Omnivore; N – Nectarivore; H – Herbivore; I – Insectivore; Aql – Aquatic Insectivore; C – Carnivore; LC – Least Concern; NT – Near Threatened; VU – Vulnerable; LD – Largely Depleted; MD – Moderately Depleted

Rövidítések

F – Gyümölcssevő; G – Magevő; O – Mindenevő; N – Nektárevő; H – Növényevő; I – Rovarevő; Aql – Vízirovar fogyasztó; C – Ragadozó; LC – Nem fenyegetett; NT – Mérsékelten fenyegetett; VU – Sebezhető; LD – Széleskörben lecsökkent; MD – Mérsékelten lecsökkent

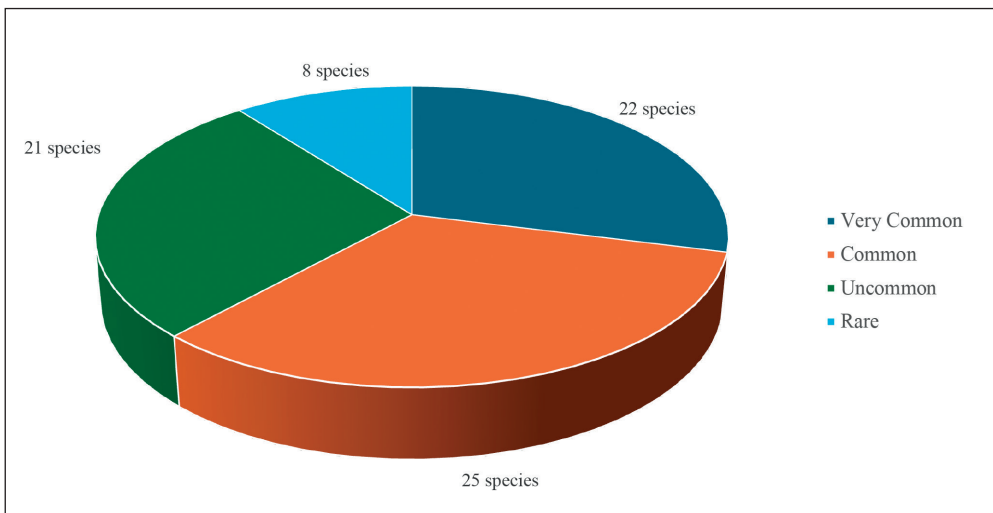


Figure 2. A pie chart representing the abundance status of recorded bird species

2. ábra A madárfajok megoszlása az egyes gyakoriság kategóriákban

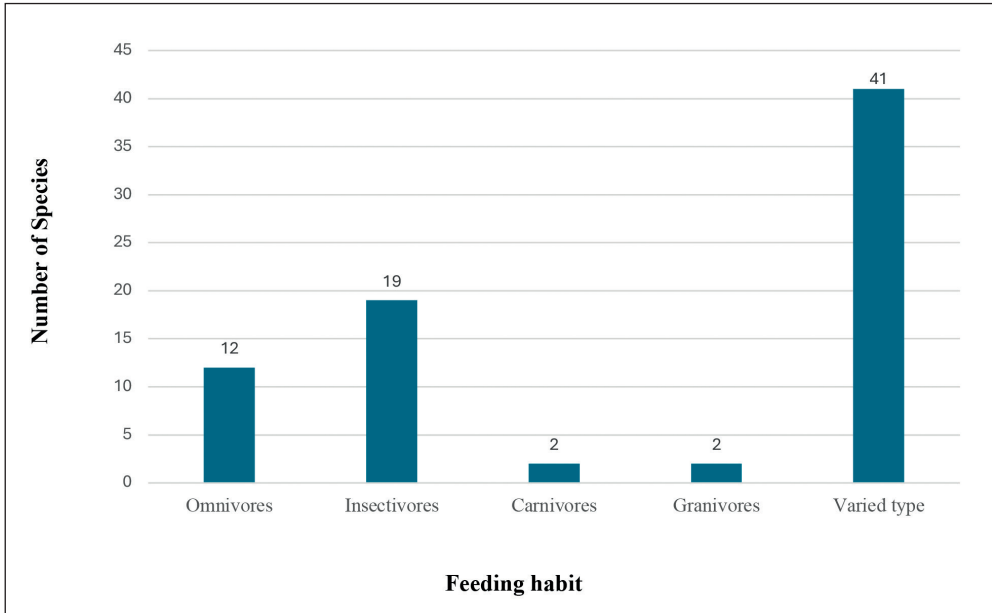


Figure 3. A column graph representing the feeding guild for recorded bird species
3. ábra A feljegyzett madárfajok táplálkozási guildek szerinti száma

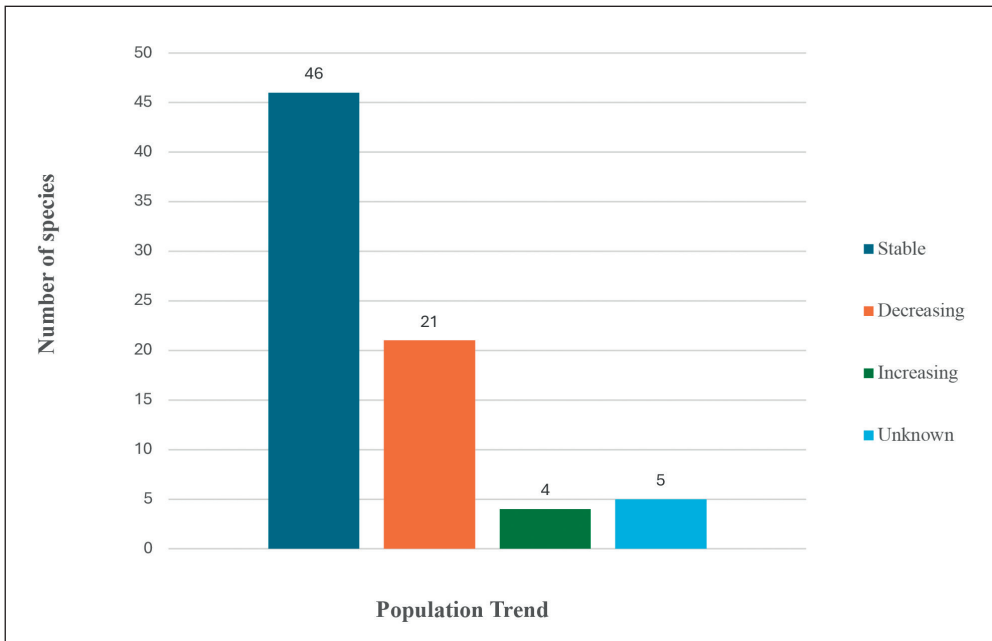


Figure 4. A column graph representing the population trend for recorded bird species
4. ábra A feljegyzett madárfajok száma populációs trendjeik alapján

46 bird species, and unknown for 5 bird species (Table 1, Figure 4). The species with a decreasing population trend worldwide reported in this sanctuary are Western Tragopan (*Tragopan melanocephalus*), Himalayan Monal (*Lophophorus impejanus*), Koklass Pheasant (*Pucrasia macrolopha*), Kalij Pheasant (*Lophura leucomelanos*), Common Cuckoo (*Cuculus canorus*), Oriental Honey-buzzard (*Pernis ptilorhynchus*), Himalayan Griffon (*Gyps himalayensis*), Eurasian Hoopoe (*Upupa epops*), Great Barbet (*Psilopogon virens*), Rufous-bellied Woodpecker (*Dendrocopos hyperythrus*), Eurasian Kestrel (*Falco tinnunculus*), Long-tailed Minivet (*Pericrocotus ethologus*), Northern Nutcracker (*Nucifraga caryocatactes*), Coal Tit (*Parus ater*), Striated Laughingthrush (*Grammatoptila striata*), Variegated Laughingthrush (*Trochalopteron variegatum*), Bar-tailed Tree-Creeper (*Certhia himalayana*), Mistle Thrush (*Turdus viscivorus*), House Sparrow (*Passer domesticus*), Tree Pipit (*Anthus trivialis*) and Common Rosefinch (*Carpodacus erythrinus*). The sanctuary is home to species of key conservation values. The Western Tragopan, which is found in the study area, has been assigned the status of vulnerable by IUCN Red List. In addition, the Himalayan Griffon Vulture is classified into the near threatened category which is commonly found in the study area.

The study area is witnessing increased anthropogenic activities which may lead to habitat destruction, fragmentation, degradation and over-exploitation, which will have a direct effect on its biodiversity values including avian diversity. The sanctuary provides shelter to threatened birds species as well as is home to 21 bird species with a global decreasing population trends. These facts highlight the importance of the study area for the conservation of avifauna in the region.

Bird composition and diversity can be influenced by vegetation structure, foraging substrates, and the availability and abundance of food (MacArthur & MacArthur 1961). Around 30% of the species have a very narrow and restricted feeding guild, indicating that they are precariously surviving on limited food supply. Since the sanctuary is located in a region of harsh and fragile climatic conditions and is home to rare and threatened species, therefore, further scientific studies and conservation planning are required to preserve these valuable species. The current study brings to light an initial checklist of avifaunal diversity in the sanctuary and will serve as a baseline reference for future research on avifauna and biodiversity of Daranghati Wildlife Sanctuary.

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Avian assemblages in the waste landfills: observations from urban sites in West Bengal, India

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Abstract Urban dumpsites, often regarded as environmental eyesores, may play an unexpected ecological role by providing habitats for avian species. In this context, the present study was conducted to justify whether dumpsites can sustain bird populations and enhance urban biodiversity. We recorded the bird abundance and species richness through surveys at four well-known urban dumpsites in Kolkata and surrounding areas between February and June 2024. We encountered a variety of common and rare bird species that use the dumpsite for feeding, breeding, and roosting using established methods. The dumping area close to the agricultural fields had a higher bird species richness compared to the urbanized area. However, the urban dumping areas exhibited comparatively higher bird abundance. House Crow (*Corvus splendens*) and Cattle Egret (*Bubulcus ibis*) were the two most common species at the dumpsites. This work emphasizes the importance of studying the bird groups concerning landfill sites and management, as a significant influx of birds could lead to various environmental challenges in metropolitan areas.

Keywords: avian diversity, landfill, waste management, feeding guilds

Összefoglalás A városi szeméttelpek kevésbé ismert ökológiai szerepet játszhatnak, mivel élőhelyet biztosítanak a madárfajok számára. Egy tanulmányt végeztek annak igazolására, hogy a hulladéklerakók képesek-e fenntartani a madárpopulációkat, és ezáltal növelni a városi biodiverzitást. Négy jól ismert városi szeméttelpek felmérése során rögzítettük a jelenlévő madárfajokat és a madarak számát. Sztenderd ornitológiai eljárások segítségével számos olyan gyakori és ritka madárfajt jegyeztünk fel, amelyek táplálkozásra, költésre és pihenésre használják a szeméttelpeket. 2024 februárja és júniusa között blokkszámlálások segítségével madárszámlálásokat végeztünk két különböző hulladéklerakó helyen. Az erdőhöz közeli lerakási területen szignifikánsan több madár volt, mint a városi területen. Azonban utóbbi mutatta a magasabb avifaunisztikai diverzitást és gazdagságot. A két leggyakoribb faj az indiai varjú (*Corvus splendens*) és a pásztorgém (*Bubulcus ibis*) volt. Ez a munka hangsúlyozza a madársoportok hulladéklerakók területén történő tanulmányozásának és kezelésének fontosságát, mivel a madarak jelentős beáramlása különböző környezeti kihívásokhoz vezethet a nagyvárosi területeken.

Kulcsszavak: hulladékgazdálkodás, sokféleség, szeméttelrakó, táplálkozási guildek

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Introduction

Urban development often misses the proper management of dump areas, which become alternative homes for various bird species. As cities grow larger and natural spaces get smaller, several birds find food, shelter, and breed in these artificial yet polluted areas. A deeper analysis of this development deepens our understanding of human and wildlife interactions (Oro *et al.* 2013). Dump areas provide optimal bird habitats in towns and cities, where natural habitats are frequently fragmented or deteriorated. Numerous species are attracted to these locations, especially those adapted to take advantage of anthropogenic resources due to the high amount of food resources. Avian species diversity is influenced by the habitat variability seen in dumpsites, which includes open areas, garbage piles, and vegetated zones. Because of the diversity of microhabitats, several species can live and use diverse ecological niches found in the dumpsites. On the other side, environmental pollution evokes growing environmental concern, especially in developing countries, where rapid growth of the human population is being observed. Municipal solid waste results from human activities, causing significant levels of pollution and health problems. Food leftovers produced by people act as food subsidies for other species, vastly changing the ecosystem structure and function, especially for leftovers from farms and dumpsites (Oro *et al.* 2013). Municipal solid waste mainly comes from places where people live, including small businesses, hospitals, clinics, and other commercial activities. Mixing medical waste with regular trash brings health hazards and environmental challenges. Therefore, it is crucial to keep medical waste separate for safe disposal following the standard rules and regulations to reduce health and environmental hazards (Pattnaik & Reddy 2009, Sing *et al.* 2011, Hoornweg *et al.* 2013). Fast-developing countries often face tricky problems with solid waste management based on population size and wealth levels. When waste is not managed well, it damages the environment and public health, so we must find sustainable solutions (Flores-Tena *et al.* 2007, Houston *et al.* 2007, Matejczyk *et al.* 2011, Mane & Hingane 2012). Particularly in developing countries, managing solid waste becomes highly important since financial resources are limited and the risks from dumping increase (Hazra & Goel 2009, Monney *et al.* 2013). In this context, understanding the feeding ecology of the fauna present in the dumpsites may help with wildlife management.

Human-modified habitats like landfills become crucial for avian species as natural habitats decline. As birds adapt, social behaviour and demography may change, leading to stable dominance hierarchies in anthropogenic food-dependent species (Saalfeld *et al.* 2013). Although birds in landfills provide essential ecological services (Devault 2016), abnormally high populations of a few bird species with poor species richness might negatively affect the overall proportions of other species (Mills *et al.* 1989, Cam *et al.* 2000). It is vital to know how bird species adjust to their environments as well as how they help to keep ecosystems balanced and diverse (Sekercioglu 2012). With this knowledge, we can generate better conservation efforts and strategies for protecting bird populations and their homes (Maurice *et al.* 2020). Food subsidies from the dumpsites have multiple effects on bird communities, including behaviour changes and health impacts (Robb *et al.* 2008, Oro *et al.* 2013). Birds feed on organic foods like meat, fish, fruits, and eggs,

available in the dumpsites, which may give essential nutrients along the food chain (Parfitt *et al.* 2010). However, these foods can also carry toxins or harmful materials that might risk bird health and human safety.

Waste management includes physical destruction, chemical cleaning, and getting rid of trash in the environment. Since early human history, waste management has worked well with bones and shells because they decompose quickly (Steffo 1991). Birds do important jobs – they are bioindicators of pollution that help reduce waste while controlling pests in natural and modified spaces (Sekercioglu 2006). This study aims to fill a significant information gap about how Indian bird populations are affected by landfills and focuses explicitly on assessing the variety and quantity of bird species in the south-eastern part of West Bengal. The trash landfills are important food sources for local birds, and this study aims to ascertain this by examining the feeding guild structures of these bird populations. This study is anticipated to shed light on how landfills affect nearby avian populations, improving our comprehension of the ecological significance of these birds in this particular location.

Materials and Methods

Study area

This study was conducted in four randomly selected sites, i.e. Duttapukur Station Road market (22.768° N, 88.540° E) (transect I) and its associated dump area, Basirhat dumping ground (22.651° N, 88.872° E) (transect II), Barasat dumping ground (22.653° N, 88.873° E) (transect III) under the district of North 24 Parganas, and Bara Bazar station road and its associated dumpsites (22.589° N, 88.350° E) (transect IV) under the district of Kolkata. These places are used as waste disposal sites with regular dumping for various wastes, such as household garbage, sludge, market wastes, and biomedical wastes, which generate a variety of food sources for the birds. The average temperature and relative humidity of these areas varied between 20–42 °C and 60–80%, respectively, with an average annual rainfall of 150 mm.

Data collection

The field observations were conducted between February 2024 and June 2024. From each of the surveyed sites, the birds were observed using a modified point count survey method (Buckland *et al.* 1993, Bibby *et al.* 2000). Each site was surveyed at least twice a week, and the data on the bird abundance and richness were collected and noted. Olympus binoculars were used to observe birds, and digital photographs were taken using a Nikon B700 camera for the species identification to the smallest taxonomic class (Ali & Ripley 1995, Grimmett *et al.* 2011). The observations at the dumpsites were conducted mainly in the morning hours, and the activities of the birds were recorded. We noted direct sighting records for each site and identified various species at that time using the standard method

(Bibby *et al.* 1998). Alongside these observations, we also documented different types of foods available in the dumpsites and the feeding behaviours of birds. Additionally, we recorded the major dump materials, distinguishing between biodegradable and non-biodegradable items.

Data analysis

Following the field surveys, the abundance of birds and species richness was measured. The species diversity indices (i.e. the Dominance index, Simpson's diversity index, Shannon's diversity index, Evenness, Margalef's index, and Fisher's alpha) were determined using PAST software (version 4.03) (Hammer *et al.* 2001). Pie chart showing the proportional representation of different orders of birds and relative abundance of feeding guilds were prepared using the *ggplot2* package in R software (version 4.3.0) (Wickham 2016, R Core Team 2021).

Table 1. The list of the birds observed in the urban dumpsites, West Bengal, India, during the study period

1. táblázat A városi személtlerakókon feljegyzett madárfajok listája a vizsgálati időszakban, Nyugat-Bengálban (India)

Sl no.	Order	Family	Species Name	Scientific Name	Guild	Acronym
1	Cuculiformes	Cuculidae	Asian Koel	<i>Eudynamis scolopaceus</i>	Omnivore	ESC
2	Columbiformes	Cuculidae	Spotted Dove	<i>Spilopelia chinensis</i>	Granivore	SCH
3	Suliformes	Phalacrocoracidae	Little Cormorant	<i>Microcarbo niger</i>	Carnivore	MNI
4	Pelecaniformes	Ardeidae	Black-crowned Night Heron	<i>Nycticorax nycticorax</i>	Carnivore	NNY
5	Pelecaniformes	Ardeidae	Mallard	<i>Anas platyrhynchos</i>	Omnivore	APL
6	Pelecaniformes	Ardeidae	Pond Heron	<i>Ardeola grayii</i>	Carnivore	AGR
7	Pelecaniformes	Ardeidae	Cattle Egret	<i>Bubulcus ibis</i>	Carnivore	BIB
8	Pelecaniformes	Ardeidae	Intermediate Egret	<i>Mesophoyx intermedia</i>	Carnivore	AIN
9	Accipitriformes	Accipitridae	Black Kite	<i>Milvus migrans</i>	Carnivore	MMI
10	Coraciiformes	Alcedinidae	White-breasted Kingfisher	<i>Halcyon smyrnensis</i>	Carnivore	HSM
11	Passeriformes	Sturnidae	Jungle Myna	<i>Acridotheres fuscus</i>	Omnivore	AFU
12	Passeriformes	Sturnidae	Common Myna	<i>Acridotheres tristis</i>	Omnivore	ATR
13	Passeriformes	Corvidae	Jungle Crow	<i>Corvus culminatus</i>	Omnivore	CCU
14	Passeriformes	Corvidae	House Crow	<i>Corvus splendens</i>	Omnivore	CSP
15	Passeriformes	Dicruridae	Black Drongo	<i>Dicrurus macrocercus</i>	Aerial insectivore	DMA
16	Passeriformes	Sturnidae	Pied Starling	<i>Gracupica contra</i>	Omnivore	GCO
17	Passeriformes	Motacillidae	White Wagtail	<i>Motacilla alba</i>	Insectivore	MAL
18	Passeriformes	Passeridae	House Sparrow	<i>Passer domesticus</i>	Ground granivore	PDO

Results

During the study period, a comprehensive survey recorded a total of 18 bird species across 10 families and 7 orders (Table 1). House Crows (*Corvus splendens*) are quite uncommon in the Basirhat region, but they predominate in landfills in Barasat, Duttapukur and Kolkata. In the Basirhat region, Cattle Egrets (*Bubulcus ibis*) were predominate, but are non-existent elsewhere. Unexpectedly, the Basirhat region has a higher prevalence of Jungle Myna (*Acridotheres fuscus*) than Common Myna (*Acridotheres tristis*). Among the 7 orders identified, Passeriformes emerged as the most dominant order, accounting for 50.2% of the species observed. This order includes a significant number of species, showcasing its prevalence in the study area. Following Passeriformes, Pelecaniformes represented 28.7% of the species. The remaining orders were less diverse, each represented by only a few species (Figure 1). Among the families of birds encountered, the Ardeidae family was the most dominant, represented by five species, followed by Sturnidae with three species, indicating a moderate presence. Cuculidae and Corvidae, each represented by two species, contributed to a relatively small but notable diversity. The remaining six families were each represented by a single species, highlighting a broad yet uneven distribution of species among the various families.

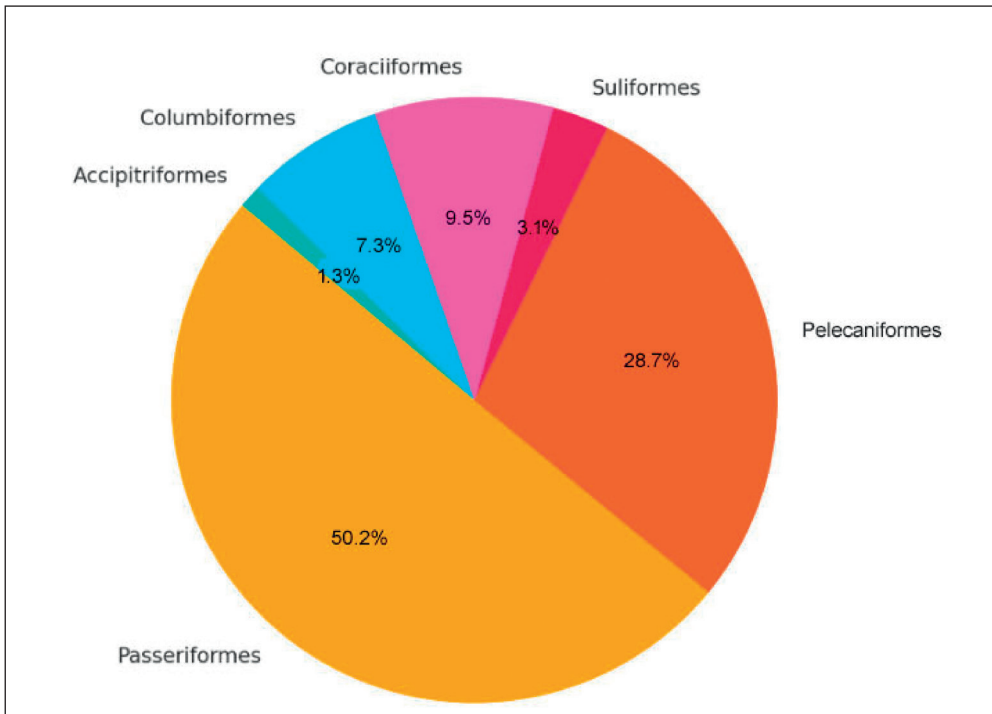


Figure 1. Pie chart showing proportional representation of different orders of birds observed in the waste landfill sites in West Bengal, India

1. ábra Nyugat-Bengál (India) területén felmért szemétklerakók madárrendjeinek arányát szemléltető kördiagram

The diversity indices, such as Simpson’s diversity index showed evenness, Shannon’s diversity index reflected community with many taxa, Margalef’s index revealed transects 1 and 2 are richer than the rest, and Fisher’s alpha confirmed this (Table 2). At the dumpsites, the bird census revealed four feeding guilds, i.e. carnivore, omnivore, granivore and insectivore (Figure 2). Of all the feeding guilds at the dumpsites, omnivores were the most prevalent, followed by carnivores, insectivores, and graminivores.

Table 2. Bird species diversity in the waste landfills in the 4 different transects (urban sites) of West Bengal, India

2. táblázat A szemétklerakók 4 transzektjének faji sokféleségét leíró mérőszámok

Diversity indices	Transect I (Duttapukur Station Road market area)	Transect II (Basirhat dumping ground)	Transect III (Barasat dumping ground)	Transect IV (Bara Bazar Station Road area)
Number of taxa	13	13	11	8
Number of individuals	461	369	345	812
Dominance index	0.168	0.2087	0.186	0.1851
Simpson’s diversity index	0.832	0.7913	0.814	0.8149
Shannon’s diversity index	2.076	1.79	1.902	1.829
Evenness	0.6131	0.4605	0.6092	0.7785
Margalef’s index	1.957	2.03	1.711	1.045
Fisher’s alpha	2.487	2.625	2.167	1.232

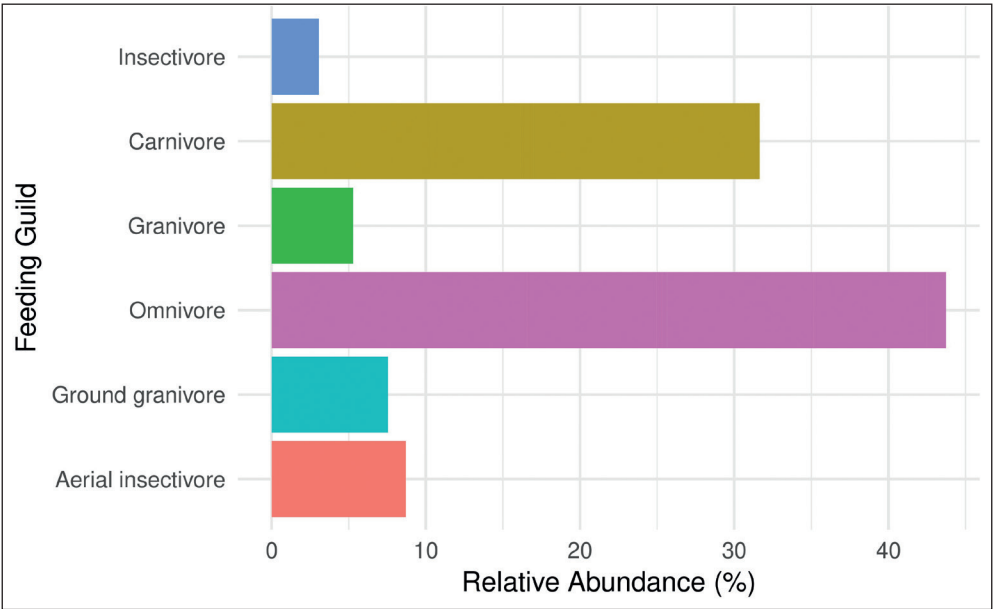


Figure 2. Relative abundance of feeding guilds in all the waste landfill sites observed in the selected urban regions of West Bengal, India

2. ábra Nyugat-Bengál (India) területén felmért szemétklerakókban megfigyelhető madarak táplálkozási guildjeinek relatív abundanciája

Discussion

The common perception of dumpsites as nothing more than environmental wastelands is a matter of debate following the earlier research on faunal diversity in these places. Instead, they highlight the intricate ecological dynamics at work, where a variety of bird species, especially scavengers and opportunists, find food and refuge. The implications of these discoveries for urban ecology, the advantages and disadvantages of using landfills as bird homes, and the larger picture of preserving urban biodiversity are covered in this regard. The present study centres on distinct species and their abundance at different dumpsites in and around Kolkata, West Bengal. The House Crow is the dominating species at two sites, while the Cattle Egret is the dominant species at two other sites. This opportunistic species can drive out other bird species from the area and is violent towards them. The research aims to examine how dominant species affect ecosystem dynamics and total biodiversity in these particular locations. Gaining knowledge about these dominating species' interactions and behaviour can be beneficial in determining the general stability and well-being of the local bird populations. The transect (II), where an agricultural area was closer, was predominated by Cattle Egrets, while House Sparrows (*Passer domesticus*) were prevalent across all sites. The results of the study demonstrate a diverse and rich avian community, with some orders and families demonstrating notable dominance. The results showed how dumpsites function as bird reservoirs, even though the dumpsites contain a variety of domestic hazardous wastes which may negatively affect the survival and conservation of several bird species. Food remains of all kinds can be found at dumpsites, including bread, fish, food grains, leftover meat, flowers, fruits, and packet meals, which draw different bird species. Matejczyk *et al.* (2011) and Abeba *et al.* (2020) found that non-biodegradable materials like metals, plastics, and paints in dumpsites lead to low bird species diversity and abundance. Plastic wastes and debris negatively impact aquatic and terrestrial bird species, affecting survival, growth, development, reproductive output, and physiology. The study recorded 18 bird species from dumpsites, which was much in line with the findings of Dutta *et al.* (2022), who identified 26 different bird species from the dumpsites of Kolkata. Variations in bird species richness may be due to food sources, human disturbances, climatic conditions, altitude, vegetation cover, water availability, urbanization, and habitat fragmentation.

Dumpsites in Barasat and Duttapukur have the highest species diversity indices, while the Basirhat transect has the highest species evenness. Variations in food availability, roosting locations, and human disturbances influence species diversity and evenness (Megersa *et al.* 2016, Haider *et al.* 2022). The type of food subsidy (Bertellotti *et al.* 2001), shelter (Burger 2001, Dementieieva *et al.* 2023), and mating season (Pons & Migot 1995, Tortosa *et al.* 2002) are some of the variables that affect the number of birds on landfills. Numerous endangered species, such as vultures, have also been observed to inhabit landfills (Houston *et al.* 2007), where they use food resources. Although more research is needed, the food subsidy is most likely the reason for the large number of birds at the garbage site. This study conducted at a landfill site highlighted the presence of several dominant bird species, offering new insights into avian ecology in such

unique habitats. Omnivores dominated all feeding guilds at the dumping locations, with carnivores, insectivores, and granivores following closely behind. The feeding guild distribution points to a wide variety of species using the resources at the waste site. The existence of omnivores, in particular, suggests that the bird populations in this habitat are highly adaptive and opportunistic (Pohajdak 1998).

The most dominant species observed at the landfill was the House Crow. This adaptable bird was frequently seen foraging across the landfill, utilizing different areas for feeding, perching, and flying. This bird's ability to thrive in such environments underscores its versatility and opportunistic behaviour in exploiting food resources provided by human activities. Cattle Egret was identified as the second most dominant species at the site. The prevalence of Cattle Egret at landfills has been well-documented globally, with studies by Plaza and Lambertucci (2017) confirming their widespread presence in similar environments. These birds are known for their scavenging habits and have become a common sight at waste disposal sites worldwide. The third most abundant species recorded was the Jungle Myna and Black Drongo (*Dicrurus macrocercus*). Unlike the Jungle Myna and House Crow, the Black Drongo is an insectivorous bird. Large numbers of drongos were observed mainly in the evening when they were drawn to the landfill by the swarms of flies that blanketed the site. The presence of this species in such high numbers at a landfill site is a notable finding, as their primary diet consists of insects, indicating that landfills can also support insectivorous birds (Annorbah & Holbech 2012). House Crows are quite uncommon in the Basirhat region, but they predominate in landfills in Barasat, Duttapukur, and Kolkata (Vuorisalo *et al.* 2003, Noreen & Sultan 2021). In the Basirhat region, Cattle Egrets predominate (Marasinghe *et al.* 2018), but they are non-existent in other urban dumpsites. Unexpectedly, the Basirhat region has a higher prevalence of Jungle Myna than Common Myna, which might be because the location is closer to the Sundarbans National Park (Dutta *et al.* 2022). Both biodegradable and non-biodegradable waste were present in the dumpsites. Crows, White Wagtails (*Motacilla alba*), and House Sparrows are found in the areas dominated by non-biodegradable wastes, while Jungle Myna, Pied Starling (*Gracupica contra*), and Black Drongo are prevalent in the organic dumps. These observations suggest that certain bird species are more adaptable to specific waste materials. Further research is needed to understand the behaviour of urban birds foraging in dump areas and their interactions with other co-existing species, which may provide insights into urban ecosystems and inform conservation efforts.

Conclusion

Birds in waste landfill areas face high risks of ingesting harmful materials like plastics, heavy metals, and other chemicals, which can lead to poisoning, organ damage, and long-term health issues, affecting the population. Further, their dependency on human-provided food can make them more vulnerable if natural food sources are scarce or removed or the dump is closed. Dumpsites also attract large numbers of birds, leading to increased competition for resources. Apparently, organic waste and discarded food items are abundant

in the waste dumps, allowing birds to sustain themselves in urban environments. However, the foraging bird community at waste landfill sites is characterized by low species diversity and dominance of a few species, such as Cattle Egrets and House Crows. Waste landfills are an emergent environmental problem requiring effective management, waste formation, and solutions to waste isolation.

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Factors influencing nest site selection of the Laughing Dove (*Spilopelia senegalensis*) in an urban area in Karaj, Iran

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Abstract Urban habitats are rapidly changing, making the conservation and management of species adapted to these environments challenging. Nest site selection is a pivotal point in the process of habitat selection and breeding for bird species. We measured 10 structural and spatial characteristics at 32 nest presence sites and 64 randomly selected nest absence sites of Laughing Doves (*Spilopelia senegalensis*) in an urban environment in Karaj, Iran. To model the nest site selection, we performed spline binary logistic regression. Two variables significantly influenced the nest site selection of Laughing Doves: nest height above ground, with an optimal range of ~286–347 cm, and nest concealment, favoring invisible places from the front and sides. Distance to opposite building, with an optimal range of ~16–34 m, was marginally significant. Additionally, we surveyed the feasibility of occupying artificial nestboxes (n = 17) by Laughing Doves between February and September 2019. The occupancy rate of the artificial nestboxes was ~35%. Overall, these results suggest that despite the relatively high structural heterogeneity of our urban environment, Laughing Doves exhibit distinct preferences for certain nest site characteristics, which may reflect a dependence on close proximity to humans for safety and access to food and water.

Keywords: human-animal interaction, nestbox, nest site preferences, spline, urban bird behavior

Összefoglalás A városi élőhelyek gyorsan változnak, ami miatt nagy kihívás az ilyen környezethez alkalmazkodott fajok megőrzése és kezelése. A fészkelőhely kiválasztása a madárfajok élőhelyválasztási és költési folyamatának egyik sarkalatos pontja. Vizsgálatunkban 10 strukturális és környezeti jellemzőt mértünk meg a pálmagerle (*Spilopelia senegalensis*) esetében 32 fészkes és 64 véletlenszerűen kiválasztott fészkek nélküli helyen az iráni Karajban. A fészkelőhely-választás modellezésére spline bináris logisztikus regressziót alkalmaztunk. Két változó befolyásolta szignifikánsan a pálmagerlek fészkelőhely-választását: a fészkek földtől számított magassága (optimális tartománya ~286–347 cm volt), és a fészkek rejtettsége (a szemből és oldalról láthatatlan helyeket részesítette előnyben). A szemközti épülettől való távolság, amelynek optimális tartománya ~16–34 m volt, csak kis mértékben volt rá hatással. Emellett megnéztük a mesterséges fészkelőládák (n = 17) pálmagerlek általi elfoglalásának valószínűségét is 2019 februárja és szeptembere között. A mesterséges fészkelő ládák elfoglalási aránya ~35% volt. Összességében ezek az eredmények arra utalnak, hogy a városi környezet viszonylag nagy szerkezeti heterogenitása ellenére a pálmagerlek bizonyos fészkelőhelyi jellemzőkkel szemben határozott preferenciákat mutatnak, ami azt tükrözheti, hogy a biztonság, valamint a táplálékhoz és vízhez való hozzáférés szempontjából függnek az emberek közelségétől.

Kulcsszavak: ember-állat interakció, fészkelő láda, fészkelőhely preferencia, spline, városi madarak viselkedése

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Introduction

Comprehending the determinants of nest site selection is essential for advancing avian conservation and management strategies. Nest site selection is an important step in the process of habitat selection and territory establishment in bird species (Suvorov *et al.* 2014). The availability of suitable nesting sites directly impacts reproductive success, making it a critical component of habitat selection (Li & Martin 1991, Jara *et al.* 2020). Identifying the factors that drive nest site selection can enhance our understanding of species' ecological requirements and improve habitat management practices (Clark & Shutler 1999).

The urban space is a permanently changing ecosystem, suffering from decreasing biodiversity, but also providing new anthropogenic habitats for some adaptable species (Sumasgutner *et al.* 2014). The availability of nest sites in native and non-native vegetation, residential houses, uninhabited buildings, and various anthropogenic structures such as metal pipes and bridges significantly influences birds' nesting behaviors in cities (Reynolds *et al.* 2019). Additionally, the provision of nestboxes has been shown to support certain bird species in urban areas (Reynolds *et al.* 2019, Dulisz *et al.* 2022).

The Laughing Dove (*Spilopelia senegalensis*) is one such species that has successfully adapted to urban habitats. Native to most parts of Africa, the Middle East, and South and Central Asia, it has also been introduced to parts of Western Australia (Frith *et al.* 1976, Baptista *et al.* 1997) and Europe (e.g. Rocha 2013, Zannetos *et al.* 2023). This species is commonly found in cultivated areas, trees (but not forests), and human habitations (BirdLife International 2019). The nest is a frail, thin platform of roots, twigs and petioles placed in bushes or trees up to 15 m above the ground (Baptista *et al.* 1997), on buildings under the eaves, on drainpipes or in cracks in walls (Snow & Perrins 1998), and is ~15 cm in diameter (Schodde *et al.* 1986). This species predominantly forages on the ground but sometimes searches for food in small trees and shrubs (Gibbs *et al.* 2010). Its diet mainly consists of fallen seeds from grasses, along with vegetable debris, fruits, nectar, succulent shoots, some insects, and even human food waste (Satheesan *et al.* 1990, Adang *et al.* 2008, Gibbs *et al.* 2010, Hanane 2015).

The presence of Laughing Doves in urban landscapes can be both beneficial and challenging. While these birds contribute to the aesthetic appeal of parks and green spaces, they also may pose health risks and create disturbances in residential areas. A study conducted on the effect of wild bird droppings as a source of *Campylobacter jejuni* in children's playgrounds showed that the presence of wild birds, including Laughing Doves, can cause acute gastroenteritis in humans, especially in children (Abdollahpour *et al.* 2015). In addition, Laughing Doves may infect humans and other animal species by transmitting Newcastle disease virus (Okpanachi *et al.* 2020, Hirschinger *et al.* 2021), West Nile virus, and Usutu virus (Ayadi *et al.* 2017). Therefore, armed with a deeper understanding of the characteristics of Laughing Dove nest sites, we can strategically modify certain locations to discourage their nesting.

Birds in urban areas consider various structural and environmental factors when selecting nest sites, such as building structure, nest tree height, proximity to green spaces, tree coverage, and distance from water sources (e.g. Jokimäki *et al.* 2017, Bressler *et al.* 2020, Ding *et al.* 2023). However, previous studies on the nest site selection of Laughing Doves have primarily focused on their tree nesting habits, particularly in agricultural areas of

Northern Africa (e.g. Boukhriss & Selmi 2009, 2019, Brahmia *et al.* 2015, Hanane 2015), leaving a gap in our understanding of their nest site selection in urban environments.

In the present study, we measured 10 structural and spatial characteristics that we hypothesized could influence Laughing Dove nest site selection. We then modeled the nest site selection to identify statistically significant factors. In addition, we examined the feasibility of Laughing Doves occupying artificial nestboxes as another objective of this study.

Methods

Study area

A portion of Karaj, situated in the Alborz province of Iran, was selected for this study (35°49'36N", 50°58'6"E; 239 ha) (*Figure 1*). Karaj is a densely populated city nestled at the foot of the Alborz mountain range, within the Palearctic realm. It has a metro area population of ~1.59 million inhabitants and a population density of ~9,815 people/km² (Macrotrends 2023). The study area, located in the central part of Karaj, encompassed main streets, side streets, and alleys. Most of the buildings were multi-story apartments, typically three or four stories high. Pedestrian traffic was more pronounced on the main streets and side streets compared to the alleys. Approximately 11% of this area was designed as green space, and the altitude varied from ~1,297–1,355 m above sea level, resulting in a gentle slope (Google 2020).

The prevailing climate in Karaj is known as a local steppe climate, with a mean annual temperature of 14.2 °C (maximum mean temperature of 32.8 °C in summer and minimum mean temperature of –4.6 °C in winter) and precipitation of ~283 mm per year. Although it varied greatly throughout each season, the driest season was summer (4 mm), whereas the rainiest season was winter (127 mm) (Climate Data 2022).

The prominent animal species observed in the study area included Laughing Doves, Rock Doves (*Columba livia*), House Sparrows (*Passer domesticus*), White-eared Bulbuls (*Pycnonotus leucotis*), White Wagtails (*Motacilla alba*), free-ranging dogs (*Canis lupus familiaris*), Eurasian Magpies (*Pica pica*), Hooded Crows (*Corvus cornix*), rats (*Rattus* spp.), and stray cats (*Felis catus*). The last four species are known as potential nest predators (e.g. Shoham *et al.* 1997, Jokimäki & Huhta 2000, Vincze *et al.* 2017, Boukhriss & Selmi 2019). Common tree species included narrow-leafed ash (*Fraxinus angustifolia*), oriental plane (*Platanus orientalis*), Arizona cypress (*Cupressus arizonica*), black locust (*Robinia pseudoacacia*), and Afghan pine (*Pinus eldarica*) (all based on our observations).

Data collection

While the breeding season of Laughing Doves varies across different regions (Baptista *et al.* 1997, Gibbs *et al.* 2010, BirdLife International 2019), our prior observations and existing studies (e.g. Boukhriss & Selmi 2009, Brahmia *et al.* 2015) led us to assume late February as the onset of the breeding season in our study area. From 25 February to 3 April 2019,

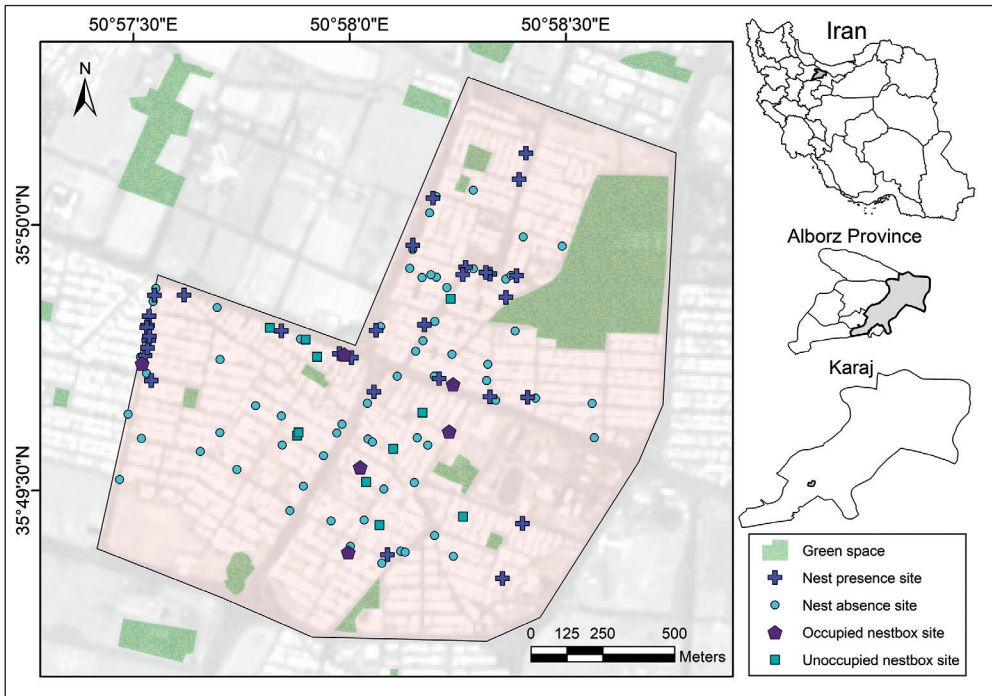


Figure 1. Study area in Karaj, Alborz province, Iran, encompassing 239 ha. Green spaces (green polygons), Laughing Dove nest presence sites (purple crosses, $n = 32$), nest absence sites (cyan circles, $n = 64$), occupied nestbox sites (dark purple pentagons, $n = 6$), and unoccupied nestbox sites (teal squares, $n = 11$) are highlighted. Contains modified Copernicus Sentinel data processed by Sentinel Hub (2022)

1. ábra A vizsgálati terület (239 ha) térképe Karajban, Alborz tartomány, Irán. Zöld területek (zöld poligonok), pálmagerle fészkelő-helyek (lila keresztek), fészeklésre nem használt helyek (kék körök), fészeklésre használt költőládák (sötétlila ötszögek) és üres költőládák (kékes-zöld négyzetek) jelölve vannak

we searched both sides of the streets and alleys, alternating between going and returning on foot from morning to afternoon, ensuring each side was searched once. Moreover, from 29 February to 15 March 2019, we directly approached 82 local residents from our own neighborhood, familiar to us, asking them to report any Laughing Dove nests they observed in the study area. Additionally, we requested that they encourage others to do the same. As a result, we believe that even more individuals indirectly contributed to our study. Based on the assumption that the abandoned nests were built in favorable places by this bird, we also recorded and accounted for them as nest presence sites. The tools and devices used in sampling included a tape measure, a Global Positioning System (GPS) device from Garmin® (USA), a sliding ladder, a compass, and a closed-circuit television (CCTV). Until 3 April 2019, we recorded 32 nest presence sites, 10 of which (31.2%) were found by local residents, and 64 randomly selected nest absence sites (Figure 1).

Ten structural and spatial variables were measured in this study (Table 1). The vertical distance from the ground to the base of the nest (HEIGHT) was measured using a measuring

Table 1. Variables measured to model the nest site selection of Laughing Dove in 2019.^aDiameter at breast height. ^bA: invisible from the front and sides; B: invisible from the front but visible from one side; C: visible from the front but invisible from the sides

1. táblázat A pálmagerle fészkelőhely-választásának modellezéséhez felmért változók. ^aMellmagaságban mért átmérő. ^bA: sem szemből, sem oldalról nem látható, B: egyik oldalról látható, de szemből nem, C: csak szemből látható

Variable	Unit	Abbreviation	Description
Nest height above ground	cm	HEIGHT	The vertical distance from the ground to the base of the nest
Nest to ceiling height	cm	CEILING	The vertical distance from the ceiling to the base of the nest
Nest horizontal distance to ceiling edge	cm	H_CEILING	The horizontal distance from the edge of the nest to the edge of the ceiling
Distance to nearest tree	m	TREE_DIST	The horizontal distance from the nest to the nearest tree > 10 cm dbh ^a
Distance to opposite building	m	BUILD_DIST	The horizontal distance from the nest to the opposite building
Distance to green space	m	GREEN_DIST	The distance from the nest to the edge of the nearest green space (area > 2 km ²)
Distance to nearest counterpart nest	m	NEST_DIST	The distance from the nest to the nearest counterpart nest
Nest concealment	#	CONCEAL	Visibility of the nest from a distance of 4 m along its horizontal plane; categorized as A, B, or C ^b
Base structure	#	B_STRUCT	A: shop boards, B: building frontispieces, C: building balconies
Base aspect	#	B_ASPECT	A: North, B: East, C: South, D: West

tape positioned directly below the nest. Similarly, the vertical distance from the ceiling to the base of the nest (CEILING) was measured with a measuring tape, with one end placed at the base of the nest and the other at the ceiling. The horizontal distance from the edge of the nest to the edge of the ceiling (H_CEILING) was also measured using a measuring tape held horizontally. The horizontal distances from the nest to the nearest tree with a diameter at breast height (DBH) greater than 10 cm (TREE_DIST) and to the nearest opposite building (BUILD_DIST) were both measured by counting steps to estimate the distances. The distances from the nest to the edge of the nearest green space (area > 2 km²; GREEN_DIST) and to the nearest counterpart nest (NEST_DIST) were both measured using a combination of GPS coordinates and mapping software. The nest concealment (CONCEAL) was assessed visually by us. We imagined our eyes were ~4 m horizontally away from the nest, in the opposite and side directions of the base structure, because it was often not possible to physically reach those viewpoints, especially when they were at a high height or when placing our sliding ladder on the street was required. The nest concealment was then categorized as A (invisible from the front and sides), B (invisible from the front but visible from one side), or C (visible from the front but invisible from the sides). The base structure of the nest (B_STRUCT) was categorized as A (shop boards), B (building

Statistical methods

Given that the influence of a continuous predictor variable on the probability of the occurrence of the binary response variable can be nonlinear, we employed the spline binary logistic regression method, specifically using linear splines, to model the nest site selection of the Laughing Dove. In this approach, the continuous predictor variable is divided into bins using knots. Consequently, we can assess any nonlinear effects of the continuous predictor variable (Harrell 2015). By utilizing the spline method, we address the issue of loss of information that arises when categorizing continuous variables (Weinberg 1995, Schuster *et al.* 2022).

We performed a knots-placing procedure based on different numbers of quantiles (2–5 quantiles; 2 quantiles = 1 knot, 5 quantiles = 4 knots) for all continuous predictor variables, using the dataset of nest presence sites ($n = 32$). To determine the most appropriate number of knots (or even no knots) for each continuous predictor variable, we conducted univariate logistic regressions for each predictor, varying the number of knots (up to four knots and no knots). Subsequently, we selected the number of knots that provided the equation with the lowest Akaike Information Criterion (AIC) value to enter into a model selection process (Harrell 2015). Since stepwise selection methods can lead to biased estimates and unreliable p-values due to their tendency to overfit models and fail to account for model uncertainty (Thayer 2002, Whittingham *et al.* 2006, Flom & Cassell 2007), we instead employed an ‘all subsets’ approach to systematically evaluate all subsets of predictor variables (Berk 1978), selecting the best model based on the lowest AIC (Akaike 1974). This method involves generating and comparing all possible combinations of predictor variables to find the model that best balances goodness-of-fit and complexity. The first level of each categorical predictor variable (A) was set as the reference level.

To identify potential correlations between/among variables and separate them into clusters that could be interpreted as single variables, we performed hierarchical variable clustering (cutoff distance value = 0.3) (Chavent *et al.* 2012). We then employed density ridgeline visualization to depict the distribution of a correlated continuous variable, nest height above ground, across various categories of a corresponding categorical variable, base structure (Wilke 2019), to interpret the relationship between different base structures and the heights.

The accuracy of the selected model (the nest site selection model) was evaluated using the area under the curve (AUC) of the receiver operating characteristic (ROC) curve (Swets 1988, Fielding & Bell 1997). The AUC was calculated based on the fitted probabilities from the model, which utilized the combined dataset of nest presence and absence points. This approach measures the model’s ability to distinguish between presence and absence data, with AUC values ranging from 0 to 1 (Phillips & Dudík 2008). A model whose predictions are 100% wrong has an AUC of 0, while a model with perfect predictions has an AUC of 1.

Due to our small sample size ($n = 96$), we used all observations from the combined nest presence and nest absence data to develop the nest site selection model. To validate the model, we employed an ‘external validation’ approach (Terrin *et al.* 2003). We used the dataset of nestbox sites ($n = 17$) as the test dataset, inputting this dataset into the nest site selection model, which generated predictions. These predictions were then

summarized and compared to the actual observed outcomes using a classification table. To determine the optimal threshold value for classification, we calculated the ROC curve for the dataset of nestbox sites using the predicted probabilities and actual outcomes. The optimal threshold was identified by maximizing Youden's J statistic, which balances sensitivity and specificity (Youden 1950, Perkins & Schisterman 2006). We assessed global spatial autocorrelation in the nest site selection model by calculating Moran's I test on the residuals, utilizing a spatial weights matrix derived from inverse distance weighting (IDW) that was based on Euclidean distance (Moran 1950, Legendre & Fortin 1989, Panigrahi 2021, Milliet *et al.* 2024).

All analyses were conducted in R v4.4.0 (R Core Team 2024). The rms R package (Harrell 2024) was used to perform spline binary logistic regression analysis. Receiver operating characteristic (ROC) analyses, along with the calculation of the optimal threshold for classification, were performed using the pROC R package (Robin *et al.* 2011). Visualization of descriptive statistics and comparisons of occupied versus unoccupied artificial nestboxes was conducted using the ggplot2 R package (Wickham 2016). Hierarchical variable clustering was performed using the ClustOfVar R package (Chavent *et al.* 2017). Additionally, the density ridgeline visualization was created using the ggridges R package (Wilke 2024). Spatial autocorrelation was assessed using the spdep R package (Bivand 2022). We utilized ArcGIS v10.2 (ESRI 2013) for map-related processes, including calculating the distance to the nearest counterpart nest and the distance to the nearest green space. Furthermore, we employed SketchUp Pro v15.2 (Trimble Navigation Limited 2014) to design the nestboxes.

Results

For the nest presence sites ($n = 32$), the median nest height above ground (HEIGHT) was 323.5 cm, ranging from 273–1,230 cm. The median nest to ceiling height (CEILING) was 29.5 cm, ranging from 16–138 cm. Nest horizontal distance to ceiling edge (H_CEILING) had a median of 104.0 cm, ranging from 18–197 cm. Distance to nearest tree (TREE_DIST) had a median of 8.0 m, ranging from 4–31 m. The median distance to opposite building (BUILD_DIST) was 19.0 m, ranging from 9–50 m. Distance to green space (GREEN_DIST) had a median of 177.75 m, ranging from 19.3–312.6 m. Distance to the nearest counterpart nest (NEST_DIST) had a median of 86.45 m, ranging from 7.9–408.8 m. Nest concealment (CONCEAL) was predominantly in category A (invisible from the front and sides, $n = 24$), followed by categories B (invisible from the front but visible from one side, $n = 4$) and C (visible from the front but invisible from the sides, $n = 4$). Base structure (B_STRUCTURE) was mostly in category A (shop boards, $n = 26$), with fewer nests in category B (building frontispieces, $n = 2$) and C (balconies, $n = 4$). Base aspect (B_ASPECT) was evenly distributed among categories A (North, $n = 10$) and D (West, $n = 10$), with fewer nests in categories B (East, $n = 7$) and C (South, $n = 5$) (*Appendices 1, 2a–j*).

Following the knots-placing procedure and univariate logistic regressions, the most appropriate number of knots was determined to be two for three variables: nest height above

Table 2. Summary of the nest site selection model for the Laughing Dove. The table presents the estimated coefficients (β), standard errors (SE), Wald z values, and p-values ($\text{Pr} > |z|$) for each predictor variable, including their spline terms or categories. The overall effect of each variable is assessed using the Wald statistics (χ^2), degrees of freedom (df), and p-values

2. táblázat A pálmagerle fészkelőhely-választásának modell eredményei. A táblázat tartalmazza a becsült modell paramétereket (β), azok sztenderd hibáját (SE), Wald z-értékeket és p-értékeket minden magyarázó változóhoz. Az egyes változók hatását Wald-féle statisztikával (χ^2) határoztuk meg, feltüntetve a szabadsági fokokat és p-értékeket

Variable	β	SE	Wald z	$\text{Pr}(> z)$	Wald Statistics (Overall Effect)		
Intercept	-36.567	11.160	-3.28	0.001	χ^2	df	p
HEIGHT (<313cm)	0.099	0.031	3.21	0.001	11.53	3	0.009
HEIGHT' (313–367cm)	-0.172	0.052	-3.34	0.001			
HEIGHT'' (>367cm)	0.074	0.025	2.93	0.003			
CONCEAL = B	-1.463	0.833	-1.76	0.079	6.87	2	0.032
CONCEAL = C	-1.815	0.787	-2.31	0.021			
BUILD_DIST (<16m)	0.211	0.207	1.02	0.309	7.09	3	0.069
BUILD_DIST' (16–25m)	-0.111	0.283	-0.39	0.695			
BUILD_DIST'' (>25m)	-0.211	0.143	-1.47	0.141			
CEILING (<24cm)	0.263	0.167	1.57	0.116	6.07	3	0.109
CEILING' (24–49cm)	-0.397	0.203	-1.96	0.050			
CEILING'' (>49cm)	0.156	0.066	2.36	0.018			
TREE_DIST	0.117	0.078	1.51	0.131	2.28	1	0.131

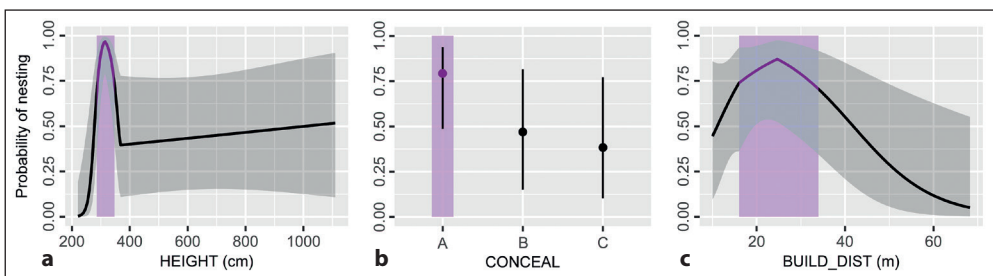


Figure 3. Influence of two significant variables and one marginally significant variable on the probability of Laughing Dove nesting based on the predictions of the nest site selection model (adjusted to the median of other predictor variables). Probability values greater than 0.7 are highlighted using purple bands for the following variables: (a) nest height above ground (HEIGHT), (b) nest concealment (CONCEAL), and (c) distance to opposite building (BUILD_DIST)

3. ábra A két szignifikáns és egy közel szignifikáns változó hatása a pálmagerle fészkelési valószínűségére a fészkelőhely-választás modellje alapján. A 0,7-nél nagyobb valószínűség értékek lila sávval vannak jelölve a következő változók esetében: (a) a fészek talaj feletti magassága (HEIGHT), (b) fészek rejtettsége (CONCEAL) és (c) a szemköztí épület távolsága (BUILD_DIST)

ground, nest to ceiling height, and distance to opposite building. For the other variables, it was zero (no knots). After assessing the AIC for all possible combinations of predictor variables ($n = 1,023$), the selected model was constructed using five predictors: two significant variables, nest height above ground ($\chi^2 = 11.53$, $df = 3$, $p = 0.01$) and nest concealment ($\chi^2 = 6.87$, $df = 2$, $p = 0.03$); a marginally significant variable, distance to opposite building ($\chi^2 = 7.09$, $df = 3$, $p = 0.07$); and two non-significant variables, nest to ceiling height ($\chi^2 = 6.07$, $df = 3$, $p = 0.11$) and distance to nearest tree ($\chi^2 = 2.28$, $df = 1$, $p = 0.13$) (Table 2).

According to the nest site selection model, the optimal range for nest height above ground (with probability values adjusted to the median of other predictor variables > 0.7) was ~286–347 cm (Figure 3a). Regarding nest concealment, Laughing Doves preferred places that were invisible from the front and sides (probability value = 0.79) (Figure 3b). Additionally, the optimal range for distance to opposite building, although marginally significant, was ~16–34 m (Figure 3c).

Following the implementation of hierarchical variable clustering, a strong correlation was observed between the two variables, nest height above ground and base structure, with a distance value of 0.15 (Appendix 3). The density ridgeline plot revealed that the category of shop boards within the base structure variable had a higher density within the optimal range of the nest height above ground variable (Appendix 4).

The nest site selection model demonstrated good performance with an area under the ROC curve (AUC) of 0.91. The optimal threshold for classification was found to be 0.52,



Figure 4. An artificial nestbox erected on a balcony, occupied by Laughing Doves. The picture was captured from CCTV footage on 11 May 2019.

4. ábra Kihelyezett költőláda egy erkélyen, amelyben pálmagerle költ. A kép a CCTV felvételéből készült 2019. május 11-én

with the classification table revealing that the nest site selection model correctly predicted 76.5% of the artificial nestbox site cases (*Appendix 5*). We observed no indication of spatial autocorrelation (Moran's $I = 0.01$, $z = 0.88$, $p = 0.19$).

As of 22 September 2019, six of the 17 artificial nestboxes erected were occupied by Laughing Doves (*Figure 1*). Four nestboxes were occupied in the vicinity of the shops, and two on the balconies of the buildings (*Figure 4, Appendix 6a–j*).

Discussion

Our findings revealed that Laughing Doves prefer nesting at a specific height above the ground. Additionally, nest concealment is a significant factor, with Laughing Doves favoring sites that are invisible from the front and sides. The distance to the opposite building is also marginally significant, with an optimal range identified. These results suggest that despite the structural heterogeneity of our urban study area, Laughing Doves exhibit distinct preferences for certain nest site characteristics. Furthermore, the feasibility of occupying artificial nestboxes has been confirmed.

Given the wide range of Laughing Dove nest heights – spanning from 273–1,230 cm – and the presence of outliers, we believe that considering the optimal range obtained from the nest site selection model is more informative than relying solely on the mean value, with the median being the next best option. The optimal nesting height – which fell within the range of ~286–347 cm – was slightly higher than the mean nest height reported for this species in most studies. Hanane (2015) found a mean height of 2.29 ± 0.05 m in an agricultural area in Tadla, Morocco. Boukhriess and Selmi (2019) observed a mean height of 2.59 ± 0.09 m in a grove in the oasis of Kettana, Tunisia, with a range of 0.80–5.98 m. Almalki (2023) documented a mean height of 2.74 ± 0.10 m, with a range of 1.05–5.20 m, around Taif City, Saudi Arabia. However, it was ~1 m lower than the mean nest height reported by Brahmia *et al.* (2015) in an olive orchard in the Guelma region of Algeria, which was 4.21 ± 0.08 m, with a range of 2.94–5.76 m. According to Jennings (2010), Laughing Dove nests in Saudi Arabia had heights ranging from 30 cm to 12 m, with one nest found at a height of 25 m on a building. Despite the structural variability in urban environments, we infer that Laughing Doves likely exhibit preferences for specific nesting heights.

The strong correlation between the two variables of nest height above ground and base structure, along with the high density of the category of shop boards across the optimal range of nesting height, may indicate the desirability of shop boards for Laughing Dove nesting. Shoham *et al.* (1997) reported that in the busy streets of Tel Aviv, Israel, Laughing Dove nests were often built on ledges above shops, often less than 1 m above people's heads. We speculate that shop boards, in addition to having structural advantages such as suitable height and shelter, may also provide security and food for Laughing Doves because of high pedestrian traffic around them.

Our nest site selection model suggested that nest concealment was a significant factor, and Laughing Doves may have preferred places that were less likely to be easily detected by potential nest predator birds. Similar patterns have been observed in other bird species. For

instance, the Eurasian Collared Dove (*Streptopelia decaocto*) in Spanish dehesas preferred specifically the holm oak evergreen tree (*Quercus ilex*), which provides concealment (Bermúdez-Cavero *et al.* 2021). Additionally, Kövér *et al.* (2015) documented that, although Hooded Crows in an urban environment in Eastern Europe preferred taller trees for nesting, they chose more concealed trees in areas with shorter trees. Concealed nests may reduce nest predation by minimizing the transmission of visual, chemical, and auditory cues to predators (Martin 1993, Burhans & Thompson 1998).

The distance to the opposite building, which itself depended on whether the nests were on main streets, side streets, or alleys, emerged as a marginally significant factor influencing the nest site selection of Laughing Dove. We infer that Laughing Doves may prefer busy main streets with abundant shops and high pedestrian traffic for nesting. This preference is likely influenced by the availability of food scraps left by people, particularly around bakery shops, and the perceived security provided by the constant human presence. In contrast, Bermúdez-Cavero *et al.* (2021) reported a completely opposite behavior for Eurasian Collared Doves in urban and periurban areas of eastern Spain. These doves avoided areas with heavier pedestrian traffic, which coincide with buildings, restaurants, and commercial centers, preferring green spaces instead. Rao and Koli (2017) reported that spatial heterogeneity, less predation, optimal feeding ground, and a higher number of advertising and display boards of shops at roadside buildings might be crucial factors for birds' nesting in the highly disturbed city of Udaipur, Rajasthan, India. Our observations align with these hypotheses, at least for the urban-adapted Laughing Dove population in our study area.

All our observed Laughing Dove nests were situated in close proximity to pedestrian traffic, and remarkably, these birds seemed to make minimal effort to conceal themselves or their nests from human view. To investigate this behavior further, we erected two artificial nestboxes in uninhabited buildings located far away from human activity, yet none of these boxes were occupied. However, we are aware that only two artificial nestboxes are too few to test this hypothesis. Hanane (2015) reported a relatively short distance between Laughing Dove nests and human settlements in the agricultural regions of Central Morocco. The Laughing Dove's reliance on proximity to human settlements, both in urban and agricultural environments, could be attributed to its relatively short wingspan, which may limit its ability to fly long distances in search of water and food (Shoham *et al.* 1997). Furthermore, Sumasgutner *et al.* (2014) indicated a trade-off between higher nest site availability in central Vienna, Austria, and longer distances to larger open green spaces as optimal foraging grounds for the Eurasian Kestrel (*Falco tinnunculus*). In our study area, nest site availability and optimal foraging grounds may co-locationally affect nest site selection in the Laughing Dove.

Some people, especially shopkeepers, were supplementary feeding birds mainly with wheat or millet seeds. We specifically observed Laughing Doves, House Sparrows, and Rock Doves foraging on the sidewalks. In contrast, potential nest predators of the Laughing Dove, such as Eurasian Magpies and Hooded Crows, are typically regarded as ominous and nuisance by humans due to ancient cultural beliefs (Clucas & Marzluff 2012, Król & Hernik 2020). As human behavior directed at birds can affect their behavior (Clucas & Marzluff 2012), this differential treatment of humans may have influenced nest site selection in the Laughing Dove.

Given the potential for Laughing Doves to be perceived as nuisances or carriers of pathogenic agents for both humans and other animal species (Abdollahpour *et al.* 2015, Ayadi *et al.* 2017, Okpanachi *et al.* 2020, Hirschinger *et al.* 2021), understanding the characteristics of their nest sites becomes crucial. By identifying preferred nesting locations, urban planners and conservationists can take targeted measures to manage and mitigate potential damages. Strategies may include discouraging nesting in certain areas or providing alternative nesting sites away from sensitive locations. Balancing the benefits of having these birds in urban spaces with the need to minimize health risks requires a thoughtful approach that considers both human well-being and wildlife conservation (Hedblom *et al.* 2017).

In light of the fact that ~35% of artificial nestboxes were occupied by Laughing Doves, it becomes evident that designing and erecting such nestboxes is a practical approach for managing, researching, or even providing recreational opportunities for this species (Reynolds *et al.* 2019, Dulisz *et al.* 2022). Notably, as small shops give way to shopping malls, the modern infrastructure often lacks suitable nesting sites for Laughing Doves (according to our observations). Unlike their more traditional counterparts, mall shopkeepers may deliberately prevent bird nesting when designing shop boards and facilities. Additionally, Clucas and Marzluff (2012) found an inverse correlation between urbanization gradient (light to heavy urbanization) and bird feeding by humans. This finding suggests that as urban areas become more densely populated, people's indifference or even dissatisfaction toward bird-related activities may increase. Consequently, artificial nestboxes emerge as valuable tools for the management and conservation, especially for future generations of this species (Reynolds *et al.* 2019, Dulisz *et al.* 2022).

Previous studies on the Laughing Dove (e.g. Kluijver 1966, Boukhriss & Selmi 2009, 2019, Brahmia *et al.* 2015, Hanane 2015) and similar columbid species (e.g. Csathó & Bozó 2022, Ramírez-Albores *et al.* 2024) have extensively documented their nesting habits in trees. Csathó and Bozó (2022) even showed a competition for nesting trees between the Eurasian Collared Dove and the Common Wood Pigeon (*Columba palumbus*) in an urban region in Southeast Hungary. When there were trees in the vicinity of almost all observed nests, and there were green spaces in our study area, we observed no Laughing Dove nests or nesting attempts on trees. We speculate that the avoidance of tree nesting by Laughing Doves may stem from factors such as the absence of suitable trees, diminished nest concealment, and consequently, an elevated risk of nest predation by Eurasian Magpies, Hooded Crows, and stray cats. Extending the study area, ideally across an urban-rural gradient, might help address this issue.

Boukhriss and Selmi (2019) found that the presence of palm trees near Laughing Dove nest trees increased the risk of predation by nest predators, particularly Black Rats (*Rattus rattus*), which used palm trees as refuges. Shoham *et al.* (1997) also considered nest predation by various species, including Hooded Crows, to be the main cause of Laughing Dove nesting failure. In our study, despite the usual nesting of potential nest predators such as Eurasian Magpies and Hooded Crows in urban green spaces (Dupak & Telizhenko 2023), the distance to the green spaces did not significantly influence the Laughing Dove's nest site preferences. This leads us to question: Does the distance of Laughing Doves' nests from their potential nest predators affect their reproductive success? And if so, does their reproductive success impact their nest site selection?

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Data availability

The data supporting the findings of this study are available in Dryad at <https://doi.org/10.5061/dryad.gxd2547rk>.

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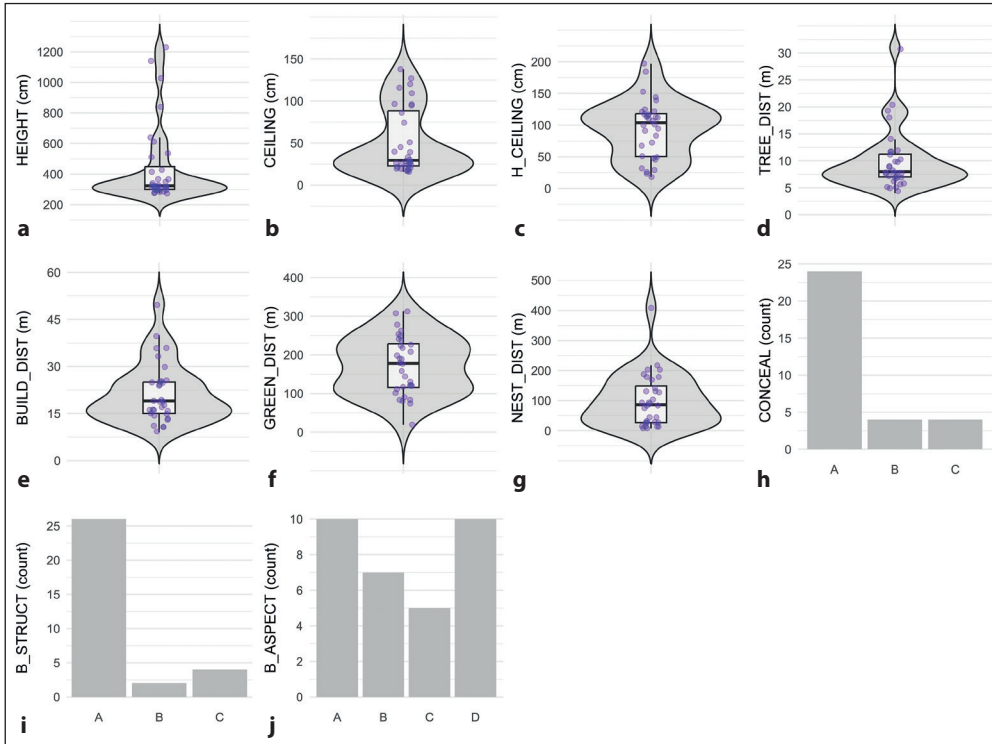
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Appendix 1. Descriptive statistics table of the Laughing Dove nest presence sites in 2019 (n = 32). For continuous variables, the table includes mean, standard deviation (SD) of the mean, median, minimum (Min), and maximum (Max) values. For categorical variables, the table shows the count (n) for each category: nest concealment (A: invisible from the front and sides, B: invisible from the front but visible from one side, C: visible from the front but invisible from the sides), base structure (A: shop boards, B: building frontispieces, C: balconies), and base aspect (A: North, B: East, C: South, D: West)

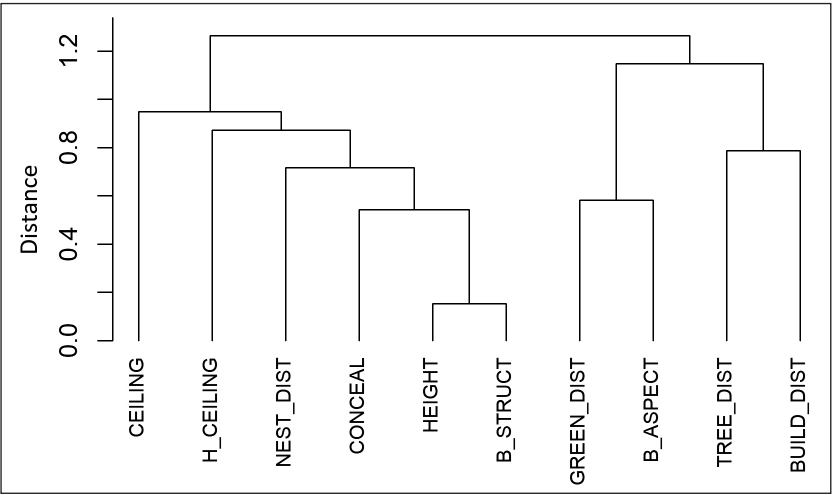
1. melléklet A pálmagerle fészkelőhelyeinek leíró mérőszámai 2019-ből. A folytonos változók esetén az átlag, a medián, a sztenderd hiba, a minimum és a maximum, míg a kategorikus változók esetén a darabszám van feltüntetve

Continuous Variables	Mean	SD	Median	Min	Max
Nest height above ground (cm)	444.91	259.03	323.50	273	1,230
Nest to ceiling height (cm)	51.62	39.37	29.50	16	138
Nest horizontal distance to ceiling edge (cm)	94.31	46.01	104.00	18	197
Distance to nearest tree (m)	9.91	5.52	8.00	4	31
Distance to opposite building (m)	21.44	9.52	19.00	9	50
Distance to green space (m)	172.10	75.28	177.75	19.3	312.6
Distance to nearest counterpart nest (m)	100.14	87.86	86.45	7.9	408.8
Categorical Variables	A (n)	B (n)	C (n)	D (n)	
Nest concealment	24	4	4	N/A	
Base structure	26	2	4	N/A	
Base aspect	10	7	5	10	



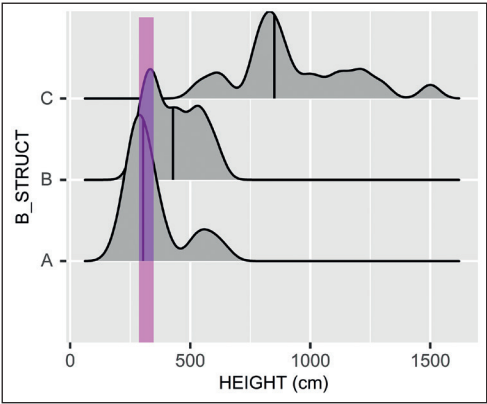
Appendix 2. Distributions of various nest site characteristics, including kernel density estimates, for Laughing Dove nest presence sites in 2019 ($n = 32$). The charts include box plots showing the median, interquartile range, and data points, as well as bar graphs displaying counts of categories. The plots (a) to (j) represent the following variables: (a) nest height above ground (HEIGHT), (b) nest to ceiling height (CEILING), (c) nest horizontal distance to ceiling edge (H_CEILING), (d) distance to nearest tree (TREE_DIST), (e) distance to opposite building (BUILD_DIST), (f) distance to green space (GREEN_DIST), (g) distance to nearest counterpart nest (NEST_DIST), (h) nest concealment categories (CONCEAL: A – invisible from the front and sides, B – invisible from the front but visible from one side, C – visible from the front but invisible from the sides), (i) base structure categories (B_STRUCT: A – shop boards, B – building frontispieces, C – balconies), and (j) base aspect categories (B_ASPECT: A – North, B – East, C – South, D – West)

2. melléklet A pálmagerle fészkelőhelyének az 1. mellékletben felsorolt tulajdonságait leíró értékek eloszlását bemutató ábrák



Appendix 3. Dendrogram of hierarchical variable clustering reveals a strong correlation between nest height above ground (HEIGHT) and base structure (B_STRUCT; distance value = 0.15)

3. melléklet A hierarchikus klaszterezéssel megállapított kapcsolatok a változók között erős összefüggést mutatnak a fészek talaj feletti magassága (HEIGHT) és a tartószerkezet (B_STRUCT) között (távolság = 0,15)



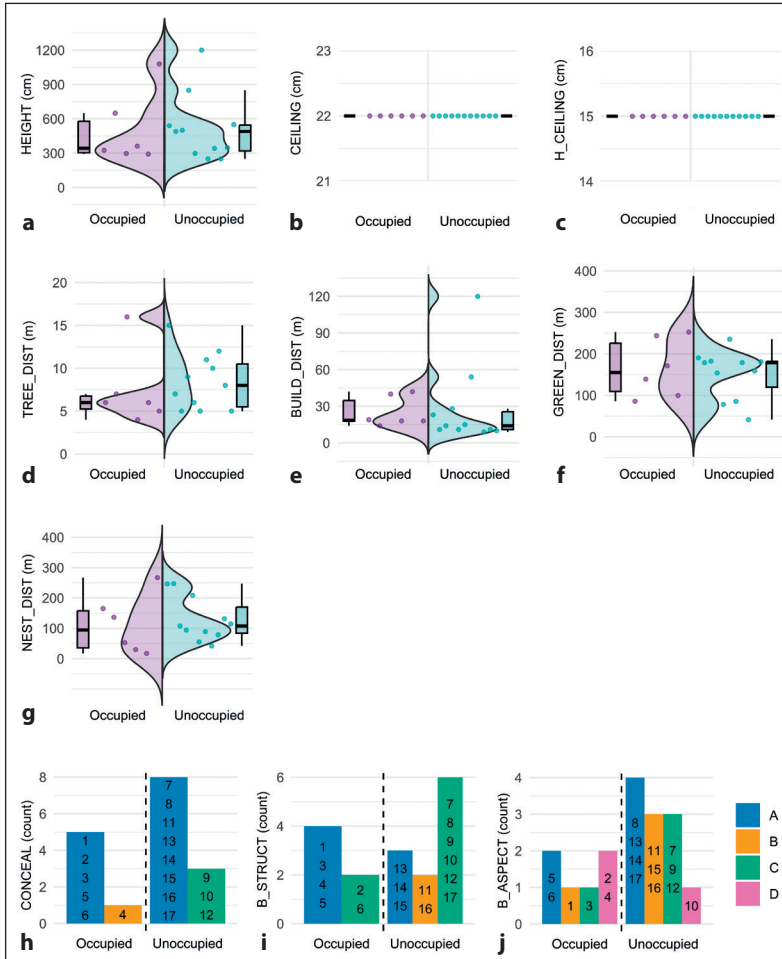
Appendix 4. Density ridgeline plot reveals that the Shop boards category (A) within the base structure (B_STRUCT) variable exhibits a higher density across the optimal range of nest height above ground (HEIGHT; 286–347 cm, as highlighted using a purple band). Vertical black lines indicate the medians

4. melléklet A fészek talaj feletti magassága (HEIGHT) és a tartószerkezet (B_STRUCT) közötti kapcsolat alapján a boltok táblái (A) magasabb denzitást mutatnak az optimális fészek magasságon belül (lila sávval jelölve). A függőleges fekete vonalak a mediánt jelölik az egyes kategóriákban

Appendix 5. The dataset from artificial nestbox sites for Laughing Doves (n = 17) was input into the nest site selection model. The model's analysis resulted in a classification table showing a correct prediction rate of 76.5% (optimal threshold = 0.52)

5. melléklet A kihelyezett költőládák helyeinek értékei a fészkelőhely-választás modellje alapján. A modell 76,5%-ban jól becsült (optimális küszöb = 0,52)

	Predicted Absence	Predicted Presence	Percentage Correct
Observed Absence	8	3	72.7%
Observed Presence	1	5	83.3%
Overall Percentage Correct			76.5%



Appendix 6. Comparisons of occupied versus unoccupied artificial nestboxes: In 2019, a total of 17 artificial nestboxes were erected, of which six were occupied and 11 were unoccupied. The variables include: (a) nest height above ground (HEIGHT), (b) nest to ceiling height (CEILING), (c) nest horizontal distance to ceiling edge (H_CEILING), (d) distance to nearest tree (TREE_DIST), (e) distance to opposite building (BUILD_DIST), (f) distance to green space (GREEN_DIST), (g) distance to nearest counterpart nest (NEST_DIST), (h) nest concealment categories (CONCEAL: A – invisible from the front and sides, B – invisible from the front but visible from one side, C – visible from the front but invisible from the sides), (i) base structure categories (B_STRUCT: A – shop boards, B – building frontispieces, C – balconies), and (j) base aspect categories (B_ASPECT: A – North, B – East, C – South, D – West). The charts include box plots showing the median, interquartile range, and individual data points, which are arranged from left to right in order of IDs (1-17) to track each nestbox across all subplots. They also display kernel density estimates of the data distribution. Bar graphs display counts of categories and, inside bars, the IDs of each nestbox. Category colors in bar graphs: A – blue, B – orange, C – green, and D – purple

6. melléklet A fészkelésre használt és nem használt költőládák összehasonlítása a korábban bemutatott tulajdonságok alapján. A kihelyezett 17 költőládából 6-ot használták a pálmagerlék 2019-ben

Urban population of the Common Wood Pigeon in Krasnodar

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Abstract Data on population density and location of nests of the Common Wood Pigeon (*Columba palumbus*) in Krasnodar is presented for the first time. The highest population density was recorded in cemeteries (10.4 ± 1.4 pairs/10 ha, reaching 12.1 pairs/10 ha in one cemetery). The population density values in green residential areas with a predominance of 5-story housing blocks and in parks are comparable 2.9 ± 1.2 and 2.7 ± 1.0 pairs/10 ha, respectively. The average Common Wood Pigeon population density in the city was 3.4 ± 0.8 pairs/10 ha. In residential areas with a predominance of 5-story housing blocks, the nests were located in trees and shrubs belonging to 13 taxa, predominantly in ash (18.4%), linden (15.8%), horse chestnut (13.2%) and elm (13.2%) trees. The nests were located at a height of 5.5 to 15 m above the ground, (9.6 ± 0.4 m). The higher location of Common Wood Pigeon nests in Krasnodar compared to other European cities is probably associated with a lower level of predation by the Hooded Crow (*Corvus cornix*). The Common Wood Pigeon began to breed in Krasnodar in the early 2000s during a period of a phenomenal upsurge in construction activity in the city when the pace of development was comparable to that in Moscow.

Keywords: city birds, breeding, synanthropization, population density, Common Wood Pigeon

Összefoglalás A Krasznodarban élő örvös galamb (*Columba palumbus*) populáció állománysűrűségének és fészkelőhelyeinek adatait jelen tanulmányban összegezzük. A legnagyobb állománysűrűséget a temetőben figyelhető meg ($10,4 \pm 1,4$ pár/10 ha, egy temetőben elérve a $12,1$ pár/10 ha értéket is). Az 5 emeletes lakótömböket körülvevő zöld övezetekben és a parkokban az állománysűrűség értékei hasonlóak: $2,9 \pm 1,2$ és $2,7 \pm 1,0$ pár/10 ha. A városban a faj átlagos állománysűrűsége $3,4 \pm 0,8$ pár/10 ha volt. Az 5 emeletes lakótömbök környezetében a fészkek 13 különböző fa-, és cserjefajon találhatók, többségében kőris (18,4%), hárs (15,8%), vadgesztenye (13,2%) és szil (13,2%) fák. A fészkek 5,5 és 15 m közötti magasságban helyezkedtek el ($9,6 \pm 0,4$ m). Más európai városokhoz képest Krasznodarban a magasabban elhelyezkedő fészkek valószínűleg a dömlyös varjú (*Corvus cornix*) jelenlétével és hatásával függnek össze. Az örvös galamb a 2000-es évek elején kezdett el költeni Krasznodarban, a városépítési tevékenységek fellendülése idején, amikor a fejlődés üteme a moszkvaihoz hasonló volt.

Kulcsszavak: állománysűrűség, költés, örvös galamb, városiasodás, városi madarak

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Introduction

Urban habitats are markedly different from suburban habitats. The major difference is the transformation of the land: from natural green areas to man-made structures and impervious surfaces. To survive in the urban habitat, birds are forced to either accept or avoid the new conditions (Isaksson 2018).

The Common Wood Pigeon (*Columba palumbus*) has been successfully colonizing urban areas in Europe and Asia for many years, forming specialized urban populations there.

Such populations have been recorded in cities in Germany, France, Poland, Belarus, Great Britain, Slovakia, Finland, Hungary and other countries (Tomiałojć 1976, 1978, 1998, Luniak *et al.* 2001, Kopij & Zendwalewicz 2009, Astafieva *et al.* 2011, Bea *et al.* 2011, Albrycht & Ciach 2013, Mošanský *et al.* 2014, Fey *et al.* 2015, Luniak & Węgrzynowicz 2019, Sakhvon & Kövér 2020, Csathó & Bozó 2022). Urban populations of the Common Wood Pigeon differ from natural ones in several significant features, including a noticeably higher population density, longer breeding period, tolerance to humans and flexibility in nest placement, including the building of nests on man-made structures (Tomiałojć 1976, Górski *et al.* 1998, Šťastný *et al.* 2005, Lykov 2009, Briedik & Šípkovský 2012). The timing of the emergence of urban populations varies significantly between different urban areas. The first urban population of the species was registered approximately in the 1830–1840s in the parks of Paris (Tomiałojć 1976). The species colonized some urban areas only in recent times, with full-fledged urban populations either having only recently formed there or being at the stage of formation (Mošanský *et al.* 2014, Fey *et al.* 2015, Faragó *et al.* 2019, Csathó & Bozó 2022).

Breeding of the Common Wood Pigeon in Russian cities has not become as common as in the cities of Central and Eastern Europe. The species was recorded breeding, for example, in Kaliningrad, Tula, as well as in the towns of Pre-Caucasus (Astafieva *et al.* 2011, Aralov 2021, Malovichko *et al.* 2021).

The species continues to show a significant population increase in Western and Central Europe, along with an expansion of its range northwards to Fennoscandia (Sattler 2020). The increase in the Common Wood Pigeon numbers is most likely related to an increased reproductive output during the breeding season combined with increased food supplies (O'Regan *et al.* 2012).

Krasnodar, where field research was carried out, at the beginning of the 21st century demonstrates phenomenally high growth rates by Russian standards, which is reflected, in particular, in population dynamics and an increase in built-up area. With a sharp increase in the developed area and at the same time in the volume of construction, the city is witnessing expansion and densification. The surge in housing construction in Krasnodar has been observed since 2005. By 2010–2012, Krasnodar had come close to Moscow in terms of the number of houses built. The urban built-up area has increased by 52.9% by 2019 compared to 2002. At the same time, new built-up areas have a higher proportion of impervious surfaces and a lower proportion of land suitable for landscaping compared to neighborhoods built in the 1960s–1970s (Pogorelov & Kiselev 2020). Phytomass losses have been noted over 26 years (1989–2015) in 60% of the residential area (Pogorelov & Lipilin 2017).

Small-house areas prevail in the city, accounting for more than 80% of all residential areas and being located in all city districts. The city's peripheral belt comprises a mosaic of small-house communities, agricultural land, farming enterprises and land owned by allotment societies (Amendments to the General Plan (<https://genplan.krd.ru/expo/>)).

Dinkevich and Korotky (2004), who studied the avifauna of Krasnodar, state that the first mass wintering of Common Wood Pigeons in the city was during the winter of 2001/2002 when low temperatures and overabundance of snow forced the birds to migrate from the Kuban River floodplain to urban habitats. In that winter, Common Wood Pigeons were

registered for the first time in the residential area of Krasnodar. In the next years, they were recorded breeding in Krasnodar in the botanical garden of the Kuban State Agrarian University (Dinkevich & Korotky 2004). Thus, at the time of our research, the urban population of the species in Krasnodar had already existed for two decades. It should also be noted that the urban population of the Common Wood Pigeon emerged during the period of phenomenally high rates of urban expansion and volume of construction in Krasnodar.

Breeding of the species in the urban environment in Crimea commenced in the same way after a cold winter, but earlier than in Krasnodar, after the 1984/1985 winter (Kostin 2020). Later, from 2008 to 2020, the Common Wood Pigeon was recorded breeding in Krasnodar in almost all parks, squares, cemeteries and residential areas with tall trees (Emtyl *et al.* 2011).

The aim of the work was to determine the main breeding parameters of the urban population of the Common Wood Pigeon in Krasnodar. In addition, the influence of the main nest predators was considered and the food supply of the Common Wood Pigeon was assessed.

Material and Methods

Study area

The Municipal Formation of the City of Krasnodar (hereinafter referred to as Krasnodar) (45°02' N 38°59' E) is located in Southern Russia and stretches along the right bank of the Kuban River. The Kuban River separates the city from the Republic of Adygea. The distance to the Black Sea and the Sea of Azov is 120–150 kilometers, and the distance to Moscow is slightly over 1,300 km. The city covers an area of about 839 km² (Amendments to the General Plan (<https://genplan.krd.ru/expo/>)). Its population is 1.2262 million people (Website of the Krasnodar Krai and Adygea Republic Office of the Federal State Statistics Service ([https://23.rosstat.gov.ru/storage/mediabank/01.01.2023\(1\).htm](https://23.rosstat.gov.ru/storage/mediabank/01.01.2023(1).htm))). The city includes five rural districts divided into 29 rural settlements.

Krasnodar is located in the steppe zone with mild continental climate. The coldest month is January with an average temperature of –0.2 °C, the average temperature of the warmest month, July, is 23.8 °C, the average annual temperature is 11.8 °C. The Kuban steppe is a typical European-type steppe. The steppes are characterized by the dominance of herbaceous vegetation, and forests are scarce.

Due to the natural geographical factors, Krasnodar, has no actual green belt. The main urbanized area does not feature any continuous structure of green spaces (Pogorelov & Kiselev 2020). Green spaces are predominantly concentrated in the old central part of the city, while new neighborhoods are characterized by poor and insufficient greening (Greenologia life quality website (<https://greenologia.ru/eko-problemy/goroda/krasnodar.html>)).

The green spaces within the boundaries of Krasnodar have a total area of 110.4 km² according to 2019 aerial photography data (Amendments to the General Plan (<https://genplan.krd.ru/expo/>)). Linden, maple, liriiodendron, cercis, plane tree, gingko, catalpa, pyramidal English oak, red oak, black pine, European yew, boxwood, barberry, spiraea, weigela, dogwood,

ninebark, as well as various species of thuja, juniper, cypress, ornamental fruit trees and many other trees and shrubs can be found in the city (Portal of the Administration of the Municipal Formation of the City of Krasnodar and the City Duma of Krasnodar (Portal of the Administration of the Municipal Formation of the City of Krasnodar and the City Duma of Krasnodar (<https://krd.ru/administratsiya/administratsii-krasnodara/departament-gorodskogo-khozyaystva-i-toplivno-energetichesk/zelenye-nasazhdeniya-goroda/>)).

Data collection

The field research was carried out within the administrative boundaries of Krasnodar as part of a program to study model bird species in Russian urban areas. Six main habitats were examined: the forest park, cemeteries, recreational parks, garden squares, small-house areas and green residential areas with a predominance of 5-story housing blocks.

The small-house areas have low-rise single-family or two-family houses built on individual land plots, which are planted mainly with fruit trees and exotic conifers. The green residential areas with a predominance of 5-story housing blocks are neighborhoods with a high proportion of greening along highways and inside yards.

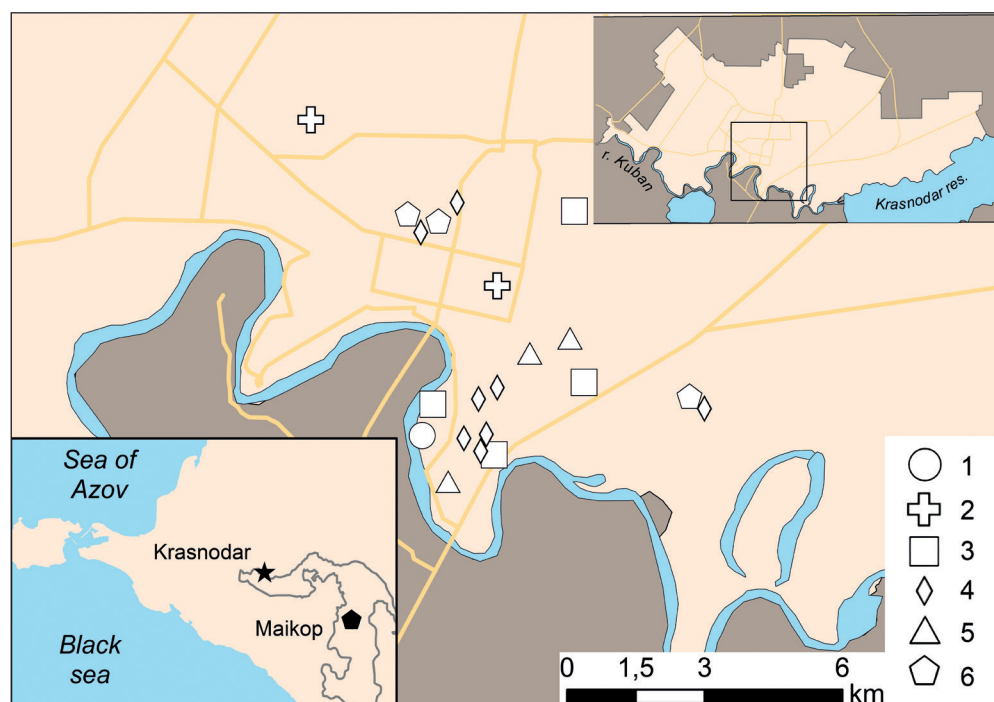


Figure 1. Placement of census plots in Krasnodar. Legend: 1 – forest park, 2 – cemetery, 3 – recreational park, 4 – garden square, 5 – small-house areas, 6 – green residential areas with a predominance of 5-story housing blocks

1. ábra A felmérési pontok elhelyezkedése Krasznodarban. Jelmagyarázat: 1 – erdős park, 2 – temető, 3 – szabadidőpark, 4 – virágos tér, 5 – alacsony házas lakóövezet, 6 – öteemeletes lakóépületek zöldövezete

Censuses of breeding pairs of the Common Wood Pigeon were carried out using the method of breeding territories mapping on designated census plots on 15–16 May 2021 and 13–14 June 2023. The census activities were carried out during the daylight hours, including the periods of maximum bird activity in the morning and evening. A total of 21 census plots were allocated (*Figure 1*), with a total area of 337.3 ha. The number of breeding pairs on each census plot was determined by the number of territorial birds, singing males and occupied nests found. Transiting birds have not been included into the registry. The 2021 and 2023 data were merged. The census activities were carried out only once on each plot. As a result, the average population density of the species was determined for each habitat and the city as a whole.

In addition, to assess the impact of species of the family Corvidae on the population of the Common Wood Pigeon, nests of the Hooded Crow (*Corvus cornix*) and Eurasian Magpie (*Pica pica*) were counted during the leafless period on 17–18 April 2020. The Hooded Crow and Eurasian Magpie nests were counted on 3 census plots with a total area of 89.5 ha in the small-house areas and on 4 census plots with a total area of 86.3 ha in the green residential areas with a predominance of 5-story housing blocks, with simultaneous observation of the Common Wood Pigeon on the same census plots.

A targeted search for Common Wood Pigeon nests, including a search with the use of 8×30 binoculars, was carried out only in the green residential areas with a predominance of 5-story housing blocks during the period of the Common Wood Pigeon breeding censuses, as well as during the complete count of Hooded Crow and Eurasian Magpie nests.

All found occupied and unoccupied Common Wood Pigeon nests were plotted on the map with an indication of their location and height above the ground.

The colonization of the Krasnodar urban areas by the species was assessed using five stages of synanthropization, which were proposed by Tomiałojć (1976):

- stage 0. Breeding in forests and groves away from populated areas;
- stage 1. Breeding of individual pairs in urban forests, as well as in peripheral parks on the outskirts of cities or smaller settlements;
- stage 2. Breeding in urban parks with a higher density than in natural habitats;
- stage 3. Breeding in urban parks with a high density, as well as regular breeding in residential areas (on the streets);
- stage 4. Breeding in urban parks with a high density, as well as breeding on the streets, including the downtown, cases of breeding on man-made structures.

Statistical analysis

The obtained data was statistically processed using the Stat Soft STATISTICA 6.0 software. Regarding the Common Wood Pigeon population density and nest height, the mean and standard error of the mean ($M \pm SE$) were calculated. The Common Wood Pigeon population densities in different habitats were compared using the Kruskal-Wallis non-parametric test.

Table 1. Population density (mean±SE) of the Common Wood Pigeon on census plots in Krasnodar in 2021, 2023*

1. táblázat Az örvös galamb állománysűrűsége (átlag±standard hiba) Krasznodarban két független felmérés alapján, 2021-ben és 2023-ban

Nº	Habitat	Census plot location	Census plot surface area, ha	Number of pairs	Population density, pairs/10 ha
1.	Forest park	30-Letiya Pobedy Park**	19.9	9	4.5
Average population density					4.5
2.	Cemetery	Vsesvyatskoe Cemetery	24.0	29	12.1
3.		Slavyanskoe Cemetery***	34.5	32	9.3
Average population density					10.4±1.4
4.	Recreational park	30-Letiya Pobedy Park**	19.8	9	4.5
5.		Gorodskoy Garden	13.3	5	3.8
6.		Chistyakovskaya Roshcha Park	49.0	11	2.2
7.		Kuban Park	9.6	0	0
Average population density					2.7±1.0
8.	Garden square	Trudovoy Slavy Square	1.9	0	0
9.		Festivalny Square	1.7	1	5.9
10.		Zhukova Square	2.5	1	4.0
11.		Druzhby Square	2.7	0	0
12.		Universitetsky Square	1.0	1	10.0
13.		Yekaterininsky Square	2.7	0	0
14.		Mariinsky Square	1.3	0	0
15.		Gagarinsky Boulevard	2.7	0	0
Average population density					1.8±1.3
16.	Small-house areas	Between Dlinnaya St., Kutuzova St., Golovatogo St., Bazovskaya St.	36.2	0	0
17.		Between Chapaeva St., Levanevskogo St., Dlinnaya St., Kommunarov St.	33.4	0	0
18.		Between Khimzavodskaya St., Koshevogo St., Zatonnaya St., Krasnodonskaya St.	19.9	0	0
Average population density					0
19.	Green residential areas with a predominance of 5-story housing blocks	Between Gagarina St., Atarbekova St., Gertsena St.	19.2	1	0.5
20.		Between Stavropolskaya St., Dimitrova St., 2-ya Pyatiletka St.	17.8	7	3.9
21.		Between Gagarina St., Vorovskogo St., Atarbekova St.	24.2	10	4.1
Average population density					2.9±1.2

* censuses of breeding pairs were conducted only once in one of the two years

** in 30-Letiya Pobedy Park, censuses were carried out separately in the recreational and forest-park parts, which were included in different habitat categories: "forest parks" and "recreational parks" respectively

*** censuses carried out on a part of the cemetery area

Results

Population density

The breeding of the Common Wood Pigeon was recorded in Krasnodar in all studied habitats, except the small-house areas. The highest population density was observed in the cemeteries (10.4 ± 1.4 pairs/10 ha, with the maximum value of 12.1 pairs/10 ha recorded for Vsesvyatskoe Cemetery). Lower population densities were observed in the forest park (4.5 pairs/10 ha), green residential areas (2.9 ± 1.2 pairs/10 ha) and recreational parks (2.7 ± 1.0 pairs/10 ha). The average population density of the Common Wood Pigeon in cemeteries exceeds the average population density in parks by 3.9 times. The maximum difference (factor of 5.5) between the population density values was noted for Vsesvyatskoe Cemetery and Chistyakovskaya Roshcha Park (Table 1).

Garden squares were less favored as breeding habitats. Of the eight squares studied, only three were occupied by only one pair of birds each (Table 1).

The censuses of breeding pairs of the Common Wood Pigeon in small-house areas are less complete than in other habitats due to the large area of this habitat in the city. The share of small-house areas is more than 80% of all residential areas in the city (Amendments to the General Plan 2020). Thus, rare breeding of birds in this habitat cannot be ruled out.

The average population density of the species in the city was 3.4 ± 0.8 pairs/10 ha.

The comparison of the Common Wood Pigeon population density between three habitats (parks, garden squares and green residential areas with a predominance of 5-story housing blocks) demonstrated the absence of significant differences ($H = 0.76$, $df = 2$; $p = 0.685$).

Nest placement

In the residential areas with a predominance of 5-story housing blocks, a total of 38 nests were found, which were built in trees and shrubs belonging to 13 taxa (Table 3). The majority of nests were located in ash (*Fraxinus excelsior*) (18.4% of the total number of nests found), linden (*Tilia* sp.) (15.8%), horse chestnut (*Aesculus hippocastanum*) (13.2%) and elm (*Ulmus* sp.) (13.2%) (Table 2) trees.

The Common Wood Pigeon nests in Krasnodar were located at a height of 5.5 to 15 m above the ground (mean \pm SE: 9.6 ± 0.4 m).

Main nest predators of the Common Wood Pigeon

Such nest predators of the Common Wood Pigeon as the Hooded Crow and Eurasian Magpie breed in Krasnodar. In general, their breeding density in the city is relatively low, with the Hooded Crow being the more numerous breeder. At the same time, the population density of both species in the small-house areas is very low. The breeding density of the Hooded Crow in the green residential areas with a predominance of 5-story housing blocks can reach 2.8 pairs/10 ha in some parts of the city (Table 3).

Table 2. Placement of Common Wood Pigeon nests in trees and shrubs in the green residential areas with a predominance of 5-story housing blocks in Krasnodar

2. táblázat Az örvös galamb fészkeinek elhelyezkedése a különböző fa- és cserjefajok alapján Krasznodar ötemeletes lakóépületeinek környezetében

Species of tree or shrub	Number of nests	%
Ash (<i>Fraxinus excelsior</i>)	7	18.4
Linden (<i>Tilia</i> sp.)	6	15.8
Horse chestnut (<i>Aesculus hippocastanum</i>)	5	13.2
Elm (<i>Ulmus</i> sp.)	5	13.2
Cherry plum (<i>Prunus cerasifera</i>)	2	5.3
Black locust (<i>Robinia pseudoacacia</i>)	1	2.6
Birch (<i>Betula</i> sp.)	1	2.6
Maple (<i>Acer</i> sp.)	1	2.6
Ash-leaved maple (<i>Acer negundo</i>)	1	2.6
Oriental plane (<i>Platanus orientalis</i>)	1	2.6
Pine (<i>Pinus</i> sp.)	1	2.6
Thuja (<i>Thuja</i> sp.)	1	2.6
Fruit tree (unidentified)	1	2.6
Unidentified tree	5	13.2

Table 3. Breeding density (nests/10 ha) of the Hooded Crow and Eurasian Magpie on census plots in Krasnodar, 2020

3. táblázat A dolmányos varjú és a szarka állománysűrűsége (fészkek száma/10 ha) Krasznodar területén 2020-ban

Habitat	Breeding density (min-max)		Average breeding density (mean±SE)	
	Hooded Crow	Eurasian Magpie	Hooded Crow	Eurasian Magpie
Small-house areas	0–1.2	0–0.3	0.4	0.1
Green residential areas with a predominance of 5-story housing blocks	0.4–2.8	0–0.8	1.4±0.5	0.2

Discussion

Population density

The Common Wood Pigeon population density varies markedly in the studied habitats of Krasnodar. The analysis conducted on the available literature data at our disposal (Biadun 1994, Dombrowski & Łuczak 1998, Kasprzykowski & Łuczak 2000, Tomiałojć 2007, Kopij 2010, Mošanský *et al.* 2014, Tomiałojć *et al.* 2020) revealed a number of patterns. A pairwise comparison (24 comparisons made) of Common Wood Pigeon population densities in the same year (in the event of absence of data for some year, data for the previous or subsequent year was used) in selected parks and cemeteries in 5 cities (Lublin, Legnica, Wrocław,

Siedlce, Košice) showed, that for slightly more than a half of the comparisons (54.2%) the population density is higher in parks (maximum difference by a factor of 5.3), for 37.5% it is higher in cemeteries (maximum difference by a factor of 8.4) and for 8.3% almost no difference was observed.

In Krasnodar, the Common Wood Pigeon population density is almost equal in the recreational parks and green residential areas with a predominance of 5-story housing blocks (*Table 1*). However, in three cities for which comparative data are available (Wrocław, Zielona Góra, Białystok), the population density of the Common Wood Pigeon is significantly higher in parks compared to the built-up area (Tomiałojć 2011, Bocheński *et al.* 2013, Zbyryt 2014, Kopij 2017, Tomiałojć *et al.* 2020). In particular, the Common Wood Pigeon population density in Białystok and Zielona Góra is higher in the parks than in the built-up area by a factor of 1.9 and 8, respectively (Bocheński *et al.* 2013, Zbyryt 2014).

Thus, the urban population of the Common Wood Pigeon in Krasnodar can be distinguished by a comparable population density in recreational parks and in green residential areas with a predominance of 5-story housing blocks.

The species was not observed in Krasnodar in the small-house areas during the research period. The information on breeding in small-house areas and the population density of the Common Wood Pigeon in this habitat in European urban areas is fragmentary. In Białystok, the population density in the small-house areas was 1 pair/10 ha (Zbyryt 2014). In Kaliningrad, 7.7% of all nests found were located in the small-house areas (Lykov 2009).

Its population density in most European urban areas is noticeably higher than the average population density in Krasnodar. But at the same time, the population density of the species in some locations was noticeably lower compared to Krasnodar (*Table 4*).

Thus, during the existence of the urban population of the species in Krasnodar, its population density reached relatively high values, however, in most other European urban areas this value has climbed even higher. This difference is apparently linked to a relatively recent formation of an urban population of the Common Wood Pigeon in Krasnodar compared to the urban populations in most other European urban areas, which have a longer history. Additionally, the population density of the Common Wood Pigeon may be determined by habitat structure, food source availability, weather factors and predation (Faragó *et al.* 2019).

Nest placement

The conditions for sheltering Common Wood Pigeon nests related to the structural features of trees are the main factor that determines the degree of predation pressure. This species tends to build nests in areas with dense foliage (Tomiałojć 1979).

Data on preferred nesting tree species in Krasnodar is comparable to the information published for other locations. Thus, in other cities for which data on the taxa of preferable nesting trees is available, the Common Wood Pigeon most often nested in linden trees: Minsk (53.5% of the total number of nests; Sakhvon & Kövér 2020), Liverpool (32%; Slater 2001), Legnica (26.4%; Tomiałojć 1979), Kaliningrad (19.8%; Lykov 2009), Galway, Ireland (19.5%; Ó hUallacháin 2014). In Debrecen, the linden is the second favored nesting tree after the common hackberry (*Celtis occidentalis*), with 15.3% of nests located in linden trees (Juhász & Varga 2019). In

Table 4. Comparative population density of the Common Wood Pigeon in European urban areas
4. táblázat Az örvös galamb állománysűrűsége más európai városok összehasonlításában

Population density, pairs/10 ha	City/town, country (year, habitat, location)	Source
More than 100	Legnica , Poland (2003–2004, down-town Central Park)	Tomiałojć 2007
50–99	Ślupsk , Poland (1990–1997, urban parks); Legnica , Poland (1966–1967, 1972, down-town Central Park); Wrocław , Poland (1970–1973, Słowacki Park)	Górski <i>et al.</i> 1998, Tomiałojć 2007, Tomiałojć <i>et al.</i> 2020
40–49	London , Great Britain (1974, St James's Park); Zielona Góra , Poland (parks)	Tomiałojć 1978, Bocheński <i>et al.</i> 2013
30–39	Legnica , Poland (2006, cemetery)	Jędro <i>et al.</i> 2008
20–29	Ślupsk , Poland (1985–1989, urban parks); Legnica , Poland (2004–2005, cemetery); Wrocław , Poland (2009, 2013, Południowy Park; 2007, Botanic Garden)	Górski <i>et al.</i> 1998, Jędro <i>et al.</i> 2008, Kopij & Zendwalewicz 2009, Kopij 2010, Tomiałojć <i>et al.</i> 2020
10–19	London , Great Britain (1974, Kensington Gardens); Wrocław , Poland (2009–2010, Szczytnicki Park; 2008–2010, Słowacki Park; 2013, Botanic Garden); Warsaw , Poland (2015, Krasińskich Garden); Mezőhegyes , Hungary (2022)	Tomiałojć 1978, Tomiałojć 2011, Luniak & Węgrzynowicz 2019, Tomiałojć <i>et al.</i> 2020, Csathó & Bozó 2022
8–9.9	Ślupsk , Poland (1974–1977, 1990–1997, suburban parks); Siedlce , Poland (1999, cemetery at Cmentarna str.)	Górski <i>et al.</i> 1998, Kasprzykowski & Łuczak 2000
6–7.9	Lublin , Poland (1988–1990, Akademicki Park; 1989–1991, Botanic Garden; 1989–1991, area planted with trees in Czechów); Ślupsk , Poland (1985–1989, suburban parks); Liverpool, Great Britain (1996–1998, Sefton park); Legnica , Poland (1965, Peripheral park); Košice , Slovakia (2011–2012, Cemetery of St. Rozália)	Biadun 1994, Górski <i>et al.</i> 1998, Slater 2001, Tomiałojć 2007, Mośański <i>et al.</i> 2014
4–5.9	Lublin , Poland (1988–1990, Park "Saski Garden"); Legnica , Poland (2007, Peripheral park); Wrocław , Poland (2011, Biskupiński Park; 2008, Osobovitskoe Cemetery; 2016, city center); Zielona Góra , Poland (housing estates); Warsaw , Poland (2000, cemetery-mausoleum of Soviet soldiers, estate near the street Dombrowsky; 2004–2005, 2014, Skaryszewski Park); Battonya , Hungary (2022)	Biadun 1994, Luniak <i>et al.</i> 2001, Tomiałojć 2007, Bocheński <i>et al.</i> 2013, Luniak & Węgrzynowicz 2019, Tomiałojć <i>et al.</i> 2020, Csathó & Bozó 2022
2–3.9	Lublin , Poland (1982–1984, Lipowa Str. Cemetery; 1987–1989, Unicka Str. Cemetery; 1987–1990, Bronowicki Park; 1988–1990, Majdanek Cemetery, Kalinowszczyzna Cemetery); Ślupsk , Poland (1974–1977, urban parks); Siedlce , Poland (1985, Main Siedlce Park; 1984, 1986, Wild Siedlce Park; 1998, cemetery at Cmentarna str.; 1985–1986, 1998–1999, cemetery at Janowska str.); Kaliningrad , Russia (2008–2010); Kraków , Poland (2012, Rakowice Cemetery); Košice , Slovakia (2011–2012, park in Barca); Białystok , Poland (2013, parks); Wrocław , Poland (2017, block-buildings area); Warsaw , Poland (2006, 2013, Żeromskiego Park; 2013, Krasińskich Garden); Końskie , Poland (2016–2017, multi-family housing plot)	Biadun 1994, Górski <i>et al.</i> 1998, Dombrowski & Łuczak 1998, Kasprzykowski & Łuczak 2000, Astafieva <i>et al.</i> 2011, Albrycht & Ciach 2013, Mośański <i>et al.</i> 2014, Zbyryt 2014, Kopij 2017, Luniak & Węgrzynowicz 2019, Dębowski 2020
1–1.9	Lublin , Poland (1982–1984, Ludowy Park); Siedlce , Poland (1984, 1986, Main Siedlce Park; 1985–1986, cemetery at Cmentarna str.); Dijon , France (1980–1984, Parc des Sports); Košice , Slovakia (2011–2012, Public Cemetery); Białystok , Poland (2013, city centre, old block of flats area, detached houses estate); Warsaw , Poland (2004, 2017, Żeromskiego Park; 2013, Kępa Potocka Park)	Biadun 1994, Dombrowski & Łuczak 1998, Tomiałojć 1998, Kasprzykowski & Łuczak 2000, Mośański <i>et al.</i> 2014, Zbyryt 2014, Luniak & Węgrzynowicz 2019
less than 1	Dijon , France (1980, Parc de La Colombiere); Warsaw , Poland (late 1980s, Łazienki Park, Rakowiec, Wierzbno districts; 2006, Kępa Potocka Park); Košice , Slovakia (2011–2012, park on the Komenského street); Wrocław , Poland (2017, industry area); Białystok , Poland (2013, modern block of flats area, industrial areas, suburban rural area)	Tomiałojć 1998, Luniak <i>et al.</i> 2001, Mośański <i>et al.</i> 2014, Kopij 2017, Zbyryt 2014, Luniak & Węgrzynowicz 2019

Minsk and Legnica, the Common Wood Pigeon, in addition to linden, also preferred to build nests on Horse Chestnut (18.4% and 18.3% respectively) (Tomiałojć 1979, Sakhvon & Kövér 2020). In other urban areas, the Common Ash and Elm species were also used by the Common Wood Pigeon for nesting, however, the number of nests in trees of these taxa was insignificant (Tomiałojć 1979, Ó hUallacháin 2014). In Kaliningrad and Minsk, Common Wood Pigeon nests were not found in elm trees (Lykov 2009, Sakhvon & Kövér 2020).

Nests of the Common Wood Pigeon in the green residential areas with a predominance of 5-story housing blocks in Krasnodar were on average located higher (9.6 m) than ones in the residential areas of Debrecen (7.2 m) (Juhász & Varga 2019), parks and residential areas of Minsk (7.1 m) (Sakhvon & Kövér 2020), in the university area in Galway (6.0 m) (Ó hUallacháin 2014) and in the residential areas of Kaliningrad (5.4 m) (Lykov 2009). In Słowacki Park in Wrocław, the average nest height above the ground of 9.9 m is comparable to our figures obtained in Krasnodar. In other Polish parks (Szczynicki Park in Wrocław and City Park in Legnica), the average height of nests of the Common Wood Pigeon was greater (10.9 and 14.3 m respectively) than in Krasnodar (Tomiałojć 1979). According to Tomiałojć (1979), the height of Common Wood Pigeon nests above the ground depends on the degree of predation by the Hooded Crow. When the level of predation by the Hooded Crow is high, the Common Wood Pigeon places its nests lower above the ground (Tomiałojć 1979). Common Wood Pigeon nests in Debrecen, Minsk, Galway and Kaliningrad may be placed lower due to higher degrees of predation by crows in these cities compared to Krasnodar. In addition, an important factor influencing the height of nest placement is the height of trees.

Main nest predators of the Common Wood Pigeon

The main nest predators include the Hooded Crow and, to a lesser extent, the Eurasian Magpie (Tomiałojć 1979, Górski *et al.* 1998, Faragó *et al.* 2019, Varga & Juhász 2020). The high numbers of the Hooded Crows are regarded by Tomiałojć (1979) as one of the factors that complicate the colonization of the urban environment by the Common Wood Pigeon. Therefore, the breeding Hooded Crow and Eurasian Magpie have an impact on the population of the Common Wood Pigeon in Krasnodar. However, taking into account the low density of both corvid species in Krasnodar, their impact on the Common Wood Pigeon is not so significant compared to urban areas where the population density of those species is significantly higher. It should be noted that in some urban areas, the breeding success of the Common Wood Pigeon can decrease by up to 3–7% due to predation pressure (Tomiałojć 2021).

Thus, the predation pressure in Krasnodar is deemed to be in general insignificant. In the green residential areas with a predominance of 5-story housing blocks, the total breeding density of the main nest predators (Hooded Crow, Eurasian Magpie) is much lower than the population density of the Common Wood Pigeon.

Food supply assessment

Breeding urban populations of the Common Wood Pigeon are characterized by long-distance foraging movements to nearby farmland, where the birds receive additional food,

supplementing the food found in the urban area itself (Tomiałojć 1976, Jędro *et al.* 2008, Bendjoudi *et al.* 2015). During the breeding period, Common Wood Pigeon can make regular foraging movements over distances of up to 15 km or more (Tomiałojć 1976). Food supply provided by cultivated crops with different sowing dates, vegetation periods and, consequently, different harvesting dates partially complements the food source (Faragó *et al.* 2019).

No special studies have been carried out on the foraging movements of the Common Wood Pigeon to the farmland in the suburbs of Krasnodar. However, taking into account the importance of farmland for bird nutrition, it is necessary to provide a summary about this food source and of the state of agriculture in the areas around Krasnodar.

The Common Wood Pigeon feeds on a variety of plant foods, including weed seeds, shrub berries, seeds of crops (e.g. pea, sunflower, wheat, barley, corn, rice) ripening or lost during the harvest and seeds sown. It also eats green parts of herbaceous plants, tree fruits, soft shoots and pine seeds. Sometimes the species eats earthworms and snails (Faragó *et al.* 2019).

Analysis of data provided on the analytics platform “Unified Center for Remote Satellite Monitoring of the Agro-Industrial Complex of Krasnodar Krai” showed the following

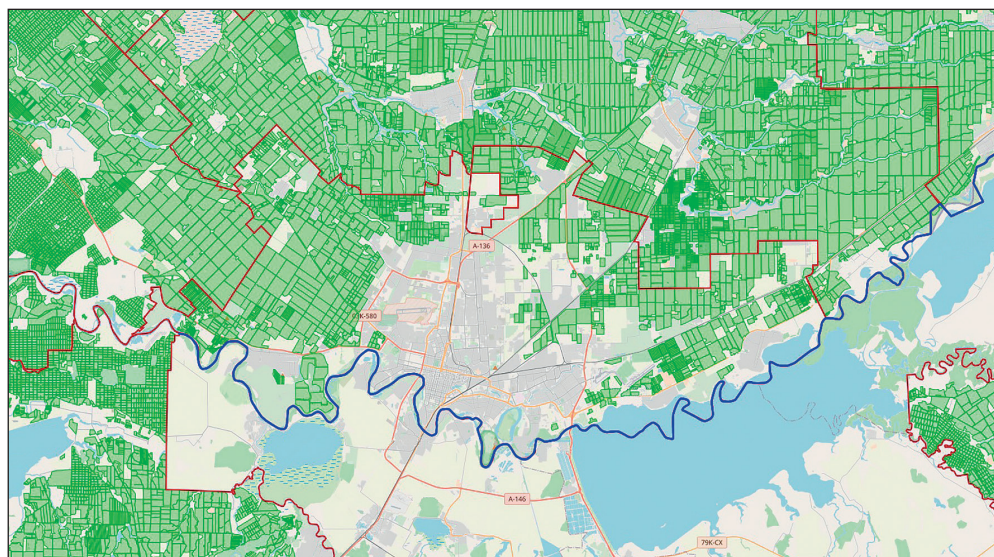


Figure 2. Distribution of farmland in Krasnodar and the surrounding area (indicated in green on the map). For the territory of the Republic of Adygea, farmland is not shown. The boundaries of the municipal districts of the Krasnodar Krai are indicated by a red line, the border between the Krasnodar Krai and the Republic of Adygea is indicated by blue.

Source: website of the analytics platform “Unified Center for Remote Satellite Monitoring of the Agro-Industrial Complex of Krasnodar Krai” https://maps.krasnodar.ru/agrofields_pub/regions/23

2. ábra A mezőgazdasági területek kiterjedése Krasznodarban és a környező területeken (a térképen zöld színnel jelölve). Adigeföld területén nincsenek jelölve ezek a területek. A közigazgatási kerületek határait Krasznodarszk vidékén piros vonal jelöli, míg Adigeföld határát kék vonal mutatja

patterns. The agricultural lands where the birds breeding in Krasnodar can move for feeding are located both within the city itself and in the territories of Krasnodar Krai adjacent to the city (*Figure 2*) (Website of the analytics platform “Unified Center for Remote Satellite Monitoring of the Agro-Industrial Complex of Krasnodar Krai” (https://maps.krasnodar.ru/agrofields_pub/regions/23)). Farmland is also present in the nearby Republic of Adygea in Takhtamukaisky Municipal District (Official website of the state executive bodies of the Republic of Adygea (<http://www.adygheya.ru/about/information/selskoe-khozyaystvo/?ysclid=Irs3vp2qya806799149>)). Takhtamukaisky Municipal District adjoins the southern part of Krasnodar. The minimum distance between the conventional city center and the nearest farmland landscapes located in Krasnodar is 6–9 km.

Agricultural lands occupy a significant part of Krasnodar (36%). The total cultivated area within the city was 282 km² in 2018 (Amendments to the General Plan 2020).

In 2023, the most widely cultivated crops in Krasnodar Krai were wheat (44% of the total cultivated area in the region), sunflower (13%) and corn (11%; Website of the Krasnodar Krai and Adygea Republic Office of the Federal State Statistics Service. (https://23.rosstat.gov.ru/agriculture_kk)), while in the Adygea Republic, they comprised wheat (40% of the total cultivated area in the region), corn (8%) and rapeseed (6%; Website of the Krasnodar Krai and Adygea Republic Office of the Federal State Statistics Service. (https://23.rosstat.gov.ru/agriculture_ra)). In Takhtamukaisky Municipal District of the Republic of Adygea that adjoins Krasnodar, rice, sunflower and wheat have the largest share (Adygea Today news agency website (<https://adigeatoday.ru/news/38464.html?ysclid=Irs4m3senl72428284>)). The crops specified for Krasnodar Krai and the Adygea Republic can be used by Common Wood Pigeon as food.

Thus, an assessment of the food supply of the Common Wood Pigeon showed the presence of vast areas of agricultural crops in the city and its surroundings that can be used by birds as food. As noted by Tomiałojć (1976), the location of the city in a river valley among fertile soils with developed agriculture in the form of cultivated field lands is one of the conditions conducive to the formation of a stable urban population of the Common Wood Pigeon.

Assessment of the colonization of urban areas by the Common Wood Pigeon

Using the stages of synanthropization specified by Tomiałojć (1976), the urban population of the Common Wood Pigeon in Krasnodar can be conditionally classified as one of the synanthropization stage 3: breeding in urban parks with a high density, as well as regular breeding in residential areas (on the streets). The studied population, however, seems to not reach the last synanthropization stage 4, when the species breeds at a high density in urban parks, as well as in residential areas, and individual pairs start to build nests on man-made structures.

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Population density and distribution of the Eurasian Eagle-Owl (*Bubo bubo*) in the Republic of Mordovia (Middle Volga)

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Abstract We determined the population density and number of Eurasian Eagle-Owl in one of the regions of Russia (Republic of Mordovia). The work was carried out over sixteen years (2009–2024). The studies were conducted using methods of direction finding of vocalizing birds with subsequent search for nests and with the use of digital voice recorders. The Eurasian Eagle-Owl reliably lives in 16 out of 22 districts (72.7%). During our research, we built more than 60 artificial nests, in which Eurasian Eagle-Owls subsequently settled. In total, during the research period, 89 Eurasian Eagle-Owl habitats (of which 38 in four stationary sites), 125 nesting cases, and 214 chicks were identified. Over the entire period, the birds laid 282 eggs. Productivity for couples was 76%. The most successful years in terms of the number of eggs and chicks in nests were 2014, 2018, 2022, 2023. The density of the Eurasian Eagle-Owl population ranges from 3.1 to 5 pairs per 100 km². The total population density in the open spaces of the region can reach 583 habitats. The presence of a large number of ravines with stands determines the high population density of Eurasian Eagle-Owl in Mordovia. Making artificial nesting boxes in ravines improves the suitability of breeding sites and promotes better survival of young individuals.

Keywords: *Bubo bubo*, Strigidae, density, reproduction, forest-steppe area, Russia

Összefoglalás Oroszország egyik régiójában (Mordvin Köztársaság) meghatároztuk az uhu állománysűrűségét és számát. A munkát tizenhat éven keresztül végeztük (2009–2024). A vizsgálatokat a hangoskodó madarak irány-meghatározásának módszereivel, a fészkek későbbi felkutatásával és digitális hangrögzítők alkalmazásával végeztük. Az uhu a 22 körzetből 16-ban (72,7%) megbízhatóan él. Kutatásaink során több mint 60 mesterséges fészket építettünk, amelyekben az uhu később megtelepedett. Összesen a kutatási időszak alatt 89 uhu élőhelyet (ebből 38-at négy helyhez kötött helyszínen), 125 fészkelést és 214 fiókát azonosítottunk. A teljes időszak alatt a madarak 282 tojást költöttek. A párok termelékenysége 76%-os volt. A fészekben lévő tojások és fiókák számát tekintve a legsikeresebb évek a következők voltak: 2014, 2018, 2022, 2023. Az uhu populáció sűrűsége 3,1–5 pár/100 km² között mozog. A régió nyílt területein a teljes populáció sűrűsége elérheti az 583 élőhelyet. A nagyszámú, állományokkal rendelkező szurdok jelenléte határozza meg az uhu nagy populációs sűrűségét Mordvin-földön. A szurdokokban mesterséges fészkelőhelyek kialakítása javítja a költőhelyek alkalmasságát és elősegíti a fiatal egyedek jobb túlélését.

Kulcsszavak: *Bubo bubo*, Strigidae, sűrűség, szaporodás, erdő-sztyepp terület, Oroszország

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Introduction

The vulnerability of the Eurasian Eagle-Owl (*Bubo bubo*) and the influence of humans on it (poaching, disturbance during nesting period) in many European countries: Germany (Brauneis & Hormann 2005, Rockenbach 2005), Austria (Liditznig 2005), Spain (Delgado & Penteriani 2005) is of ongoing research interest. The current number of the species in Europe, including the European part of Russia, ranges from 20,130 to 28,952 pairs and represents a relative estimate of the size of the European population (Penteriani & Delgado 2019). Moreover, in the latest monograph on the Eurasian Eagle-Owl (hereafter, Eagle-Owl), the above estimate has some subjectivity, since, as its authors themselves admit, there is no reliable data on the number of Eagle-Owls in Russia and Central Asia. The range of the Eagle-Owl is estimated at 32 million km². Territory of Russia (without the Arctic regions), where the Eagle-Owl lives, accounts for more than one third of the entire range. Based on this, the ratio is 17,630–23,452 pairs for European countries (less than 1/5 of the range) and 2,500–5,500 pairs only for the European part of Russia looks very underestimated. In addition, the Eagle-Owl population status is currently unknown in 39% of the countries, where the species occurs (Penteriani & Delgado 2019). An increase in population size is observed in 30.2%, a decrease in 21%, and stabilization in 4% of the countries. The share of the declining Eagle-Owl population in Europe is lower than in the late 1990s (Hagemeijer & Blair 1997). Colleagues report positive trends in the increase in the number of Eagle-Owls in Germany, France and Spain. At the same time, it is reported that in the European part of Russia and the northernmost countries of Scandinavia (Norway, Finland) the number of Eagle-Owls is decreasing. Difficulties in objectively assessing the number of Eagle-Owls in Russia by ornithologists can be explained by the small number of publications on vast territory of Russia. Indeed, despite the significance and priority of studying the ecology and biology of the Eagle-Owl, there are very few works devoted to its study in Russia. As a rule, the authors only indicate individual cases of records of the Eagle-Owl in certain regions (Pukinsky 1993, Karyakin 1998). Unfortunately, in Russia there has still been no targeted work to identify the number and population density of the species by area of individual plots, as was done in other countries (Penteriani & Delgado 2019). There are few Russian ornithologists, who specifically study the Eagle-Owl, and detailed data on individual regions is of great importance. According to molecular genetic data, 13–14 subspecies are distinguished (Wink *et al.* 2009). More data were known for the subspecies *B. b. ruthenus* (1,271 to 1,505 pairs) and *B. b. interpositus* (from 241 to 376 pairs) (Penteriani & Delgado 2019). The largest number of Eagle-Owl subspecies have been recorded exactly in Russia (*B. b. bubo* L., 1758; *B. b. ruthenus* Zhitzkov and Buturlin, 1906; *B. b. interpositus* Rothschild and Hartert, 1910; *B. b. sibiricus* Gloger, 1833; *B. b. yenisseeensis* Buturlin, 1915). Therefore, research of the Eagle-Owl in different regions of Russia should be conducted to assess the total number in the country. In most regions, abundance estimates are based on extrapolation and are relatively subjective, which indicates insufficient knowledge of local populations of the species.

A lot of work has been devoted to studying the dependence of population density and productivity of the Eagle-Owl on various factors, including diet (Brambilla & Bionda 2013, Andreychev *et al.* 2016, Harms 2017, León-Ortega *et al.* 2017, Milchev & Spassov 2017,

Milchev & Georgiev 2020, Obuch 2021, Pearson & Husby 2021, Tobajas *et al.* 2021, Hadad *et al.* 2022, Husby & Pearson 2022), however, it is not always possible to establish the main reason for the higher numbers in some territories or lower numbers in others. Therefore, it is important to identify the population density of the Eagle-Owl in individual areas in different parts of the range. Each study aimed at this deserves attention for a comprehensive assessment of the state of the species.

Along with the great influence of the availability of food, the success of Eagle-Owl breeding is also affected by the presence of nesting sites that can be improved artificially. It is widely known that for the Eagle-Owl, artificial nests, caves, and cornices on rocks and cliffs of ravines are arranged, or natural nest sites are cleared and improved (Olsson 1979, Gorner 1983). However, until now, not enough attention has been paid to comparing the success of Eagle-Owl reproduction in natural and artificial nests. It is especially important whether this affects the population density.

The aim of our research is to determine the population density and number of Eagle-Owls in one of the regions of central Russia. Targeted work to achieve this goal has begun in 2009 and is ongoing. In total, observations covered most districts of the region. A separate task was to compare the success of reproduction in natural and artificial nests. One of the important objectives of the study was to assess possible positive and negative factors influencing population density and reproduction of Eagle-Owl in the region.

Material and Methods

Study site

The studies were carried out from 2009 to 2024 in the Republic of Mordovia. The region is located in the center of the East European Plain (53°38' – 55°11'N and 42°11' – 46°45'E). The length from west to east is 298 and from north to south is from 57 to 140 km. The area of Mordovia is 26,200 km², which is comparable to the territories of individual European countries (Belgium, Slovenia, Macedonia, Albania). Most of Mordovia is located in the northwestern part of the layered Volga Upland, which in the west turns into the layered Oka-Don lowland. This geographical location of the region causes a decrease in the activity of erosion-denudation processes from the southeast to the northwest. The republic is located in the interfluvium of rivers Moksha and Sura (the right tributary of the Volga), on the southwestern periphery of the Volga. The study area is characterized by a temperate climate. The average annual temperature is about 15 °C, the average annual precipitation is from 700 to 800 mm. The warmest months are July and August (average temperature 24–26 °C), the coldest months are January and February (average temperature 7–9 °C). Mordovia is located on the border of broad-leaved, mixed and forest-steppe zones. Most of Mordovia is occupied by the forest-steppe zone. Natural vegetation is dominated by oak forests and meadow steppes. Three stationary sites in the southeastern and central parts of the region were selected as the main ones. The southeastern and southern part of Mordovia is occupied by an erosion-denudation plain, characterized by high erosional dissection. As a result of

erosion, linearly elongated relief forms – ravines and gullies – were formed. The ravines often have a V-shape with slope heights of up to 40 m. The watershed areas of this plain in the territory of Mordovia can reach significant heights above sea level (from 280 to 320 m), the depth of the erosion incision is from 100 to 120 m. The density of linear erosion forms on in some areas exceeds 1 km/km².

The main studies took place at four stationary sites: Prisursky, Kochkurovsky, Chamzinsky, and Kovylninsky (Figure 1). Prisursky stationary site is located in the southeastern part of the region. Its area is 900 km². The territory of the stationary site is limited to the north by the highway from the village of Chamzinka (54°23.553 N, 45°47.858 E) through the villages Kochkurovo, Dubenki, and Morga to the border of Mordovia with the Ulyanovsk oblast (54°25.035 N, 45°31.477 E). The border from the southeast runs along the edge of the Prisursky forest (54°25.035 N, 45°31.477 E) to Bol'shie Berezniki (54°11.289 N, 45°59.468 E). The border from the west runs along the road from the village of Bol'shiye Berezniki through the villages Mariupol, Elizavetinka, Petrovka, Penkozavod, Picheury, Ivanova Polyana to Chamzinka. The stationary site is a complex of field-type agricultural land with a small amount of island forests.



Figure 1. Map of the Republic of Mordovia with habitats (red circle) of Eagle-Owl at four stationary sites (A – Prisursky, B – Kochkurovsky, C – Chamzinsky, D – Kovylninsky)

1. ábra A Mordvin Köztársaság térképe az uhu élőhelyeivel (piros kör) négy állandó helyen (A – Prisursky, B – Kochkurovsky, C – Chamzinsky, D – Kovylninsky)

Kochkurovsky stationary site is located in the southern part of the region. The area of the hospital is 450 km². The territory of the stationary site is limited from the north by a straight horizontal line running from the village of Yalga (54°06.456 N, 45°09.089 E) through the village Goryainovka to the village Podlesnaya Tavla (54°06.401 N, 45°27.408 E), from the east – a straight line running from the Podlesnaya Tavla through the villages Kochkurovo, Novaya Pyrma, Staraya Pyrma, Starye Turdaki to the border of Mordovia with the Penza oblast, from the south – along the border of Mordovia, from the west – a vertical line passing from the border of Mordovia (Ovechkin forest) (53°57.853 N, 45°09.013 E) to village of Yalga (54°06.456 N, 45°09.089 E). The stationary site is a complex of agricultural land alternating with shallow ravines and gullies.

Chamzinsky stationary site is located in the central part of the region. The area of the hospital is 100 km². The territory of the stationary site is limited from the north by a straight horizontal line running from the village Kurilovo (54°27.113 N, 45°29.899 E) to the village Kochkushi (54°27.104 N, 45°39.159 E), from the east – a straight vertical line passing from the Kochkushi through the villages Otradnoye, Syryatino, Bolshoye Maresevo (54°21.727 N, 45°39.142 E), from the south – a straight vertical line running from the Bolshoye Maresevo to the Pyatina (54°21.732 N, 45°29.908 E), from the west – along a vertical line running from the Pyatina to the Kurilovo. The stationary site is a complex of predominantly open slopes and gullies on the right bank of the Amorda River.

Kovylkinsky stationary site is located in the western part of the region. The area of the hospital is 200 km². The territory of the stationary site is limited from the north by a straight horizontal line running from the village Matveevka (54°16.005 N, 45°34.981 E) through the villages Staraya Potma, Zaitsevo, Vorona (54°16.008 N, 45°43.994 E), from the east – a straight vertical line passing from the Vorona through the villages Malyj Azyas, Izosimovka, Potma (54°05.012 N, 43°44.007 E), from the south – along a straight horizontal line passing from the Potma through the Ezhovka to the Pokrovsk (54°05.009 N, 43°34.998 E), from the west – a vertical line running from the Pokrovsk through the Mikhailovskoe to the Matveevka. The stationary site is a complex of agricultural land with a small number of forested ravines.

Stationary sites included the territories of several municipal districts (Bolshebereznikovsky, Dubensky, Chamzinsky, Kochkurovsky, Ruzaevsky, Romodanovsky, Kovylkinsky and suburbs of the city of Saransk). In addition to stationary sites, research was carried out in the territories of other districts of the northwestern, central and eastern parts of Mordovia (Temnikovsky, Krasnoslobodsky, Insarsky, Kadoshkinsky, Ardatovsky, Ichalkovsky, Atyashevsky, Lyambirsky), where separate habitats of Eagle-Owls were found. Thus, in Mordovia the Eagle-Owl reliably lives in 16 out of 22 districts (72.7%).

Artificial nests sites in the amount of 60 were arranged in Bolshebereznikovsky, Chamzinsky, Dubensky, Kochkurovsky, Ardatovsky districts. The analysis was conducted from them on 17 nests in which reproduction was detected. Artificial nests with a roof, and with a roof and walls were used by the Eagle-Owl equally. Artificial nests dug in the ground were used to a lesser extent.

Data collection

Initially, the identification of Eagle-Owl habitats was carried out by direction finding of vocalizing birds, followed by a search for nests. Then the method was applied using portable digital voice recorders Olympus VN-416PC, VN-406PC, VN-712PC (Andreychev *et al.* 2017, Lapshin *et al.* 2018). The detection radius of Eagle-Owl calls in open areas was 1–3 km, depending on the wind direction. More than 10,000 h of audio recordings were processed each year. Most of them were made during the spring-summer period (more than 7,000 h/year). The Eagle-Owl nests found on the ground were marked with dots when using a GPS navigator.

To increase the population density of Eagle-Owls, we carried out biotechnical measures in suitable areas. We arranged more than 60 artificial nests, which were subsequently occupied by Eagle-Owls (Lapshin *et al.* 2022, 2023). We studied the productivity of the Eagle-Owl in natural nest sites (1) and three types of artificial nest sites: with a roof (2), with a roof and walls (3), dug in the ground without a roof and walls (4). Since the last three types of artificial nest sites had similar significance for birds, the analysis of the results is given for a combined sample of them.

Population productivity

The productivity of Eagle-Owl pairs was assessed by the generally accepted method as the definition is number of fledgling/all breeding attempts (including failed ones). We also determined the ratio of the number of nesting events, max number of eggs/chicks per pair/year, mean number of chicks per successful pair, number of chicks for all years. Nesting event was taken as a reproduction resulting in the laying of at least one egg. Breeding is considered successful, if at least one young fledged.

Data analysis

The distance between the nearest nests, between the Eagle-Owl nest and the nearest settlement, automobile field dirt roads, paved roads, an agricultural land was set using the OziExplorer software. The calculation of the population density of Eagle-Owl on the area was carried out using the observed distance to the nearest neighbor (a statistical method used in GIS, Sergio *et al.* 2004). Data analysis was performed using MS Excel (Microsoft Corporation, Redmond, Washington, DA, USA). For proportions, 95% confidence interval (CI) was calculated using Quantitative Parasitology software, Qpweb version 1.0.15 (<https://www2.univet.hu/qpweb/qp10/index.php>).

Results

Results of studying the Eagle-Owl population for individual periods

In the study of the number and distribution of Eagle-Owl from the territory belonging to Mordovia, several periods can be distinguished:

I. period – end of the 19th century – information was given that the Eagle-Owl was a common species in the coniferous and mixed forests of the eastern part of the region. Eagle Owl calls were often recorded in the river valley Alatyr. Zhitkov and Buturlin (1906) reported that the Eagle-Owls were numerous in the southwestern corner of Ardatovsky district, in deciduous dachas near the villages of Syryatino and Maresevo (currently Chamzinsky district), where they nested in remote forest ravines. In the deciduous forests of the steppe north of the Alatyr valley, the Eagle-Owl was found much less frequently, and was not found at all in the steppe ravines and gullies. The Eagle-Owl was a common species in the forests of the river valley Sura. However, the number of breeding pairs was not known, the approximate number was not estimated, and the nests were not described.

II. period – 20th century. Until the 1960s, the number of Eagle-Owls in Mordovia was relatively high. By the middle of the second half of the 20th century, the Eagle-Owl has become rare in the republic, which was facilitated by several factors: direct persecution by humans and a reduction in the number of hare (Lugovoy 1975). The Eagle-Owl was a common species on the territory of the Mordovian Nature Reserve, where the bird mostly stayed in coniferous areas, especially where spruce grows (Ptushenko 1938). At that time, there was no data on the discovery of Eagle-Owl nests in Mordovia (Lugovoy 1975). The only nest was discovered on July 18, 1982, in the area of the village. Redkodubye (Ardatovsky district). It was located on the edge of a pine forest in a bend of the river Alatyr. There were three well-feathered chicks in the nest (Lysenkov 1994). In the 1990s, the Eagle-Owl population started to recover.

III. period – beginning of the 21st century (2000–2014). By 2005, it was known from game managers, foresters and the local inhabitants about 25 Eagle-Owl territories in the regions of Mordovia. As our subsequent surveys using voice recorders in these areas showed, Eagle-Owls were not present in all reported areas. The reason for this was that many people confused the Eagle-Owl with the Long-eared Owl (*Asio otus*), Ural Owl (*Strix uralensis*) and other owl species. However, breeding territories were already known in the vicinity of the villages Redkodubye and Lunga in the Ardatovsky district, village Nikolaevka in the Dubensky district and village Papulevo in the Ichalkovsky district (Barakhmanovskoye forestry in the Smolny National Park). New nests were discovered in the vicinity of the villages of Simkino and Engalychevo. In subsequent years, dead individuals were found, or mating calls were heard, therefore, we suppose that new territories were established in the vicinity of the villages of Supodeevka, Krasnaya Polyana, Ekonomicheskije Polyanki, and Sarast. Thus, using classical methods of counting owls until 2014, six nests and four territories of Eagle-Owls were known. These territories were subsequently subject to annual observations to study the biology and ecology of Eagle-Owls.

IV. period – first quarter of the 21st century (2015–2024). Detection of Eagle-Owl territories in Mordovia using digital portable voice recorders began in 2015 (Andreychev *et al.* 2017,

Lapshin *et al.* 2018) and by 2024, nine times more habitats were identified in the region with this method than with previously used traditional methods. By using the acoustic method, it was possible to identify the distribution and population density of the Eagle-Owl more effectively.

Results of studying the Eagle-Owl population using acoustic method

During the research period (2015–2024), 640 expeditions were carried out, 920 sound recordings were obtained and processed, with a total length of more than 84,500 hours. In total, during the research period, 89 Eagle-Owl territories, 125 nesting events, and 214 chicks were identified. Over the entire period, the birds laid 282 eggs. The most successful years in terms of the number of eggs and chicks in nests were 2014, 2018, 2022, and 2023 (*Figure 2*).

Three types of biotopes were identified, in which Eagle Owls in Mordovia live: 1) ravine-gully complexes, 2) slopes of river terraces, 3) abandoned quarries. In most cases, Eagle-Owls prefer to inhabit ravine-gully complexes (*Figure 3*), and, to a lesser extent, the slopes of river terraces and undeveloped quarries. Most of the nests were located in the spurs of the upper part of ravines, which have steeper slopes. In almost all cases, the Eagle-Owl chose the upper part of the slopes for nesting; less often, its nests were found in the middle part. The steepness of the slopes of the ravines on which the nests were located varied from 35 to 80°. The nests themselves were located on the steeper part of slopes, which were about 70–80° steep. Of the atypical ones, noteworthy is the nest located in the vicinity of the village Starye Naimany (Bolshebereznykovsky district), which was located on a flat surface 20 m from the edge of the ravine. Many nests were found in ravines overgrown with woody

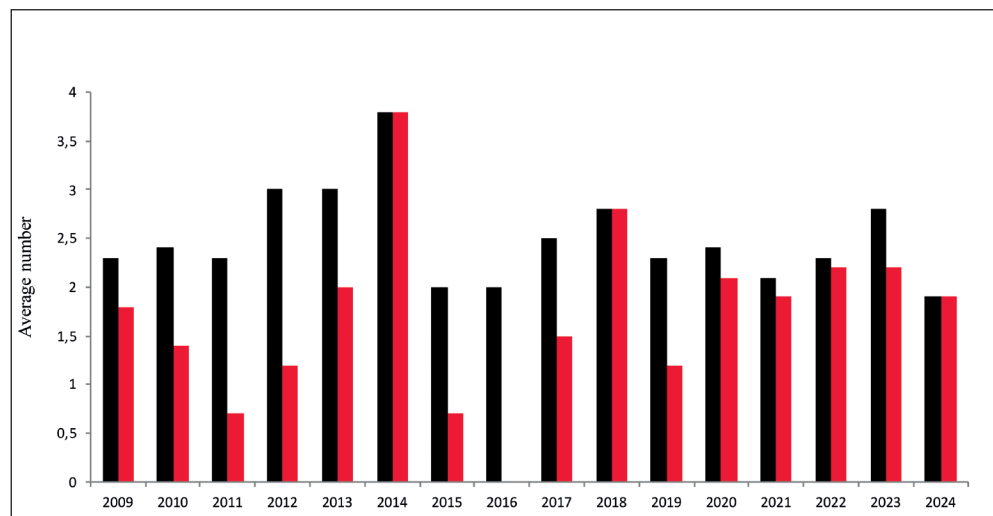


Figure 2. Average number of eggs (black bar) and fledglings (red bar) of the Eagle-Owl in Mordovia in different years (2009–2024)

2. ábra Az uhu tojásainak (fekete) és fiókáinak (piros) átlagos száma a különböző években Mordvinföldön (2009–2024)



A. Typical habitat of an Eagle-Owl in the vicinity of the village of Tazino

A. Az uhu tipikus élőhelye Tazino község környékén



B. Female at a nest in the vicinity of the village of Makolovo

B. Egy tojó a Makolovo falu közelében lévő fészkenél



C. Natural nest in the vicinity of the village of Bulgakovo

C. Természetes fészkek Bulgakovo falu közelében



D. Artificial nest box in the vicinity of the village of Cheberchino

D. Mesterséges fészkek (költőláda) Cheberchino falu közelében

Figure 3. Habitats and nests of the Eagle-Owl in Mordovia

3. ábra Az uhu élőhelyei és fészkei Mordvinföldön

vegetation. The ravines were surrounded by pine or birch plantings. In a single case, a nest was located in open ravine. In most cases, Eagle-Owls tried to settle on slopes with a southern exposure, and only in one with a northern exposure.

In the Prisursky stationary area, 38 Eagle-Owl territories were identified, in Kochkurovsky – 14, in Chamzinsky – 5, in Kovylkinsky – 7. Outside the stationary areas, habitats were noted in 25 cases, most of them were located in the Ichalkovsky, Ardatovsky and Atyashevsky districts. Productivity for couples was 76%. In 51% of nesting events, pairs laid three eggs, compared to two eggs in 43% or four eggs or one egg each in 3%. The average clutch size was 2.54 ± 0.08 ($\mu \pm \text{SE}$; $n = 125$) eggs. Over the years, the study pairs in Mordovia raised one (13.6%), two (36.4%), or three chicks (50%). The number of chicks per successful nest averaged 2.36 ± 0.11 . The mortality rate in nests was lower in artificial nesting sites (30%) ($n = 17$) than in natural ones (40.6%) ($n = 21$). The average nesting success on artificial and natural nesting sites in different periods of research in Mordovia was comparable (*Figure 4*).

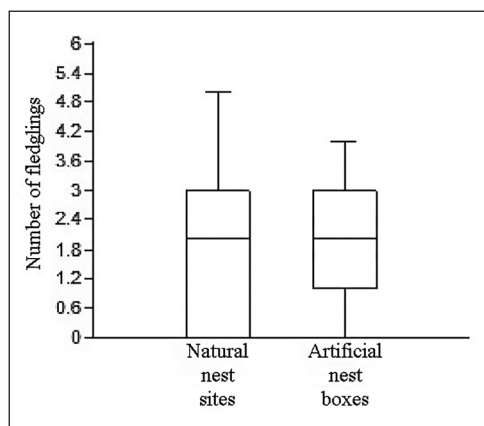


Figure 4. Number of fledglings the Eagle-Owl from natural and artificial nesting sites in Mordovia for the period of 2009–2024. Note: 'strip on box' is the median, the boundaries of the box are 25–75% quantiles, whiskers – minimum and maximum values

4. ábra Mordvinföld természetes és mesterséges fészkelőhelyeiről származó fiókák száma a 2009–2024-es időszakban

Forty-one breeding pairs were identified. At the Prisursky stationary site, 19 breeding pairs were registered, at Kochkurovsky – 9, at Chamzinsky – 3, at Kovylkinsky – 1 (Table 1).

The Eagle-Owl population density at the Prisursky station ranges from 4.3 pairs per 100 km² (1 pair/23.1 km²). The distance between neighboring nesting sites ranges from 1.2 to 9.2 km, the average is 3.2 km. The Eagle-Owl population density at the Kochkurovsky station is 3.1 pairs/100 km² (1 pair/32.1 km²). The distance between neighboring nesting sites ranges from 1.8 to 3.2 km, the average is 2.6 km. The Eagle-Owl population density at the Kovylkinsky station is 3.5 pairs/100 km² (1 pair/28.6 km²). The Eagle-Owl population density at the Chamzinsky station is 5 pairs / 100 km² (1 pair/20 km²). The distance between neighboring nesting sites ranges from 1.3 to 4.8 km, the average is 2.7 km.

Based on the degree of cover, nests were well, partially and completely covered. The degree of cover of the nest was determined by the distance from which the nest could be seen during the nesting period. Three nests had good cover, which could only be seen from several meters. Five nests had partial cover, visible from one side from 10–20 m. Most of the nests were built openly, were not covered by trees and shrubs and were clearly visible from the opposite side of the ravine, in some cases from more than 100 m.

In Mordovia, Eagle-Owl nests were located in relative proximity to populated areas. The average distance from nests to settlements was 1,550 m. The greatest distance from a nest to a settlement was 3,700 m (Engalychevo). The nests closest to populated areas were located in the vicinity of the villages of Veyse and Yanguzhinskiy Maydan (400 m).

The average distance from nests to automobile field dirt roads was 436 m. The shortest distance between a nest in the vicinity of Griboedovo and the road was 40 m. The greatest distance between a nest in the vicinity of Yanguzhinskiy Maydan and the road was 1,370 m. There were no nests located within 1 km of paved roads.

Eagle-Owl nests were located near cultivated agricultural lands. The average distance from the nest to the field was 627 m. The greatest distances from nests to agricultural land were 3 nests located in the north-eastern part of Mordovia, on the slope of the Alatyr River valley – the average distance was 1,936 m. The shortest distance from nests to fields was in the vicinity of villages Novye Turdaki (48 m), Nikolaevka (50 m), Kaybichevo (71 m), Tazino (73 m), Sosnovyj Gart (100 m).

Table 1. Breeding parameters of an Eagle-Owl population in Mordovia monitored from 2009 to 2024
 1. táblázat A mordvinföldi uhu populáció költési paraméterei 2009 és 2024 között

Nº	Nearest town	Year of first record of the territory	Number of nesting events	Number of successful nesting	Breeding success (%) as of successful / all nesting attempts	Total number of young	Mean number of young / successful pairs	Mean productivity (young / all breeding attempts)
Prisursky stationary site								
1	Simkino	2005	7	5	67	8	1.6	1.1
2	Tazino	2015	9	7	76	16	2.3	1.8
3	Chernaya Promza (north)	2015	1	0	0	0	0	0
4	Veyse	2016	3	3	100	6	2	2
5	Sosnovyj Gart	2015	3	3	100	8	2.7	2.7
6	Starye Najmany (west)	2017	3	3	100	6	2	2
7	Bol'shie Berezniki	2016	2	1	50	3	3	1.5
8	Parakino	2016	2	2	100	4	2	2
9	Starye Najmany (east)	2016	0	0	0	0	0	0
10	Chernaya Promza (west)	2016	0	0	0	0	0	0
11	Shugurovo (north)	2016	0	0	0	0	0	0
12	Chernaya Promza (north-east)	2016	0	0	0	0	0	0
13	Chernaya Promza (south)	2017	0	0	0	0	0	0
14	Starye Najmany (south-west)	2016	0	0	0	0	0	0
15	Kosogory	2016	0	0	0	0	0	0
16	Engalychevo (east)	2010	5	3	60	12	4	2.4
17	Nikolaevka (north-east) – ravine Mishin	2010	9	5	55.56	16	3.2	1.8
18	Kaybichevo (north-east)	2016	4	3	75	5	1.3	1.3
19	Engalychevo (north)	2016	3	1	33.34	2	1	0.7
20	Krasnoe Pol'co	2017	2	0	0	0	0	0
21	Cheberchino (north-east)	2016	6	6	100	14	2.3	2.3
22	Nikolaevka (near the village)	2017	4	4	100	9	2.3	2.3
23	Nikolaevka (north) – ravine Gorodok	2018	1	1	100	3	3	3
24	Purkaevo (north-west) – ravine Markov	2016	1	1	100	2	2	2
25	Yavlejka	2018	1	0	0	0	0	0
26	Krasino	2016	0	0	0	0	0	0
27	Morga	2016	0	0	0	0	0	0
28	Kaybichevo (north) – Urochishche Babrenkin	2016	0	0	0	0	0	0
29	Kaybichevo (east)	2015	0	0	0	0	0	0

№	Nearest town	Year of first record of the territory	Number of nesting events	Number of successful nesting	Breeding success (%) as of successful / all nesting attempts	Total number of young	Mean number of young / successful pairs	Mean productivity (young / all breeding attempts)
30	Engalychevo (south)	2016	0	0	0	0	0	0
31	Purkaevo (south-west) – ravine Ramzinskij	2017	0	0	0	0	0	0
32	Purkaevo (north) – river Ashnya	2017	1	0	0	0	0	0
33	Nikolaevka (north-west) – ravine Izvestnyj	2018	0	0	0	0	0	0
34	Machkazerovo (east)	2016	0	0	0	0	0	0
35	Picheury	2018	0	0	0	0	0	0
36	Ivanova Polyana	2016	0	0	0	0	0	0
37	Machkazerovo (west)	2017	0	0	0	0	0	0
38	Medaevo	2024	0	0	0	0	0	0
Kochkurovsky stationary site								
39	Bulgakovo (west)	2017	6	6	100	12	2	2
40	Novye Turdaki (east)	2017	2	2	100	4	2	2
41	Novye Turdaki (west)	2017	1	1	100	2	2	2
42	Bulgakovo (east)	2020	2	2	100	6	3	3
43	Vnukovka	2020	1	1	100	3	3	3
44	Semilej	2021	0	0	0	0	0	0
45	Novotyaglovka	2021	0	0	0	0	0	0
46	Karnaj	2021	0	0	0	0	0	0
47	Starye Turdaki (west) – ravine Zhivyarki	2021	0	0	0	0	0	0
48	Starye Turdaki (south)	2022	0	0	0	0	0	0
49	Monastyrscoe	2018	4	4	100	10	2.5	2.5
50	Kulikovka	2017	1	1	100	2	2	2
51	Griboedovo	2019	2	1	50	2	2	1
52	Akshenas	2021	1	1	100	2	2	1
Chamzinsky stationary site								
53	Makolovo	2017	4	3	75	5	1.7	1.3
54	Bol'shie Remezenki	2020	1	1	100	2	2	2
55	Kul'mino	2020	1	1	100	2	2	1
56	Ogaryovka	2020	0	0	0	0	0	0
57	Bol'shoe Chufarovo	2020	0	0	0	0	0	0
Kovylninsky stationary site								
58	Ezhovka	2017	4	4	100	8	2	2

№	Nearest town	Year of first record of the territory	Number of nesting events	Number of successful nesting	Breeding success (%) as of successful / all nesting attempts	Total number of young	Mean number of young / successful pairs	Mean productivity (young / all breeding attempts)
59	Izosimovka	2018	0	0	0	0	0	0
60	Staraya Samaevka	2018	0	0	0	0	0	0
61	Sejtyanovka	2018	0	0	0	0	0	0
62	Novaya Sazonovka	2018	0	0	0	0	0	0
63	Vysokoe	2018	0	0	0	0	0	0
64	Staraya Derganovka	2018	0	0	0	0	0	0
Habitat areas outside stationary sites								
65	Gart	2017	0	0	0	0	0	0
66	Krasnyj Poselok	2012	0	0	0	0	0	0
67	Tarhanovo	2017	2	2	100	4	2	2
68	Vedyancy	2020	1	1	100	2	2	1
69	Petrovka	2020	1	0	0	0	0	0
70	Bol'shaya Pestrovka	2020	3	3	100	5	1.7	1.7
71	Permeevo	2020	0	0	0	0	0	0
72	Lada	2020	0	0	0	0	0	0
73	Barahmanovskoe lesnichestvo	2002	7	2	28.57	5	2.5	0.7
74	Redkodubye	1982	5	5	100	8	1.6	1.6
75	Lun'ga	2005	6	5	83.34	11	2.2	1.8
76	Supodeevka	2008	0	0	0	0	0	0
77	Krasnaya Polyana	2008	0	0	0	0	0	0
78	Oktyabr'skij	2010	0	0	0	0	0	0
79	Yanguzhinskiy Maydan	2016	2	1	50	3	3	1.5
80	Payovo	2015	0	0	0	0	0	0
81	Ekonomicheskie Polyanki	2006	0	0	0	0	0	0
82	Dyurki	2015	0	0	0	0	0	0
83	Alovo	2013	0	0	0	0	0	0
84	Chukaly na Vezhne	2021	1	1	100	2	2	2
85	Sarast	2009	0	0	0	0	0	0
86	Staryj Gorod	2015	0	0	0	0	0	0
87	Srednee Pole	2015	0	0	0	0	0	0
88	Novye Verhissy	2015	0	0	0	0	0	0
89	Novaya Uda	2024	0	0	0	0	0	0
	Total	1982–2024	125	97	77.6	214	2.2	1.8

Discussion

The average number of nestlings of Eagle-Owl in Mordovia slightly less than in southeast Spain (2.84 ± 1.17 , Pérez-García *et al.* 2012) and in Israel (2.67 ± 0.13 , Hadad *et al.* 2022), but more than in France (1.69 ± 0.76 , Penteriani *et al.* 2004) and in Belarus (2, Gritschik & Tishechkin 2002). In this regard, it should be recognized that the quantitative characteristics of Eagle-Owl reproduction are quite stable across regions.

High rate of association of Eagle-Owl with ravines in Mordovia is explained by their high density of ravines – more than 1 km of ravines per 1 km² in the study area and the relatively large number of nesting sites in them. Such selectivity when choosing a place to occupy nest site a nest may be due to the inaccessibility of Eagle-Owl nests to humans and the presence of forest cover in ravines. The presence of a large number of ravines with stands determines the high population density of Eagle-Owl in Mordovia. The influence of local conditions on the population density of Eagle-Owl has been shown for other regions as well. In southeast Spain, a high-density population of the Eagle-Owl in Alicante (100–120 pairs, approximately 2.03 pairs/100 km², Martínez & Zuberogoitia 2003) and Murcia (182–220 pairs, approximately 1.78 pairs/100 km², Sánchez-Zapata *et al.* 1995) has been reported. Somewhat later, an even higher population density (22.01 territories/100 km²) was revealed for the southeast of Spain across 99 different Eagle-Owl territories (Pérez-García *et al.* 2012).

Given the population density of the Eagle-Owl on stationary sites in Mordovia, it can be determined that one pair account for from 20 to 32.1 km² of territory. These results are consistent with those obtained by researchers using different methods in other parts of the range. Based on the satellite-tracking data in Hungary, the habitat of one Eagle-Owl can cover an area of about 18 km² (Prommer *et al.* 2018). According to other researchers in the Caucasian Mineral Waters region in the Stavropol oblast (Russia), the habitats in some cases can be 60–80 km² (Ilyukh 2017).

The high density of the Eagle-Owl population in Mordovia in stationary sites can be explained by the fact that previously artificial nest sites were established here in suitable areas. We know that the dispersal distance of juveniles is usually between 4 and 8 km from the natal site, although some data suggest that it may be much greater (13 to 72 km) (Nygård *et al.* 2023). A study of Eagle-Owls radio-tagged in the Sierra Morena, 20 km north of Seville, Spain, showed that young Eagle-Owls remained in their parents' range until the end of August, and then dispersed over distances ranging from 1.6 to 34.9 km (average 9.1 km) (Delgado & Penteriani 2005). Therefore, we believe that the presence of conditions for nesting in the nearest ravines determines a shorter distance for the settlement of young individuals and, accordingly, the optimal population density at stationary sites.

Considering that on a total area of four stations of 1,650 km², 64 habitats of the Eagle-Owl were registered, by extrapolating this number, up to 1,032 habitats can be located on the entire area of Mordovia of 26,200 km². However, we understand that this number is not achievable, since when choosing stations for surveys, we were guided by optimal conditions for the Eagle-Owl. The Eagle-Owl population density in habitats outside of stationary sites is somewhat lower. In addition, we identified most of the Eagle-Owl's habitats in open spaces that surround populated areas, agricultural lands, or on the edges of forests. The

Eagle-Owl settles in the forests of Mordovia much less frequently than in open landscapes. Taking into account the fact that the area of open spaces in Mordovia is 18,800 km², and research was carried out on an area of open spaces of 4,000 km² (89 sites were identified), the possible population density in the open spaces of the region can reach 583 habitat sites.

We can conclude that by analyzing literature sources on other regions of Russia, the population density of the Eagle-Owl in Mordovia is one of the highest of the surveyed populations. In particular, in Kalmykia, 36 habitats of the Eagle-Owl are known (the possible number in the region could be 293 pairs), which corresponds to 1.29 pairs/100 km² (Abushin 2021). The total number of Eagle-Owls in the Stavropol oblast is estimated at 150–180 breeding pairs (Fedosov 2019), in the Rostov oblast up to 200–300 pairs (Belik 2014) in the Volgograd oblast

Table 2. Population estimates (number of pairs) of Eagle-Owls in different countries

2. táblázat Az uhupopulációk becslése (párok száma) a különböző országokban

Country	Area, km ²	Number of pairs	References
Macedonia	25,333	100–300	Velevski <i>et al.</i> 2010
Albania	28,748	50–100	Penteriani & Delgado 2019
Slovenia	20,253	120–140	Mihelič & Marčeta 2000
Montenegro	14,026	225–300	Cochet 2006
Turkey	23,900	3000–6000	Penteriani & Delgado 2019
Moldova	33,843	5–10	Cochet 2006
Belgium	32,528	80–85	Vangeluwe <i>et al.</i> 2010
Slovakia	48,845	300–400	Danko & Karaska 2002, Dravecky & Guziová 2012
Bosnia and Herzegovina	51,129	400–500	Kotrošan & Hatibović 2012
Poland	312,685	250–280	Wójciak <i>et al.</i> 2008
Italy	301,340	250–340	Brichetti & Fracasso 2006
Bulgaria	110,910	420–490	Hristov <i>et al.</i> 2007
Greece	131,940	200–500	Penteriani & Delgado 2019
Portugal	92,082	250–580	Cabral <i>et al.</i> 2005, Lourenço <i>et al.</i> 2015
Serbia	88,361	330–450	Penteriani & Delgado 2019
Spain	498,508	2350	Martínez & Zuberogoitia 2003
France	547,030	1649 950–1500	Cochet 2006 Martin 2010
Romania	237,500	1000–1500	Penteriani & Delgado 2019
Germany Schleswig-Holstein	357,021 1900	1100 58	Lanz & Mammen 2005 Klose & Koop 2007
Croatia	56,542	800–1200	Barišić <i>et al.</i> 2016
Czech Republic	78,866	600–900	Štastný <i>et al.</i> 2006, Hora <i>et al.</i> 2010
Belarus	207,600	400–500	Gritschik & Tishechkin 2002
Norway	385,186	451–681	Øien <i>et al.</i> 2014
Sweden	449,964	474	Hellström & Helander 2012

up to 200–300 pairs (Gugueva & Belik 2013). The number of Eagle-Owl for the Sverdlovsk oblast – 850 pairs (147 nesting territories identified), the Chelyabinsk oblast – 150 pairs (135 nesting territories identified) (Karyakin 1998). In the regions of the Volga, the population differs significantly. The region with a known high number of Eagle-Owls is the Republic of Bashkiria – 850 pairs (identified 374 nesting territories). A slightly smaller number is known for the Perm kray. According to one estimate, 500 pairs may live here (225 nesting territories have been identified (Karyakin 1998). According to other estimates, the number varied in different years from 120 to 330 pairs (Shepel 2011). In the Saratov oblast, there were 125–140 pairs of Eagle-Owl (Zavyalov & Tabachishin 2006). A relatively small number (up to 15 pairs) is known for the Chuvash Republic and the Mariy El Republic (Shepel 2011).

In countries with similar territorial areas, the number of Eagle-Owls is comparable to Mordovia, for example, in Macedonia (25,333 km²) from 100 to 300 pairs (Velevski *et al.* 2010), Albania (28,748 km²) from 50 to 100 pairs (Penteriani & Delgado 2019), in Slovenia (20,253 km²) from 120 to 140 pairs (Mihelič & Marčeta 2000), in Montenegro (14,026 km²) from 225 to 300 pairs (Cochet 2006). The various data can be seen in Table 2. High Eagle-Owl population (3–6 thousand pairs) is typical for Turkey (23,900 km²) (Penteriani & Delgado 2019). Lower numbers in relation to area are noted for Moldova (from 5 to 10 pairs on an area of 33,843 km²) (Cochet 2006) and Belgium (from 80 to 85 pairs on an area of 32,528 km²) (Vangeluwe *et al.* 2010).

Assessing not only the factors that have a positive impact, but also the negative impact, it is necessary to characterize the relationship of the Eagle-Owl with other species in the region. Considering our previous results in the region, based on studies of co-vocalization (Andreychev 2023, Andreychev *et al.* 2023) and the diet of competitive owl species (Andreychev & Lapshin 2017), we can conclude that the Eagle-Owl has no enemies. The large owl, the Ural Owl, is found in the Eagle-Owl's diet, but Eagle-Owl remains are not found in its diet. This is also indirectly evidenced by the fact that the Eagle-Owl vocalizations are recorded during the vocalizations of predatory mammals in the family Canidae.

Thus, the abundance of Eagle-Owl in Mordovia has been determined. This can be used to estimate the total number of the subspecies *B. b. ruthenus* throughout its range. This study provides data on the density of the Eagle-Owl in Mordovia, but further research is needed to obtain more information on the ecology of the species in the Middle Volga region. For example, it would be particularly interesting to test whether Eagle-Owl population densities are comparable in other regions during the same study years. Each study on monitoring the status of the Eagle-Owl population is important, as it contributes to the general knowledge of the species and allows us to evaluate the effectiveness of conservation measures.

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Vulnerable bird species and avian mortality due to predation by domestic cat (*Felis catus*) in anthropogenically transformed landscapes of the Bukovynian Carpathians

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Abstract The domestic cat (*Felis catus*) poses a significant threat to bird fauna, particularly in the urbanised, anthropogenically transformed landscapes of the Bukovynian Carpathians, Ukraine. This study aimed to determine the annual bird mortality rate due to cat predation and identify vulnerable species to this threat. Using the Monte Carlo method with 10,000 iterations, mortality rates were calculated. Data were gathered from sociological surveys, field observations, and scientific publications. The methodology included estimating the number of rural pets and feral cats across 877.9 km², calculating population density, and assessing bird predation. Annually, an average of 509,000 birds are killed, with a 95% mortality estimate range between 188.6 thousand and 1,008.6 thousand, which highlights the cats' impact on local bird populations. Linear regression models were developed to predict bird mortality based on cat population size. The regression formulae are $nBFC = 44.10 * nFC$ for feral cats and $nBRP = 32.92 * nRP$ for rural pets. A total of 62 vulnerable bird species were identified in these landscapes of the Bukovynian Carpathians.

Keywords: Ukrainian Carpathians, cat predation, Monte Carlo simulation, cat population density, bird species conservation.

Összefoglalás A házimacska (*Felis catus*) jelentős veszélytető tényező a madárvilágra nézve, különösen a Bukovinai-Kárpátok (Ukrajna) urbanizált, ember által módosított tájain. Jelen tanulmány célja a házimacska okozta éves madárpusztulási arány meghatározása és az e veszélynek kitett fajok azonosítása volt. A 10 000 iterációval végzett Monte Carlo-módszerrel számoltuk ki a mortalitási rátákat. Az adatokat szociológiai felmérésekből, terepi megfigyelésekből és tudományos publikációkból gyűjtöttük. A módszertan magában foglalta a vidéki háziállatok és elvadult macskák számának becslését 877,9 km²-es területen, a populációsűrűség kiszámítását és a madár zsákmányolás értékelését. Évente átlagosan 509 000 madarat pusztítanak el a macskák, a 95%-os becsült mortalitás 188 600 és 1 008 600 közé esik, ami rávilágít a macskák helyi madárpopulációra gyakorolt jelentős hatására. Lineáris regressziót használtunk a madárpusztulás előrejelzésére a macskapopuláció nagysága alapján. A regressziós képletek a következők: $nBFC = 44,10 * nFC$ az elvadult macskák esetében és $nBRP = 32,92 * nRP$ a háziállatok esetében. A Bukovinai-Kárpátok tájain összesen 62 veszélyeztetett madárfajt azonosítottunk.

Kulcsszavak: Ukrán Kárpátok, macskák predációja, Monte Carlo szimuláció, macskák populációsűrűsége, madárfajok védelme

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Introduction

Since the 1600s, at least 33 bird species have gone extinct due to predation by domestic cat (Winter & Wallace 2006). Cats pose a serious threat, and research from various countries indicates significant bird mortality due to predation. In the United States, between 1 and 4 billion birds die annually (Loss *et al.* 2013), in Canada between 105 and 348 million per year (Blancher 2013), 27 million annually during the spring-summer period in the United Kingdom (Woods *et al.* 2003), and between 100,000 and 300,000 during the spring each year in Switzerland (Tschanz *et al.* 2011). Cat predation is one of the most significant factors in bird mortality resulting from human activity (Loss *et al.* 2013), surpassing mortality from collisions with glass surfaces and other engineering structures (Erikson *et al.* 2005), industrial solar power stations (Walston *et al.* 2016, Visser *et al.* 2019, Kosciuch *et al.* 2020), wind power stations, or traditional sources of electrical energy (Sovacool 2009). Domestic cats rank second in the number of birds preyed upon (29% of the total) after the Northern Goshawk (*Accipiter gentilis*) in the US and Southern Canada (Dunn & Tessaglia 1994). In several countries, the domestic cat is classified as an invasive alien species (Winter & Wallace 2006). The number of domestic cats in Western Europe has increased since the 1970s, resulting in up to 26.3% of birds being killed due to predation (Pavisse *et al.* 2019).

To more accurately estimate the prey hunted by domestic cat, researchers distinguish between owned cats and stray cats based on their conditions and behavioral characteristics. A stray cat is an animal without care or an owner, living on its own and foraging for food independently. P. Baker and colleagues use the term «semiferal/feral cats» to describe this group of animals (Baker *et al.* 2010), while other definitions include «un-owned cats» (Loss *et al.* 2013) or «free-ranging cats» (Winter & Wallace 2006). Owned cats, in turn, are divided into urban (urban pet) and rural (rural pet) categories (Blancher 2013).

It is known that accurately counting instances of domestic cat predation is challenging due to the difficulty in documenting such events. Only a certain percentage of prey is brought to the homes by cats and can be identified, while the rest of the prey is consumed in natural conditions (Toner 1956, Barratt 1998, Woods *et al.* 2003, Tschanz *et al.* 2011, Thomas *et al.* 2012, Blancher 2013). Therefore, adjustment factors are introduced for a more accurate estimation of bird mortality (George 1974, Kays & DeWan 2004, Thomas *et al.* 2012). Recent studies using cameras on collars have found that only 23% of prey is brought to homes by domestic cats (Lloyd *et al.* 2013). Analysis of stomach contents has shown that the ratio of consumed to brought prey is 11.4 (Krauze-Gryz *et al.* 2012).

Currently, research on domestic cat predation has been conducted in the following countries: Australia (Barratt 1998), Canada (Blancher 2013), USA (Loss *et al.* 2013), United Kingdom (Churcher & Lawton 1987, Woods *et al.* 2003, Thomas *et al.* 2012), Sweden (Liberg 1984), Switzerland (Tschanz *et al.* 2011), Poland (Krauze-Gryz *et al.* 2012, 2018), France, and Belgium (Pavisse *et al.* 2019), Croatia (Lanszki *et al.* 2016), Hungary (Biró *et al.* 2005). A comparative analysis of the diet of feral and house cats, and wildcat in Europe, was the focus of a recent study (Széles *et al.* 2018). A global synthesis and assessment of free-ranging domestic cat diet was made by Lepczyk *et al.* (2023).

As of 2013, in Canada, the number of urban and rural cats was 8.5 million, while the number of feral cats ranged from 1.4 to 4.2 million (Blancher 2013). Thus, the percentage of feral cats in Canada varied from 14% to 33% of the total number. In the USA, there are approximately 70 million feral cats, and nearly 25% of their diet consists of birds (Web1), with the total number of pet cats in the USA reaching up to 82 million (Web2). Thus, feral cats constitute over 46% of the total number in the USA. In Western Europe, the number of feral cats remains completely unknown as of 2019 (Pavisse *et al.* 2019).

In Ukraine, data on bird mortality due to domestic cat predation is currently unavailable, and thorough studies have not been conducted, except for some individual reports (Merzlikin 1998, Koshelev & Osytkovska 2017).

The aim of this study is to establish the probable annual range of bird mortality due to domestic cat predation in the anthropogenically transformed landscapes (ATL) of the southwestern Bukovynian Carpathians, within the former Putyla district, now encompassing the Konyatyn, Putyla, Selyatyn, and Ust-Putyla territorial communities (TC) of the Vyzhnytsia district in the Chernivtsi region, Ukraine. This is the first time that a preliminary assessment of mortality and identification of the most affected species are being conducted. The results of the study will help understand the scale of the problem and identify the most vulnerable species in need of protection. The ultimate goal is to identify bird species vulnerable to domestic cat predation, based on research from other countries and criteria for species vulnerability developed as a result. Data analysis will provide recommendations for mitigating the negative impact of domestic cats on bird populations and will contribute to the conservation of biodiversity in these areas.

Material and Methods

Calculation of bird mortality due to domestic cat predation

In this study, all cats owned by people are considered rural pets, while animals without owners that live independently are referred to as feral cats. The approach used to calculate bird mortality due to domestic cat predation in the ATL of the southwestern Bukovynian Carpathians involved estimating the number of rural pets and feral cats in the area and the average number of birds hunted by one cat per year (Blancher 2013). The total number of bird fatalities was calculated using the formula:

$nB = nBRP + nBFC$, where $nBRP$ is the number of birds hunted by the rural pet population per year, and $nBFC$ is the number of birds hunted by the feral cat population per year.

$nBRP = nRP * BpRP * koefRP$, where nRP is the population size of rural pets in the ATL of the southwestern Bukovynian Carpathians; $BpRP$ is the average number of birds hunted by one rural pet per year; and $koefRP$ is the correction coefficient for rural pet.

$nBFC = nFC * BpFC$, where nFC is the population size of feral cats in the ATL of the southwestern Bukovynian Carpathians; $BpFC$ is the average number of birds hunted by one feral cat per year.

The above formulas are similar to Blancher's equations (2013), but we did not use the proportion of rural pets that are allowed outdoors. All surveyed owners reported that their cats are not kept isolated indoors but have free access to the surrounding environment, which poses a threat to small mammals and birds. Therefore, the proportion of rural domestic cats with access to the outdoors is considered to be 1.

Estimation of the rural pet population

Given that there is no official census of rural or urban pets in Ukraine, the author had to employ a sociological approach to estimate rural pets population. This approach was based on a survey of residents of private households and apartments in multi-story buildings, primarily in Putyla township and in nearby villages, to determine the presence and number of cats in each household. Similar surveys, though on a much larger scale, have been conducted by colleagues in Poland (Krauze-Gryz *et al.* 2018) and in the USA (APPA 2019). The number of private households in the southwestern Bukovynian Carpathians was determined by the number of household consumers recorded by the Putyla Central Operations Centre of Chernivtsi Regional Energy Supply Company LLC («CRESC» LLC). Based on the survey results, a statistical calculation was made of the number of rural pets per household in Putyla township and in the aforementioned TC. The average number of rural pets per household in Putyla township and in the southwestern Bukovynian Carpathians was taken as the mean value of the survey sample for these areas. The obtained values were extrapolated to the entire study area by multiplying the average number of pets per household by the number of households in Putyla township or in the specified TC according to the formula:

$nRP = nH * nRPPH$, where nRP is the total number of rural pets in the study area; nH is the number of households; and $nRPPH$ is the number of rural pets per household.

To ensure the representativeness of households, a stratified sample was used that included households from different socio-economic groups and settlements. Such an approach allowed us to reduce the error and maintain the representativeness of the results.

Statistical analysis of the obtained results

Samples formed through monitoring and surveys were analyzed using statistical methods. The calculation of the minimum sample size for Putyla township and the southwestern Bukovynian Carpathians was performed using the online calculator from the Laboratory of Sociological Research at Vinnytsia National Technical University, Ukraine (<https://socio-lab.vntu.edu.ua/calculator/>). The following criteria were set to obtain the result: confidence level – 99%; margin of error – $\pm 10\%$.

The mean value of the sample obtained from the study was determined using the formula $\bar{X} = \sum x_i n_i / n$, where x_i represents the variant values (individual values in the sample), n_i is the frequency of the variant (the number of times the value x_i appears in the sample), and n is the total number of values in the sample (the sum of all frequencies).

The deviation was calculated using the formula $d_i = a_i - \bar{X}$, where d_i is the difference between the current and mean values; a_i is the current value in the sample; \bar{X} is the mean value.

The mean deviation was calculated using the formula $\sigma_a = \sqrt{\frac{1}{N-1} \sum_1^N d_i^2}$, where N is the number of sample elements, and d_i is the deviation.

The standard deviation was calculated using the formula $\sigma_a = \sigma_a / \sqrt{N}$, where σ_a is the mean deviation and N is the number of sample elements.

The relative error was calculated using the formula $\Delta a = \frac{\sigma_a}{\bar{X}} 100\%$, where Δa is the relative error ($\pm\%$), σ_a is the standard deviation, and \bar{X} is the sample mean (Ploskonos 2012).

To automate the calculation processes, these formulas were interpreted according to the syntax of the R programming language, using the free software RGui (R Console) version R-4.4.1 as the environment for statistical computations.

Estimation of the average identified number of birds hunted by rural pets per year

This parameter is crucial for a reliable assessment of bird mortality due to cat predation. Considering that no thorough studies have been conducted in Ukraine, it is reasonable to rely on data from other temperate climate countries where such information is available.

Studies on birds hunted by rural pets are usually based on the number of prey detected by owners, but this method is not accurate since some prey is consumed by the cat without being brought home. The average number of identified birds killed per year in Canada ranges from 2.8 to 14 birds per year for rural pets, while feral cats kill between 24 and 64 birds per year (Blancher 2013). These ranges were used for the research in this study.

Given the lack of data from Ukraine, especially from the mountainous areas of the Ukrainian Carpathians, the author applied the method of extrapolating results from countries with a temperate climate where such studies have been conducted. The extrapolation method involves transferring known patterns to areas where such patterns are unknown.

Blancher's (2013) research, specifically the establishment of ranges for identified bird fatalities per year, was based on an analysis of research from various countries. Of course, this method has certain drawbacks, as there may be climatic differences and other local factors. The diet of rural pets and feral cats can also include small mammals, which can sometimes be more significant than birds. Therefore, this may reduce the proportion of birds killed by cats in different territories or countries.

Assuming that the diet of cats in the Bukovynian Carpathians is different and may somewhat vary from the data of our Canadian colleagues, these inaccuracies are mitigated by the application of the Monte Carlo method, which accounts for uncertainty and variations in the data. The Monte Carlo method employs random modeling to estimate the probability of different outcomes, allowing for a wide range of possible scenarios and variations in the data to be considered.

Thus, even if the diet of cats in the Bukovynian Carpathians differs from that in Canada (Blancher 2013), the Monte Carlo method helps ensure the accuracy of our estimates by reducing the impact of potential inaccuracies in the data and providing a clearer understanding of the actual impact of cats on bird populations in this region.

Determining the value of the correction factor for calculating instances of successful hunting by rural pets (koefRP)

Findings of rural pet prey near private residences do not provide a complete picture of their numbers because it is difficult to investigate all instances of successful hunting, as cats bring only a small percentage of their prey (Krauze-Gryz *et al.* 2012, Loyd *et al.* 2013). G. Toner reports that observations of 4 cats on a farm in Canada produced the following results: one cat brought all the prey, another about half, and the other two did not bring their prey but consumed it in close proximity to the hunting site (Toner 1956). Various authors estimate the value of the correction factor for counting bird mortality due to rural pet predation, ranging from 2 to 5.8 (George 1974, Krauze-Gryz *et al.* 2012), with most leaning towards a value of 3.3 (Baker *et al.* 2005, 2008, Maclean *et al.* 2008, Thomas *et al.* 2012). Studies using video cameras embedded in cat collars yielded a result of 4.34 (23%) (Loyd *et al.* 2013). In analyzing bird mortality due to rural pet predation in Canada, the correction factor was chosen in the range of 2 to 5.8 (Blancher 2013). Considering that the studied area of the Bukovynian Carpathians is mountainous, like much of Canada, and the fact that the upper limit of the correction factor (5.8) is based on stomach content analysis (Krauze-Gryz *et al.* 2012), which is considered the most accurate method, a correction factor range for rural pets (koefRP) from 2 to 5.8 was chosen.

Estimation of the feral cat population

Determining the population of feral cats is scientifically challenging, but this estimate is crucial for assessing mortality, as these animals are the primary predators of birds. This study used personal observations and local surveys to estimate the feral cat population in the specified area.

Extrapolation and spatial modeling methods were employed to assess the feral cat population. A representative model area in the center of Putyla township, with an area of 1.0 km², was selected. This area included all potential habitats for feral cats, such as residential buildings (private homes and apartment blocks), commercial infrastructure (shops, eateries), administrative buildings, a park predominantly consisting of deciduous trees, a stadium and sports ground, coniferous forest and riverbank areas. Feral cats are known to prefer locations with access to resources, particularly food (Tennent & Downs 2008). Based on this criteria, potential gathering spots for feral cats within the model area were identified (places where food waste is generated, apartment buildings, high-traffic areas, such as shops). Observations and local surveys were conducted to determine the number of feral cats at these locations (*Table 1*). The population density of feral cats in the control area was calculated using the formula:

$D_{FCCP} = N_{FCI} / S_{CP}$, where: D_{FCCP} – feral cat population density in the control area; N_{FCI} – total identified number of feral cats in the control area; S_{CP} – area of the control area (1.0 km²). This value was extrapolated to the total area of Putyla township using the formula:

$N_{FCP} = D_{FCCP} * S_{PUT}$, where: N_{FCP} – total number of feral cats in Putyla township; S_{PUT} – area of the settlement.

Table 1. Results of the study on the population size and density of rural pet and feral cat in the southwestern Bukovynian Carpathians

1. táblázat A háziállatként tartott és elvadult macskák populációmérete és denzitása a Délnyugat-Bukovinai-Kárpátok különböző területein

Settlement, community	Area, km ²	Number of house-holds	Number of rural pets, ind.	Density of rural pets, ind. /km ²	Number of feral cats, ind.	Density of feral cats ind. /km ²	Number of domestic cats, indiv.	Density of domestic cats ind. /km ²
Putyla town-ship (Putyla TC)	13.7	2,192	2,234±357	163±26.08	343±19	25±1.38	2,577±554	188±40.42
Konyatyn TC	122.0	Not examined in terms of communities.						
Putyla TC	231.6							
Selyatyn TC	367.7							
Ust-Putyla TC	156.6							
Together across communities:	877.9	9,539	13,599±1,904	15.49±2.17	1,493±82	1.70±0.09	15,092±3,245	17.19±3.70

To determine the number of feral cats per dwelling in Putyla township, the formula was used: $N_{FCPH} = N_{FCP} / nH_p$, where: N_{FCPH} – number of feral cats per dwelling; nH_p – total number of dwellings in Putyla township. To extrapolate the number of stray cats to the scale of ATL in the southwestern Bukovynian Carpathians, the formula used was:

$nFC = N_{FCPH} * nH$, where: nFC – feral cat population size in ATL of the southwestern Bukovynian Carpathians; nH – number of dwellings in the ATL of the specified areas.

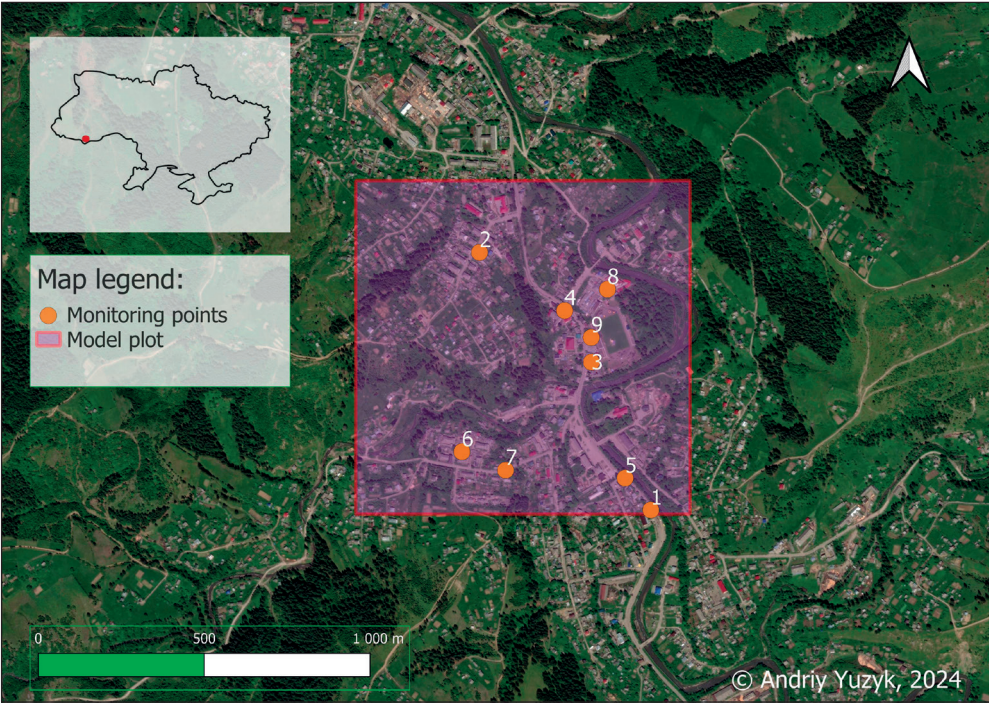


Figure 1. Map-scheme of the model area for feral cat monitoring in Putyla township
1. ábra Az elvadult macskák mintaterülete Putyla körzetében

For mapping the model area and monitoring points, the free software QGIS ver. 3.18.1 Zürich, QuickMapServices plugin, and Bing Satellite raster data were used. Coordinates of the vertices of the model area: 25.08920738, 47.99324544; 25.08920738, 48.00225444; 25.07570438, 47.99324544; 25.07570438, 48.00225444 (*Figure 1*).

To prevent the same individuals of feral cats from being counted multiple times in the study area, the author focused on the physical characteristics of the cats, such as size, fur color, and the presence or absence of certain features (e.g. spots, stripes). Additionally, traits like the absence of a collar and the overall health condition of the cats (e.g. scars, injuries) were taken into account for the identification of stray cats. Whenever possible, staff members of the institutions were surveyed to determine whether the cat was stray or had an owner and was simply roaming the streets.

The counts were conducted using a route method at least 4 times a month. Each time, the number of animals at the monitoring point was recorded, after which the average value for the month was calculated as the sum of all detections divided by the number of observations.

In the case of using information from third-party observers, the author attempted to independently verify the data. In the event of discrepancies, preference was given to personal observations.

Table 2. Encounters of feral cat from January to June 2024 at the model site in Putyla township
2. táblázat Az elvadult macskák megfigyelései 2024 január-június időszakában Putyla területén

Monitoring point number	Location	Monthly abundance, ind.						Source of information
		1	2	3	4	5	6	
1.	Ukrains'ka Street, "Norma" Store	2	1	1	2	1	1	Personal observation
2.	Meshkova Street, Building 4, 8, 12	12	11	10	12	12	16	Balan, S., oral pres.
3.	Ukrains'ka Street, Library	2	2	2	2	2	2	Boychuk, S., oral pres.
4.	Ukrains'ka Street, "Horyanka" Restaurant	6	3	4	5	4	4	Matsenko, M., oral pres., personal observation
5.	Corner of Ukrains'ka Street and Alekseev Street, Pizzeria	1	2	1	1	2	1	Personal observation
6.	Franko Street (Apartment Buildings)	0	0	0	0	0	0	Mykyrtuchyan, L., oral pres.
7.	Franko Street, Grocery Store	0	0	0	0	0	0	Personal observation
8.	Ukrains'ka Street, Market	2	1	1	2	2	2	Personal observation
9.	Ukrains'ka Street, "Classic" Store (near the stadium)	5	4	2	0	0	2	Personal observation
Total by months:		30	24	21	24	23	28	
Average number for the period:		25 ± 1.38						

The basis for estimating the population of feral cats in the specified areas was settlements. The presence of feral cats in forests is unlikely, as these animals tend to search resources, particularly food (Tennent & Downs 2008), and rarely venture deep into the forests (Kays & DeWan 2004), unless it is an isolated area where there are human dwellings.

Coniferous or mixed coniferous-deciduous forests that cover the Bukovynian Carpathians are poorly suited for the permanent presence of feral cats for several reasons. There are no anthropogenic food sources here, which often consist of garbage dumps or feeding by humans. In the forests, there are significantly fewer places for shelter, in contrast to populated areas where animals can stay in abandoned buildings, basements, attics, agricultural structures, etc., which provide protection from predators and adverse environmental conditions. Forests are habitats for predatory mammals capable of preying on feral cats, such as the Brown Bear (*Ursus arctos*), Grey Wolf (*Canis lupus*), domestic dog (*Canis familiaris*) (mostly feral individuals), European Wildcat (*Felis silvestris*), and Eurasian Lynx (*Lynx lynx*), among others. Small mammals, particularly rodents, tend to be more numerous in human settlements, which in turn attract cats. These assertions are supported by the author's personal observations, as he has not encountered feral cats in the forests of the Bukovynian Carpathians during extensive expeditions.

Determining the population density of rural pets

The population density of rural pets was assessed for the largest settlement, which is Putyla township. The area of this settlement was determined according to the official source (letter from the State Geocadastr of the Chernivtsi Region No. III-72/0-68/0/63-20 dated October 16, 2020) and for the southwestern Bukovynian Carpathians as a whole. The area of the settlements in these mountainous regions was determined as the total sum of the areas of Koniatyn, Putyla, Selyatyn, and Ust-Putyla TC (<https://decentralization.ua>).

The population density of rural pets in these areas was calculated using the formula: $D_{RP} = n_{RP} / S$, where: D_{RP} – population density (number of rural pets per unit area, km²); S – area of the studied territory.

Similarly, the population density for feral cats (D_{FC}) and the overall population density (D_{TOTAL}) were calculated.

Application of the Monte Carlo method for the overall assessment of bird mortality and the construction of linear regression models

To achieve a more accurate estimation of the annual bird mortality rate due to domestic cat predation in the ATL of the southwestern Bukovynian Carpathians, the Monte Carlo method was applied. For this purpose, a random value of a specific parameter was selected from the range of probable values listed in *Table 3*. These random parameters were then multiplied according to the formulas mentioned above, and all the multiplication results were recorded. A total of 10,000 iterations were performed, followed by calculations of the average value and relative error of the obtained samples for rural pets and feral cats, frequency distribution of all iterations, and the construction of regression models. To achieve this, the bird mortality

Table 3. Parameters and value ranges used for calculating bird mortality due to domestic cat predation in the anthropogenically transformed landscapes of the southwestern Bukovynian Carpathians

3. táblázat A házimacska okozta madárpusztulás kiszámításához használt paraméterek és értéktartományok a Délnyugat-Bukovinai-Kárpátok területén

Parameters	Acronym	value ranges	Data source
Number of rural pets, ind.	nRP	11695–15503	Original research
Percentage of rural pets with access to the outdoors	pORP	100% (all surveyed owners, was not considered)	Original research
Number of feral cats, ind.	nFC	1411–1575	Original research
Average identified number of birds preyed upon by rural pets per year	BpRP	2.8–14.0	Blancher 2013
Correction factor for calculating successful hunting incidents by rural pets	koefRP	2.0–5.8	Krauze-Gryz <i>et al.</i> 2012, Blancher 2013
Average number of birds preyed upon by feral cats per year	BpFC	24–64	Blancher 2013

Table 4. Preliminary assessment of annual bird mortality in the ATL of the southwestern Bukovynian Carpathians based on values from *Table 3* and using the Monte Carlo method (10,000 iterations)

4. táblázat Az éves madárpusztulás előzetes becslése a Délnyugat-Bukovinai-Kárpátok területén a 3. táblázat értékei alapján Monte Carlo-módszerrel (10 000 iteráció) számolva

	Total	Rural pet	Feral cat
Cat population, thousands	15.1±21.5%	13.6±14.0%	1.5±5.5%
Bird mortality, thousands	509,0 (188.6–1008.6)	443,0 (127.0–942.5)	66,0 (37.6–95.1)

calculation formulas for domestic and stray cats were interpreted according to the syntax of the R programming language, and the code was executed in the aforementioned software environment. The results are presented in *Table 4* and *Figures 2–4*. The specialized ggplot2 library for RGui was used to create the graphs.

As is known, the general formula for linear regression is: $y = \beta_0 + \beta_1 x$. In this formula, y is the dependent variable (annual number of birds killed by feral cats or rural pets), x is the independent variable (population size of feral cats or rural pets), β_0 is the intercept, representing the value of y when $x=0$, β_1 is the slope coefficient, which indicates how much y changes with a one-unit change in x . In predicting the number of bird killings based on cat population size, it is logically assumed that if the predator population size x is 0, the number of victims would also be 0, as there would be no cats to kill birds. Therefore, in our linear regression models for estimating bird mortality due to feral cats and rural pets, the intercept β_0 should be 0 (or simply absent), making the general equation $y = \beta_1 x$.

The next step in deriving the linear regression formula is to estimate the relationship between mortality values and cat population size based on predicted data (β_1 based on the analysis of 10,000 Monte Carlo simulations). This can be done using the least squares method, which was used to build the linear regression model without an intercept, seeking

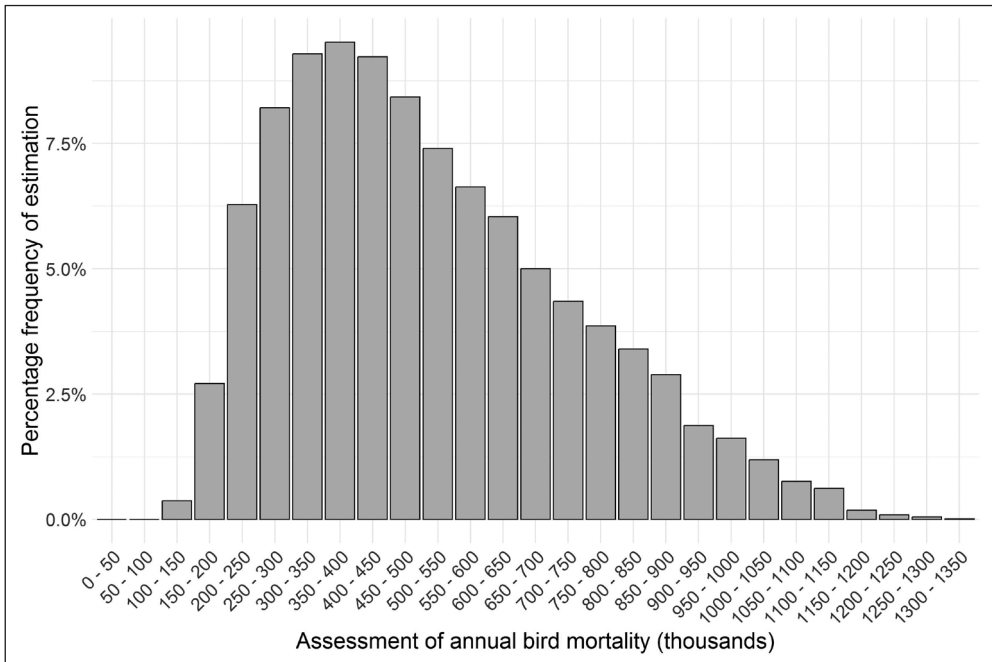


Figure 2. Distribution of the percentage frequency of annual bird mortality estimation due to cat predation in the ATL of the southwestern Bukovynian Carpathians

2. ábra A macskák által okozott éves madárpusztulás becsült százalékos gyakoriságának megoszlása a Délnyugat-Bukovinai-Kárpátok területén

the line that best describes the relationship between cat population and bird mortality. The quality of the model was assessed by the coefficient of determination R^2 , which indicates the proportion of variation in the dependent variable explained by the model. The closer R^2 is to 1, the better the linear regression model explains the variation in the dependent variable (in this case, annual bird mortality) based on the independent variable (cat population). Thus, a coefficient of determination R^2 close to 1 indicates that the linear model well describes the dependency of mortality on population size.

The use of Artificial Intelligence (AI) for working in the R programming environment

To make efficient use of time, the author employed the AI software product ChatGPT 4.0 by OpenAI for syntax analysis and self-learning in R programming, selecting appropriate commands according to the calculation methodology, interpreting statistical formulas in R, and working with the ggplot2 library. The R scripts, created using AI and used in this study, are provided as appendices at the end of the article. The author formulated queries, composed code blocks, identified incorrect responses of AI, and set the code according to the tasks set within this study.

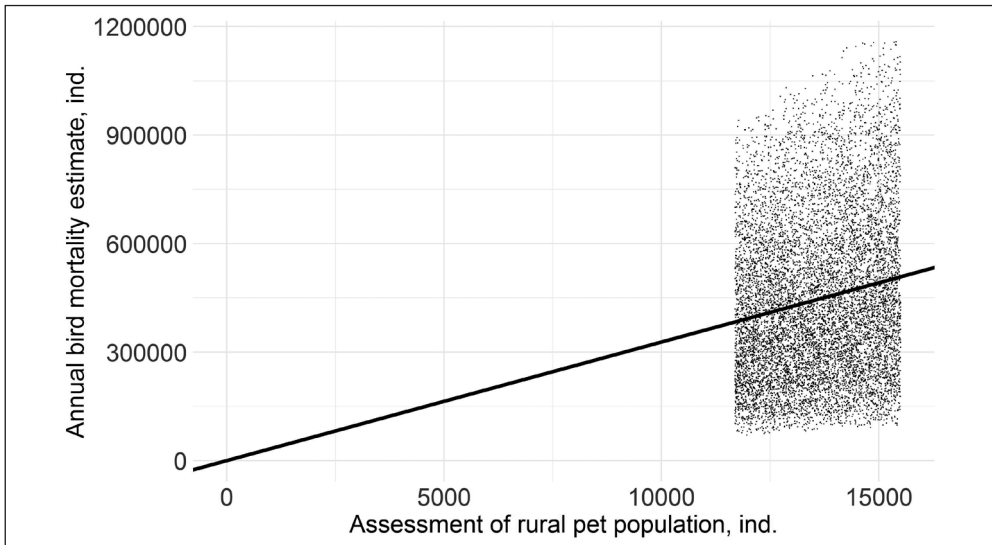


Figure 3. The relationship between the estimated annual number of birds killed by rural pet and the size of the cat population. The shaded area on the graph represents the range of results from multiplying the parameters over 10,000 iterations

3. ábra A vidéki háziállatok által elpusztított madarak becsült éves száma és a macskapopuláció nagysága közötti kapcsolat. Az árnyékolt terület a paraméterek 10 000 iteráción keresztüli szorzásából származó eredmények tartományát jelöli

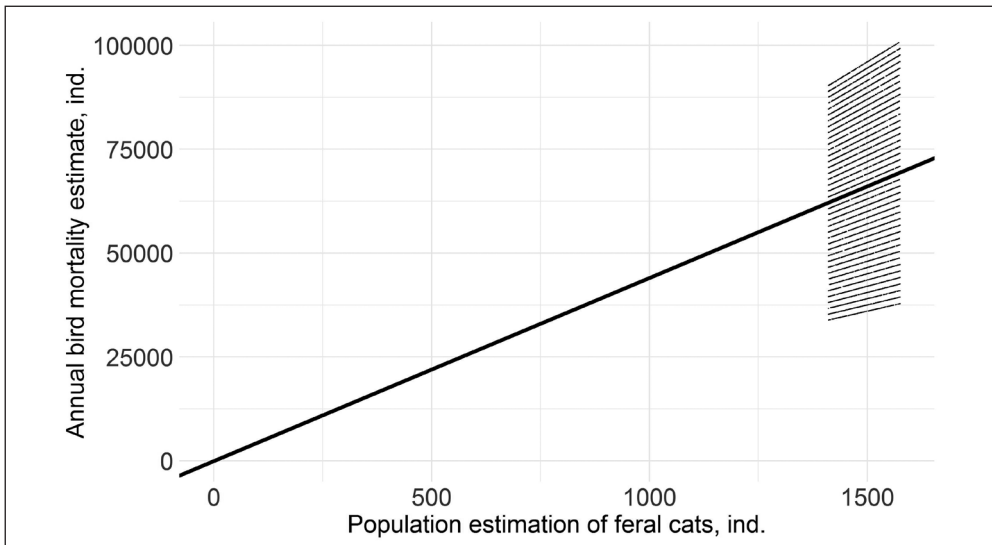


Figure 4. The relationship between the estimated annual number of birds killed by feral cat and the size of the cat population. The shaded area on the graph represents the range of results from multiplying the parameters over 10,000 iterations

4. ábra Az elvadult macskák által elpusztított madarak becsült éves száma és a macskapopuláció nagysága közötti kapcsolat. Az árnyékolt terület a paraméterek 10 000 iteráción keresztül történő szorzásából származó eredmények tartományát jelöli

Criteria for identifying vulnerable bird species

It is evident that not all bird species identified in the Bukovynian Carpathians can be hunted by cats. Forest species face a significantly lower threat from predators, as cats rarely venture deep into forests (Kays & DeWan 2004). Observations of the Wood Thrush (*Hylocichla mustelina*) nests in Ontario, Canada, which were located in forests near human settlements and farms, recorded 12 predator species, but domestic cats were not among them (Blancher 2013). Wetlands also serve as a natural barrier to cat predation, making birds that nest there much less vulnerable, except for species that nest on the edges of wetlands (Winter & Wallace 2006). The body mass of a bird is a significant factor. Studies at bird feeders have shown that domestic cats prefer smaller birds weighing between 14 and 42 grams, but a few larger birds, such as Mourning Doves (*Zenaida macroura*) weighing over 100 grams, were also hunted (Dunn & Tessaglia 1994). The nesting location is also crucial to a cat's hunting success. Species that nest or feed on the ground are more vulnerable (Winter & Wallace 2006) than those that nest in tree canopies or anthropogenic environments (artificial nests, roofs of buildings and structures, power line poles, etc.) and forage in the air. Another important criterion is the food base. Species whose diet includes berries and fruits of trees, terrestrial molluscs and invertebrates, earthworms, etc., are more vulnerable due to their proximity to the ground during foraging, which makes them easy prey for predators (Churcher & Lawton 1987, Crooks & Soulé 1999, Hawkins *et al.* 2004, Baker *et al.* 2005, 2008, van Heezik *et al.* 2010, Balogh *et al.* 2011). In Western Europe, vulnerable bird species are those frequently found on the ground or visiting feeders in winter (Pavisse *et al.* 2019).

Thus, the following criteria were selected to identify potentially vulnerable species due to cat predation in the anthropogenically transformed landscapes of the Bukovynian Carpathians:

1. Established presence of the species in these areas (Yuzyk & Yuzyk 2016, 2022, 2024, Yuzyk 2021).
2. Bird body mass between 10 and 150 g (Blancher 2013).
3. Ground-nesting habits or nesting in locations potentially accessible to cats.
4. Foraging behavior associated with obtaining food on the ground or in bushes or shrubs.

Result

Determining the population of rural pets in the study areas

The number of households is crucial for estimating the population of cats kept by owners. Upon the author's request, information was obtained on the number of household consumers from the Putyla Central Operations Centre «CRESC» LLC within the former Putyla district and in Putyla township (*Table 1*). According to the aforementioned calculation criteria (see Methods), the minimum sample size for the town of Putyla was 155, and

for the southwestern Bukovynian Carpathians, it was 164. In this study, residents from the following settlements were surveyed: Putyla township, the villages of Dykhtynets, Kyselytsi, Malyi Dykhtynets, Ploska, Ploshchi, Serhiyi, Sokoli, Stebni, Toraky, and the hamlet of Khrabusna.

In total, 158 residents from different addresses in Putyla township and nearby hamlets were surveyed, while 195 were surveyed in the southwestern Bukovynian Carpathians. The extrapolated estimates of the rural pet population in the town of Putyla and across the entire study area are presented in *Table 1*.

The average number of rural pets per household in Putyla township is 1.02 ± 0.163 , whereas for the ATL of the southwestern Bukovynian Carpathians, this value is higher (1.43 ± 0.2).

A survey of rural pet owners revealed that none of the animals are kept exclusively indoors; all (or 100%) have free access to the outside world, and therefore pose a threat to birds.

Estimating the population of feral cats in the studied areas

The population of feral cats significantly affects bird mortality, as these animals hunt more frequently to survive (see Methods).

The results of the monitoring of feral cats in the model area in Putyla township are presented in *Table 2*, with the monitoring points shown in *Figure 1*. Data on the estimated population size and density of feral cats in Putyla township and in the ATL of the southwestern Bukovynian Carpathians are presented in *Table 1*.

From the results obtained (*Tables 1, 4*), it can be established that the population of feral cats constitutes about 10% of the total population, yet these cats kill over 14% of all birds, which is related to their higher food requirements.

Bird mortality due to domestic cats predation

Anthropogenically transformed landscapes create new ecological conditions for the local fauna. One of the most influential anthropogenic factors is the presence of domestic cats, which, due to their hunting instincts, significantly impact local bird populations. The Bukovynian Carpathians, as a biodiversity hotspot, are an important area for studying this impact.

To accurately calculate the number of birds killed by cats in these areas annually, the Monte Carlo method with 10,000 iterations was used (see methods). This method provides a more accurate estimate and accounts for the high variability in cat behavior and ecological living conditions. The raw data and ranges for calculating 10,000 probable annual bird mortality values due to cat predation are presented in *Table 3*.

The analysis of the results from each iteration allowed for the distribution of the percentage frequency of the estimated annual bird mortality due to cat predation in the southwestern Bukovynian Carpathians (*Figure 2*) and shows a wide range of possible impacts on the bird fauna. *Table 4* presents the median values for the obtained sample and the 95% ranges of mortality estimates due to predation by feral cats, rural pets, and the total number of birds killed per year (values are given in parentheses).

Determination of linear regression models for estimating bird mortality due to predation by rural pets and feral cats

The relationships between the estimated total number of birds killed annually by rural pets and feral cats and their population sizes are shown in *Figures 3, 4*. The points on the graph represent the distribution of random values resulting from 10,000 iterations within the specified ranges. The line represents the computed linear regression model.

The computed linear regression model for feral cats is: $nBFC = 44.10 * nFC$ (acronyms as defined above, see methods). Thus, with a one-unit increase in the feral cat population, the annual bird mortality in the southwestern Bukovynian Carpathians will change approximately by 44 birds. $R^2 = 0.9325$. The linear regression model for rural pets is: $nBRP = 32.92 * nRP$. With a one-unit increase in the rural pet population, the annual bird mortality in the specified areas will change approximately by 33 birds. $R^2 = 0.8063$.

Based on the slope coefficients of the linear regressions, it is possible to determine how many more birds are killed by feral cats compared to rural pets. Calculation: $((44.1 / 32.9) - 1) * 100\% = 34\%$. Thus, feral cats kill 34% more birds compared to rural pets.

Bird species vulnerable to domestic cat predation

In accordance with the selection criteria (see Methods), a list of bird species vulnerable to domestic cat predation in the ATL of the Bukovynian Carpathians has been compiled (*Table 5*).

Discussion

Cats are opportunistic carnivores that consume a wide variety of animals, including mammals, birds, reptiles, amphibians, and insects. The study identifies 2,084 species consumed by cats, of which 347 species are of conservation concern. Notably, islands have a higher proportion of species affected by cat predation compared to continents. Cats prey on approximately 9% of bird species, 6% of mammal species, and 4% of reptile species globally. These findings underscore the ecological threat that free-ranging domestic cats pose, as they prey on vulnerable and endangered species across a wide geographical range (Lepczyk *et al.* 2023).

This study found that the average number of cats per household in villages and remote homesteads is higher compared to Putyla township. According to the author, this is related to the larger number of cats kept in homesteads and remote villages, where almost every private household keeps livestock and poultry, and cats help control small mammals, particularly rodents. Such cats potentially kill more birds, as Polish colleagues note that domestic cats on Polish farms are mainly kept to control mice and are poorly fed, which intensifies their impact on biodiversity and birds (Krauze-Gryz *et al.* 2018).

In the village of Vakalivshchyna, Sumy region, Ukraine in 1992, there were 18 rural pets in 10 households, and by 1996, there were 25 rural pets in 18 households. Between 1989 and 1996, 72 cases of bird mortality (22 species) due to cat predation were recorded

Table 5. Vulnerable bird species to domestic cat predation in the Bukovynian Carpathians

5. táblázat A házimacska zsákmányolás által veszélyeztetett madárfajok listája a Bukovinai-Kárpátokban

Species ¹	Criteria of vulnerability			Conservation status
	Body mass up to 150 g	Ground nesting	Feeding on the ground or in shrubs (bushes)	
<i>Coturnix coturnix</i>	+	+	+	Annex II of the EU Birds Directive, Annex II of the Bonn Convention, Annex III of the Bern Convention, IUCN, LC
<i>Cuculus canorus</i>	+	-	+	EU Birds Directive (Article 1), Annex III of the Bern Convention, IUCN, LC
<i>Columba oenas</i>	+	-	+	RDBU ² , Annex II of the EU Birds Directive, Annex III of the Bern Convention, IUCN, LC
<i>Streptopelia decaocto</i>	+	-	+	Annex II of the EU Birds Directive, Annex III of the Bern Convention, IUCN, LC
<i>Charadrius hiaticula</i>	+	+	+	RDBU, EU Birds Directive (Article 1), Annex II of the Bern Convention, Annex II of the Bonn Convention, Annex II of the AEWA, IUCN, LC
<i>Charadrius dubius</i>	+	+	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, Annex II of the Bonn Convention, Annex II of the AEWA, IUCN, LC
<i>Actitis hypoleucos</i>	+	+	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, Annex II of the Bonn Convention, Annex II of the AEWA, Helsinki Convention, IUCN, LC
<i>Tringa totanus</i>	+	+	+	Annex II of the EU Birds Directive, Annex III of the Bern Convention, Annex II of the Bonn Convention, Annex II of the AEWA, IUCN, LC
<i>Upupa epops</i>	+	-	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, IUCN, LC
<i>Dryobates minor</i>	+	-	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, IUCN, LC
<i>Dendrocopos major</i>	+	-	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, IUCN, LC
<i>Picus canus</i>	+	-	+	Annex I of the EU Birds Directive, Annex II of the Bern Convention, IUCN, LC
<i>Oriolus oriolus</i>	+	-	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, IUCN, LC
<i>Lanius collurio</i>	+	+	+	Annex I of the EU Birds Directive, Annex II of the Bern Convention, IUCN, LC

Species ¹	Criteria of vulnerability			Conservation status
	Body mass up to 150 g	Ground nesting	Feeding on the ground or in shrubs (bushes)	
<i>Poecile montanus</i>	+	+	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, IUCN, LC
<i>Coloeus monedula</i>	+	-	+	Annex II of the EU Birds Directive, IUCN, LC
<i>Bombycilla garrulus</i>	+	-	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, IUCN, LC
<i>Periparus ater</i>	+	-	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, IUCN, LC
<i>Poecile palustris</i>	+	+	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, IUCN, LC
<i>Cyanistes caeruleus</i>	+	-	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, IUCN, LC
<i>Parus major</i>	+	-	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, IUCN, LC
<i>Remiz pendulinus</i>	+	-	+	EU Birds Directive (Article 1), Annex III of the Bern Convention, IUCN, LC
<i>Lullula arborea</i>	+	+	+	Annex I of the EU Birds Directive, Annex III of the Bern Convention, IUCN, LC
<i>Hirundo rustica</i>	+	-	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, IUCN, LC
<i>Phylloscopus sibilatrix</i>	+	+	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, Annex II of the Bonn Convention, IUCN, LC
<i>Phylloscopus trochilus</i>	+	+	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, Annex II of the Bonn Convention, IUCN, LC
<i>Phylloscopus collybita</i>	+	+	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, Annex II of the Bonn Convention, IUCN, LC
<i>Hippolais icterina</i>	+	-	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, Annex II of the Bonn Convention, IUCN, LC
<i>Sylvia atricapilla</i>	+	+	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, Annex II of the Bonn Convention, IUCN, LC
<i>Sylvia borin</i>	+	-	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, Annex II of the Bonn Convention, IUCN, LC

Species ¹	Criteria of vulnerability			Conservation status
	Body mass up to 150 g	Ground nesting	Feeding on the ground or in shrubs (bushes)	
<i>Curruca curruca</i>	+	+	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, Annex II of the Bonn Convention, IUCN, LC
<i>Regulus regulus</i>	+	-	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, Annex II of the Bonn Convention, IUCN, LC
<i>Troglodytes troglodytes</i>	+	+	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, IUCN, LC
<i>Sitta europaea</i>	+	-	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, IUCN, LC
<i>Sturnus vulgaris</i>	+	-	+	IUCN, LC
<i>Turdus philomelos</i>	+	-	+	Annex II of the EU Birds Convention, Annex III of the Bern Convention, IUCN, LC
<i>Turdus viscivorus</i>	+	-	+	Annex II of the EU Birds Convention, Annex III of the Bern Convention, IUCN, LC
<i>Turdus merula</i>	+	+	+	Annex II of the EU Birds Convention, Annex III of the Bern Convention, IUCN, LC
<i>Turdus pilaris</i>	+	-	+	Annex II of the EU Birds Convention, Annex III of the Bern Convention, IUCN, LC
<i>Erithacus rubecula</i>	+	+	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, Annex II of the Bonn Convention, IUCN, LC
<i>Luscinia svecica</i>	+	+	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, Annex II of the Bonn Convention, IUCN, LC
<i>Ficedula parva</i>	+	-	+	Annex I of the EU Birds Directive, Annex II of the Bern Convention, Annex II of the Bonn Convention, IUCN, LC
<i>Ficedula albicollis</i>	+	-	+	Annex I of the EU Birds Directive, Annex II of the Bern Convention, Annex II of the Bonn Convention, IUCN, LC
<i>Phoenicurus ochruros</i>	+	-	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, Annex II of the Bonn Convention, IUCN, LC

Species ¹	Criteria of vulnerability			Conservation status
	Body mass up to 150 g	Ground nesting	Feeding on the ground or in shrubs (bushes)	
<i>Phoenicurus phoenicurus</i>	+	-	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, Anex II of the Bonn Convention, IUCN, LC
<i>Cinclus cinclus</i>	+	-	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, IUCN, LC
<i>Passer montanus</i>	+	-	+	EU Birds Directive (Article 1), Annex III of the Bern Convention, IUCN, LC
<i>Passer domesticus</i>	+	-	+	EU Birds Directive (Article 1), IUCN, LC
<i>Motacilla cinerea</i>	+	+	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, IUCN, LC
<i>Motacilla alba</i>	+	-	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, IUCN, LC
<i>Anthus pratensis</i>	+	+	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, IUCN, LC
<i>Anthus trivialis</i>	+	+	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, IUCN, LC
<i>Anthus spinoletta</i>	+	+	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, IUCN, LC
<i>Fringilla coelebs</i>	+	-	+	EU Birds Directive (Article 1), Annex III of the Bern Convention, IUCN, LC
<i>Coccothraustes coccothraustes</i>	+	-	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, IUCN, LC
<i>Pyrrhula pyrrhula</i>	+	-	+	EU Birds Directive (Article 1), Annex III of the Bern Convention, IUCN, LC
<i>Chloris chloris</i>	+	-	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, IUCN, LC

Species ¹	Criteria of vulnerability			Conservation status
	Body mass up to 150 g	Ground nesting	Feeding on the ground or in shrubs (bushes)	
<i>Linaria cannabina</i>	+	-	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, IUCN, LC
<i>Carduelis carduelis</i>	+	-	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, IUCN, LC
<i>Serinus serinus</i>	+	-	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, IUCN, LC
<i>Spinus spinus</i>	+	-	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, IUCN, LC
<i>Emberiza citrinella</i>	+	+	+	EU Birds Directive (Article 1), Annex II of the Bern Convention, IUCN, LC

Note. ¹ Classification and scientific species names of birds according to the IOC World Bird List (version 14.1) (<https://www.worldbirdnames.org/new/>).

² RDBU – the Red Data Book of Ukraine, according to the Order of the Ministry of Environmental Protection and Natural Resources of Ukraine No. 29 dated January 19, 2021.

(Merzlikin 1998). However, annual data is not provided, making it impossible to estimate the average number of birds hunted per year by rural pets in this area. Additionally, data on the number and hunting cases of feral cats in the mentioned locality is not provided. According to O. Koshelev and O. Osytkovska, in 2017, Ukraine ranked 9th in the world in terms of cat population (7.5 million individuals) and second in the number of cats per 100 residents (17 cats per 100 people). On average, one cat kills 30–47 birds per year (Koshelev & Osytkovska 2017). However, it was not specified whether this figure applies to urban, rural pets, or feral cats, nor the source or calculation method were not provided.

The population of feral cats varies significantly over time and is difficult to accurately estimate, as these animals reproduce uncontrollably, which can lead to a sharp increase in their numbers. At the same time, they die from diseases, malnutrition, traffic, and other factors, which has the opposite effect. Therefore, the population changes dynamically.

The number of feral cats in the study area is significantly influenced by the fact that feral animals have much larger home ranges compared to rural pets. The home range of feral cats greatly exceeds that of cats kept by owners, due to the constant need to search for food. Studies from various countries support this view. Recent research on feral cats in New Zealand found that the home range of males varies from 22.1 to 3,232 hectares, while for females, it ranges

from 9.6 to 2,078 hectares (Nottingham *et al.* 2022). Research conducted in winter 2005 in New South Wales, Australia, indicates that the average home range of feral cats was 248 hectares (Molsher *et al.* 2005). Data from the Mauna Kea Forest Reserve (Hawaii, USA) show significant home ranges for local feral cats. For females, the area was 772 hectares, while for males, it ranged from 1,418 to 2,050 hectares, making it one of the highest recorded figures (Goltz *et al.* 2008). A two-year study from central Illinois (USA) demonstrated that feral cats have much larger home ranges than rural pets. The largest home range for a male was 547 hectares, while for rural pets, it was around 2 hectares. Additionally, rural pets were active only 3% of the time, while ferals were active 14% of the time (Horn *et al.* 2011). Researchers in Europe, particularly in Croatia, distinguish between rural pets and feral cats based on their dependency on human settlements. Accordingly, rural pets are dependent on human homes, while feral cats are independent of them (Lanszki *et al.* 2016). In the Bukovynian Carpathians, it is currently difficult (or nearly impossible) to draw such a parallel, as feral cats are associated with settlements. Prolonged stay of a domestic cat in forests seems unlikely, as under such conditions, the cat would transform from predator to prey, given that forests have a greater number of natural enemies compared to areas near human dwellings (see methods). To either confirm or refute this idea, it is worth conducting a study of the home ranges of feral cats in the Bukovynian Carpathians using high-tech equipment.

With the increase in the proportion of feral cats, overall bird mortality will rise. Therefore, effective measures are necessary to control the feral cat populations.

The coefficients of determination R^2 , which are 0.9325 and 0.8063 for feral cat and rural pet respectively, indicate that approximately 93.25% and 80.63% of the annual bird mortality variations can be explained by domestic cat population size variations, suggesting a fairly high level of model accuracy. However, 6.75% and 19.37% of the variations remain unexplained by the models, pointing to the presence of other factors and random variables influencing bird mortality. One such factor is that a cat may catch a bird weakened by the use of pesticides, or a bird poisoned from feeding at a landfill, or injured by collisions with glass surfaces of windows (Erickson *et al.* 2005), solar panels (Visser *et al.* 2019, Kosciuch *et al.* 2020, Yuzyk 2024), or vehicles. Another factor is the need for more precise investigation of the koefRP value for these areas. Therefore, further research is needed, including the use of advanced technologies (collars with cameras, GPS trackers, etc.) (Lloyd *et al.* 2013, Nottingham *et al.* 2022), to shed more light on domestic cat predation and its home range in the specified areas.

Among bird species vulnerable to predation by domestic cat, 2 species are listed in the Red Data Book of Ukraine, the Stock Pigeon (*Columba oenas*) and Common Ringed Plover (*Charadrius hiaticula*). 44 species are listed in Annex II, 15 in Annex III of the Convention on the Conservation of European Wildlife and Natural Habitats (the Bern Convention). 18 species are listed in Annex II of the Convention on the Conservation of Migratory Species of Wild Animals (the Bonn Convention). 5 species are listed in Annex I, 9 in Annex II, and 47 in Article 1 of the EU Birds Directive. 4 species are listed in Annex II of the Agreement on the Conservation of African-Eurasian Migratory Waterbirds (the AEWA). 1 species is listed in the Helsinki Convention. The data from this study align with the results of a comparative analysis of the diet of feral and house cats, and wildcat in Europe, where rural pets, feral cats, and wildcats hunted the following number of bird species: 71, 42, and 23

species, respectively (Széles *et al.* 2018). Small mammals were the main prey for all cat groups (including feral domestic cats, wild cats, and their hybrids), with birds constituting the second most important prey in Hungary (Biró *et al.* 2005).

To more accurately assess species vulnerability, detailed studies of parameters such as the total population size and the proportion of individuals falling victim to cat predation are required. These data allow for evaluating the impact of domestic cats on various bird species and developing strategies for biodiversity conservation in the future.

Conclusion

The population of feral cats and rural pets in the ATL of the southwestern Bukovynian Carpathians was estimated at $1,493 \pm 82$ and $13,599 \pm 1,904$ individuals, respectively. The total population of domestic cats was determined to be $15,092 \pm 3,245$ individuals. Annually, feral cats (smaller population compared to rural cats) kill an average of 66.0 thousand birds, with a 95% mortality estimate range from 37.6 thousand to 95.1 thousand, while rural pets kill an average of 443.0 thousand birds annually (ranging from 127.0 thousand to 942.5 thousand). The total bird mortality rate due to domestic cat predation in the ATL of the southwestern Bukovynian Carpathians is 509.0 thousand (ranging from 188.6 thousand to 1,008.6 thousand).

Linear regression models were developed for feral cats and rural pets, described by the formulas: $nBFC = 44.10 * nFC$; $nBRP = 32.92 * nRP$. Thus, with a change in the feral cat population by 1 individual, the annual bird mortality in the ATL of the southwestern Bukovynian Carpathians would change by approximately 44 birds, $R^2 = 0.9325$. With a change in the rural pet cat population by 1 individual, the annual bird mortality would change by approximately 33 birds, $R^2 = 0.8063$.

It was found that feral cats in the southwestern Bukovynian Carpathians kill 34% more birds compared to rural pets. A total of 62 bird species vulnerable to domestic cat predation were identified.

Research indicates the need to develop comprehensive strategies for protecting birds from domestic cat predation in anthropogenically transformed landscapes. This includes not only population control measures for cats but also environmental education for the public and the promotion of responsible pet ownership. Further research and monitoring are necessary to understand the dynamics of this process and to develop effective biodiversity conservation measures.

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Appendix 1. R Script for Monte Carlo simulation of bird mortality due to domestic cats predation, data analysis and visualization

1. függelék R Script a házimacsák okozta madárpusztulás Monte Carlo szimulációjához, adatelemzéshez és vizualizációhoz

```
# Setting the number of iterations
n <- 10000

# Generating random numbers within specified ranges
range1 <- runif(n, 1411, 1575)
range2 <- runif(n, 24, 64)
range3 <- runif(n, 11695, 15503)
range4 <- runif(n, 2.8, 14.0)
range5 <- runif(n, 2.0, 5.8)

# Setting the total number of bird mortality
results <- (range1 * range2) + (range3 * range4 * range5)

# Rounding the sorted multiplication results to whole numbers
rounded_results <- round(results)

# Outputting the sorted multiplication results as a single line, separated by spaces
cat(paste(rounded_results, collapse = " ", "\n"))

# Mean value
mean_value <- mean(results)

# Deviations
deviations <- results - mean_value

# Mean deviation
mean_deviation <- sqrt(sum(deviations^2) / (length(results) - 1))

# Standard deviation
standard_deviation <- mean_deviation / sqrt(length(results))

# Relative error
relative_error <- (standard_deviation / mean_value) * 100

# Outputting results
cat("Mean value of the sample:", mean_value, "\n")
cat("Mean deviation:", mean_deviation, "\n")
cat("Standard deviation:", standard_deviation, "\n")
cat("Relative error (%):", relative_error, "\n")
```



```

# Calculating the percentage distribution of values across ranges
bins <- seq(0, 1300000, by = 50000)
percentage <- sapply(bins, function(b) {
  sum(results >= b & results < b + 50000) / length(results) * 100
})

# Outputting percentage distribution
cat("Percentage distribution of values across ranges:\n")
for (i in seq_along(bins)) {
  cat(sprintf("From %d to %d thousands: %.2f%%\n", bins[i], bins[i] + 50000,
percentage[i]))
}

# Creating a bar chart of the ranges against percentages
library(ggplot2)

# Creating a dataframe for the chart
distribution_data <- data.frame(
  Range = paste(bins / 1000, "-", (bins + 50000) / 1000),
  Percentage = percentage
)

# Building the chart
p <- ggplot(distribution_data, aes(x = factor(Range, levels = Range), y = Percentage))
+
  geom_bar(stat = "identity", fill = "gray", color = "black") +
  labs(x = "Assessment of annual bird mortality (thousands)", y = "Percentage
frequency of estimation") +
  theme_minimal() +
  scale_y_continuous(labels = scales::percent_format(scale = 1)) +
  theme(axis.text.x = element_text(angle = 45, hjust = 1, size = 16),
        axis.text.y = element_text(size = 16),
        axis.title.x = element_text(size = 20),
        axis.title.y = element_text(size = 20),
        plot.title = element_text(size = 20))

# Saving the chart to a JPEG file
ggsave("D:/percentage_distribution.jpg", plot = p, width = 12, height = 8, dpi = 300)

# Opening the saved file in Windows
shell.exec("percentage_distribution.jpg")

```

```
# Calculating quantiles for the interval covering 95% of values
lower_bound <- quantile(results, 0.025)
upper_bound <- quantile(results, 0.975)

# Outputting the minimum and maximum values for the interval
cat("95% of values are within the range from", lower_bound, "to", upper_bound,
    "\n")
```

Appendix 2. R Script for Monte Carlo simulation of bird mortality due to feral cats predation, data analysis, visualization and linear regression model

2. függelék R Script az elvadult macskák okozta madárpusztulás Monte Carlo szimulációjához, adat-elemzéshez, vizualizációhoz és lineáris regressziós modellhez

```
# Loading the necessary library
library(ggplot2)

# Setting the number of iterations
n <- 10000

# Generating random integers within specified ranges
set.seed(123) # Fixing seed for reproducibility of results
range1 <- sample(1411:1575, n, replace = TRUE)
range2 <- sample(24:64, n, replace = TRUE)

# Multiplying random numbers
results <- range1 * range2

# Creating a dataframe for plotting
data <- data.frame(
  range1 = range1,
  range2 = results
)

# Creating a linear regression model without an intercept
model_no_intercept <- lm(range2 ~ range1 + 0, data = data)
summary(model_no_intercept)

# Outputting the R-squared value to the console for the model without an intercept
r_squared_no_intercept <- summary(model_no_intercept)$r.squared
cat("R-squared value for the model without an intercept:", r_squared_no_intercept,
    "\n")

# Regression formula without an intercept
```

```

coefficients_no_intercept <- summary(model_no_intercept)$coefficients
slope_no_intercept <- coefficients_no_intercept[1, 1]

cat("Linear regression formula without an intercept: range2 =", slope_no_intercept,
    "* range1\n")

# Creating a scatter plot with a thicker regression line without an intercept
p <- ggplot(data, aes(x = range1, y = range2)) +
  geom_point(color = "black", size = 0.1) +
  geom_abline(intercept = 0, slope = slope_no_intercept, color = "black", size = 1.5)
+ # Regression line without an intercept
  labs(x = "Population estimation of feral cats, ind.", y = "Annual bird mortality
estimate, ind.") +
  theme_minimal() +
  theme(
    axis.text = element_text(size = 20), # Font size for axis labels
    axis.title = element_text(size = 20) # Font size for axis titles
  ) +
  coord_cartesian(xlim = c(0, max(data$range1)), ylim = c(0, max(data$range2)))

# Saving the plot to a JPEG file
ggsave("D:/ENG_FC_R2_no_intercept.jpg", plot = p, width = 10, height = 6, dpi =
300)

# Opening the saved file in Windows
shell.exec("D:/ENG_FC_R2_no_intercept.jpg")

```

Appendix 3. R Script for Monte Carlo simulation of bird mortality due to rural pets predation, data analysis, visualization and linear regression model

3. függelék R script a vidéki háziállatok által okozott madárpusztulás Monte Carlo szimulációjához, adatelemzéshez, vizualizációhoz és lineáris regressziós modellhez

```

# Loading necessary libraries
library(ggplot2)

# Setting the number of iterations
n <- 10000

# Generating random integers in specified ranges
range1 <- sample(11695:15503, n, replace = TRUE)
range2 <- runif(n, min = 2.8, max = 14.0)
range3 <- runif(n, min = 2.0, max = 5.8)

```

```
# Calculating the result for variable2
variable2 <- range1 * range2 * range3

# Creating a dataframe for plotting
data <- data.frame(
  variable1 = range1,
  variable2 = variable2
)

# Creating a linear regression model without an intercept to obtain R squared
model <- lm(variable2 ~ variable1 + 0, data = data)
r_squared <- summary(model)$r.squared

# Building a scatter plot
p <- ggplot(data, aes(x = variable1, y = variable2)) +
  geom_point(color = "black", size = 0.1) +
  geom_abline(intercept = 0, slope = summary(model)$coefficients[1, 1], color =
    "black", size = 1.5) +
  labs(x = "Assessment of rural pet population, ind.", y = "Annual bird mortality
estimate, ind.") +
  theme_minimal() +
  theme(
    axis.text = element_text(size = 20), # Font size for axis labels
    axis.title = element_text(size = 20) # Font size for axis titles
  ) +
  coord_cartesian(xlim = c(0, max(data$variable1)), ylim = c(0, max(data$variable2)))

# Saving the plot to a JPEG file
ggsave("D:/ENG_scatter_plot.jpg", plot = p, width = 10, height = 6, dpi = 300)

# Opening the saved file in Windows
shell.exec("D:/ENG_scatter_plot.jpg")

# Printing the value of R squared to the console
cat("R squared value:", r_squared, "\n")

# Printing the linear regression formula to the console
slope <- summary(model)$coefficients[1, 1]
cat("Linear regression formula without intercept: variable2 =", slope, "* variable1\n")
```

The global population status and distribution of the Indian Swiftlet: Implications for Conservation

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Abstract The current study aims to comprehend the population status and distribution of the Indian Swiftlet (*Aerodramus unicolor*, family: Apodidae) using primary and secondary data. We acquired the population data from the secondary literature and presence data from open source (GBIF.org). We conducted surveys to document the breeding and foraging locations of the species in the Western Ghats, West Coast and Offshore islands of Maharashtra. After estimating the population, we used the presence data to create a model predicting species distribution in current and future scenarios. The Indian Swiftlet's current distribution is from Southwest Maharashtra to Kerala and Sri Lanka. In future, the changing climate might restrict it to the southern Western Ghats and some pockets in Sri Lanka. Burnt Island, home to the largest known colony, deserves conservation attention. We recommend population surveys and immediate conservation efforts to ensure the survival of the Indian Swiftlet endemic to India and Sri Lanka.

Keywords: Apodidae, cave bird, climate change, conservation actions, edible nest, aerial insectivore

Összefoglalás Jelen tanulmány célja, hogy elsődleges és másodlagos adatok felhasználásával megvilágítsa a malabári szalángána (*Aerodramus unicolor*, Apodidae család) populációs helyzetét és elterjedését. A populációs adatokat a másodlagos szakirodalomból, a jelenléti adatokat pedig nyílt forrásból (GBIF.org) szereztük be és terepi felméréseket végeztünk a faj költő- és táplálkozóhelyeinek dokumentálására India Nyugati-Ghátok régiójában, a nyugati partvidéken és Maharashtra állam partmenti szigetein. A populáció becslését követően a faj jelenlegi elterjedését becslő és a jövőbeli elterjedését előrejelző modellt építettünk a jelenléti adatok felhasználásával. A malabári szalángána jelenlegi elterjedési területe Maharashtra állam délnyugati részétől Keraláig és Sri Lankáig terjed. A modell szerint a jövőben az elterjedés a változó éghajlat miatt a déli Nyugati-Ghátokra és Sri Lanka néhány zónájára korlátozódhat. A legnagyobb ismert kolóniának otthont adó Burnt-sziget külön természetvédelmi figyelmet érdemel. Az Indiában és Sri Lankán endemikus malabári szalángána fennmaradásának biztosítása érdekében az állományok részletes felmérését és azonnali védelmi erőfeszítéseket javasolunk.

Kulcsszavak: Apodidae, barlangi madár, ehető fészkek, klímaváltozás, levegőben vadászó rovarrevő, természetvédelmi beavatkozások

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Introduction

Most swiftlets of the family Apodidae and order Caprimulgiformes assemble their nests using saliva as a binding agent. However, saliva is the most crucial or the sole nesting material for a few species that built economically significant edible nests. The edible nests of swiftlets are among the costliest animal products in the world, costing anything from \$4,000 to \$10,000 per kilogram depending upon the saliva quality and quantity (Lau & Melville 1994 Sankaran 2001). Among the 39 species of swiftlets known worldwide (Ibrahim *et al.* 2009), many are exploited for profit (Hobbs 2004, Thorburn 2015). Because of their economic significance and large breeding colonies known from across their distribution range, the edible-nest swiftlets are never included in the list of conservation priority species, even after severe declines and local extinctions are documented (Sankaran 2001, Manchi & Sankaran 2014).

India is home to (i) the Himalayan Swiftlet (*Aerodramus brevirostris*) which is found throughout the Himalayas and mountains of North India from Himachal Pradesh East to South China (W Yunnan), Myanmar and Thailand and winters in Bangladesh, SW Thailand and the Malay Peninsula, and possibly Sumatra, (ii) the Plume-toed Swiftlet (*Collocalia affinis*), and (iii) the Andaman Edible-nest Swiftlet (*Aerodramus inexpectatus*) which are found throughout the Andaman and Nicobar Islands (Chantler & Boesman 2020), and iv) the Indian Swiftlet (*Aerodramus unicolor*) which occurs in the coastal region and the Western Ghats complex from about Ratnagiri in south Konkan region of Maharashtra to southward through Goa, western Karnataka, Tamil Nadu, Kerala (Chantler & Kirwan 2020). The Indian Swiftlet is endemic to India and Sri Lanka. Under British colonialism, edible nests in India were traded frequently by the early 1800s. Later, uncontrolled and illegal nest harvesting caused rapid population drop (Sankaran 2001, Mahabal *et al.* 2007). Conservation efforts largely halted the illegal edible nest collection in the Andaman and Nicobar Islands to restore the population (Manchi & Sankaran 2014, Manchi *et al.* 2022). The inaccessible breeding sites and difficulties in identifying the species in flight have limited scientific exploration of several swiftlet species (Kawalkar & Manchi 2023). Nonetheless, several studies undertaken over the years (Abdulali 1940, 1942, Gunawardana 1997, Katdare 2001, Pande 2001, Kumar 2006, Mahabal *et al.* 2007) were concluded independently, however, primarily concentrated on the population status, the scenario of illegal commerce and some aspects of biology. Apart from a few focused studies, information on the occurrence of the species throughout its distribution range was gathered through a citizen science approach (for example eBird forum), which can be found in the Global Biodiversity Information Facility (GBIF) database (GBIF.org, last access: 30 December 2023). Though a considerable quantity of data on species distribution is available, no attempts were made to collate the data to comprehend the species' population status, distribution pattern and habitat. The Indian Swiftlet's habit of nesting in dark caves and other similar habitats, the inaccessibility of these breeding sites, and the lack of expertise and interest to explore the inaccessible and overlooked cave habitats is the biggest constraint in exploring and estimating the breeding population of the cave-dwelling Indian Swiftlet in the WGS biodiversity hotspot.

Hence, we attempted to explore the Indian Swiftlet's current global population status, understand distribution pattern, the effect of climate on the species' distribution in current and future scenarios, and identify the potential habitats.

Materials and Methods

Study area

Our study was limited to the known global distribution range of the Indian Swiftlet, i.e., the West Coast & Western Ghats of India and Sri Lanka. The Western Ghats of India and Sri Lanka (WGSL), one of the 34 global biodiversity hotspots (UNESCO 2012), is extraordinarily rich in biodiversity and unique plants (Myers *et al.* 2000). The Western Ghats is a series of mountains that run through Kerala, Tamil Nadu, Karnataka, Goa, Maharashtra, and Gujarat, around 30–50 km inland and parallel to India's western coast. The sole breadth in this 1,600 km long mountain track, which encompasses around 140,000 km², is the 30 km Palghat Gap at about 11°N (UNESCO 2012). While the eastern part of the Western Ghats is in the rain-shadow region of the Peninsula, the western side is on the verge of southwest monsoon and receives rainfall of 203–254 cm. Despite being 400 km apart, the Western Ghats of India and the highlands in southwest Sri Lanka are remarkably similar in geology, climate, and evolutionary history. The WGSL is a geographically complicated biodiversity hotspot (Myers *et al.* 2000, Bossuyt *et al.* 2004) with a dense human population farming and altering the landscape for thousands of years (Bawa *et al.* 2007). The WGSL has a mountainous terrain (up to 2,700 m elevation) and a geographic barrier, the Palk Strait, which divides India's Western Ghats and Sri Lanka.

The West Coast of Peninsular India forms a narrow belt of low land, geologically of laterite formation, lying between the sea and the Western Ghats, which extends the whole length of the Peninsula, varying in width from 20 to 100 miles inland. The Western Ghats (WG) runs parallel to the entire western coast of the Indian peninsula and extends across Maharashtra, Karnataka, Goa, Kerala, and Tamil Nadu (Paul *et al.* 2018) (Figure 1). Paul *et al.* (2018) also

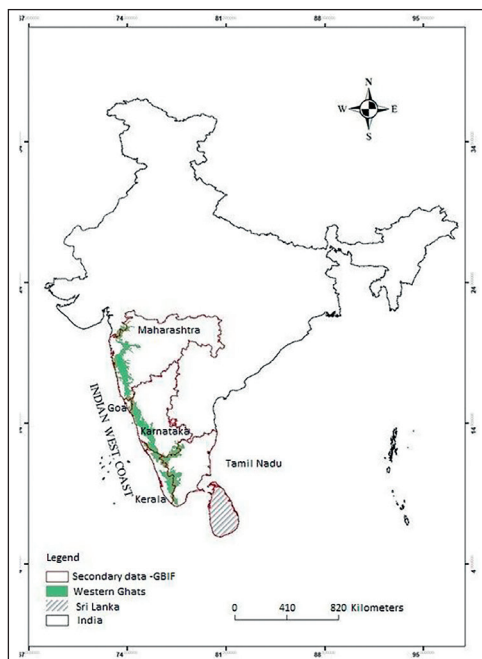


Figure 1. The global distribution range of the Indian Swiftlet is restricted to Western Ghats of India and Sri Lanka

1. ábra A malabári szalangána globális elterjedése India Nyugati-Ghátok régiójára és Sri Lankára korlátozódik

suggest that the forests of the West Coast in India supply 25–40% (average) of moisture to the southwest monsoon rainfall over the East Coast. The WGSL have five distinct regional forest types: (i) Tropical wet evergreen forest (comprising lowland and hill rain forests); (ii) Tropical semi-evergreen (monsoonal) and moist deciduous forest; (iii) Tropical dry deciduous forest and woodland; (iv) Tropical thorn scrub; (v) Tropical lower and upper montane forests (Champion & Seth 2005, Ashton *et al.* 2014).

Population status

We reviewed the secondary literature to understand the population status of the Indian Swiftlet across its distribution. Due to funding constraints, we could only conduct primary surveys in Western Ghats, West Coast and offshore islands of Maharashtra to document the roosting locations of the Indian Swiftlet. We surveyed the potential habitats (natural caves, rock-cut caves, forts) and rigorously checked for the direct and indirect evidence (for example nests or bird droppings) to confirm the birds' presence. Since the survey was conducted during the breeding season, wherever we documented Indian Swiftlet's presence, we meticulously searched the site/structure (for example cave) to locate the swiftlet nests. Once located, we used the nest count method to estimate the population (Sankaran 2001). Since swiftlets are known as monogamous, each nest was considered equal to a breeding pair, i.e. two birds (Manchi & Sankaran 2014, Gurjarpadhye *et al.* 2021). It allowed us to estimate the colony-wise breeding population of the species. These counts were conducted at the end of every month during the breeding season (from December to June) of the Indian Swiftlet. The maximum nest count obtained in a season was considered as the breeding population of the colony.

Species Distribution Modelling (SDM)

We collected the primary data (foraging and breeding location) from Western Maharashtra viz. Konkan region comprises six coastal districts (Palghar, Mumbai, Thane, Raigad, Ratnagiri, and Sindhudurg).

We conducted the distribution survey in Maharashtra between December 2020 and December 2022. Since swiftlets are well-known diurnal foragers (Manchi & Sankaran 2010, Brinkløv *et al.* 2013), the distribution survey was carried out during daylight hours, between 06.00 h and 18.00 h (Patil *et al.* 2021) to confirm the species' presence and identify the foraging locations. The survey's beginning and ending times were determined upon sunrise and sunset. We surveyed a total of 3,210 km in the six districts, using vehicles (two and four-wheeler) on the road and machine dingy to access the offshore islands. The aerial insectivores were tracked down and eventually identified at the species level. A handheld GPS (Garmin Montana 680) was used to mark these encounter locations, further referred to as the survey points. Totally we could mark 350 survey points during the study. We utilized a spotting scope (Endurance ED 20×85 Hawke) and binoculars (Nikon Monarch 10×42) to observe and identify the different aerial insectivorous birds foraging, including the study species. Additionally, we used cameras (Nikon D500 and Nikon Coolpix P900)

to photograph the foraging aerial insectivores whenever we could not confirm the species encountered using binoculars or spotting scope.

Based on the survey points located (foraging and roosting locations) and data from GBIF.org (n=28,049), we modeled the current and future distribution of the Indian Swiftlet in both India and Sri Lanka. For error-free analyses, we eliminated the stray (n=7; five from India and two from Maldives) and misidentified (n=5; three from Maharashtra, one from Gujarat, and one from Andhra Pradesh) records.

We used the SDM Toolbox v2.5 to obtain spatially rarefied occurrence data for SDMs to reduce spatial autocorrelation for thinning the entire data to reduce any possible bias. To minimize the impact of biased sampling, a spatial thinning of 10 km for India and Sri Lanka (Kawalkar & Manchi 2023) was performed between each occurrence point. After eliminating the duplicate records and minimising spatial autocorrelation, we obtained a total of 549 presence records for the modeling. We confirmed that this sample size would allow capturing the broadest possible range of species-habitat associations to make reliable predictions (Santini *et al.* 2019, Gaul *et al.* 2020).

Based on the most recent climatic data (1970–2000), retrieved from the WorldClim database v2.1 at a spatial resolution of 30 seconds (www.worldclim.org), we employed 15 bioclimatic variables (*Table 2*) out of the 19 (except Bio 8, 9, 18 and 19) to create the SDM (Booth *et al.* 2022) (*Table 1*). We utilized the spatial resolution of 30 seconds (~1 km) for the detailed and accurate measures of the climate variables. Swiftlets forage in a variety of habitats, including open, scrub, wetland, paddy land, forest habitats (dry deciduous, subtropical/tropical moist lowland, subtropical/tropical moist montane), shrubland (subtropical/tropical high altitude) and rocky offshore islands (Koon & Cranbrook 2002, Manchi & Sankaran 2010, IUCN 2016, Petkliang 2017, Chantler & Kirwan 2020). Swiftlets forage close to their nesting colonies (caves) throughout the incubation and nestling period (March-June) (Nguyễn & Voisin 1998, Manchi 2009); as a result, we additionally evaluated the Terrain Roughness Index as a covariate to represent the presence of the caves and crevices. To forecast the occurrence of the species, we employed 6 categorical habitat covariates (Evergreen Forest, Deciduous Forest, Mosaic forest, Cultivated and Managed Vegetation, Elevation, and Terrain Roughness/Ruggedness Index) (*Table 1*) from the EarthEnv repository (www.earthenv.org) with a spatial resolution of 30 arc-second per pixel (~1 km per pixel at the equator). Our analysis used the future climate data (2021–2040 and 2081–2100) for the minimum and maximum emission representative for carbon dioxide concentration predicted for two Shared Socio-economic Pathways (SSPs), 126 and 585, as predictors. The CMIP6 downscaled from the WorldClim v2.1 as baseline climate (www.worldclim.org) was used to download 15 global climate models (GCM-HadGEM3-GC31-LL) from a spatial resolution of 30 seconds. The Hadley Global Environment Model-3 (HadGEM3) is employed since it is a comparatively better model (Adhikari *et al.* 2024, Meher & Das 2024).

The 15 bioclimatic variables were initially checked for collinearity using the Pearson correlation variable coefficient analysis (Pearson 2007). Any variable with an r-value exceeding 0.6 was not considered for further analysis (*Table 1*). SDM was done using MaxEnt software Version 3.4 (Phillips *et al.* 2006). We used 10,000 random background

Table 1. The bioclimatic and habitat variables used for the Species Distribution Modelling of Indian Swiftlet (Note: The bioclimatic variables in bold were considered for modelling after eliminating the correlated variables)

1. táblázat A malabári szalagána faj-elterjedési modellezésében használt bioklimatikus és élőhelyi változók (Megjegyzés: a félkövérrel szedett bioklimatikus változókat használtuk a korreláló változók kizárását követően)

Codes	Environmental Variables	Unit
Bio1	Annual Mean Temperature	°C
Bio2	Mean Diurnal Range of monthly Temperature	°C
Bio 3	Isothermality (BIO2/BIO7) (*100)	°C
Bio 4	Temperature Seasonality (standard deviation *100)	°C
Bio 5	Max Temperature of Warmest Month	°C
Bio 6	Min Temperature of Coldest Month	°C
Bio 7	Temperature Annual Range (BIO5-BIO6)	°C
Bio10	Mean Temperature of Warmest Quarter	°C
Bio11	Mean Temperature of Coldest Quarter	°C
Bio12	Annual Precipitation	mm
Bio13	Precipitation of Wettest Month	mm
Bio14	Precipitation of Driest Month	mm
Bio15	Precipitation Seasonality (Coefficient of Variation)	mm
Bio16	Precipitation of Wettest Quarter	mm
Bio17	Precipitation of Driest Quarter	mm
Class 2	Evergreen Forest	Continuous
Class 3	Deciduous Forest	Continuous
Class 4	Mosaic forest (a mosaic of mixed forest, shrubland, and woody savannah)	Continuous
Class 7	Cultivated and Managed Vegetation (Agricultural land, plantations)	Continuous
Elev	Elevation	m
TRI	Terrain Roughness/Ruggedness Index (Topographic heterogeneity)	m

points as pseudo-absences, followed the sub-sampling technique, the Receiver Operating Characteristics curve (ROC) and 1,000 imitations to determine the Area Under the Curve (AUC). The model was run for five replicates with 5,000 iterations, with 70% of the species presence sites serving as training data for the model's evaluation and the remaining 30% serving as test data for the statistical significance (Parolo *et al.* 2008). The developed models were assessed based on the AUC statistics of ROC plots (Swets 1988). Following Wordley *et al.* (2015), we discarded all variables with poor predictive capabilities ($AUC \leq 0.5$) and retained the remaining variables for modeling. We continued this pruning process until the model yielded high AUC performance variables. The resulting SDMs were deemed to match the data well when their AUC value was >0.9 and poorly when it was <0.7 (Parolo *et al.* 2008). As MaxEnt models' outputs, the resulting suitability

classes were categorized into the default categories. To measure each variable’s relative and absolute contribution to the model, the jackknife test permitted the independent evaluation of the model’s accuracy.

Results

Population status of Indian Swiftlet

Even though the forts, human-made caves and rock-cut caves hold potential habitats for the nesting of the Indian Swiftlet, the renovation activities under regular maintenance have disrupted the suitable and potential habitats for birds and bats over the years. Another colony was known to be at Tiracol Fort (Maharashtra-Goa border), which was destroyed by a natural disaster (Mahabal *et al.* 2007). The Swiftlet cave (locally known as Pakoli Dhol) on Burnt Island is a sea cave, naturally formed by the interaction between the sea waves and

Table 2. Locations and sizes of known populations of the Indian Swiftlet in India and Sri Lanka
2. táblázat A malabári szalángána ismert állományainak helye és nagysága Indiában és Srí Lankán

Sr. No	Location	Region, State/ Province	Country	Population	Year	References
1	Burnt Island, Vengurla Rocks	Sindhudurg, Maharashtra	India	4,220	2023	Kawalkar & Manchi 2024
2	Old Lighthouse Island, Vengurla Rocks	Sindhudurg, Maharashtra	India	116	2023	
3	Tiger cave	Ooty, Tamil Nadu	India	250	2002	Katdare, 2002
4	Sacrifice rock	Thalassery, Kerala	India	200	1849	Jerdon 1862
5	South Travancore	Kanyakumari, Kerala	India	500	1953	Pravin Jaydevan 2006 (Personal comm with Mahabal <i>et al.</i> 2007)
6	Pakshi Pathalam caves	Wayanad, Kerala	India	Not estimated		Jerdon 1862
7	Netrani Is. or Pigeon Island	Uttara Kannada, Karnataka	India			
8	Mandaramnuwara Cave	Nuwara Eliya, Central Province	Sri Lanka	68	2013	Munasinghe & Ranawana 2013
9	Kosgala Dark Cave	Ratnapura, Sabaragamuwa	Sri Lanka	Not estimated		LICAS, 2017
10	Belilina tunnel	–	Sri Lanka			
11	Ravana tunnel	Ella, Uva	Sri Lanka			
12	Sthreepura Lena Cave	Ratnapura, Sabaragamuwa	Sri Lanka			
13	Railway tunnels	Ohiya to Idalgashinna, Uva	Sri Lanka	Not estimated		Ekanayake & Kudavidanage 2013

the hard igneous rock, whereas the Old Lighthouse is an old structure built using laterite and cobblestones (Manchi *et al.* 2022, Kawalkar & Manchi 2024, Manchi *et al.* 2024). The colony at Burnt Island, Vengurla Rocks, has been known at least since 1865 for more than 140 years (Mahabal *et al.* 2007), and the Old Lighthouse Island colony was discovered in 2001 by Mahabal *et al.* (2007). Worldwide, 13 known breeding colonies of the Indian Swiftlet are formally documented (seven in India and six in Sri Lanka), but only four of them in India and a single one in Sri Lanka have their populations estimated (Table 2). Recently, Kawalkar and Manchi (2024) documented that the Burnt Island of Vengurla Rocks in Sindhudurg, Maharashtra (India), is the largest colony of the Indian Swiftlet ever known. This population is known to have reached its peak and is identified as the ‘K’ population (Kawalkar & Manchi 2024). Therefore, it has the highest priority for conserving the Indian Swiftlet.

Distribution of the Indian Swiftlet

The present survey, from December 2020 to December 2022, throughout all the coastal districts of Maharashtra and comparatively much higher and repetitive sampling efforts (3,210 km and 350 survey points), confirmed that the current distribution of the Indian Swiftlet in Maharashtra is restricted to the coastal South Sindhudurg region ($n=11$ locations). Simultaneously, the GBIF database (1800–2022; 222 years) created based on the presence records from the region largely confirms the findings of the present study. Hence, the Indian Swiftlet’s current distribution in Maharashtra can be inferred from the presence data (Figure 4a).

Based on the results of Species Distribution Modelling, the Indian Swiftlet is currently widely distributed in India and Sri Lanka ($AUC=0.88\pm0.002$). In India, the species is distributed southward from Central Ratnagiri in Western Maharashtra through Goa, western Karnataka, Tamil Nadu, Kerala, and rocky offshore islets in the Arabian Sea (Figure 4a). Presently, in Sri Lanka it is distributed throughout except in the few regions of Northern and North Central Provinces. The Indian Swiftlet is predicted to have a wide distribution in the Western Ghats of India and mostly towards the southern region during 2021–2040 [ssp126

Table 3. The climatic factors with high percent contribution affecting the distribution of Indian Swiftlet under different scenarios

3. táblázat A malabári szalagána elterjedését magas százalékos magyarázóerővel befolyásoló klimatikus tényezők a különböző scénáriókban

Variables	Current (1970–2000)	2021–2040		2081–2100	
		ssp126	ssp585	ssp126	ssp585
Class 2 (Evergreen Forest)	37	-	-	-	-
Bio 3 (Isothermality)	20	-	-	-	-
Class 7 (Cultivated and Managed Vegetation)	16.2	-	-	-	-
Bio 4 (Temperature Seasonality)	-	58.3	56.5	46.4	76.2
Bio 5 (Max Temperature of Warmest Month)	-	24.1	27.9	20.5	-
Bio 12 (Annual Precipitation)	-	-	-	13.9	-

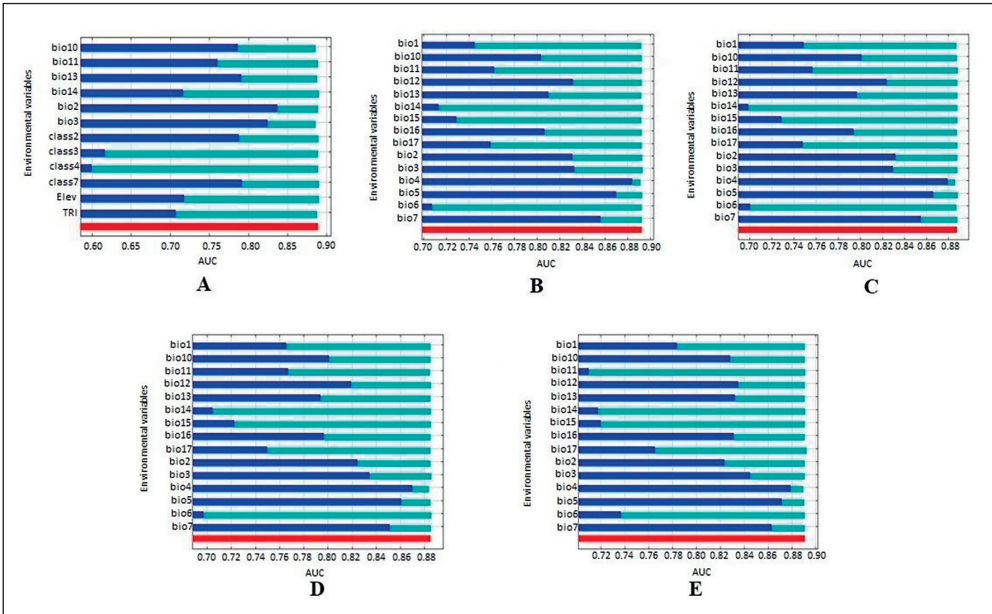


Figure 2. Results of the Jackknife test between the variables: (a) Current scenario, (b) 2021–2040 ssp126 (c) 2021–2040 ssp585, (d) 2081–2100 ssp126 and, (e) 2081–2100 ssp585

2. ábra A változók közötti jackknife-tesztek eredményei: (a) jelenlegi elterjedés alapján, (b) 2021–2040 között az ssp126 szcenárió alapján, (c) 2021–2040 között az ssp585 szcenárió alapján, (d) 2081–2100 között az ssp126 szcenárió alapján és (e) 2081–2100 között az ssp585 szcenárió alapján

(0.89 ± 0.003) and ssp585 (0.88 ± 0.004)]. The species distribution might shrink in Karnataka to concentrate in Tamil Nadu and Kerala during 2081–2100 [ssp126 ($AUC=0.88\pm0.004$) and ssp585 ($AUC=0.89\pm0.002$)]. In Sri Lanka, the distribution is concentrated in the central and southern regions, during both scenarios (Figure 4b, 4c). The Indian Swiftlet distribution, in the current scenario, is most influenced by the Evergreen Forest (Class 2), Cultivated and managed vegetation (Class 7) and Isothermality (Bio 3) (Table 3, Figure 2a–2e).

Discussion

The present study confirms that the current distribution of the Indian Swiftlet in Maharashtra is restricted to the coastal South Sindhudurg region. In contrast to Mahabal *et al.* (2007), the species was not encountered in Raigad and Ratnagiri districts or anywhere in the further northern parts of the coastal South Sindhudurg region of Maharashtra. However, the GBIF database (1800–2022; 222 years) created based on the contribution of several birders (Citizen Science) from the region largely confirms the findings of the present study. Based on the results of Species Distribution Modelling, the Indian Swiftlet is currently widely distributed in India and Sri Lanka. The Indian Swiftlet is predicted to have a wide distribution in the Western Ghats of India and mostly towards the southern region in the future scenarios.

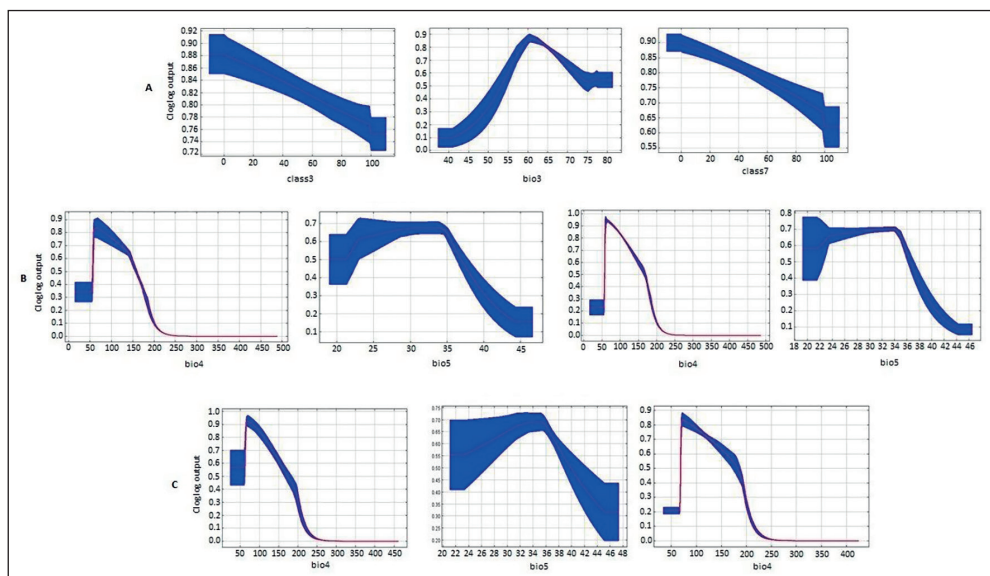


Figure 3. The area under the curve (AUC): (a) Current scenario, (b) 2021–2040 (ssp126 and ssp585), (c) 2081–2100 (ssp126 and ssp585)

3. ábra A görbe alatti terület (AUC): (a) a jelenlegi helyzet alapján, (b) 2021–2040 között az ssp126 és ssp585 scenáriók alapján, és (c) 2081–2100 között az ssp126 és ssp585 scenáriók alapján

The Indian Swiftlet distribution, in the current scenario, is most influenced by the Evergreen Forest (Class 2), Cultivated and managed vegetation (Class 7) and Isothermality (Bio 3). The Indian Swiftlet is recorded from the lowlands to 2,200 m over various habitats commonly seen over scrub and dry deciduous forests, and it frequents small rocky offshore islands and mainland habitats across its distribution range in India (Chantler 2010). In Sri Lanka, it is mostly an upland species (Chantler 2010), found in cloud forest (0.26%), cloud forest die-back (0.90%), and grasslands (Chandrasiri *et al.* 2018). During the primary survey in western Maharashtra, the Indian Swiftlet is primarily observed foraging with Little Swift and Red-rumped Swallow over mangroves, rocky islands, seashores (sandy beach, rocky shores), inland wetlands, laterite plateaus and mosaic of plantations (mango, cashew, areca-nut, and coconut) and various forests. Several studies (Lourie & Tompkins 2000, Nguyễn *et al.* 2002, Ponak 2004, Manchi & Sankaran 2010) concerning various members of *Aerodramus* also indicated the significance of the forest. As Nguyễn *et al.* (2002) mentioned, it might be due to food availability in the forest habitat, an important source of Hymenoptera, a significant diet component for swiftlets. According to Abdulali (1942), the gut content of four Indian Swiftlet individuals captured from Burnt Island, Vengurla Rocks was primarily composed of Hemiptera, Diptera and Hymenoptera. Moreover, the Hymenoptera comprised the largest proportion of food boluses collected from nestlings fed by swiftlets foraging over the forest canopy in eastern Thailand (Lourie & Tompkins 2000, Ponak 2004). As Manchi (2009) described, the Edible-nest swiftlet requires large areas to forage, which means that land-use changes at the landscape level may affect their population and the breeding chronology. Contrarily, the habitat alteration around the caves may not

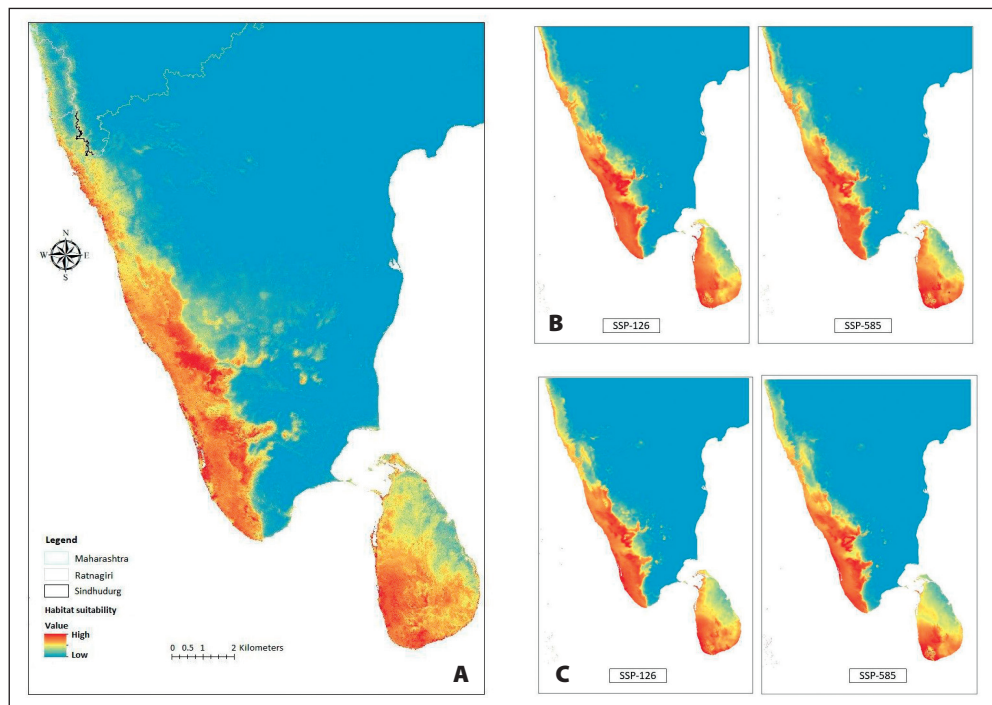


Figure 4. The global distribution of the Indian Swiftlet (a) current distribution, (b) future distribution in 2021–2040 (ssp126 and ssp585 scenarios) and (c) 2081–2100 (ssp126 and ssp585 scenarios)

4. ábra A malabári szalagána globális elterjedése (a) jelenleg, (b) 2021–2040 között (ssp126 és ssp585 scenáriók alapján) és (c) 2081–2100 között (ssp126 és ssp585 scenáriók alapján)

have a visible effect on the foraging activities of Plume-toed Swiftlet, but could interfere with their natural routine and related biology and physiology (Manchi 2009, Manchi & Sankaran 2010). All these findings indicate the importance of forests for the survival of the swiftlets across their range of distribution. Hence, the usage of the forest habitat in different seasons should be assessed in greater detail.

Along with the Evergreen Forest, the Indian Swiftlet also shows affinity towards the cultivated and managed vegetation (Class 7) (Table 3). In Malaysia, where 99% of all bird nest farms are geared towards producing white edible nests, recently started to integrate swiftlet rearing in the oil palm plantations (Ibrahim & Baharun 2009). This highlights the benefits to the swiftlet nest farmer from the management of the plantations (Kamarudin 2009), particularly the availability of insects these birds feed on and support pest control (Rahman *et al.* 2016, Mursidah *et al.* 2020). Based on the observations during the survey in Sundhudurg, Maharashtra, Indian Swiftlet was seen foraging mostly on the mosaic of forest and cash crop plantations, which include mango (*Magnifera indica*), cashew (*Anacardium occidentale*) and coconut (*Cocos nucifera*). Also, 80% of Indian Swiftlet's gut content sampled from Vengurla Rocks by Ali and Ripley (1970) comprised two species of harmful mango hoppers (*Idiocerus niveosparsus*, *I. atkinsoni*), amply available on the

mango orchards in and around Sindhudurg. According to Munje and Kumar (2022), when forests are rapidly being cleared for plantations, the cultivated and managed vegetation serves as an important surrogate habitat for forest birds in the northern Western Ghats of India. However, further studies are apparent to understand the role of plantations (mango, coconut, cashew, etc.) as the foraging habitat and Indian Swiftlet's role in the pest management in the region.

In the future scenarios, 2021–2040 (ssp126 and ssp585) and 2081–2100 (ssp126 and ssp585) when the Indian Swiftlet distribution starts concentrating to the southern Western Ghats, the major factor which would influence the distribution is Bio 4 (Temperature seasonality). Temperature seasonality measures change in temperature over the year (O'Donnell & Igniziom 2012), and the isothermality (Bio 3) refers to the degree to which temperature varies throughout the year. Both temperature variables are one of the most critical bioclimatic predictors for understanding species distribution (Reside *et al.* 2010). The isothermality is also proven to affect the various breeding parameters in several species of bee-eaters in Europe (Abdul-Wahab *et al.* 2024). This is apparent as the changes and degree of change in climate over the year will directly affect the insect availability and have an adverse impact on the breeding success, chronology (Dunn & Møller 2019, Halupka *et al.* 2023). Hence, it is important to conduct long-term studies to understand influence of various temperature variables on the breeding biology of the swiftlets, particularly the Indian Swiftlet.

Temperature seasonality plays a vital role in shaping species distribution patterns, influencing the geographic ranges of species and the composition of ecological communities. According to many studies (Janzen 1967, Chan *et al.* 2016, Frishkoff *et al.* 2016, Srinivasan *et al.* 2018), tropical species that depend on forests are often associated with a narrow range of temperatures leading to significant change in the species' probability of occupancy with increasing variation in temperatures. According to Ramesh *et al.* (2022), after examining ~78% of species, the likelihood of birds' occupancy (n=43 out of 55) in the southern Western Ghats, is significantly associated with temperature seasonality. As a result of reduced temperature seasonality in the tropics, relative to temperate regions, the species exhibit narrow thermal niches. Hence, these birds are likely to be unable to shift their distributions to track future climate changes (Janzen 1967, Tewksbury *et al.* 2008). This helps us to understand why the Indian Swiftlet populations in southern Western Ghats might not get much affected. However, since no such studies are available for the northern Western Ghats, it is difficult to pinpoint the reason for the predicted shifting of the Indian Swiftlet population from the northern to the southern Western Ghats.

During the future climate scenarios (2021–2040 ssp126 and 585, and 2081–2100 ssp126 and ssp585), Bio 4 (Temperature seasonality) and 5 (Max Temperature of Warmest month) (Table 2, Figure 3) also influences the distribution of the Indian Swiftlet in its entire distribution range. In India, the warmest months are April-May, the peak of the Indian Swiftlet's breeding season, when the eggs hatch and growth of the nestlings begin. As swiftlets are insectivorous, and their foraging entirely depends on food availability, the temperature during the warmest month might play a crucial role in the growth of the chicks. However, this can have both direct and indirect impacts on the swiftlets. Johnson (1974)

suggested insects move upwards with increased ambient temperature. Hence, based on food availability, the birds also fly higher to forage (Tompkins 1999, Manchi & Sankaran 2010). This alteration in feeding habits can also influence its breeding chronology and success (Dunn & Møller 2019). The limited available knowledge is a caveat to comment firmly on this aspect, therefore, further studies are crucial.

Our study modelled the distribution and identified the potential/suitable habitats of the Indian Swiftlet based on the secondary and primary datasets for the current and future climate change scenarios. The results show that the species distribution is affected by the changing climate; species will henceforth remain in the southern Western Ghats and some pockets in Sri Lanka. This also indicates the disappearance of the largest known colony of the species from Burnt Island, located in the northern Western Ghats. The quality of the potential habitat decreases; hence, it is of utmost importance to draft the required strategies and implement the required measures to avoid future detrimental effects on the Indian Swiftlet distribution. The southern Western Ghats is the potential future habitat for the species. Hence, we recommend the identification of the breeding colonies and detailed documentation of the population status of the Indian Swiftlet in other states of India (Goa, Karnataka, Tamil Nadu, and Kerala) and Sri Lanka as paramount to undertake any conservation measures. The Indian Swiftlet colony at Burnt Island, Vengurla Rocks (Sindhudurg, Maharashtra), is the largest and most crucial colony globally. So, focused conservation attention must be provided to the colony to safeguard and manage. We recommend further population genetics studies and ex-situ measures for this oldest known colony of the species. Also, the isolation of the Sri Lankan population from the Indian populations and its considerable confinement to specific habitats (Sri Lankan uplands) makes it more vulnerable to ecological changes, anthropogenic pressures, and climate change, highlighting an Indian Swiftlet a priority species for research and conservation. Therefore, along with local and national efforts, we recommend international collaborative efforts from both India and Sri Lanka for the conservation of this unique cave-dwelling bird, a source of a distinctive natural resource.

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European Serin (*Serinus serinus*) expands nesting area to south and east

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Abstract In the context of anthropogenic and climatic changes in the faunas of numerous regions, species that were previously absent from there are now observed, accompanied by alterations in the number of animal populations and disruption of the usual trends of their dynamics. The changes in the range of the European Serin (*Serinus serinus*) within Ukraine were identified by creating a four-layer GIS map in the DIVA-GIS software. The analysis revealed that European Serin continues to disperse in Europe, with a notable shift towards the south and east. The current southern border of the breeding range of the species passes through Central Ukraine, encompassing the right bank of the Dnipro river. However, the status of European Serin in the Black Sea region remains unclear. In the territories inhabited in the 20th century, the density of European Serin in the breeding season has increased, although decreasing population size is described for old populations of Europe. It is relevant to study the comparative dynamics of bird numbers in old habitats and in newly developed areas, with a view to identifying new wintering grounds for birds and populations that have shifted to a sedentary lifestyle.

Keywords: population changes, species distribution, Ukraine

Összefoglalás Az antropogén és éghajlati változásokkal összefüggésben számos régió faunájában megfigyelhetők olyan fajok, amelyek korábban hiányoztak onnan. Egyik ilyen faj a csicsörke (*Serinus serinus*), amelynek az Ukrajnán belüli elterjedési területén bekövetkezett változásokat a DIVA-GIS programban négyrétegű GIS-térkép készítésével azonosítottuk. Az elemzés kimutatta, hogy a csicsörke továbbra is terjed Európában, jelentős elmozdulással dél és kelet felé. A faj költőterületének jelenlegi déli határa Közép-Ukrajnán halad keresztül, magába foglalva a Dnyipro folyó jobb partját. A Fekete-tenger térségében azonban továbbra is tisztázatlan a státusza. A 20. században nőtt a faj denzitása költési időszakban, de ezzel szemben a régebbi európai populációinak csökkent az állománya. Annak érdekében, hogy megállapíthassuk a vonulási viselkedés változását (telelőhelyek változása, áttelelő madarak számának emelkedése), fontos a madarak számában bekövetkező változások nyomon követése a régi és az újonnan kolonizált területeken is.

Kulcsszavak: csicsörke, fajok elterjedése, Ukrajna

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Introduction

In recent decades, there has been a notable shift in the number of animal populations and a disruption of the usual trends in their dynamics (Kinzelbach 2011, Masoero *et al.* 2016). There has been a shift in the ranges of species, with the disappearance of species from the composition of breeding birds in different territories (Huntley *et al.* 2008). Palaeontological records indicate that modern northern bird species, along with their associated vegetation types, have shifted southwards, while open country species have shifted from the south-east to the north-west (Holm & Svenning 2014). This has resulted in species that were previously absent from the faunal lists of many regions becoming established. The distribution of birds is influenced by a variety of factors, with the initial position of the primary centre of origin of the species being the most important (Payevsky 2015). Species have been moving for a long time, but the movement of people, intensification of trade and economic development have accelerated this process. Changes in the species composition of the fauna of many regions are associated with an increase in the abundance and range expansion of some species. In the 1970s, the planet faced the issue of global climate change. The warming of northern regions has contributed to range changes in many species (Huntley *et al.* 2008, Holm & Svenning 2014, McLaughlan *et al.* 2014). Some species are at risk of population and range declines (e.g. Shupova & Tytar 2022), while others are conversely at risk of increases (e.g. Londei 2020). In order to make informed conservation decisions for biotic communities and species, it is crucial to ascertain the extent to which changes in the distribution of biota are attributable to natural causes and the extent to which they are due to human activities (Huntley *et al.* 2008).

In the context of climatic and anthropogenic changes, several new species of birds have emerged in the modern fauna of Ukraine. The European Serin (*Serinus serinus*), for instance, was absent from the country for almost the entire 20th century (Parkhomenko 1950, Strautman 1963, Charlemagne 2020). The birds entered Ukraine independently, occupying urbanised biotopes. In urbanised biotopes, anthropogenic changes have been demonstrated to reduce the competitiveness of local species (Evans 2010, Díaz *et al.* 2011, Møller *et al.* 2015). This phenomenon may have contributed to the settlement of European Serin in Ukraine. Fortunately, alien bird species have a minimal impact on changes in local diversity, and respond to abiotic habitat gradients in a manner similar to native bird species (Andrikou-Charitidou & Kallimanis 2021, Lazarina *et al.* 2023). The European Serin is currently undergoing dispersal, and the breeding range boundaries of the species are not yet stable (Knaus 2020).

The primary range of the European Serin was the Canary Islands and the Mediterranean, including southern Europe and Asia Minor (Dementev *et al.* 1954). The species had already started to disperse in the 16th century, but stopped due to the influence of the Little Ice Age. At the end of the 16th century, there were no European Serin populations in Europe north of the 48°N. Nevertheless, the birds lived in Carinthia (Austria), Trentino (Italy), Ticino and the Jura (Switzerland), Provence and Gascony (France), Tyrol and the Rhône Valley (Kinzelbach 2004). By the end of the 18th century, the European Serin's range extended to Spain, Italy, parts of Switzerland and the Balkans (Payevsky 2015).

Since 1800, its range has undergone a resurgence of expansion (Mayr 1926, Kinzelbach 2004). In the early years of the 19th century, the European Serin occupied the area north of the Alps. By the mid-20th century, it had populated all of Central Europe and reached southern England (Payevsky 2015). From 1927 to 1943, the species expanded its range into the countries of the eastern Baltic (Kumari 1958). From 1950, it reached Belarus. From 1960 to 1972, it spread to Finland and the Leningrad region (Mal'chevskii & Pukinskii 1983). This second wave of bird dispersal continues to the present day (Kinzelbach 2011, Knaus 2020).

The European Serin is a thermophilic migratory bird, with the exception of southern Europe, where it is sedentary and winters on the coastal regions of France, the Iberian Peninsula, North Africa, and the northern Mediterranean coast (Ottaviani 2008, Senar *et al.* 2011). In winter, occasionally, the species has been observed in the vicinity of Gdansk and Kaliningrad (Dementev *et al.* 1954), Galicia (Bokotey 1995), and Bukovina (Skilsky 1998).

The average lifespan of the species is 1.98 years (Senar & Copete 1990). In different regions, the birds have one or two broods per breeding season (Ottaviani 2008), with instances of three broods (Knysh & Malyshok 2015, Haraszthy 2019). In the natural landscape, the species prefers forest edges and glades, ravines, and gorges of mountains, where there are trees and shrubs. In Europe, the European Serin is currently found mainly in cultural landscapes (Gil-Delgado & José 1981, Díaz *et al.* 2011, Knysh 2011), including open cultivated areas, hedgerows, gardens, and urban parks. It also occurs in large cities and small towns (Ottaviani 2008). It has been observed that birds tend to favour biotopes with individual coniferous trees, although they have been known to place nests in both coniferous and deciduous trees (Kumari 1958, Knysh & Malyshok 2015, Haraszthy 2019). The nests of the European Serin are located at heights of 1.50–10.0 m (Ouarab *et al.* 2007, Knysh & Malyshok 2015). The mean clutch size in western Europe was found to be 3.70–3.87 (Gil-Delgado 1981), in Germany (Gnielka 1978) and Hungary (Haraszthy 2019) 4.0, while in the east, it was 4.37 eggs (Strautman 1963, Knysh & Malyshok 2015). In western Europe, egg-laying occurs in early March and early July (Gil-Delgado & José 1981). In Hungary, the first clutches are complete in mid-May, the second clutches in late June – early July, while the third clutches in mid-August (Haraszthy 2019). In eastern Europe, it occurs in the second half of May and the beginning of June. However, there is a description of nest construction on 19 July in 2011, and a new nest on 4 August 2011 (Knysh & Malyshok 2015). The breeding productivity of the species ranges from 1.49 (Senar & Copete 1990) to 2.18 fledglings per nesting attempt (Knysh & Malyshok 2015). Predation, starvation and tree pruning have been identified as factors that affect the abundance and nesting success of the species (Gil-Delgado & José 1981, Hoi-Leitner *et al.* 1999).

The objective of this study is to elucidate the distribution of the European Serin in Ukraine and to ascertain the status of the species and its current range in different regions of the country.

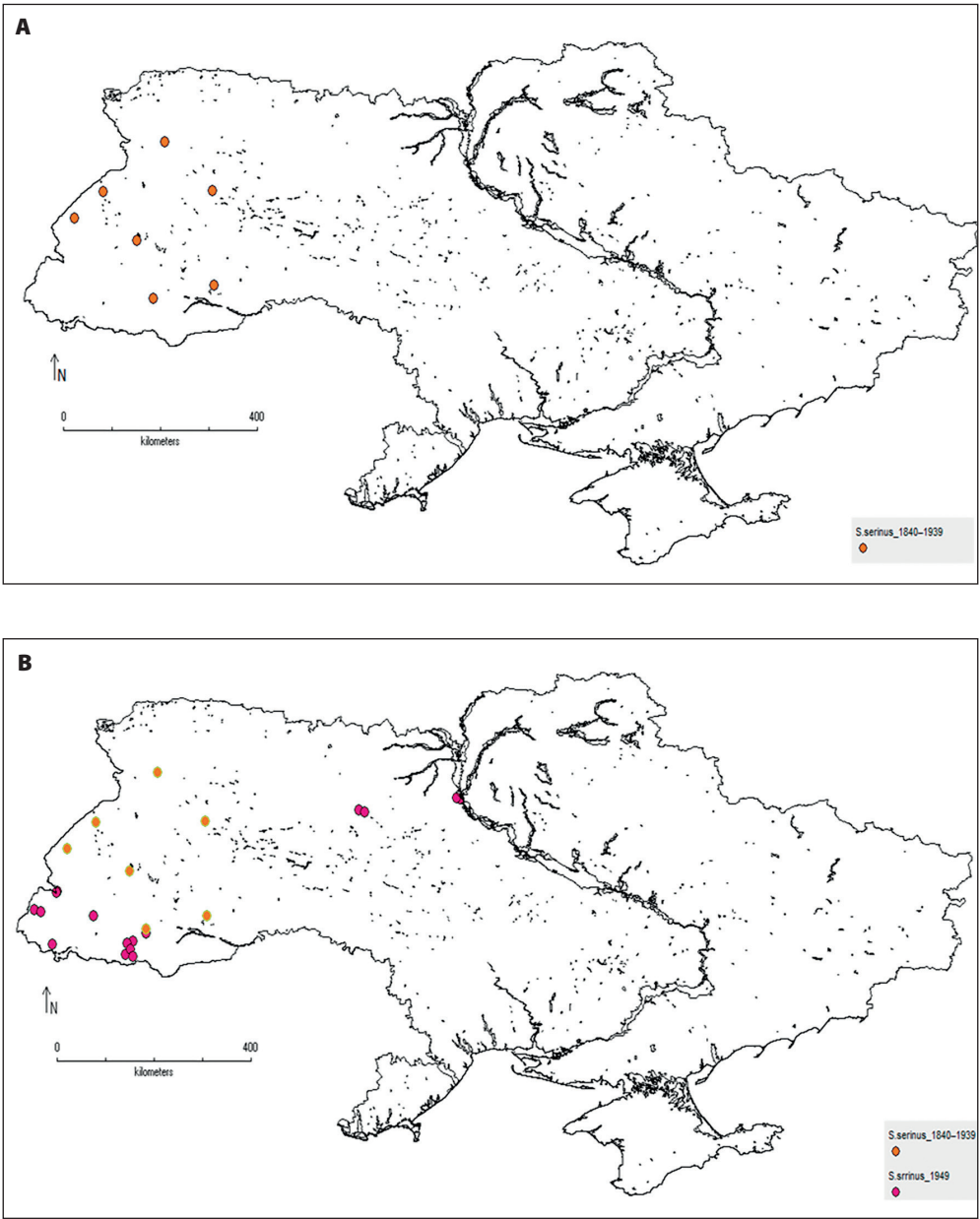
Methods

To clarify the changes in the European Serin range in Ukraine, we created a GIS map in the DIVA-GIS software (<http://www.diva-gis.org/>). The map was constructed using geographic coordinates obtained during our own observations and indicated in the analysed literature sources and museum collections. These coordinates were subsequently determined by recalculation with a geocalculator in DIVA-GIS. A total of 45 literature sources containing information about European Serin occurrences in Ukraine and 13 in adjacent territories were analysed. Articles devoted to the European Serin, catalogs of museum collections, and articles that contain lists of the regional faunas were analysed. For map construction, 39 of these sources (Parkhomenko 1950, Strautman 1963, Talposh 1974, 1976, Belik & Moskalenko 1993, Bokotey 1995, 2004, Guziy 1996, Kogut *et al.* 1996, Mistryukova 1996, Godovanets 1997, Knyazev 1997, Bashta 1999, Lugovoi *et al.* 2001, Nadtochiy 2002, Grischenko 2003, Senyk & Hornyak 2003, Gavriilyuk 2004, Gavriilyuk *et al.* 2005, Lugovoi 2005, Sizhko 2006, Gavrys *et al.* 2007, Domashevskiy 2008, Matveev 2008, Peklo 2008, 2018, Matviychuk & Serebryakov 2010, Knyish, 2011, Materials of ornithological observations 2011, Nankinov 2011, Skilskyi *et al.* 2014, Knysh & Malyshok 2015, Novak 2015, Yakovlev 2015, Yanenko *et al.* 2015, Pisulinska *et al.* 2016, Biletska 2017, Charlemagne 2020, Ghryb 2022), in addition to 83 author's registration points and data from databases, namely UaBirds, iNaturalist, UkrBIN, were used. In the absence of the coordinates of the sightings in the literature source, GIS coordinates were determined by searching on the map for the indicated locality and the distance from it to the place of bird registration. All the bird sightings described in the literature were divided into four GIS map layers. The first layer comprises records of the species during the first century of ornithological research in Ukraine before the beginning of bird expansion in Ukraine (1840–1939). The second layer represents the beginning of expansion in Ukraine and includes the distribution of the European Serin in the first half of the 20th century (1940–1950). The third layer encompasses the range of the European Serin by the end of the 20th century (1951–1999), and the fourth layer represents the distribution of the species at the beginning of the 21st century (2000–2022). A total of 399 GIS coordinates were utilised in the construction of the maps.

Results

Analyses of European Serin records during the breeding period revealed that in the late 19th and early 20th centuries, the species inhabited the western and south-western regions of the territory that is part of modern Ukraine (*Figure 1a*). By the middle of the 20th century, the number of nestings in the south-west increased, and the boundaries of the breeding range shifted to the centre of the country, reaching Kiev (*Figure 1b*). By the end of the 20th century, the European Serin had established a breeding population in Central and Northeastern Ukraine, with a possible expansion into the Odessa region (*Figure 1c*).

In the territories developed in the 20th century, the density of European Serin records during the breeding season increased. In the 21st century, the boundary of the breeding range of the species shifted further to the south and east (*Figure 1d*).



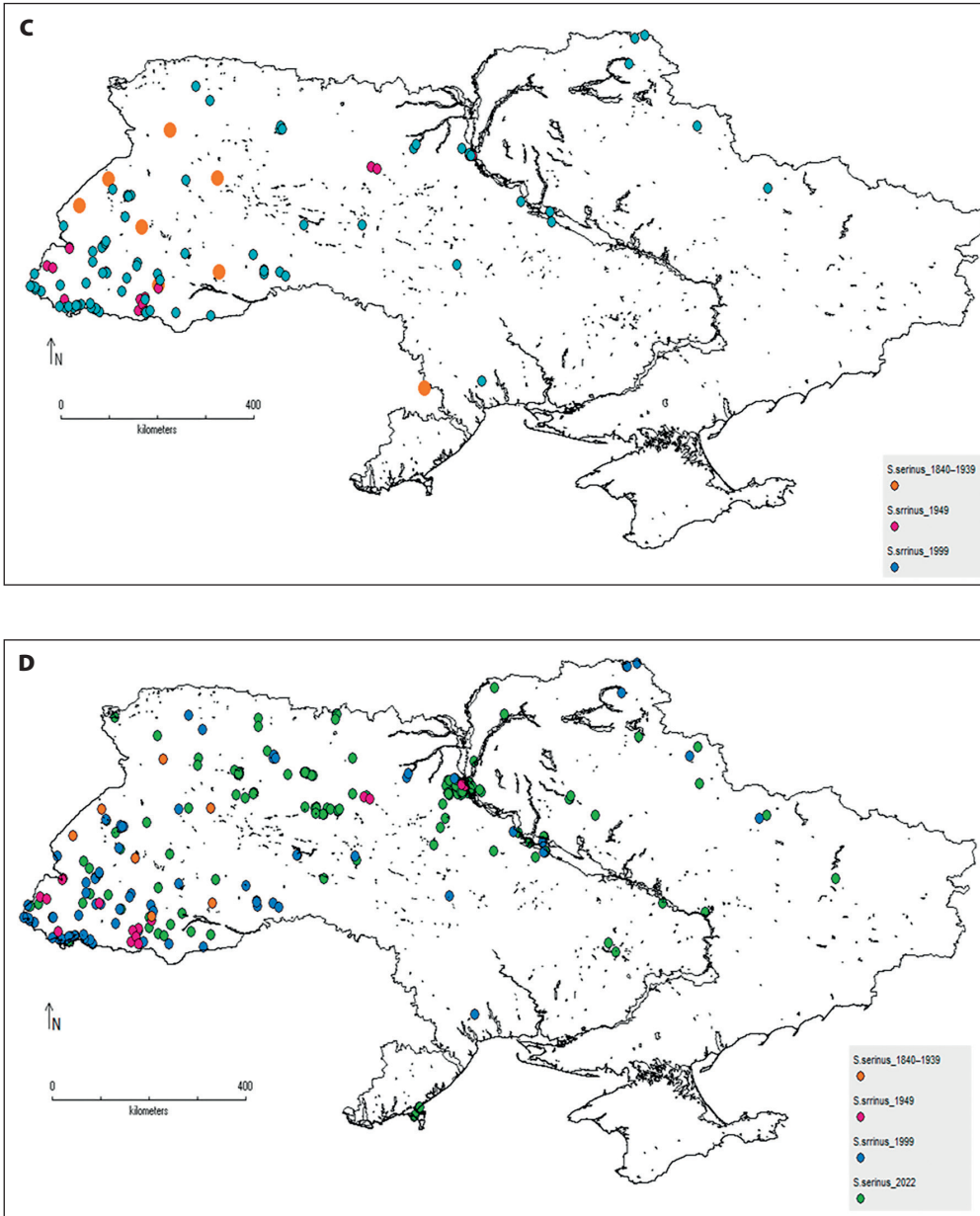


Figure 1. The changes in the breeding range of the European Serin in Ukraine during the period 1840–1939 (a), 1940–1949 (b), 1950–1999 (c) and 2000–2022 (d): stable area: 1840–1939 (orange); new nesting sites in 1940–1949 (pink), and in 1950–1999 (blue); area of European Serin in Ukraine in 2000–2022 (green)

1. ábra A csicsörke fészkelőterületeinek változása Ukrajnában az 1840–1939 (a), 1940–1949 (b), 1950–1999 (c) és a 2000–2022 (d) közötti időszakban

Discussion

The study of the European Serin in Ukraine commenced in the mid-20th century. The initial description of the species was provided by Charlemagne (2020), who indicated that the bird was not previously documented until 1939. From 1939 onwards, the species was observed in Volyn, and in Kiev, it was first recorded as breeding in 1948 (Parkhomenko 1950, Charlemagne 2020). However, the information provided by Charlemagne (2020) is contradicted by the data presented by Strautman (1963), who indicates that the species was common in Galicia between 1840 and 1880 and was distributed in Galicia, Romania, the vicinity of Tiraspol, Bukovina, the upper Prut river, and the northern foothills of the Carpathians. Additionally, records indicate the birds' dispersal in western Ukraine (in Volyn, Lviv, Zakarpattia, and Ivano-Frankivsk regions) (Strautman 1963, Pisulinska *et al.* 2016). Subsequently, the European Serin continues to expand its breeding range in an eastern direction. The birds establish new regions through the formation of focal settlements, from which they subsequently disperse. This process was observed in the Baltics, where the European Serin initially distributed in small, scattered groups in new areas (Kumari 1958). A similar pattern can be observed in Central and Eastern Ukraine, where the species has penetrated in the last two decades and currently inhabits localised subpopulations. In the western region, where the species has been breeding for over a century, the frequency of observations is evenly distributed across the entire territory (*Figure 1*).

In established populations of the species in Ukraine, an increase in numbers has been observed since the middle of the 20th century (Bashta 1999, Knaus 2020). At the same time, decreasing population size has been observed in Europe since 1982 (BirdLife International 2024). Bird abundance have declined particularly significantly in Portugal, although in some regions of Europe, for example in Slovenia, it has increased (Knaus 2020). This intriguing discrepancy in the species' population trends across different regions necessitates a comprehensive investigation.

In Ukraine, the area inhabited by the European Serin continues to expand. The modern breeding range of the species encompasses the entirety of the west and north of Ukraine, with its southern border traversing the Cherkassy, Dnepropetrovsk, and Donetsk oblasts. The status of the species in Odessa oblast requires further clarification. The species has already been documented in the Black Sea region, and although birds were first observed in the Odessa region in 1990 (Yakovlev 2015), the status of the species remains uncertain, as all descriptions of the species in this region pertain to the migratory period. Nevertheless, adult birds and fledglings are regularly recorded in the region (Yakovlev 2015). In Poltava (Nankinov 2011, UaBirds), Dnepropetrovsk (Sizhko 2006, author's data) and Donetsk (author's data) regions, European Serin appeared already in the 21st century.

The primary factor driving the range expansion of numerous species, including the European Serin, is the shift in the European climate towards less continental (Huntley *et al.* 2008). The range of the European Serin has been expanding northwards in a gradual manner (Kinzelbach 2004). It is postulated that, having crossed the Alps, the birds encountered optimal conditions devoid of competitors. Further northward dispersal was attributed by Mayr (1926) to the emergence of mutants with an increased desire for expansion, with

the number of dispersing individuals accounting for approximately 20% of the population. The introduction of European Serin into the northwest occurred in a broad front and at a greater rate than into the southeast (Mal'chevskii & Pukinskii 1983). Over the course of 170 years, the European Serin expanded its range by 2.5 million km², with an average rate of 14,300 km² per year. This expansion was punctuated by shifts in the habitat boundary, which occurred at a distance of 100–270 km from previous records (Payevsky 2015). By the end of the 20th century, the species was observed in almost all of continental Europe during the summer period. The subsequent slowdown in dispersal rate is attributed to the challenging climatic conditions of northern regions (Alimova & Bogutskaya 2004). We posit that the slow dispersal of European Serin in the eastern direction is due to the necessity for the birds to develop the Carpathian region and to settle mountainous terrain. The total area of mountain valleys and uplands requires a larger number of individuals than the flat areas of Central and Northern Europe. For example, European Serin inhabited the Ukrainian Carpathians for approximately 100 years, first appearing in 1840 in Bukovina and only in 1939 in Volyn and Kremenets (Strautman 1963). Subsequently, the birds migrated to the plains, occupying Transcarpathia in a southern direction within 10 years and reaching Kiev in an eastern direction (Peklo 2008, Pisulinska *et al.* 2016, Charlemagne 2020). Over the consecutive 50 years, its range expanded to encompass the middle reaches of the Dnieper river (*Figure 1*) along both its banks. By 2022, only the southern and south-eastern regions of Left-Bank Ukraine remained uninhabited. It can be predicted that by the middle of this century, the European Serin's breeding range may extend to the northern coasts of the Black and Azov Seas. It is also important to recognise that military actions in this region may both accelerate and slow down the process of bird dispersal.

The occupation of new territories by a species is most often initiated by young individuals during the 'prolongation of migration', who then nest further away from the former range boundary with each spring (Payevsky 2015). It is believed that the dispersal of the European Serin was a completely natural process, without any influence of random introductions, and accompanied by a change in attachment to specific biotopes (Payevsky 2015). From a preference for forest edges, clearings and small groups of trees among open landscapes, the species migrated to urbanised biotopes, including parks, cemeteries, gardens and suburban areas with sparse vegetation. As a native of the Mediterranean, it found in European settlements a habitat that was highly similar to its original environment (Belik 1977).

It is also important to consider the impact of human activity on the landscape when analysing the factors influencing the dispersal of the species. Urban biotopes, which occupy more and more living space every year, represent a mosaic landscape of islands of green areas in buildings (Croci *et al.* 2008, Miroshnyk *et al.* 2021). European Serin have become well adapted to living in these urbanised green areas. Thanks to their presence in steppe areas that have always been deficient in tree plantations, the birds have been able to populate regions rich in food resources for grain-eating birds but previously unsuitable for dendrophiles.

In conjunction with the expansion of the breeding range, the wintering grounds of birds are undergoing a transformation. Traditionally, most populations of European Serin gather in flocks and migrate in September (Knysh & Malyshok 2015). In Europe, the species winters in suburban areas with ruderal herbaceous communities, often in vineyards, olive

or almond groves and other non-irrigated lands, where its average density reaches from 100 to 240 birds/km² and it is less often found in urban parks. Some populations of birds from different regions of Europe winter in the southern regions of the continent, while 10% of populations are sedentary (Senar *et al.* 2011). They leave Ukraine for the winter in October. However, in western Ukraine, European Serin was already recorded in small numbers in winter during the 20th century (Mistryukova 1996, Skilsky 1998). These birds are kept in winter in botanical gardens and arboretums. In the 21st century, the species was described wintering in Armenia too (Ananyan & Boyadzhyan 2016). It is believed that since European Serin has been hunted to extinction, there are no threats to the species in Europe. Its numbers are negatively affected only by shortcomings in agricultural policy (Senar *et al.* 2011). It is proposed that fallow lands and wastelands be developed to provide the birds with food during the winter months.

Conclusion

The European Serin continues to disperse southwards and eastwards in Europe. The current breeding range of the species encompasses the entirety of the west and north of Ukraine, with the southern border of the species traversing Central and Eastern Ukraine, extending to the right bank of the Dnieper river and the Black Sea regions. However, the status of this population in this region requires further clarification. It is similarly important to examine the comparative dynamics of bird populations in old habitats and in newly developed areas. This can help identify new wintering grounds and populations that have shifted to a sedentary lifestyle.

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Domed nests do not offer additional environmental protection in some songbirds

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Abstract Nest architecture in birds is highly variable ranging from simple scrapes through to elaborate woven constructions. In species that nest in open situations the nest can have a cup open to the elements or the nest has a dome positioned over the cup. The functional properties of domed nests have yet to be explored and this study compared the thermal insulation and rainproofing properties of nests built by four species of European songbird, two of which build domed nests whereas the others build open cup nests. Insulatory values were recorded using temperature loggers. Nests were exposed to a simulated rain event and the amount of water absorbed and the minutes for the nest to dry were determined. The nests were then deconstructed into their component parts. Materials used to build the nests differed among the species. Differences in thermal insulation were associated with the species and the size of the nest. The amount of water absorbed by a nest during a simulated rainfall event was negatively associated with the size of the nest. Feathers and moss had significant positive effects on thermal insulation and rainproofing, respectively. There was no significant difference between domed and open nests in terms of thermal insulation or rainproofing, except for the time taken for a nest to dry, which showed a significant interaction between nest mass and type of nest. Insulatory values and degree of rainproofing were like data from previous reports for songbird nests of comparable size. That no differences observed between domed and open cup nests in the species studied may reflect similarity among species, although it may be due to a paucity of data from a wider range of species building domed nests.

Keywords: cup nest, domed nest, insulatory value, Long-tailed Tit, Northern Wheatear, rainproofing, Reed Bunting, Wood Warbler

Összefoglalás A madárfészkek formája és szerkezete igen változatos, az egyszerű kaparékoktól a bonyolultan szövött szerkezetekig terjed. A nyílt helyen fészkelő fajoknál a fészkek lehet az időjárásnak jobban kitett nyitott csésze vagy a csésze fölött kupola helyezkedhet el. A kupolás fészkek funkcionális tulajdonságainak feltáráshoz még további vizsgálatok szükségesek. Ez a tanulmány négy európai énekesmadárfaj fészkeinek hőszigetelő és esőszigetelő tulajdonságait hasonlította össze, amelyek közül két faj kupolás fészket, míg a másik kettő nyitott fészket épít. A szigetelési értékeket hőmérséklet-nyilvántartó készülékekkel rögzítettük. A fészkeket szimulált esőzésnek tettük ki, és meghatároztuk a felszívódó víz mennyiségét, valamint a fészkek száradásához szükséges időt. Ezután a fészkeket alkotóelemeikre bontottuk. A fészkek építéséhez használt anyagok a fajok között különböztek. A hőszigetelésben mutatkozó különbségek a fajhoz és a fészkek méretéhez kapcsolódtak. A fészkek által egy szimulált esőzés során elnyelt vízmennyiség negatívan függött a fészkek méretétől. A tollak és a moha jelentős pozitív hatást gyakoroltak a hőszigetelésre, illetve az esőszigetelésre. Sem a hőszigetelés, sem az esőszigetelés tekintetében nem volt szignifikáns különbség a kupolás és a nyitott fészkek között, kivéve a fészkek száradási idejét, amely szignifikáns kölcsönhatást mutatott a fészkek tömege és a fészkek típusa között. A szigetelési értékek és az esőszigetelés mértéke megegyezett a hasonló méretű énekesmadár fészkekben mért adatokkal. Az, hogy a vizsgált fajoknál nem észleltünk különbséget a kupolás és a nyitott fészkek között, a fajok közötti hasonlóságot tükrözheti, bár lehet, hogy ez a kupolás fészket építő fajok szélesebb köréből származó adatok hiányának tulajdonítható.

Kulcsszavak: csésze-alakú fészkek, esőszigetelés, hantmadár, kupolás fészkek, nádi sármány, őszapó, síségő fűzike, szigetelési érték

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Introduction

Nest construction is a critical aspect of successful reproduction in birds because it is the location for incubation (Deeming 2016) and, in many cases, rearing of chicks through to fledging (Deeming 2023). Whether reproduction is successful will depend on, *inter alia*, the nest minimising energy loss, providing protection from adverse environmental conditions and from possible predation, and remaining physically secure during incubation and rearing (Hansell 2000, Deeming 2023). Over recent years there have been many studies that have explored the key role of the materials used to construct songbird nests in terms of maintaining structural integrity (Biddle *et al.* 2017, 2018b), providing thermal insulation (Crossman *et al.* 2011, Mainwaring *et al.* 2012, 2014, Deeming *et al.* 2020a), maintaining an appropriate nest humidity (Deeming *et al.* 2020b), and providing adequate protection from rainfall (Biddle *et al.* 2019). Despite these studies, our understanding of functional properties of nests remains limited to relatively few species of songbirds despite the observed diversity in nest architecture (Hansell 2000, Deeming 2023).

For thermal insulation, studies have mainly focussed on European species that build cup nests in open environments or, in a few cases, in cavities (see Deeming *et al.* 2020). However, many passerine species build domed nests, which may have been the primitive condition for nest architecture (Price & Griffith 2017). Several macroecological studies have suggested that a domed nest confers an advantage to the species that this is linked to reduced predation and increased environmental protection (Hall *et al.* 2015, Medina *et al.* 2022, Vanadzina *et al.* 2024). A nest cup with a domed roof could potentially offer increased protection against heat loss, increase waterproofing of the cup, or hide the incubating bird, its eggs and/or chicks from predators approaching the nest from above. For instance, Duursma *et al.* (2018) found that in Australia domed nests were more prevalent in hotter, drier regions and likely provided shade to the nest contents where there were sparse plant canopies. Indeed, Zebra Finches (*Taeniopygia guttata*) appear to modify their nest architecture to ameliorate the effects of high environmental temperatures (Griffith *et al.* 2016). Moreover, the pendent nests of some weaverbird (Ploceidae) and icterid (Icteridae) species may offer structural and environmental protection to the eggs and chicks (Street *et al.* 2022). Comparing temperate and tropical locations, Martin *et al.* (2016) concluded that enclosed nests offered important thermal benefits compared with open nests, but this conclusion was only based on indirect evidence associated with factors such as nest size and presumed cooling rates of eggs. All these things could confer an advantage to the parent or its offspring during incubation or chick rearing, although Humphries *et al.* (2007) found that a domed nest seemed to offer no advantage with respect to protection from predation of flooding to clutches of Saltmarsh Sharp-tailed Sparrows (*Ammodramus caudacutus*). The studies above all offer only indirect evidence that domed nests confer an advantage to the birds that build them and to date there has been no study that has tested the thermal properties of domed nests in the same way as

for open-nesting species (Deeming *et al.* 2020a). Developing a better appreciation of the functional properties of domed nests could improve our understanding of variation in nest architecture is being exploited by birds when reproducing.

This study describes the detailed composition of domed nests from two species of European passerine in comparison with two species of European passerine that build open cup nests. Data are available for insulatory values for all species except the Northern Wheatear (*Oenanthe oenanthe*) although methodologies vary among studies (Deeming *et al.* 202a). Limited data are available for composition of the Long-tailed Tit (*Aegithalos caudatus*) nest (Hansell 2000, McGowan *et al.* 2004) but this falls short of other more detailed studies of nest composition (e.g. Biddle *et al.* 2018a, Briggs & Deeming 2016, 2012, 2022, Dickinson *et al.* 2022). There are no compositional data for the other species in this study, which determined values for thermal insulation as an insulatory value (McGowan *et al.* 2004, Deeming *et al.* 2020a), and measures of the degree of rainproofing, i.e. water absorbed and minutes to dry (Biddle *et al.* 2019), were also determined for all nests. The effect of species was explored in the analysis but the main hypotheses were that: (1) the roof of the domed nests would offer more physical protection and so provide better thermal insulation and rainproofing than open nests; and (2), as has been demonstrated in other nests of songbirds, amount of materials, such as feathers, would contribute to thermal insulation, whereas amounts of moss would be associated with the degree of rainproofing during a simulated rain event. Data on detailed nest composition could also help improve our understanding of factors that affect the degree of inter- and intra-specific plasticity in nest construction.

Methods

Data were collected for nests from four species of passerine bird: two species that build domed nests, i.e., the Long-tailed Tit (*Aegithalos caudatus*, Aegithalidae; body mass = 8.6 g; n = 17 nests) and the Wood Warbler (*Phylloscopus sibilatrix*, Phylloscopidae; body mass = 9.2 g; n = 5), and two species that build open cup nests, i.e. the Northern Wheatear (*Oenanthe oenanthe*, Muscicapidae; body mass = 24.0 g; n = 7) and the Reed Bunting (*Emberiza schoeniclus*, Emberizidae; body mass = 17.2 g; n = 8). The Long-tailed Tit nests were all collected in Sheffield, South Yorkshire, Great Britain, during 2021 but the other nests were supplied from various locations within Great Britain between 2013 and 2021. Prior to being tested, all nests were stored dry in plastic bags kept in cardboard boxes held at room temperature and humidity.

Thermal insulation of the nests was determined using temperature loggers (McGowan *et al.* 2004, Mainwaring *et al.* 2012, 2014, Deeming & Gray 2016, Deeming & Campion 2018, Deeming *et al.* 2020a). Briefly, for open nests, a temperature logger heated in a water bath to a temperature of 80 °C (iButtons, Maximintegrated.com, accuracy ± 1 °C from -30 °C to $+70$ °C in 0.5 increments) was placed on a small polystyrene plinth and the nest was placed upside down over the logger so that it contacted the base of the nest. A second logger heated at the same time was placed on a similar plinth next to the nest. Data collection took place within a closed box (which contained a third, unheated logger that recorded the temperature

in the box during the experiment) that controlled for air movements as the loggers cooled (Gray & Deeming 2017). For the domed nests, the heated logger was pushed through the entrance hole into the depths of the nest and a block of polystyrene was pushed into the nest cavity to hold the logger next to the nest wall. Otherwise, the domed nests were treated in the same way as the open cup nests. The difference in cooling rates of the two loggers (i.e. the one under or in the nest and the one adjacent to the nest) was the insulatory value (IV in $^{\circ}\text{C}\cdot\text{min}^{-1}$) (McGowan *et al.* 2004). For each nest, data were collected three times and the mean IV calculated, with larger values indicating better insulation.

Each nest was weighed (in g) using an electronic balance (Sartorius) prior to being exposed to simulated rainfall using the method described by Biddle *et al.* (2019). Briefly, for nests of the Reed Bunting and Common Wheatear, a nest was placed in a metal household sieve placed over a bowl. To stimulate a rainfall event, 250 ml of water was poured into a plastic cup with holes in its bottom, which was held and rotated about 30 cm above the nest. Water fell as droplets on to the nest for about three minutes and percolated through the nest wall to drain into the bowl. A sheet of 3 mm thick polystyrene was cut to the shape of the nest cup and placed over the cup to exclude water during the simulated rain. After draining at room temperature ($\sim 22^{\circ}\text{C}$) and humidity ($\sim 50\%$ RH) for 10 minutes from the start of the 'rainfall', nests were weighed. Thereafter, nests were kept at room temperature and humidity and weighed 1, 2, 4, and 6 hours after the rain event. The only difference for domed nests was that they were placed in the sieve with the opening to the interior of the nest facing downwards thereby preventing water from directly entering the nest cavity and pooling in the nest cup. The amount of water absorbed (WA) was determined by subtracting the original dry mass of the nest from its mass after 10 minutes. The WA value was also expressed as a percentage of the dry nest mass (%WA). The time in minutes taken for nests to dry out (MTD) was determined by using an exponential equation from Biddle *et al.* (2019) to predict when the nest returned to its original dry mass.

After thermal and rainproofing testing the nests were deconstructed into their component materials, which were allocated to either the outer nest wall or the cup lining (Biddle *et al.* 2018a). Nests were deconstructed by systematically pulling the nest wall apart with forceps and separating the various materials (Britt and Deeming 2011, Biddle *et al.* 2018a, Dickinson *et al.* 2022). Materials, such as droppings or bones of dead neonates, were considered as being added to the nest after initial construction so dry nest mass (in g) was determined after deconstruction by summing all the materials deemed to be part of the original nest. There were three categories for animal-derived materials: feathers, hair, and invertebrate silk. Plant-derived materials were separated into: grass (including reed leaves), moss, leaves, lichen, bark, roots, and plant stems (woody and more flexible stems combined). Any mineral material was classified as mud.

Prior to analysis, to ensure that values of zero (for example, many nests lacked certain nest materials) could be analysed, all data values had 1 added before being \log_{10} -transformed. Linear modelling performed in R version 2.4.3 (R Core Development Team 2023) tested for effect of species on the variables of interest, including nest materials, IV, WA, etc. Analysis of covariance tested for the effects of species as a fixed factor and total nest mass as a covariate using linear models in R. Where there was a significant effect of species pairwise

comparisons were conducted (with Bonferroni correction) to identify where significant difference lay. A linear model was also used to determine the effects of the different nest materials on IV or measures of rainproofing; the model included all material types and was applied to data for all nests irrespective of species.

Linear mixed effects regression analysis was carried out in R using the R packages *lme4* (Bates *et al.* 2015) and *lmerTest* (Kuznetsova *et al.* 2017) to test for the effect of total nest mass as a covariate and type of nest (i.e., domed versus open) as a fixed factor, on insulatory value and measures of rainproofing. Species was included as a random factor in the model.

Results

On average, the masses of the nests were similar for the four species. Although Long-tailed Tits built the heaviest nest (*Figure 1*), the differences between species were significant ($F_{3,33} = 2.92$, $P = 0.048$; $R^2 = 0.21$) and none of the pairwise comparisons were statistically significant (*Figure 1*). The outer nest formed the bulk (65–77%) of the total nest mass but was significantly affected by species ($F_{3,33} = 6.68$, $P = 0.001$; $R^2 = 0.38$). This effect was due to data for the Northern Wheatear nests where the outer wall was only ~50% of the total mass, which was significantly lower than values for the Long-tailed Tit and Wood Warbler (*Figure 1*). Similarly, cup lining mass was significantly affected by species ($F_{3,33} = 6.58$, $P = 0.001$; $R^2 = 0.37$) with the cup lining of Reed Bunting nests being significantly lighter in mass from that of the Long-tailed Tit and Northern Wheatear (*Figure 1*).

Not all materials were found in nests, nor in the outer nest, nor in the cup lining, of all species (*Figure 2*). Reed Bunting nests were dominated by grass in the outer nest and the cup lining (*Figure 2*). Northern Wheatear nests were characterised by a large amount of hair found in the cup lining with a range of plant materials in the outer nest (*Figure 2*). Long-tailed Tit nests had a lot of moss and lichen (especially *Parmelia sulcata*, *Parmelia subaurifera*, and *Xanthoria parietina*) held together by silk in the outer nest but the cup lining was almost all feathers, which were largely absent in the other species' nests (*Figure 2*). It was interesting to note that simulated rainfall hydrated the moss and lichen on outside of Long-tailed Tit nests changing the colour to a dull grey-green to a more vibrant green within only a few seconds (DCD *per. obs.*). Wood Warblers built nests that had a wide range of plant materials but had no animal-derived materials present (*Figure 2*). This distribution of materials meant that linear models used to make among-species comparisons showed that mass of all the material categories was significantly affected by species (*Table 1*).

Insulatory value was higher for Long-tailed Tit nests compared with the other species and there was a significant effect of species ($F_{3,33} = 9.64$, $P < 0.001$; $R^2 = 0.47$) with significant pairwise differences between the Long-tailed Tits and the Reed Bunting and Northern Wheatear nests, but not Wood Warbler nests (*Figure 3*). Linear models to test for the effect of species did not show any significant effect of species on WA, %WA, or MTD ($F_{3,33} = 2.09$, $P = 0.12$; $R^2 = 0.16$; $F_{3,33} = 0.48$, $P = 0.700$; $R^2 = 0.04$; and $F_{3,33} = 1.58$, $P = 0.213$; $R^2 = 0.13$, respectively).

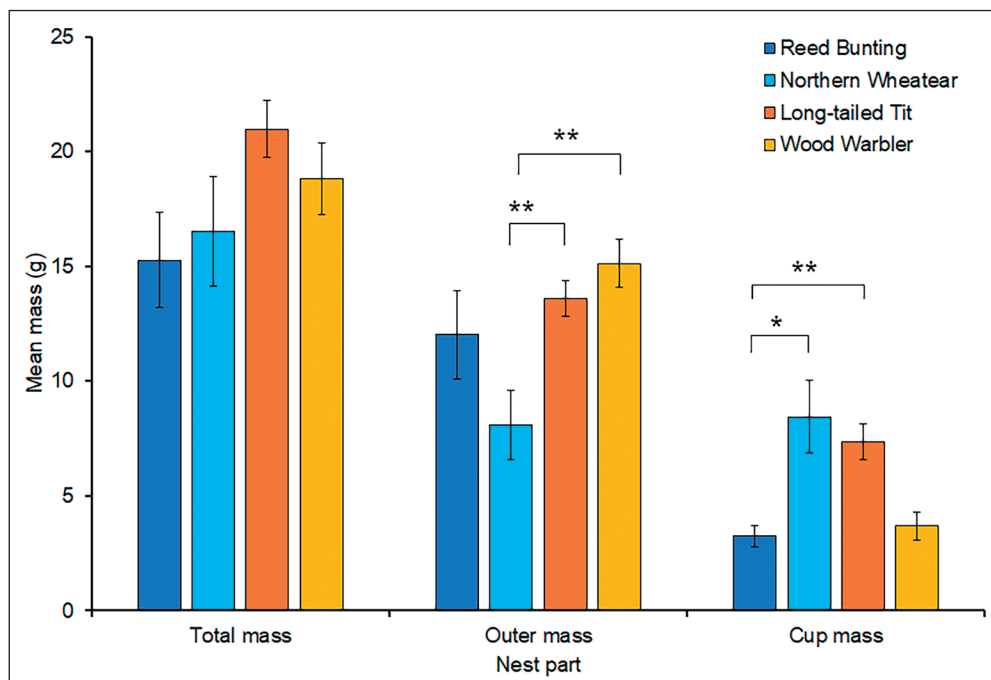


Figure 1. Mean (\pm SE) values for total nest mass, outer nest mass and mass of the cup lining for the species studied. Blue columns indicate species that have open nests with orange/yellow columns indicating species with domed nests. Significant ($P < 0.05$) differences between species are indicated by brackets and * above the relevant columns. Other comparisons between columns were not significant

1. ábra A fészkek teljes tömegének, a fészkek külső tömegének és a csésze bélése tömegének átlagértékei (\pm sztenderd hiba) a vizsgált fajoknál. A kék oszlopok a nyitott fészkek fajokat, a narancssárga/sárga oszlopok a kupolás fészkek fajokat jelölik. A fajok közötti szignifikáns ($P < 0,05$) különbségeket zárójel és * jelzi a vonatkozó oszlopok felett. Az oszlopok közötti egyéb összehasonlítások nem voltak szignifikánsak

Given this result, further linear models were run to test for the effect of total nest mass, species, and any interaction. For IV, there was a significant effect of species and total nest mass plus there was a significant interaction between these two terms (Table 2). By contrast, both WA and %WA were significantly affected by total nest mass but not species and there was no significant interaction (Table 2). Values for MTD were unaffected by nest mass or species and there was no significant interaction (Table 2).

To test for the type of nest on IV or WA, linear mixed effect modelling showed that there was no effect of total nest mass or nest type with no significant interaction (Table 2). By contrast, the total nest mass exhibited a significant negative relationship with %WA (Figure 4) but there was no significant interaction with nest type (Table 2). For MTD, type of nest was significant as was the interaction with total nest mass (Table 2); domed nests had higher values for MTD as nest mass increased but for open cup nests this relationship was negative (Figure 5).

Linear models showed that different nest materials contributed to IV and the various measures of rainproofing. For IV, the mass of feathers was the most significant covariate but

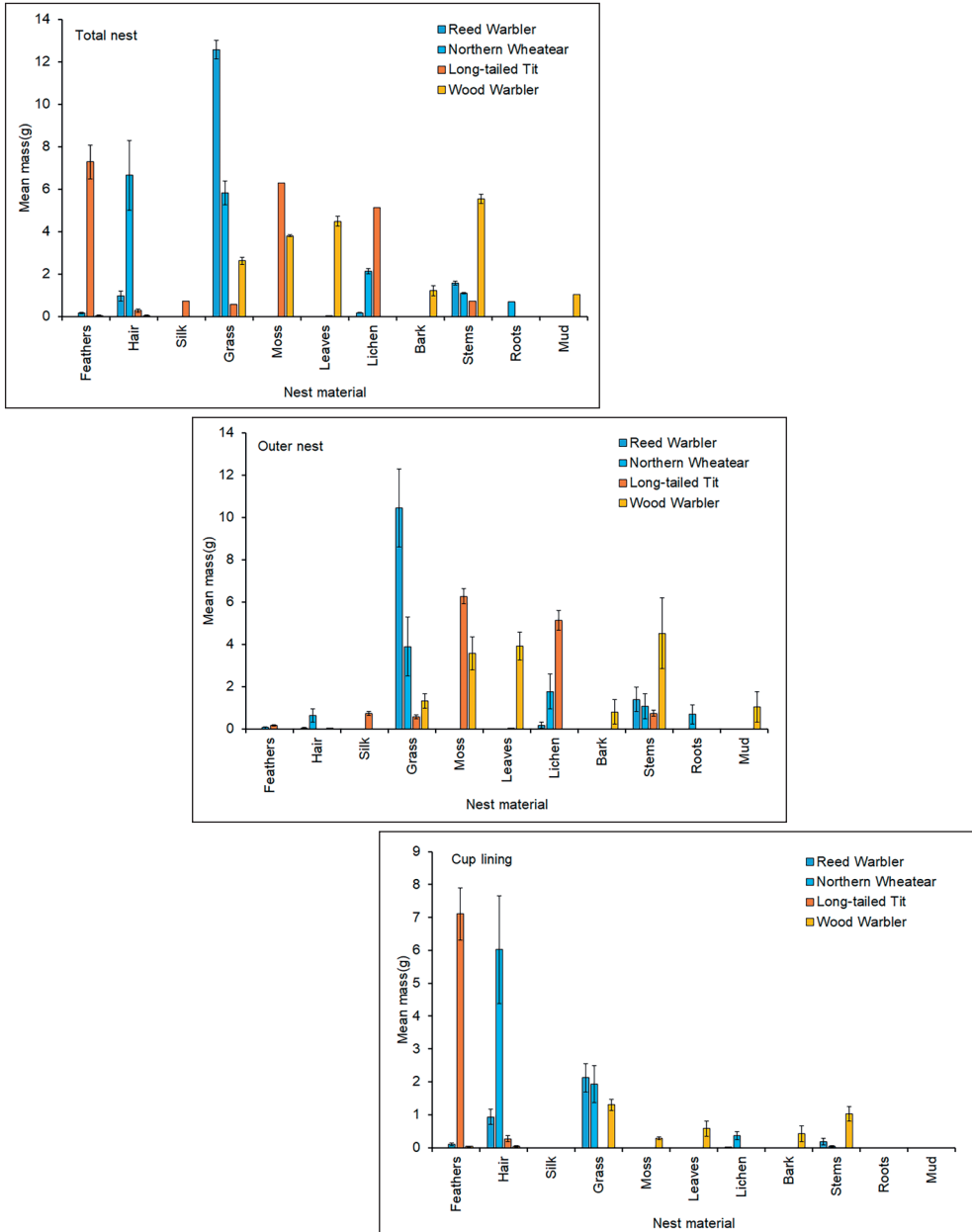


Figure 2. Mean (\pm SE) values for the eleven categories of materials found in the total nest (top panel), the outer nest (middle panel) and the cup lining (bottom panel) for four species of songbird. Blue columns indicate species that have open nests with orange/yellow columns indicating species with domed nests

2. ábra A teljes fészkekben (felső panel), a fészkek külsejében (középső panel) és a csésze bélésében (alsó panel) található tizenegy anyagkategória átlagos (\pm szterderd hiba) értékei négy énekesmadárfaj esetében. A kék oszlopok a nyitott fészű fajokat, a narancssárga/sárga oszlopok a kupolás fészű fajokat jelölik

Table 1. Results of analysis of variance to test the significance of species as a categorical factor on log₁₀-transformed data for the various nest materials found in the total nest, the outer nest wall, and the cup lining. R² = coefficient of determination

1. táblázat A faj mint kategorikus tényező hatásának vizsgálatára szolgáló varianciaanalízis eredményei a teljes fészket, a külső részt és a csésze bélését felépítő különböző anyagok log-transzformált adatainak felhasználásával. R² = determinációs együttható

Nest material	Total nest		Outer nest		Cup lining	
	F _{3,33} (P-value)	R ²	F _{3,33} (P-value)	R ²	F _{3,33} (P-value)	R ²
Feathers	207.15 (< 0.001)	0.95	9.57 (< 0.001)	0.47	198.01 (<0.001)	0.95
Hair	55.19 (< 0.001)	0.83	10.54 (< 0.001)	0.49	36.31 (< 0.001)	0.77
Silk [#]	51.00 (< 0.001)	0.82	51.00 (< 0.001)	0.82	-	-
Grass	51.08 (<0.001)	0.82	41.31 (< 0.001)	0.79	24.39 (< 0.001)	0.69
Moss	296.63 (< 0.001)	0.96	271.88 (< 0.001)	0.96	115.71 (< 0.001)	0.91
Leaves	323.59 (< 0.001)	0.97	271.95 (< 0.001)	0.96	22.28 (< 0.001)	0.67
Lichen	47.87 (< 0.001)	0.81	49.44 (< 0.001)	0.82	18.43 (< 0.001)	0.63
Bark	10.99 (< 0.001)	0.50	6.34 (0.002)	0.37	8.71 (< 0.001)	0.44
Stems	7.29 (0.001)	0.40	5.11 (0.005)	0.32	32.19 (< 0.001)	0.75
Roots [#]	3.50 (0.026)	0.24	3.50 (0.026)	0.24	-	-
Mud [#]	8.68 (< 0.001)	0.31	8.68 (< 0.001)	0.44	-	-

#dash (-) indicates no analysis because the component was absent in the cup lining.

amounts of hair, stems, roots, and mud also had significant effects (Table 3). By contrast, WA values were only significantly affected by the significant positive relationship with the mass of moss in nest (Table 3). None of the nest materials had any significant effect on %WA or MTD (Table 3).

Discussion

The materials used to build the nests of the four species were very different from each other. Differences in thermal insulation were associated with the species and the size of the nest whereas the amount of water absorbed by a nest during a simulated rainfall event had a negative relationship with the size of the nest. Contrary to expectations there was little effect of nest type, i.e. domed versus open cup, on thermal insulation or rainproofing and a range of materials significantly contributed to IV and WA but not for other measures of rainproofing. As predicted materials, especially feathers and moss, used in the nests had significant effects on thermal insulation and rainproofing.

Nest composition is a species-specific characteristic (Deeming 2023). Data for a range of passerine species show that whilst some closely related species, e.g., thrushes (Turdidae), can have similar nest compositions, other species that are closely related, e.g. finches (Fringillidae), can differ greatly in nest composition and architecture (Biddle *et al.* 2018a, Dickinson *et al.* 2020). Nest architecture for any species is broadly comparable among individuals, which means that a range of field guides for nests can be produced (e.g. Beruldsen

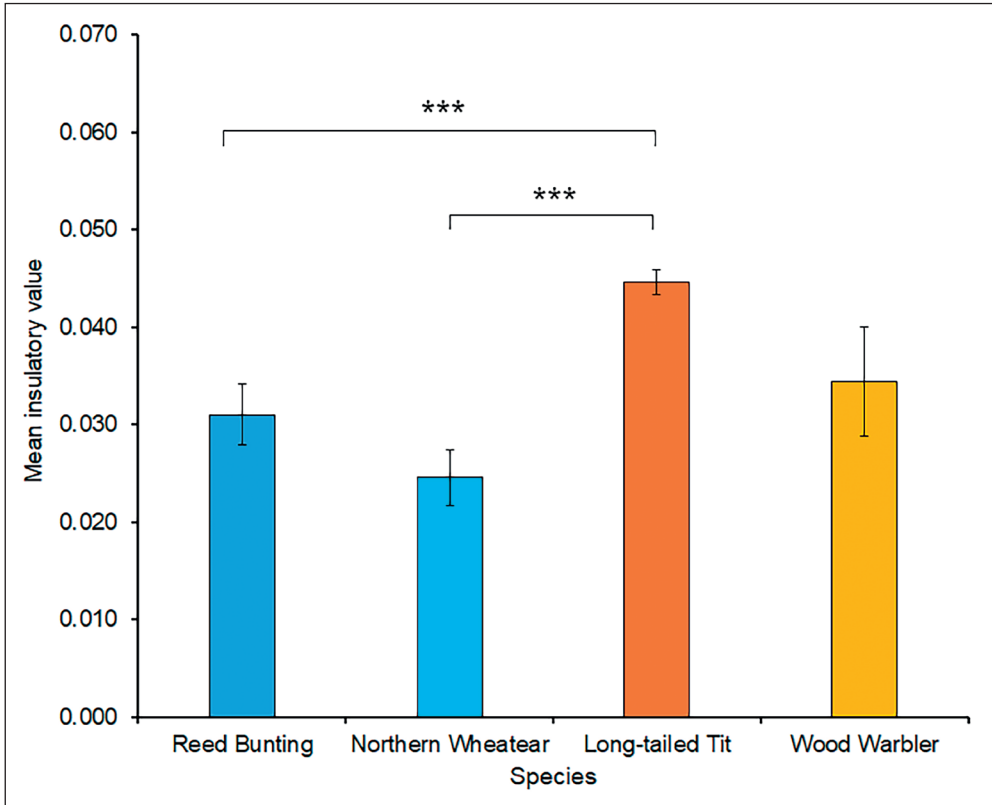


Figure 3. Mean (\pm SE) values for insulatory value in four species of songbird. Blue columns indicate species that have open nests with orange/yellow columns indicating species with domed nests. Significant ($P < 0.001$) differences between species are indicated by brackets and *** above the relevant columns. Other comparisons between columns were not significant

3. ábra A szigetelési értékek átlagos (\pm sztenderd hiba) értékei négy énekesmadárfajnál. A kék oszlopok a nyitott fészekű fajokat, a narancssárga/sárga oszlopok a kupolás fészekű fajokat jelölik. A fajok közötti szignifikáns ($P < 0,001$) különbségeket a zárójelek és a *** jelzi a vonatkozó oszlopok felett. Az oszlopok közötti egyéb összehasonlítások nem voltak szignifikánsak

2003, Ferguson-Lees *et al.* 2011, McFarland *et al.* 2021). However, choice of nesting material appears to be very plastic. For instance, intra-specific variation in nest composition can be affected by geographical location across several hundred kilometres (e.g. Crossman *et al.* 2011, Briggs & Deeming 2016), or locally across less than 10 km (e.g. Briggs & Deeming 2021, 2022). Indeed, when building a nest birds will take advantage of material types found nearby close at hand (Briggs & Deeming 2016, 2022) or use materials that resemble materials used previously, such as Pied Flycatchers (*Ficedula hypoleuca*) using plastic string instead of mammal hair (Briggs *et al.* 2023). For the Long-tailed Tit, nest composition described here was comparable to previous reports in terms of the birds using feathers to line the nest cavity (Hansell 2000, McGowan *et al.* 2004). However, the nest built almost entirely out of artificial insulation materials reported by Broughton and Parry (2020) rather than lichen, moss, and feathers, further highlights the plasticity of nest construction behaviour in birds. Nest

Table 2. Mean (\pm SE) values for insulatory value, water absorbed 10 minutes after simulated rain, the amount of water absorbed expressed as percentage of nest mass, and the time in minutes that the nest was predicted to dry back to its original dry condition. At the base of each column there the results of (1) linear models to test for the effect of total nest mass as covariate, species as a fixed factor plus the interaction; and (2) linear mixed effects regression analysis using Satterthwaite's method that tested for the effect of total nest mass as a covariate, type of nest (domed versus open cup) as a fixed factor, and the interaction whilst controlling for species as a random factor. Significant values are highlighted in bold. For the linear model the F-value and R^2 = coefficient of determination for the model is included

2. táblázat A szigetelési érték, a szimulált eső után 10 perccel elnyelt víz, a fészek tömegének százalékában kifejezett elnyelt vízmennyiség és a száradási idő (percben kifejezve) átlagértékei (\pm sztenderd hiba). Az egyes oszlopok alján (1) a fészek teljes tömege mint kovariáns, a faj mint fix tényező és a kettő kölcsönhatását vizsgáló lineáris modellek eredményei; és (2) a Satterthwaite-féle módszert alkalmazó lineáris kevert regressziós elemzés, amely a fészek teljes tömege mint kovariáns, a fészek típusa (kupolás vagy nyitott) mint fix tényező és ezek kölcsönhatását vizsgálta, miközben a faj mint véletlen tényező szerepelt a modellben. A szignifikáns értékek félkövérrel szedettek. A lineáris modell esetében az F-érték és az R^2 , vagyis a modell determinációs együtthatója szerepel

	Insulatory value (°C·min ⁻¹)	Water absorbed (g)	%Water absorbed	Minutes to dry
Open				
Reed Bunting	0.031 (0.003)	14.85 (1.50)	105.17 (12.39)	3036.3 (637.3)
Northern Wheatear	0.025 (0.003)	17.21 (2.72)	111.86 (10.07)	1705.8 (252.8)
Domed				
Long-tailed Tit	0.045 (0.001)	19.31 (1.47)	95.48 (7.76)	2206.7 (517.0)
Wood Warbler	0.034 (0.006)	21.97 (2.18)	110.22 (9.34)	2204.6 (449.44)
Linear model				
Total nest mass	20.15 (< 0.001)	14.71 (0.001)	9.79 (0.004)	0.11 (0.745)
Species	12.30 (< 0.001)	0.64 (0.595)	0.64 (0.594)	1.61 (0.208)
Total nest mass * Species	3.68 (0.023)	0.32 (0.813)	0.32 (0.813)	1.64 (0.203)
F _{7,29} (P-value) for model	9.66 (< 0.001)	2.51 (0.038)	1.81 (0.123)	1.41 (0.240)
R ²	0.70	0.38	0.30	0.25
Linear mixed effects model				
Total nest mass	3.37 (0.076)	6.06 (0.019)	10.56 (0.003)	1.56 (0.221)
Type	0.06 (0.814)	0.62 (0.435)	0.62 (0.435)	5.62 (0.024)
Total nest mass * Type	0.29 (0.594)	0.49 (0.491)	0.49 (0.491)	5.33 (0.028)

composition for the Reed Bunting, Northern Wheatear, and Wood Warbler were generally like nest composition reported for closely related species (Biddle *et al.* 2018a, Deeming *et al.* 2020a, Dickinson *et al.* 2022). For instance, like the Reed Bunting, Yellowhammer (*Emberiza citrinella*) nests were mainly built from grass (Deeming *et al.* 2020a), and Northern Wheatear

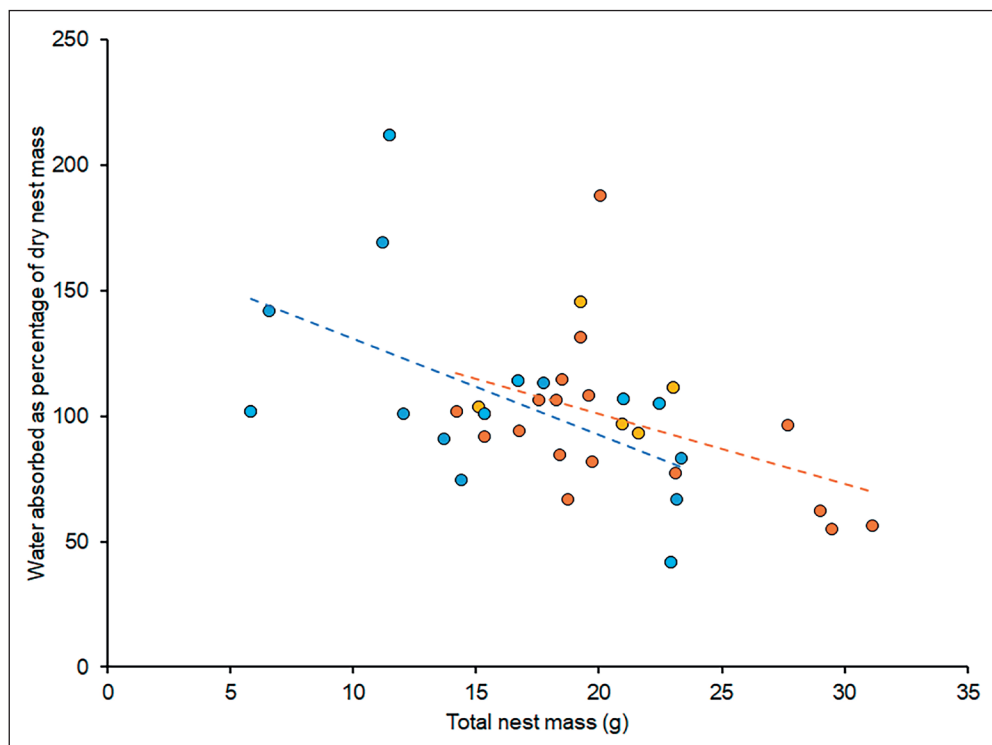


Figure 4. Relationships between total nest mass and water absorbed during simulated rainfall, expressed as a percentage of dry nest mass, for open cup nests (blue symbols) and domed nests (orange symbols). Dashed lines represent least squares linear regression relationships calculated by Excel to illustrate the pattern for the two nest types

4. ábra A fészkek teljes tömege és a szimulált esőzés során elnyelt vízmennyiség közötti összefüggés, a fészkek száraz tömegének százalékában kifejezve, a nyitott fészkek (kék szimbólumok) és a kupolás fészkek (narancssárga szimbólumok) esetében. A szaggatott vonalak az Excel programmal kiszámított legkisebb négyzetek lineáris regressziós összefüggéseit jelölik, amelyek a két fészektípusra vonatkozó mintázatot külön szemléltetik

nests were similar in composition to those of Stonechats (*Saxicola torquatus*) (Biddle *et al.* 2018a). Like Wood Warblers, Willow Warblers (*Phylloscopus trochilus*) built nests with a mix of grass, moss but with more feathers and hair (Dickinson *et al.* 2022). It would be interesting to systematically compare dissimilarity in nest composition across a range of passerine species to determine whether closely related species are building nests that more similar than we perhaps expect.

The mean insulatory value for Long-tailed Tit nests reported here was about a third lower than values reported by McGowan *et al.* (2004). This may reflect differences in methodology because McGowan *et al.* (2004) used thermistors attached to the surface of heated clay 'eggs' left in the nest cup rather than self-recording data loggers. Moreover, in the present study the nest was upside-down, which suggests that the IV value should have been greater because convection from the temperature logger would be minimal compared to a dummy egg placed in the base of the nest (see also Boulton & Cassey 2012). In any case, the IV

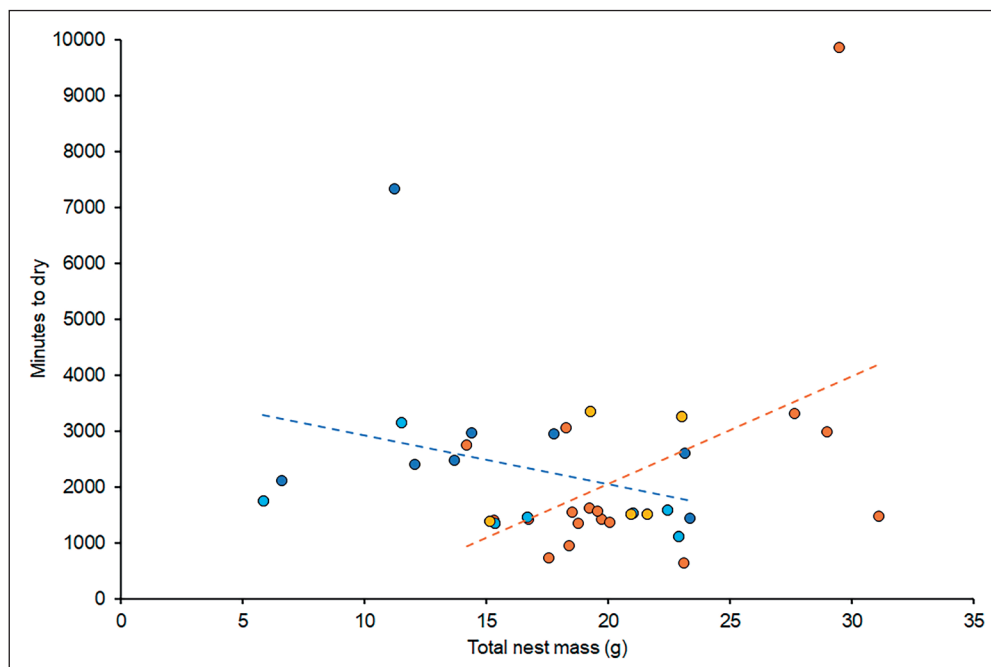


Figure 5. Relationships between total nest mass and time in minutes for a nest to dry after simulated rainfall for open cup nests (blue symbols) and domed nests (orange symbols). Dashed lines represent least squares linear regression relationships calculated by Excel to illustrate the pattern for the two nest types

5. ábra A fészkek teljes tömege és a fészkek kiszáradásához szükséges idő (percben kifejezve) közötti összefüggés a szimulált esőzés után a nyitott fészkek (kék szimbólumok) és a kupolás fészkek (narancssárga szimbólumok) esetében. A szaggatott vonalak az Excel programmal kiszámított legkisebb négyzetek lineáris regressziós összefüggéseit jelölik, amelyek a két fészektípusra vonatkozó mintázatot külön szemléltetik

values for species studied here were comparable to species means reported for a range of similar-sized songbirds, i.e. $0.03\text{--}0.08\text{ }^{\circ}\text{C}\cdot\text{min}^{-1}$ (Deeming *et al.* 2020a). Exceptionally, IV for the Wood Warbler nests were much lower than a previous report, i.e. 0.034 vs $0.120\text{ }^{\circ}\text{C}\cdot\text{min}^{-1}$ (Deeming & Biddle 2015), but this difference was almost certainly due to the methodologies used. Instead of having the temperature logger lying in the nest cup, the techniques used by Deeming and Biddle (2015) required the logger to be pushed into the cup lining, which is known to correlate with higher IV values in other species (Deeming *et al.* 2020a). The mean IV values for Willow Warbler and Reed Bunting nests reported by Deeming *et al.* (2020a) were slightly higher than that reported here, whereas values for the Northern Wheatear nests were lower than reported for other chats and robins in the Muscicapidae (Deeming *et al.* 2020a). Although these differences may reflect methodological difference, other studies suggest that they reflect natural variation between species (Deeming *et al.* 2020a), or indicate that nests where a nest is built is important. It is known that latitude significantly affects thermal insulation in a range of species due to differences in nest composition (Mainwaring *et al.* 2012, 2014). New data reported here are within the range of data for IV for nests

Table 3. Results of linear regression analysis that tested for the effect of the log-transformed values for mass of different nest materials as covariates irrespective of species. Significant values are highlighted in bold. For the linear model the F-value and R² = coefficient of determination for the model is included at the base of each relevant column

3. táblázat A lineáris regresszió eredményei, amely a különböző fészekanyagok tömegének log-transzformált értékei mint kovariánsok hatását vizsgálta, függetlenül a fajoktól. A szignifikáns értékek félkövérrel szedettek. A lineáris modell esetében az F-érték és az R², vagyis a modell determinációs együtthatója minden vonatkozó oszlop alján szerepel

Dependent variable	Insulatory value		Water absorbed		Water absorbed as percentage of mass		Minutes to dry	
	Coefficient (SE)	F-value (P-value)	Coefficient (SE)	F-value (P-value)	Coefficient (SE)	F-value (P-value)	Coefficient (SE)	F-value (P-value)
Intercept	-1.648 (0.122)		0.638 (0.185)		2.328 (0.203)		3.569 (0.348)	
Feathers	0.069 (0.147)	42.76 (< 0.001)	-0.002 (0.222)	1.74 (0.199)	-0.515 (0.244)	2.48 (0.128)	0.305 (0.418)	0.49 (0.491)
Hair	-0.074 (0.093)	12.29 (0.002)	0.211 (0.139)	0.72 (0.406)	-0.082 (0.153)	0.40 (0.535)	-0.495 (0.263)	1.80 (0.191)
Silk	-0.176 (0.262)	0.03 (0.875)	-0.428 (0.340)	0.44 (0.513)	-0.317 (0.435)	0.00 (0.967)	-0.346 (0.746)	0.77 (0.390)
Grass	0.065 (0.092)	2.81 (0.106)	0.382 (0.139)	0.56 (0.462)	-0.230 (0.153)	0.94 (0.341)	0.030 (0.263)	1.61 (0.216)
Moss	0.123 (0.180)	2.03 (0.166)	0.560 (0.272)	12.67 (0.002)	0.272 (0.299)	0.06 (0.804)	-0.558 (0.512)	1.84 (0.187)
Leaves	-0.565 (0.174)	3.97 (0.057)	0.178 (0.262)	0.21 (0.654)	-0.145 (0.288)	1.70 (0.204)	0.249 (0.494)	0.20 (0.656)
Lichen	0.147 (0.096)	0.01 (0.935)	0.171 (0.144)	1.64 (0.213)	0.014 (0.159)	0.08 (0.779)	0.038 (0.272)	0.11 (0.748)
Bark	0.489 (0.192)	154 (0.227)	-0.072 (0.290)	0.15 (0.702)	-0.190 (0.319)	0.00 (0.954)	-0.077 (0.546)	0.02 (0.898)
Stems	0.230 (0.069)	11.02 (0.003)	0.125 (0.104)	1.45 (0.241)	-0.157 (0.114)	1.98 (0.177)	-0.128 (0.195)	0.42 (0.521)
Roots	-0.541 (0.151)	11.15 (0.003)	0.012 (0.228)	0.01 (0.941)	-0.204 (0.251)	0.82 (0.374)	0.215 (0.430)	0.25 (0.620)
Mud	0.569 (0.221)	6.62 (0.016)	-0.428 (0.333)	1.65 (0.211)	-0.313 (0.367)	0.73 (0.402)	0.004 (0.629)	0.00 (0.994)
F _{11,25} (P-value)		8.57 (< 0.001)		1.98 (0.084)		0.83 (0.613)		0.68 (0.742)
R ²		0.79		0.46		0.27		0.23

constructed by songbird species of similar body masses and so offer no surprises. Deeming (2016) suggested that as birds get heavier their reliance on the nest to insulate the clutch of eggs is reduced. Further research, therefore, should perhaps focus on measuring the IV of nests built by much larger passerines, e.g. the Corvidae or Cracticidae, than those studied to date to assess the role of the nest in providing nest insulation.

In relation to nest materials the four species studied here had IVs affected by positive relationships for the amounts of feathers, stems, and mud but negative relationships with hair and roots. By contrast, using data for many more species Deeming *et al.* (2020a) showed that insulation related to the increasing amounts of only feathers and moss. These differences between studies may reflect dissimilarity in composition of the four species studied here than was exhibited by the larger sample size of species and nests used by Deeming *et al.* (2020a). Given that different materials will vary in their thermal properties (see Hilton *et al.* 2004, Deeming *et al.* 2020c), further studies could focus on the insulation offered by nests mainly built from materials other than plants or derived from animals, e.g. the mud nests of swallows (Hirundinidae) (Ferguson-Lees *et al.* 2011).

In terms of rainproofing, the nests of the four species studied here had values for water absorbed and minutes to dry within the range reported by Biddle *et al.* (2019). Similarly, despite nests substantially increasing in mass the amount of water absorbed was less than 10% of the mass of water that rained upon them suggesting that the nests had quite effective drainage. Larger domed nests absorbed more water and take longer to dry although, contrary to the report by Biddle *et al.* (2019), for the two species with open cup nests there seemed little effect of nest mass. As was also shown by Biddle *et al.* (2019) the amounts of moss in the nest significantly affected water absorbed, but not as a percentage of dry nest mass. It is acknowledged by Biddle *et al.* (2019) and here that testing rainproofing in a laboratory environment has its limitations because it does not account for other forms of shelter that may be present around the nest *in situ*. Development of a methodology that allows investigation of nests at their construction site would certainly provide a better understanding of how the immediate surroundings affect not only nest function but also how this impacts upon the success of incubation of eggs and rearing of nestlings.

Simulated rainfall reduced the IVs of a range of songbird nests due to a higher internal rate of nest cooling (Deeming & Campion 2018). It is interesting to note that the smallest change in IV was for nests of the Common Chaffinch (*Fringilla coelebs*; Fringillidae) which, compared to the other species studied by Deeming and Campion (2018), has a high proportion of feathers lining its nest cup (Biddle *et al.* 2018a, Deeming *et al.* 2020a). These feathers help contribute to 90% of the thermal insulation of the nest despite the cup lining only being a third of the total nest mass (Deeming & Humphries 2020). Given the hydrophobic nature of feathers (Lui *et al.* 2008) it is possible that a lining of feathers in a nest cup may not only provide thermal insulation but also, they help repel water in the outer nest wall and so keep the cup dry. It would be interesting to explore this possibility in Long-tailed Tit nests and other species, e.g. the European Goldfinch (*Carduelis carduelis*), where the cup lining has a proportion of feathers (Biddle *et al.* 2018a), or to experimentally add feathers to nests.

The results of the present study conflict with the conclusions of previous studies. In particular, Martin *et al.* (2017) suggested that enclosed nests conferred thermal benefits to the

eggs and chicks, that were more important than reduced predation risks. However, Martin *et al.* (2017) drew their conclusions based on interpretation of nest size, attentiveness of parents during incubation, presumed rates of egg cooling and rates of nestling growth, rather than measuring the actual insulation offered by the nests. Other studies that infer advantages to domed nests are also based on indirect measures of nest success (Griffith *et al.* 2016, Duursma *et al.* 2018, Street *et al.* 2022). It is acknowledged that the study described here has a very limited sample but it does suggest that nest architecture may not play such an important role in providing thermal insulation as has been suggested. However, there needs to more research in a wider range of species and nesting locations in order to test this hypothesis further.

This study has confirmed that nest insulation and rainproofing properties varies between species but the data suggested that there is no consistent difference between domed and open nests. Although the dome covering the top of the nest may have little effect on thermal insulation or rain protection, the lack of difference may reflect limitations of the study due to small samples of nests and/or species, nest composition, or geographical variation. During incubation the dome may not have much impact on the thermal profile of the clutch of eggs, but it could confer insulation/rainproofing to the parent during incubation, or to the nestlings during rearing. The presence of a dome may reflect a degree of protection from predators but for many open cup nesting species the construction of a nest in a location surrounded by vegetation may serve the same role. Given that domed nests are commonly built by passerines (Price & Griffith 2017), more research is needed on the functional properties of domed nests to determine whether they differ from those of open cup nests.

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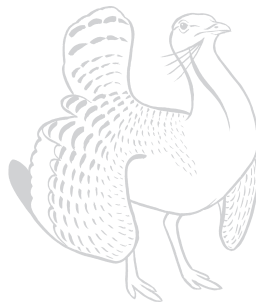
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Migration and wintering of the Mistle Thrush (*Turdus viscivorus*) in SE Hungary

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Abstract The migration and wintering of the Mistle Thrush (*Turdus viscivorus*) is the least known of the other common nesting thrushes in Europe. It is also an understudied species in Hungary, detailed studies having been carried out nearly 50 years ago. The aim of the present study was to summarise the data on migration and wintering of this species that have been collected over the last 20 years in Kevermes, southeastern Hungary. There is no breeding population of the species either here or in the surrounding area. The species was observed on 371 of the 2,807 days of field observations. The first individuals typically arrived in the second week of October, but there are also data from late September and early October. Its migration is characterized by several peaks. In spring, it was still migrating intensively in the first half of April, with the last individuals leaving the area by the end of the month. In recent years, they have appeared earlier in the autumn and left later in the spring, which may coincide with the increasing breeding population of the species in Hungary. The number of individuals observed was higher in spring than in autumn and winter, similar to the situation in other parts of the country. The number of birds was high every second or third year. The number of observations in the same seasons of different years showed different patterns as well as the autumn observations did not allow any conclusions to be drawn about the number of observations later in winter and spring. The most important food was the common hackberry (*Celtis occidentalis*), which was protected against the Fieldfares (*T. pilaris*). Comparing the two species, the migratory patterns were basically similar, but the seasonal abundance of the two species differed in certain years.

Keywords: breeding success, *Celtis occidentalis*, Fieldfare, food supply

Összefoglalás A léprigó (*Turdus viscivorus*) vonulása és telelése a többi, Európában gyakori fészkelő rigófaj közül a legkevésbé ismert. Magyarországon is alulkutatott faj, a részletesebb vizsgálatok közel 50 évvel ezelőtt történtek. Jelen tanulmánynak az volt a célja, hogy összefoglalja a faj vonulásával és telelésével kapcsolatos adatokat, amelyek az elmúlt 20 évben összegyűltek a délkelet-magyarországi Kevermesen. Sem itt, sem a környező térségben nem költ a léprigó. A 2,807 terepi megfigyeléssel töltött napból 371 napon figyeltem meg a fajt. Az első példányok jellemzően október második hetében jelentek meg, de van szeptember végi és október eleji adata is. Vonulását több csúcs jellemzi. Tavasszal még április első felében is intenzíven vonult, az utolsó példányok a hó végéig hagyták el a területet. Az utóbbi években ősszel egyre korábban jelentek meg, míg tavasszal egyre később vonultak el, ami a faj növekvő állományával magyarázható. A megfigyelt példányok száma tavasszal nagyobb volt, mint ősszel és télen, hasonlóan az ország más részein tapasztaltakhoz. A madarak száma minden második vagy harmadik évben kimagasló volt. A különböző évek azonos időszakaiban történt megfigyelések száma eltérő mintázatot mutatott. Az őszi megfigyelések száma alapján nem lehetett következtetni a téli és tavaszi megfigyelések számára. Legfontosabb tápláléka a nyugati ostorfa (*Celtis occidentalis*) termése volt, amit védelmezett a fenyőrigókkal (*T. pilaris*) szemben. Utóbbi fajjal összehasonlítva alapvetően hasonló volt a faj vonulási mintázata, de a két faj mennyisége bizonyos években elért egymástól

Kulcsszavak: fenyőrigó, költési siker, nyugati ostorfa, táplálékkínálat

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Introduction

The Mistle Thrush (*Turdus viscivorus*) is a polytypic species with 3 subspecies in its range. The nominate *T. v. viscivorus* is found from Europe to western Siberia and northern Iran, *T. v. deichleri* in northwest Africa, Corsica and Sardinia, while *T. v. bonapartei* nests from southcentral Asia to Turkmenistan and western Nepal (Collar 2020, Gill *et al.* 2023). It nests most parts of Europe except treeless regions. The largest populations occur in the taiga forests of Scandinavia and in the mountainous regions of western and central Europe. From the 19th century onwards, it colonised lowland areas, expanding into northern Scandinavia, Ukraine, Hungary and Italy. In England, Ireland and the Netherlands, however, there has been a significant population decline in recent decades, for which the exact reasons are not known (Knaus 2020).

Partial migrant, with a decreasing migratory propensity southward (Ashmole 1962, Huttunen 2004). At least 50 km from the ringing site, only 1,087 individuals have been encountered in Europe, which is at least an order of magnitude less than other European nesting species, except the Ring Ouzel (*T. torquatus*). The birds with the highest number of encounters are those marked in the Netherlands, France and Germany, while there are very few records from eastern European countries (Spina *et al.* 2022). The species migrates southwestwards from most regions (Bønløkke *et al.* 2006, Spina & Volponi 2009, Budinski 2013, Spina *et al.* 2022), but there are exceptions, such as birds from Finland and the UK, whose recaptures are widely dispersed but show an average southerly orientation (Ashmole 1962, Huttunen 2004). The birds ringed and later recaptured in Sweden wintered further south than the Dutch and Belgian ones, which probably explains that the species is a leapfrog migrant (Ashmole 1962). Most of the Mistle Thrushes that have been encountered at least 50 km from the ringing site have flown a maximum of a few hundred km, but some have flown 2,000–3,000 km (Spina *et al.* 2022). Analysis of data on recaptured birds during autumn migration in France over the last 70 years showed that Dutch and Belgian Mistle Thrushes were mainly observed in the west of France, while German individuals are widely distributed across the country (Lahournat *et al.* 2021). It spends the winter in southwestern Europe, but a few birds ringed in central Europe have been found in northwestern Africa (Spina *et al.* 2022).

It mainly breeds at the edge of old pine and mixed forests, but also prefers oak forests, planted forests and parks. In winter, it occurs in places where it finds sufficient food. During this period, it feeds mainly on berries, with a particular preference for mistletoe (Snow & Snow 1984). Studies in Italy have found that it avoids food-poor and grassy areas, but habitat use is 60% similar to that observed during the breeding season (Chiatante 2022). In winter, Mistle Thrushes maintain small territories on wintering grounds, which they protect not only from their conspecifics but also from other species (Snow & Snow 1984). This often involves protecting one or a few berry trees that are tall and solitary (Snow & Snow 1984). Cramp (1988) suggests that the species avoids dense forests, but this was not confirmed in Poland (Skórka & Wójcik 2005). In years when food was abundant, birds did not behave territorially, but fed in flocks. The extent of territorial behaviour decreased as winter progressed, probably due to a decrease in the birds' density (Cuadrado 1995, Skórka & Wójcik 2005).

In Hungary, it nests mainly in the Transdanubian region, in the hills and mountains of northern Transdanubia and in the central and southern parts of the Danube-Tisza basin (Gyurácz & Csörgő 2021). While it used to nest in hilly and mountainous oak forests, it now breeds in lowland acacia and pine forests, floodplains, shrub-wooded pastures and parks (Haraszthy 2019). The first lowland nesting was recorded in 1962 near Sárszentmihály (Pátkai 1963), while the first urbanised pairs were observed in the 1970s (Horváth 1976, Bozsko 1985a, Novák 1985). Later, it also bred in Budapest (Rózsa 2003). In the early 2020s, 3–4 pairs bred in the castle park in Fehérvárcsurgó (Kovács 2020).

Due to the absence of recaptures, no information is available on the migration and wintering sites of the Hungarian Mistle Thrushes (Csörgő & Gyurácz 2009). In autumn, the first migrants appear in the second half of September, with most observations from November. After that, the overwinterers can be seen. In spring, the last migrants leave Hungary by the end of April at the latest (Csörgő & Gyurácz 2009, Gyurácz & Csörgő 2021). According to Schmidt (2000), the birds wintering in Hungary probably originate from the surrounding high mountains, while the local population winters in the vicinity of the nesting sites. In contrast, Horváth's (1972) studies over several years have shown that the local population have most certainly left the breeding sites. These birds may be on the breeding grounds in spring as early as February. Pairs are formed during the winter, and nesting is sometimes complete by the end of March, but typically only in the first half of April. They usually breed only once a year, but sometimes twice (Haraszthy 2019). Between 2014 and 2018, the national nesting population was estimated at 12,000–14,000 pairs, with an increasing trend. In contrast, the wintering population has decreased moderately (Gyurácz & Csörgő 2021).

The migration of the species in Europe has only been studied in detail in very few works (Ashmole 1962, Huttunen 2004, Lahournat *et al.* 2021), and the national migration atlases provide only limited information too (Bønløkke *et al.* 2006, Spina & Volponi 2009). This is due to the fact that, compared to other European nesting thrush species, the number of individuals ringed is orders of magnitude lower (typically less than 100 individuals per year per country), and the number of recaptures is much lower than for other species. There is also a lack of knowledge on the timing of migration. In Italy, for example, there is a distinct autumn peak because this is when hunting of the species was most intense (Spina & Volponi 2009).

In Hungary, sporadic data on the breeding (Thibaut de Maisières 1940, Farkas 1951, 1955, Szijj 1955, Pátkai 1963, Horváth 1972, Varga 1976, 1979, 1982a, 2003, Anonym 1978a, 1978b, 1979, 1980, Barbácsy 1978, 1981a, 1981b, 2006, Homoki-Nagy 1980, Kasza 1983, Schmidt 1983, Szenek 1984, Bozsko 1985a, Molnár 1985, Novák 1985, Balsay 1986, Mogyorósi 1988, 2021, Moskát 1988, Varga 1989, Németh 2000, Rózsa 2003, Aczél 2011, 2012, Gyurácz 2012, Illés & Kóta 2012, Faragó & Szentirmai 2014), autumn and spring migration (Barta 1978, Szvezsényi 1978, Dénes 1981, 1982, Schmidt 1992, Gyurácz *et al.* 2007), and winter occurrences (Varga 1928, Simig 1978, Bali 1980, Ács 1981, Király & Varga 1981, Schmidt 1981, Szalai 1981, Varga & Király 1981, Nagy 1982, Varga 1982b, Bozsko 1985b, Matyikó 1985, Kalivoda 1986, Zsoldos 1986, Kalotás 1987, Bali 1990, Király 1992, Fintha & Szabó 1993, Kárpáti 2003, Kelemen 2003, Bedőcs

2013) of the species can be found in various journals. However, these usually contain only a few or very few observations, which do not provide a clear picture of the migration and wintering of the species.

The only study dealing with the species in detail was published nearly 50 years ago (Horváth 1972), showing that until then it was considered a resident bird in the Carpathian Basin, migrating from the Carpathian mountains to the plains for the winter (e.g. Chernel 1899). However, Horváth (1972) pointed out the fact that the species migrates and winters in Hungary in much larger numbers than the nesting population. He also mentioned that the *Aquila* journal (Herman 1894), in its 75 volumes, had only two articles with a few lines written on the species. The situation has not changed in the 50 years since then, so I thought it important to present my findings on the migration of the species. In this study, I have discussed the migration of the species in southeastern Hungary. The results are mainly based on field observation data. I was interested in the timing of the migration of the species and the differences in the number of birds that appear in each season.

Material and Methods

The study was carried out in the area of Kevermes and Lőkösháza in the southeastern part of Hungary. I considered the administrative boundaries of the two villages with an 80 km² total area as the territory of my research (Hevesi 2005). The landscape is dominated by agricultural lands with some planted forests, artificial lakes, canals and steppes (for details see Bozó 2018). The core area was the surroundings of Fenyves forest, with an approximate 50 ha total area. This area included the lake gravel pit (open water surface with dense lakeside vegetation), the forest of the former pheasant station (oleaster (*Elaeagnus angustifolia*) plantation), the Fenyves forest (dominated by common hackberry (*Celtis occidentalis*), common oak (*Quercus robur*), acacia (*Robinia pseudoacacia*) and a poplar (*Populus* spp.) plantation).

The study seasons covered the period between 1 January 2005 and 30 April 2024 in every year. The data collection was based on field observations with binocular. Because of the data collection was non-standard between 2005 and 2011, I do not considered the data of these period from the detailed analyses. For the description of migration periods and winter movements, I only considered the data from 2012 to 2024, when the intensity of data collection was similar. During this period, I visited the study site on 2,807 different days and I have seen the species on 371 different days.

I was interested in the earliest autumn and latest spring data, the length of time spent per season, and the observation date of the maximum number of individuals in a given season. For annual earliest and latest observations, I aimed to see if there were changes in the date of arrival and departure. For this goal, I used General Linear Models (dependent variable: season, independent variables: date of first autumn arrivals, date of last spring departures). I plotted the number of observations per 10-day interval and the number of individuals observed (in the latter case, cumulative and 10-day average values). I also plotted the number of observations per autumn, winter and spring seasons annually. In this

context, I used Spearman’s correlation to see if the number of observations in autumn was correlated with the number of observations in winter and spring. I looked at how the number of observations varied in the different seasons between 1 September and 30 April.

Results

Based on the data between 2012 and 2024 the first Mistle Thrushes arrive to the area in mid-October, rarely in late September and early October (earliest observation: 28 September 2023). Then, the number of observations increases and peaks in the first week of November. After that there is a decrease in mid-November followed by another peak between late November and mid-December. From late December to mid-February, the number of observations is constant, with a minimum in late February. The spring observations peak in late March and early April. The species has been rarely seen in the second half of April (latest observation: 28 April 2014) (Figure 1).

The 10-day average number of individuals observed in the months when the species was recorded varied between 1 and 4.3 individuals. Based on cumulative data, most individuals were observed in November and the first half of December, but there was also a small peak in late January and the mid-March (Figure 2).

Between 2005 and 2024, I observed the species on 371 different days. The number of observations varied between seasons. The fewest number of sightings (n = 10) was in 2017/18, while the highest number of sightings was in 2023/24 (n = 53) (Figure 3). Figure

The number of observations in the autumn (September – November), winter (December – February) and spring (March – April) seasons also differed (Figure 4). The number of observations in autumn was not correlated with the number of observations in winter ($R = -0.267$, $P = 0.426$) and spring ($R = 0.194$, $P = 0.566$).

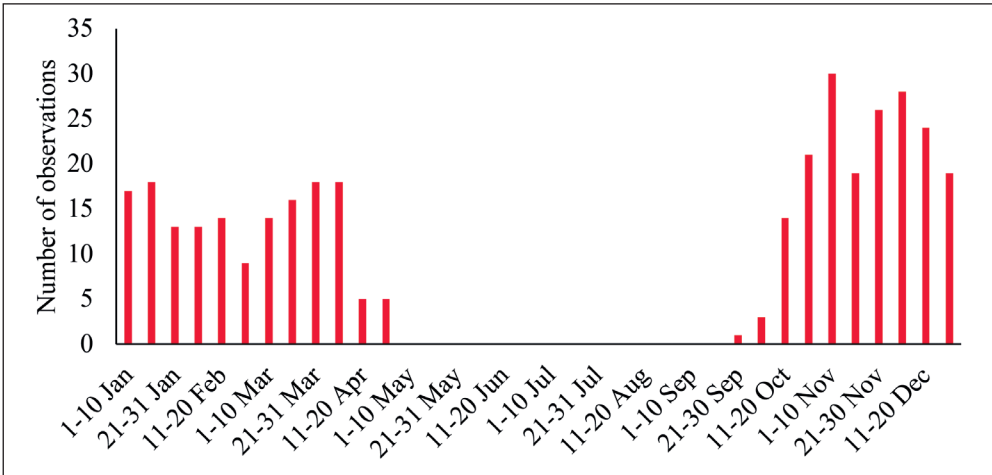


Figure 1. Distribution of the Mistle Thrush observations by 10-day intervals
1. ábra A léprígó megfigyelések 10 naponkénti eloszlása

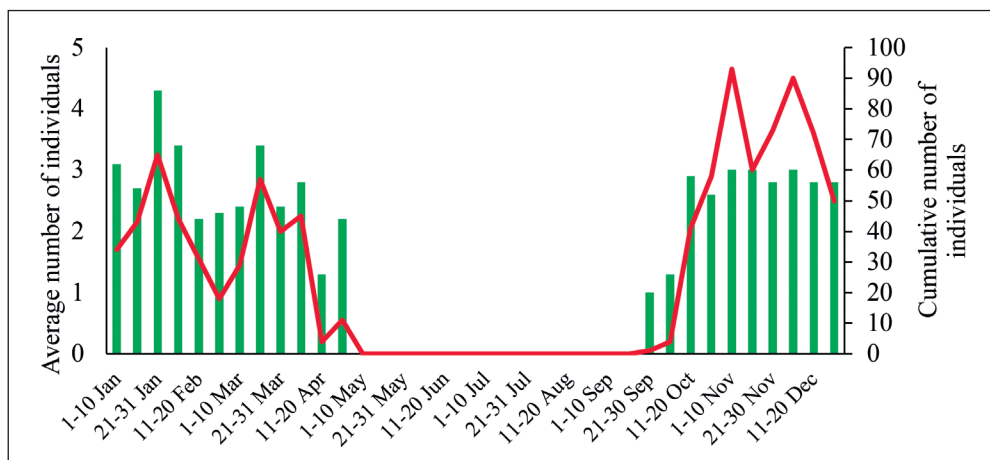


Figure 2. Average (column) and cumulative (line) number of individuals observed per 10 days
2. ábra A 10 naponként megfigyelt példányok átlagos (oszlop) és kumulált (vonal) száma

The dates of the birds' autumn arrival and spring departure also differed between years. The date of autumn arrival shifted earlier between 2011 and 2023 (Coeff. = -0.211 , Std. error = 0.095 , $t = -2.211$, $R^2 = 0.26$, $P = 0.051$), while the date of spring departure shifted later between 2012 and 2024 (Coeff. = 0.13 , Std. error = 0.083 , $t = 1.571$, $R^2 = 0.315$, $P = 0.147$), but the change was not significant in either case (Figure 5, Table 1).

The seasonal maximum varied between 4 and 17 individuals. On average, they spent 182.5 days in the area (minimum: 160 days, maximum: 201 days). The largest observed flock consisted of 17 individuals (15 March 2019). The seasonal maximum was mainly in autumn ($n = 8$), but it also occurred in spring ($n = 4$) or winter ($n = 3$) (Table 1).

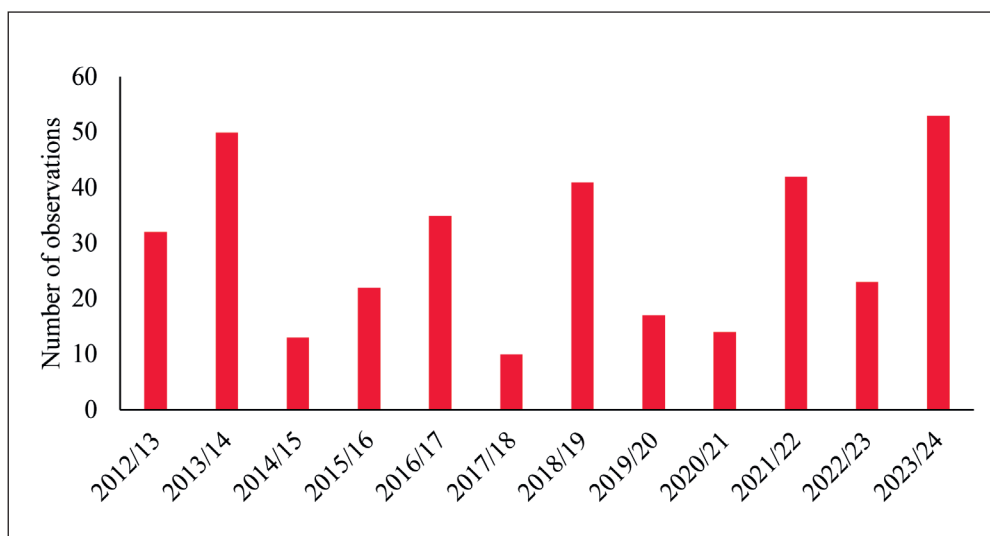


Figure 3. The number of observations by seasons
3. ábra A különböző szezonokban történt megfigyelések száma

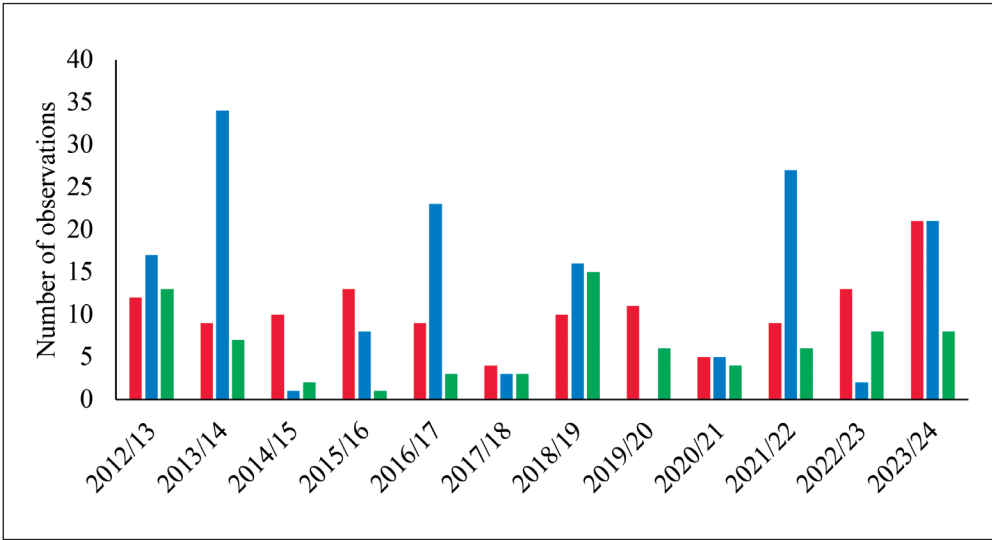


Figure 4. Number of observations in different autumns (red), winters (blue) and springs (green)
4. ábra A különböző szezonok őszi (piros), téli (kék) és tavaszi (zöld) időszakában történt megfigyelések száma

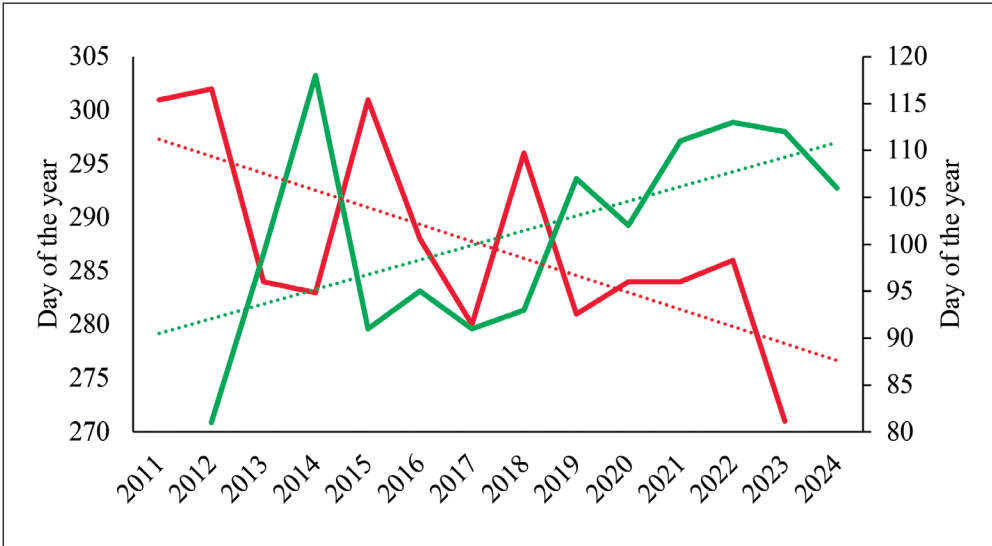


Figure 5. The dates of the earliest autumn (red line) and the latest spring (green line) records by year
5. ábra A legkorábbi őszi (piros vonal) és legkésőbbi tavaszi (zöld vonal) léprigó megfigyelések időpontja évenként

Table 1. The dates of the earliest autumn, the latest spring observations and the seasonal maximums with the maximal numbers and the number of days spent in the area in the given season

1. táblázat A legkorábbi őszi és a legkésőbbi tavaszi megfigyelések, a szezonális maximumok időpontjai, a szezonális maximum példányszámok, valamint az adott szezonban itt töltött idő hossza

Season	Earliest observation	Latest observation	Peak day	Number of birds on peak day	Days spent in the area
2012/13	29 Oct	09 Apr	04 Feb	8	163
2013/14	11 Oct	28 Apr	20 Oct	12	200
2014/15	10 Oct	01 Apr	22 Nov	5	174
2015/16	28 Oct	05 Apr	03 Nov	10	160
2016/17	15 Oct	01 Apr	27 Oct, 04 Dec, 14 Dec	10	169
2017/18	07 Oct	03 Apr	16 Nov	10	179
2018/19	23 Oct	17 Apr	15 Mar	17	177
2019/20	08 Oct	12 Apr	18 Mar	6	187
2020/21	11 Oct	21 Apr	21 Apr	6	193
2021/22	11 Oct	23 Apr	09 Apr	12	195
2022/23	13 Oct	22 Apr	04 Nov, 24 Nov	4	192
2023/24	28 Sep	16 Apr	28 Oct	16	201

Discussion

The migration of the Mistle Thrush is the least known of all European thrush species, so all research contributes to our knowledge of the species. In the Carpathian Basin, the only study so far based on a multi-year dataset was conducted in the late 1960s and early 1970s, more than 50 years ago (Horváth 1972), and since then almost only summary books have provided general information (Schmidt 2000, Ecsedi & Zöld 2004, Csörgő & Gyurácz 2009, Gyurácz & Csörgő 2021). The ornithological monographs and annotated bird lists by counties published in recent years have only defined the local status of the species, without providing detailed data on its migration (Kovács 2020, Gyurácz & Kóta 2021, Bozó 2023). Based on data from 2004 to 2019, a similar study on the Fieldfare was carried out in the study area (Bozó 2019), so it was possible to compare the migration and wintering of the two species.

The study area is located in a region where the species does not breed, and the proportion of suitable areas are also very low (Bozó 2018). Nevertheless, it has been found in varying numbers in older planted forests every year. As the species' main winter food, the mistletoe, is not present in the area at all, its most likely food source is the fruit of the common hackberries. This is also the case for the Fieldfare (*T. pilaris*), which occurs in the same habitat patches (Bozó 2019). In mixed flocks with the Fieldfare or in homogeneous groups, Mistle Thrushes can also be observed in ploughs and grasslands foraging for

insects, worms and molluscs (Schmidt 2000, B. L. pers. obs.). My observations suggest that this is a characteristics mainly for migratory birds, less so in winter, which may be due to winter territoriality (Snow & Snow 1984). Overwintering Mistle Thrushes are likely to be defending common hackberries, a secure food source, against their conspecifics and other species, rather than seeking out temporarily available ground-dwelling animals. I have observed several instances of aggressive behaviour, mainly directed at nearby Fieldfares. As the forest cover in the area is extremely low and most of the few forests are poor in food (Bozó 2018), strong competition between the two species can occur. Intraspecific and interspecific competition has been cited by others as the primary cause of territorial habit (Snow & Snow 1984, Skórka & Wójcik 2005). My own study supports the view that the species occurs in dense forests, its occurrence being influenced only by food availability (Skórka & Wójcik 2005).

In autumn, the first individuals were typically arrived in the second week of October, but there was considerable variation between years. In Hungary, they typically breed only once, and these broods become complete by mid-April at the latest. However, a small proportion of the population breeds twice, and clutches may still be present in early July. Given that the youngs become independent after about 6 weeks from egg-laying, birds from the latest brood also fledge by mid-August (Haraszthy 2019). In Göd, the members of the local population left the nesting site in late July – early August, while the first migrants appeared only at the end of September (Horváth 1972). Horváth (1972) found that the plumage of breeding birds was lighter than that of wintering and migrating birds, and that the latter did not sing in the area, so that they could be clearly distinguished from each other. On this basis, he assumed that the local ones did not winter in Hungary and that birds from the north or from the higher mountains such as the Carpathians came to winter. My own results support these assumptions. In spring, sightings were still regular in the first ten days of April, but after that the number of sightings decreased. Observations from the second half of April may even indicate nesting individuals in the area. The dates of arrival in autumn and departure in spring are similar to those previously obtained for the Fieldfare, but an important difference is that individuals from more distant populations of those species arrived in the area only from November (Bozó 2019).

The species has not been proven to breed in Békés County (Bozó 2023), and considering Horváth's (1972) studies, it is likely that members of the native population do not occur in Kevermes, but probably originate from the Apuseni Mountains or the Carpathians (Romania). Both the number of observations and the number of birds observed fluctuated within the season, with several peaks characterising the occurrence, similarly to the Fieldfare (Bozó 2019). Most of the data were from the period November–December, which is at least one month later than the peak observed near Göd (Horváth 1972), but coincides with that observed in Italy (Spina & Volponi 2009). The spring migration near Göd took place between mid-February and mid-March, with local birds returning at the end of March (Horváth 1972). Accordingly, there was no temporal overlap in the presence of nesters and winterers, similar to, for example, southern England, where breeding birds departed between early July and mid-August, while winterers arrived from the north. In contrast, in Portugal and Spain, there were also resident, migratory and winter visitors (Bannerman 1954). In the

Hortobágy, where it was still considered a rare species in the 1970s, it was mainly seen only during migration. In autumn, they typically arrived in the first week of October, and were sporadic until late November – early December. In spring, the first individuals arrived in the last week of February, migrating most intensively between 10–20 March, and occurring in small numbers until mid-April (Ecsedi & Zöld 2004). Based on my own data, spring migration in Kevermes peaked about a week later, in the last ten days of March and the first ten days of April.

The number of observations was particularly high roughly every second or third year. Furthermore, the number of observations varied significantly between the same seasons in different years. In some years it occurred in larger numbers in autumn, in others in winter or spring. Moreover, the number of observations in autumn does not allow us to predict the number of observations in winter and spring. The number of observations in winter also varied considerably between years. In half of the winter seasons, I observed individuals only occasionally, while in other seasons they were in the area throughout the winter. This fluctuation was due to breeding success, weather conditions and food availability in the year, similar to other thrush species (Svårdson 1957, Tyrväinen 1970). Svårdson (1957) found that the Fieldfare invaded Scandinavia every third and fourth year, a similar interval to that I obtained for the Mistle Thrush. However, in none of the cases of the Mistle Thrush can be regarded as an invasive species, as even the largest flocks did not reach 20 individuals, and in most cases only a few individuals were seen. The larger flocks were typically observed during migration. Comparing the data for the Mistle Thrush and the Fieldfare in the study site, it can be seen that there is only a partial overlap in the number of observations and individuals observed each year. For example, in the 2013/14 season, when there were outstanding numbers of Mistle Thrushes in the area, the opposite was true for Fieldfares. In several years, however, numbers of both species were high or even low. It is therefore likely that the origin and seasonal breeding success of the two species may be different.

The seasonal average number of birds was higher in late January and spring than in autumn and winter. There is no obvious reason why they moved in larger groups at this time. This is similar to what was observed in the Hortobágy (Hungary) (Ecsedi & Zöld 2004), but contradicts the finding in Poland (Skórka & Wójcik 2005) that the number of the birds in the area decreased as winter progressed. This is probably due to the fact that the birds found sufficient food in the area throughout the winter.

Although the change was not significant in either case, it is worth noting that in the autumn they appeared earlier and earlier in the area, while in the spring I observed the last ones later and later. It is conceivable that the species has emerged as a nesting species in the area, in parallel with the increasing national nesting population, and this may explain the earlier and later observations. According to András István Csathó (in litt.), the species has become regular in Battonya only in recent years, and was not previously present. As the number of sightings has fluctuated considerably between years, clear trend could not be detected in the last 12 years. However, taking into account the period between 2005 and 2012, there has been a clear increase in the proportion of sightings, so the species has become more common.

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Far from the Palaearctic realm: First breeding record of the Blue-cheeked Bee-eater *Merops persicus* in Peninsular India with notes on its breeding ecology

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Abstract The first breeding site of Blue-cheeked Bee-eater (*Merops persicus*) in Peninsular India was observed in the Andivillai salt pans of Kanniyakumari district, Tamil Nadu. We found a total of 28 breeding burrows distributed into three subsets in the stretch of 30 m. Among 28 nests, 16 were active and 12 were inactive/unused. The active nest-tunnels have a mean tunnel length of 1.53 ± 0.53 m and nest-mouth diameter of 9.93 ± 1.48 cm. While the females excavated tunnels and started brooding males were guarding the colony. The male very often fed the female brooding inside the nest till the hatchlings appeared, and later both parents were observed feeding their chicks during the day. The chicks were first observed peeping out of the tunnels on 22 August 2023. At the end of September few juvenile birds were seen flying along with the adults foraging in the open sky. The Blue-cheeked Bee-eater is considered a passage migrant and winter visitor to the north-western part of India and vagrant in Southern India, but this study confirmed its breeding in Southern India. Additionally, the study provides baseline information on the breeding ecology of the species in India.

Keywords: Blue-cheeked Bee-eater, *Merops persicus*, Meropidae, Peninsular India, breeding ecology

Összefoglalás A zöld gyurgyalag (*Merops persicus*) fészkelését először dokumentáltuk az indiai szubkontinensen az Andivillai sólempárló tavaknál a Kanniyakumari járásban, Tamil Nadu tartományban. Összesen 3 kisebb csoportba rendezett 28 fészket találtunk egy 30 méteres partfal szakaszon. A 28 fészekből 16 bizonyult használatnak, 12 lakatlan volt. A fészkeküreghez vezető alagutak átlagos hossza $1,53 \pm 0,53$ méter, a bejárat nyílás átmérője $9,93 \pm 1,48$ cm. Míg a tojók a fészkelő üregeket mélyítették és költeni kezdtek, a hímek a kolóniát és a párjukat őrizték. A kotlótojókat gyakran a hímek etették a fiókák kikelésig. Ezután mindkét szülő etette a fiókákat. A fiókákat először 2023 augusztus 22-én figyeltük meg a fészkek bejáratánál. Szeptember végén pár fióka már a szüleivel repült, és a levegőben repülő rovarokat kaptak el. A zöld gyurgyalagot eddig vonuló, illetve telelő madárfajként tartották számon India északnyugati részén, és ritka kóborlóként India déli részén. Azonban tanulmányunk bizonyítja a faj költését Dél-Indiában. Emellett cikkünk ismerteti a faj költésbiológiájának alapelemeit.

Kulcsszavak: zöld gyurgyalag, *Merops persicus*, Meropidae, Indiai szubkontinens, fészkelés biológia

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Introduction

Bee-eaters (Meropidae) are specialised aerial insectivores inhabiting the palaeotropical region with 31 species in three genera. The Blue-cheeked Bee-eater (*Merops persicus*) has two recognized subspecies, the first *Merops persicus chrysocercus* breeds in the periphery of Western Sahara and winters in West Africa, and the second *Merops persicus persicus* breeds along Nile Delta and in Asia and winters in tropical Eastern Africa (Fry *et al.* 1988, Fry 1992, Gunnarsson & Ekblom 2019, Fry & Kirwan 2020). The Blue-cheeked Bee-eater and European Bee-eater (*Merops apiaster*) are the two species that mostly breeds in assorted colonies in the Palearctic region. They often migrate in mixed groups to their common wintering grounds in Africa (Kossenko & Fry 1998) (Figure 1). They mainly feed on various flying insects such as odonates, lepidopterans, hymenopterans (Kossenko & Fry 1998, Gunnarsson & Ekblom 2019).

In India, the Blue-cheeked Bee-eater is a widely known passage migrant to the north-western Indian Subcontinent (Ali & Ripley 1987, Rasmussen & Anderton 2012, Grimmett *et al.* 2016). The breeding of Blue-cheeked Bee-eater was once vaguely recorded by Dharmakumarsinhji (1947) in Gujarat, but Ali (1945) and Marien (1950) stated that the species was not found breeding in Kutch, but just to south in Kathiawar Blue-cheeked

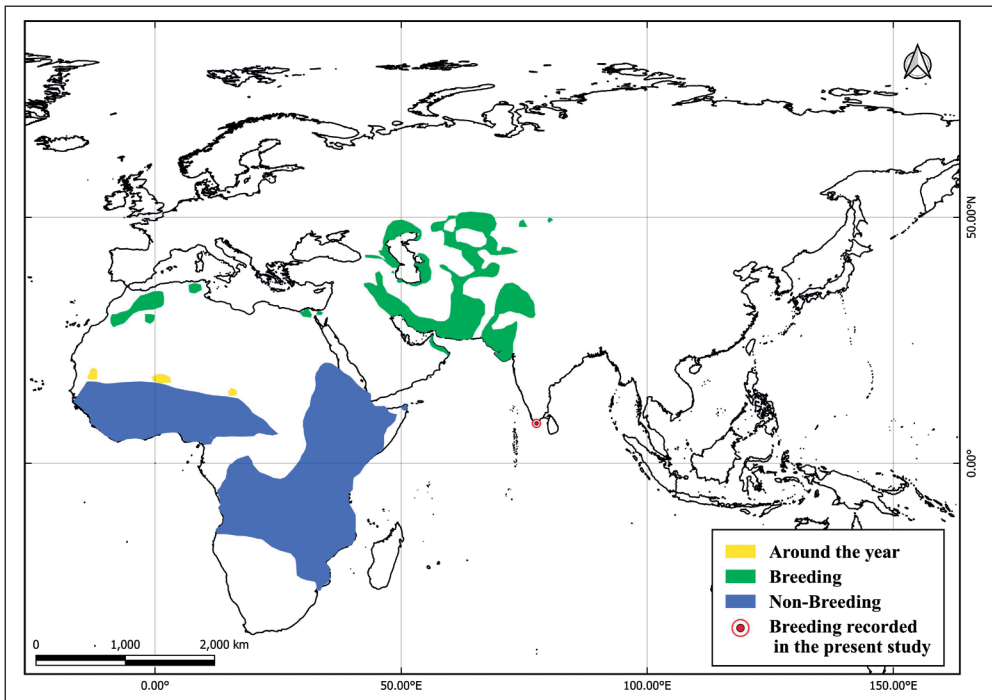


Figure 1. Global distribution map of Blue-cheeked Bee-eater with the breeding site in Kanniyakumari, Tamil Nadu, India (present study) (Source: Fry & Kirwan 2020)

1. ábra A zöld gyurgyalag elterjedési térképe, és az új fészektelep helye az indiai Kanniyakumari járásban, Tamil Nadu tartományban (Forrás: Fry & Kirwan 2020)

Bee-eater reaches its southernmost limit in India. Baker (1934) ascribes all the breeding birds of Punjab and Rajputana to Blue-cheeked Bee-eater, but that is not entirely accurate (Baker 1934, Ali 1945, Marien 1950). After 2007, they have been sparsely recorded along the south-western coast from Goa to Kerala as a passage migrant and monsoon visitor (Holt 2009, Sashikumar *et al.* 2011, Sreenivasan 2013, George 2014, Mannar & Sumesh 2015, Gosavi *et al.* 2019). They have been sighted year around in the Changaram wetland and once observed to attempt breeding, but failed before nesting due to rain and there were no further breeding evidences found in the consecutive years (Mannar & Sumesh 2015).

The breeding biology of the Blue-cheeked Bee-eater are so far studied in mixed colonies with the European Bee-eater and are very scanty in monospecific colonies (Kossenko 1994, Kossenko & Fry 1998). Until now, there are no studies on the breeding of the Blue-cheeked Bee-eater in India, and only sightings during seasonal migration have been reported. Therefore, the breeding report of the species in India is of great significance.

The study was carried out to document the breeding behaviour of the Blue-cheeked Bee-eater in Southern India and to impart the baseline details on the nesting parameters and requirements for the effective conservation of these birds in their acute habitat.

Material and Methods

Study area

The Pazhayar river, one of the prime rivers of the Kanniyakumari district originates from the slopes of Mahendragiri hills and flows southwest and reaches Arabian Sea near Manakudy mangroves. The river serves as one of the main sources of water for the Suchindram Theroor Wetland Complex, a Ramsar site and Important Bird Area (IBA) that lies at the southern tip of the Central Asian flyway of migratory birds (Ramsar 2022). The total length of the river from the origin to its outfall into Arabian Sea is 40 km.

Along the Pazhayar river basin, Periyakulam, Manakudy mangroves, Puthalam and Andivillai were frequently visited and studied from January 2022 to October 2023. These areas are mostly covered by saltpans and a few mangrove patches, and coconut coir factories have been established in their peripheries. The vegetation comprises open thorny scrubland, coconut plantations and mangroves (*Figure 2*).

Data collection

During one of our field surveys, we sighted a flock of Blue-cheeked Bee-eaters. As the birds indulged in courtship and mounting behaviour, we tried to find their nesting site. A keen observation was made over the river beds, loess banks of abandoned saltpans and all possible suitable places around the area following Look and See method (Bibby *et al.* 1992). We found some breeding burrows of Blue-cheeked Bee-eaters along the saltpans of Pazhayar river basin. We decided to observe and document the breeding of Blue-cheeked Bee-eater regularly further using Nest and Roost count method (Javed & Kaul 2002). The breeding

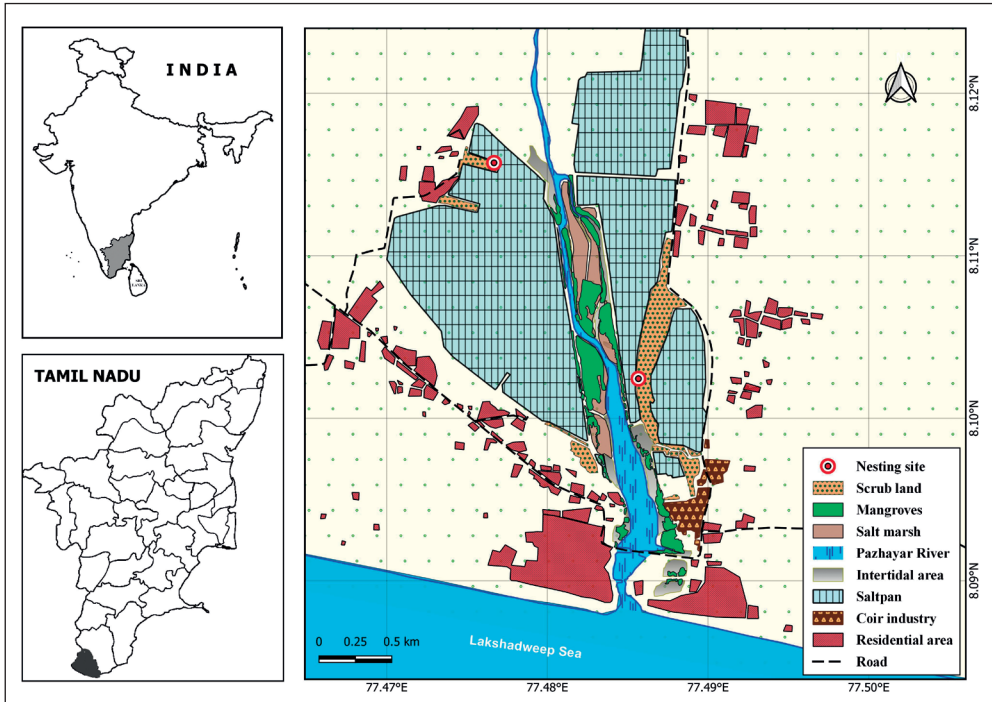


Figure 2. Locations of Blue-cheeked Bee-eater breeding sites in Kanniyakumari, Tamil Nadu, India: Red dots indicate breeding sites

2. ábra A zöld gyurgyalag fészektelepei az indiai Kanniyakumari járásban, Tamil Nadu tartományban. A piros pöttyök jelölik a költőtelepeket

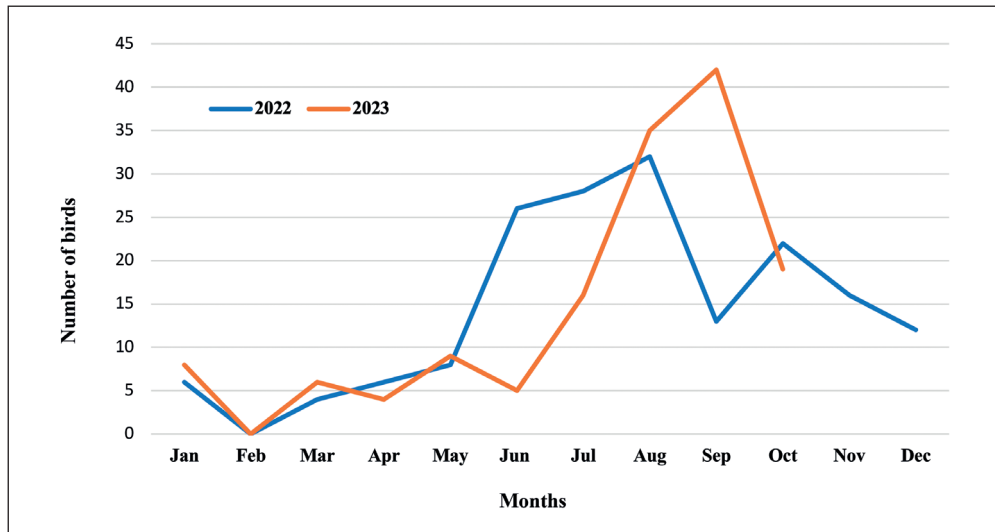


Figure 3. Variation in number of individuals of Blue-cheeked Bee-eaters throughout the year in the study site

3. ábra A zöld gyurgyalag éves állományváltozása a költőtelepen

behaviours were observed with binoculars (Olympus 8x40) and photographic documentation was carried out using a DSLR camera (Canon 200D ii). We used standard field guides and available literature (Ali & Ripley 1983, Grimmett *et al.* 2016). The observations were made on different occasions of the day light hours.

Results

The Blue-cheeked Bee-eaters were seen in the breeding sites throughout the year except February. The population fluctuates every month with a peak during the end of southwest monsoon that indicates the rise in population at the end of breeding season. The breeding season started with 28 birds and at the end we counted 48 individuals, the additional birds being juveniles, which indicates the successful breeding of the colony (*Figure 3*). The other bee-eater species in the breeding area are Blue-tailed Bee-eater (*Merops philippinus*) and Green Bee-eater (*Merops orientalis*). There were no heterospecific breeding colonies around the area. The interspecific competition for food among Blue-tailed Bee-eater and Blue-cheeked Bee-eater was observed several times.

Nesting site complex

We discovered two new nesting colonies of Blue-cheeked Bee-eaters, Puthalam and Andivillai situated on both sides of the Pazhayar river about 2 km far from each other. Puthalam (8.11572°N, 77.47668°E) is situated 350 m west of Pazhayar river, and the area is covered mainly by saltpans surrounded by coconut plantations. Andivillai (8.10243°N, 77.48566°E) is located 105 m east of Pazhayar river, adjacent to the 80 m wide linear saltpans and a small 5 m wide creek between them (*Figure 2*). The saltpans consist of three distinct pans namely; reservoir pans, evaporator pans and crystallizer pans. Each saltpan has a separate reservoir pan to stock the seawater. The top ground of the dug reservoir pans consists of thorny shrubs and few trees. The reservoir pan being devoid of water and the protruding roots of *Prosopis juliflora* and *Azadirachta indica* through the vertical surface facilitated the perch of nesting birds (*Figure 4*).

Puthalam nesting site

During our field visit along the Pazhayar river basin on 26 June 2022 to Puthalam saltpans, we saw about eight individuals catching dragonflies and presenting them to their mates, and four of them were already engaged in digging tunnels. The breeding burrows were excavated along the vertical surface of an abandoned partly dug saltpan pond (pond used for stocking saltwater in saltpans). The nesting site was 50 m and 350 m far from saltpan and Pazhayar river, respectively (*Figure 2*). There were about 11 breeding burrows among which six were utilised for nesting. Later, on 30 June 2022 the heavy rain destroyed the breeding burrows and the birds disappeared for a week and appeared again from 11 July 2022.

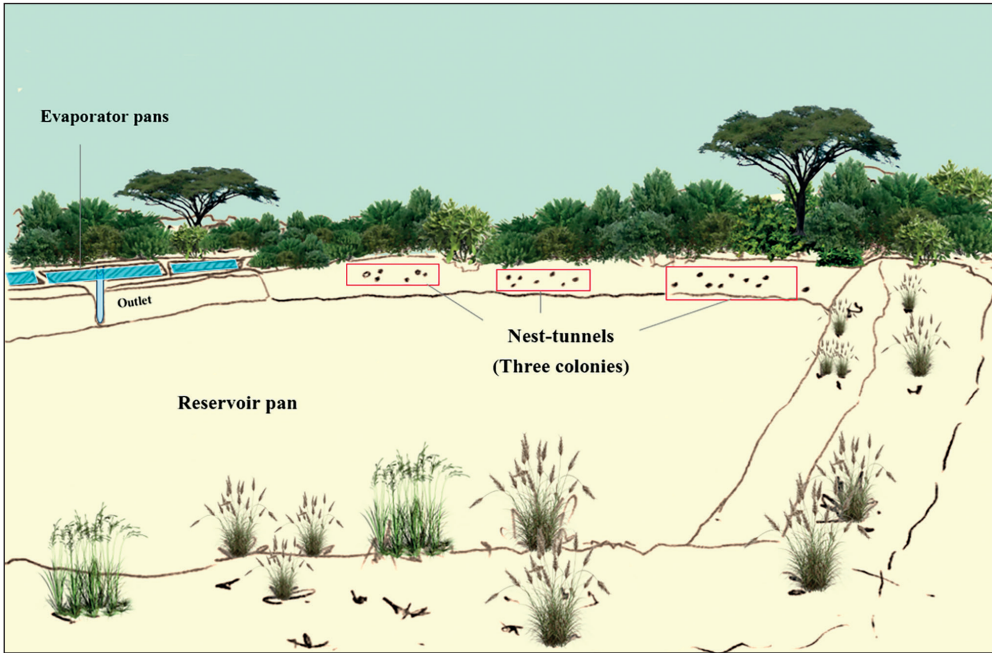


Figure 4. The breeding site of Blue-cheeked Bee-eater in the declivity face of reservoir pan along the saltpans of Andivillai, Kanniyakumari, Tamil Nadu, India

4. ábra A zöld gyurgyalagok fészektelepe a lepárló tó partoldalában az Andivillai sólepárló tavaknál az indiai Kanniyakumari járásban, Tamil Nadu tartományban

Andivillai nesting site

We found another nesting site of Blue-cheeked Bee-eater on 07 August 2023 along the Andivillai saltpans. After the birds vacated the nesting site on 10 October 2023, we examined the breeding burrows. The breeding burrows were excavated along the vertical surface of a newly dug reservoir pan with the dimensions of 40 m length, 35 m breadth and 2.7 m height. In the nesting segment, 28 breeding burrows were counted which spreads over 30 m length (*Figure 4*). The nest segment strip was about 1.65 m below the top-ground level and 0.53 m above the base ground (*Table 1*). The 28 nests altogether formed three loose subsets consisting 16 active nests and 12 inactive nests (*Table 2*). All these tunnels were freshly dug with varying depths and nest-mouth diameter ranging from 0.97 m to 2.78 m and 8 cm to 12.5 cm respectively (*Table 1*). The active nest-tunnel has a mean tunnel length of 1.53 ± 0.53 m and nest-mouth diameter of 9.93 ± 1.48 cm.

Behaviour observed

The breeding sites in Puthalam and Andivillai saltpans were frequently visited during the study. The observations during the field visits are compiled in *Table 3*.

Table 1. Physical characteristics of active nests of Blue-cheeked Bee-eaters in the colony
1. táblázat Az elfoglalt zöld gyurgyalag fészkek fizikai paraméterei

Nest number	Length (m)	Diameter (cm)	Height from ground (m)
S1N01	1.13	12.2	0.53
S1N02	1.54	9.5	0.78
S1N03	1.23	8.5	0.83
S1N04	1.73	9.2	0.93
S2N05	1.33	11.3	0.55
S2N06	2.01	12.0	0.96
S2N07	0.97	11.5	0.77
S2N08	1.07	12.5	0.93
S2N09	2.32	10.8	0.77
S2N10	2.78	11.5	0.99
S2N11	1.23	9.6	0.66
S3N12	1.20	9.2	0.81
S3N13	1.37	8.0	1.06
S3N14	1.16	8.5	1.09
S3N15	2.27	11.2	1.01
S3N16	1.20	9.8	0.90

Note: The nest number denotes the number of the subset (S1, S2, S3) and number of the nest (N1, N2, N3...)

Table 2. Nest composition of Blue-cheeked Bee-eaters in the nest subsets
2. táblázat A fészkek eloszlása a költőtelep csoportjai között

Breeding colonies	Active nest	Inactive nest	Total nest
Subset A	5	3	8
Subset B	7	4	11
Subset C	4		9
Distance between subset A&B	16 m		
Distance between subset B&C	9 m		

Diet

Throughout the period of observation from March 2022 to October 2023, the Blue-cheeked Bee-eaters were observed to feed on various flying insects in and around the Pazhayar river basin. Their diet mainly constitutes dragonflies, butterflies, hawk moths and bees (Figure 5). The insects captured by the Blue-cheeked Bee-eater around the nesting sites are butterfly species: Common Mormon (*Papilio polytes*), Lime Butterfly (*Papilio demoleus*), Common Emigrant (*Catopsilia pomona*), Mottled Emigrant (*Catopsilia pyranthe*), Common Grass Yellow (*Eurema hecabe*), Plain Tiger (*Danaus chrysippus*), Common Crow (*Eupolea core*), Tawny Coster (*Acraea violae*), Painted Lady (*Vanessa cardui*), Rice Swift (*Borbo cinnara*); dragonfly species: Ditch Jewel (*Brachythemis contaminata*), Blue-tailed Green Darner (*Anax*

Table 3. Behavioural observations of Blue-cheeked Bee-eater in the breeding site of Andivillai, Kanniyakumari, Tamil Nadu, India

3. táblázat A zöld gyurgyalag viselkedésének megfigyelése az indiai Kanniyakumari járásban talált kolónián, Tamil Nadu tartományban

Date	Time	Duration (Hours)	No. of Birds seen	Observations
07 August 2023	07:20	4.20	26	Aerial display and close perch by mates, mutual preening on nape and breast, alluring and offering treats to mates and few females started excavating nest.
09 August 2023	07:35	5.00	32	Aerial display and close perch by mates, mutual preening, males offering treats to females, mating (one observation) and females started excavating nest.
12 August 2023	08:00	4.40	30	Aerial display and close perch by mates, mutual preening, males offering food to females, frequent mating (12 observed), almost all females started excavating nest (12 individuals) and few females took nesting materials inside breeding burrows.
21 August 2023	07:10	4.30	34	Close perch by mates, mutual preening on nape, males offering food to females, mating (8 observations), several females completed nesting, few females started excavating nest and males fetching food for brooding females in 4 tunnels.
22 August 2023	07:30	4.20	36	Males were frequently fetching food for the female inside the tunnel, mates took turns in brooding (observed in two nests), still two birds were nesting, no mating observed, males involved in watch and guard duty perching along the roots near the tunnel entrance and a chick peeping from a nest.
24 August 2023	07:40	3.40	35	Males were busy fetching food for the females and hatchlings inside the tunnel, no mating and nesting was observed, males involved in watch and guard duty and few chicks peeping from the nests.
05 September 2023	08:00	5.00	38	Parents rapidly feeding the hatchlings inside the tunnel and few chicks fed near the nest entrance. Blue tailed Bee-eaters snatched the food from Blue-cheeked Bee-eaters while approaching back to the nest-tunnel (two times).
14 September 2023	08:10	3.30	36	Parents were rapidly feeding their chicks at the tunnel entrance, two fully grown chicks were seen perching on the roots adjacent to the tunnel entrance and fed by the parents.
28 September 2023	07:40	4.20	42	One chick perched outside the tunnel and fed by parents and three juvenile birds perching on the dry thorns nearby with adults and five juveniles were flying in open air trying to catch insects.
04 October 2023	07:00	5.30	16	Four juveniles were seen flying with adults and catching insects, only two pairs were feeding their chicks near the nest entrance and many nests were vacated.
05 October 2023	07:40	5.00	8	Heavy downpour on the last night damaged the breeding burrows, one male kept insect near the tunnel entrance and was waiting for the mate to retract the food inside but there was no response from inside, rather it tried again and again with different insects till noon and we left the place.



Figure 5. Blue-cheeked Bee-eater feeding on dragonfly (Lesser Green Emperor *Anax guttatus*)
5. ábra Zöld gyurgyalag zöld óriásacsa zsákmánnyal (*Anax guttatus*)



Figure 6. Mating Blue-cheeked Bee-eaters near the breeding site
6. ábra Párosodó zöld gyurgyalagok a fészektelep közelében

guttatus), Green Marsh Hawk (*Orthetrum sabina*), Coral-tailed Cloudwing (*Tholymis tillarga*), Ruddy Marsh Skimmer (*Crocothemis servilia*), Globe Skimmer (*Pantala flavescens*), Common Picture Wing (*Rhyothemis variegata*), Ground Skimmer (*Diplacodes trivialis*), Greater Crimson Glider (*Urothemis signata*), Yellow-tailed Ashy Skimmer (*Potamarcha congener*), Long-legged Marsh Glider (*Trithemis pallidinervis*), Crimson Marsh Glider (*Trithemis aurora*), Black Marsh Trotter (*Tramea limbata*); moth species: Coffee Bee Hawkmoth (*Cephonodes hylas*); bee species: Blue-banded Bee (*Amegilla zonata*), Honeybee (*Apis* sp.) and dipteran species: Green Colonel Fly (*Odontomyia* sp.), Greenbottle Fly (*Lucilia* sp.).

Courtship and mating

The observed courtship behaviour contained the following elements: flying together in air, aerial display by males, intimate close perching by mates, mutual preening especially on nape, presenting and accepting treats. During copulation the male mounts over the female

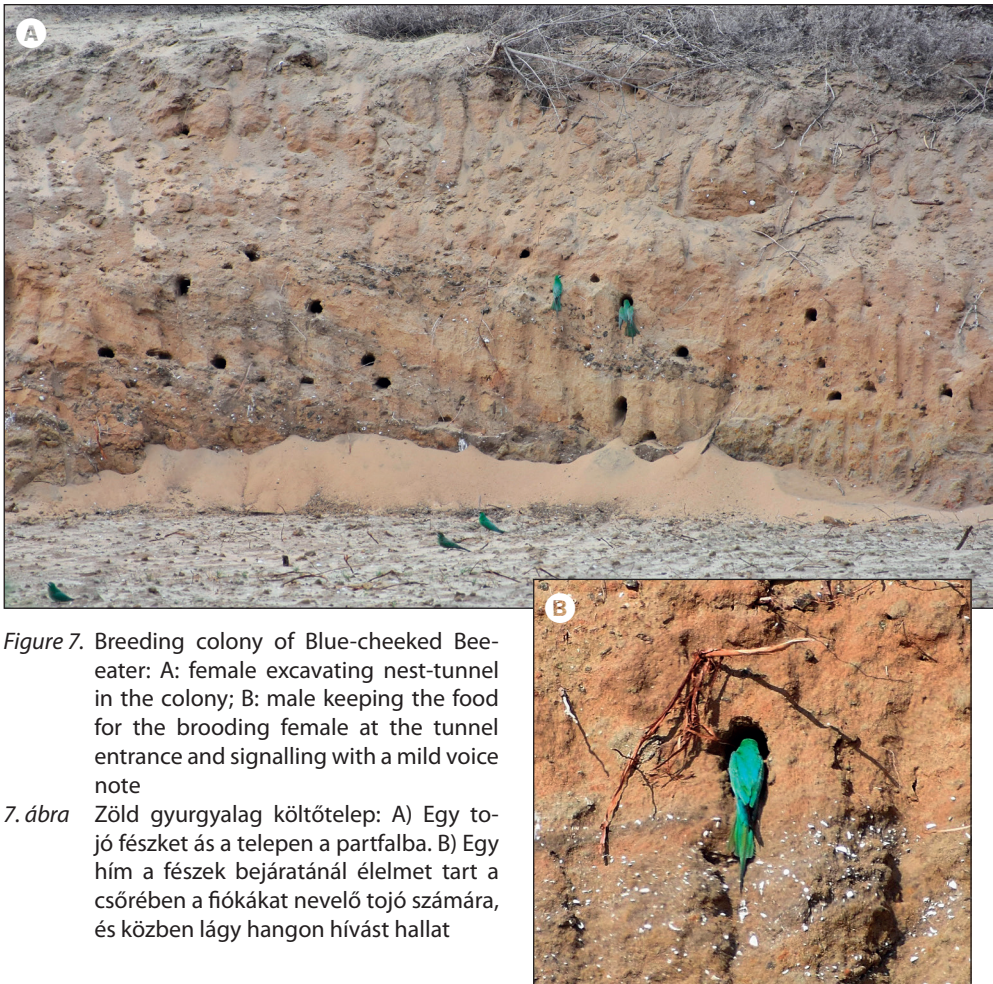


Figure 7. Breeding colony of Blue-cheeked Bee-eater: A: female excavating nest-tunnel in the colony; B: male keeping the food for the brooding female at the tunnel entrance and signalling with a mild voice note

7. ábra Zöld gyurgyalag költőtelep: A) Egy tojó fészket ás a telepen a partfalba. B) Egy hím a fészek bejáratánál élelmet tart a csőrében a fiókákat nevelő tojó számára, és közben lágy hangon hívást hallat

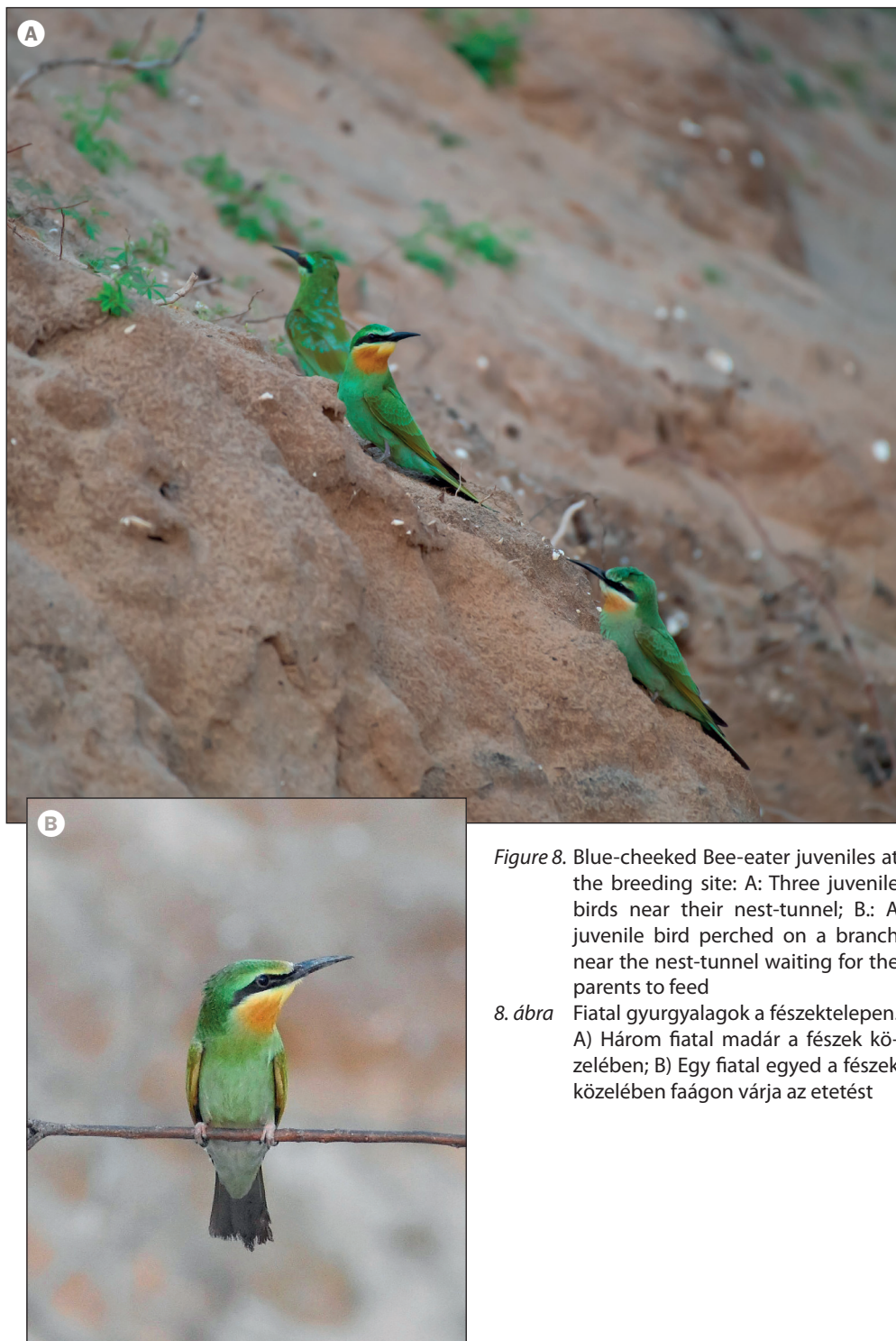


Figure 8. Blue-cheeked Bee-eater juveniles at the breeding site: A: Three juvenile birds near their nest-tunnel; B.: A juvenile bird perched on a branch near the nest-tunnel waiting for the parents to feed

8. ábra Fiatal gyurgyalagok a fészektelepen. A) Három fiatal madár a fészek közelében; B) Egy fiatal egyed a fészek közelében faágon várja az etetést

(Figure 6). After the copulation that lasted for 3–4 seconds, the male flew to the nearby perch and the female stayed in the same posture for a while. This act was repeated consecutively for 4–5 times with an interval of 20–30 minutes. The mating occurred on two substrates, the ground and dried thorny branch in close proximity (<5 m) to the nest.

Breeding burrows and brooding activities

After mating the females started excavating nest tunnels and males rarely assisted them (Figure 7a). The male mostly perched on the exposed *Prosopis juliflora* roots and dry thorns nearby, and performed mate guarding. The male occasionally offered food to the female during nest building. Subsequently, females carried nesting materials like *Prosopis* leaves and grasses inside the nest-tunnel. Males carried the food and perched on the protruding roots from the soil near the opening of the tunnels mouth and made mild call. As a brooding female from the posterior end approached the entrance of the tunnel, the male left the insect at the tunnel entrance and flew to the nearby perch. While brooding, the female accepts the insects left by the male near the entrance of the tunnel and rarely flew out of the tunnel for food (Figure 7b). The male took turns in brooding whenever the female flew out of the tunnel.

Hatchlings and juveniles

The male birds were very busy, feeding the brooding females and chicks inside the tunnel. The female fed the chicks after retracting the food inside the tunnel as it did during brooding. This is done every time when a male carried the food to the tunnel. The chicks were first observed peeping out of the tunnels on 22 August 2023. The females were seen more often outside the tunnel as the chicks grew up. Then both the males and females took turns in provisioning the chicks at the entrance of the tunnel (Figure 8a). The sky was filled with ‘tick’ sound of the busy parents gliding around to catch insects. The ‘tick’ sound was produced whenever a bee-eater whacked a prey in mid-air. Later, on 14 September 2023 two juvenile birds were seen outside the nest-tunnel and often perched on the dry thorny branches while the parents fed them (Figure 8). The juvenile birds can be easily identified by their black eyes unlike adults that have red eyes (Figure 8b). Four juvenile birds were seen flying with their parents in open sky above the saltpans on 4 October 2023.

Discussion

The mating and presence of freshly dug active breeding burrows, broken egg shells, and parents feeding the chicks at the nest entrance, confirmed the successful breeding of the species. The breeding population of Blue-cheeked Bee-eater along the Pazhayar river may be resident as they are sighted throughout the year with some fluctuations in the population during different months.

There are several studies on the ecology and breeding biology of the Blue-cheeked Bee-eater from the mixed colonies with European Bee-eater in the Palearctic regions (Kossenko

1994, Kossenko & Fry 1998). Usually, egg laying starts in the first half of May in north-west Africa and Nile Delta, in March in Mesopotamia, from April to June in Iran and in the second half of June in south Pakistan and Oman (Fry 1984). The breeding season of Blue-cheeked Bee-eater in the present study interestingly overlaps with the breeding season of its conspecifics from West Africa in North Senegal from July to September, in Eastern Sahel and Southern Mauretania from July to October (Guichard 1947, Fry 1981, Morel & Roux 1996). Though Blue-cheeked Bee-eaters are summer nesting species in the Palaearctic realm in India the breeding has been observed during the southwest monsoon.

So far, the Blue-cheeked Bee-eater was considered a passage migrant and winter visitor to North-western India, but our study confirmed its breeding in Southern India. The present record of the Blue-cheeked Bee-eater is the first breeding record of the species in Peninsular India and southernmost limit of its occurrence in the Indian Subcontinent. Furthermore, this study significantly contributes a baseline information on the breeding ecology of Blue-cheeked Bee-eater in India.

Conclusion

The Kanniyakumari coast and Pazhayar river basin are increasingly under pressure from developmental activities. The present breeding site of the Blue-cheeked Bee-eater along the Pazhayar river basin is threatened by habitat destruction due to anthropogenic pressure. In addition to anthropogenic pressure, natural events like floods, high tides, erosion and unprecedented rainfall have adverse effects on these birds and the riverine ecosystem. Privately owned lands around these ecologically sensitive areas are prone to rapid unpredictable changes and exploitation.

Though the newly discovered breeding site of Blue-cheeked Bee-eater lies within the Coastal Regulation Zone (CRZ) and No Development Zone (CRZ-III NDZ), the area is highly subjected to developmental activities. Being in close proximity to the Ramsar site and IBA the area requires special protection status for the conservation of wetland birds. Therefore, we have to protect the breeding site and ensure the feeding and nesting requirements are not compromised as this serves as the only hub of breeding population in the country.

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Autumn diet of Little Owls (*Athene noctua*) in southern Tunisia

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Abstract The Little Owl (*Athene noctua*) is a widespread nocturnal raptor whose diet has extensively been studied. In the Mediterranean, a tendency towards increased ingestion of insects has been highlighted, but this tendency still needs more empirical validation. Here, we provide data on the autumn diet of Little Owls in an arid area in southern Tunisia. We analyzed 621 pellets from 30 different territories and identified 4,267 prey items, with insects being by far the most ingested prey. However, based on biomass, the diet was more evenly distributed across the identified prey categories. Vertebrates accounted for 63% of the biomass ingested, while the remaining part of prey composition was almost equally shared between insects and arachnids. The importance of arachnids looks like a characteristic dietary trait of Little Owls in south Mediterranean deserts, where arachnids represent a main component of the community of potential prey. In summary, our findings highlight the great food opportunism of the Little Owl due to its capability of adapting its diet to the availability of potential prey.

Keywords: arid areas, Mediterranean, nocturnal raptors, pellet analysis, trophic ecology

Összefoglalás A kuvik (*Athene noctua*) széles körben elterjedt éjszakai ragadozómadár, amely táplálkozása alaposan tanulmányozott. A Mediterráneumban a faj megnövekedett rovarfogyasztását emelték ki, azonban ez további megerősítést igényel. Jelen tanulmányban a kuvik Dél-Tunézia száraz régiójában jellemző őszi étrendjéről közlünk adatokat. A 30 különböző területről származó, összesen 621 köpet elemzésével 4267 zsákmányt azonosítottunk, amelyek közül messze a rovarok jelentették a legtöbbet fogyasztott zsákmányt. A biomassza alapján azonban a zsákmánykategóriák eloszlása egyenletesebb volt az étrendben. Az elfogyasztott biomassza 63%-át a gerincesek tették ki, míg a zsákmány-összetétel fennmaradó része közel egyenlő arányban oszlott meg a rovarok és a pókfélék között. A pókfélék fontossága egy jellemző táplálkozási jelleg a dél-mediterrán sivatagokban előforduló kuvikok esetén, ahol a pókfélék a potenciális zsákmány közösség fő alkotóelemei. Összefoglalva, eredményeink rávilágítanak a kuvik nagy táplálkozási opportunizmusára, mivel képes étrendjét a potenciális zsákmány elérhetőségéhez igazítani.

Kulcsszavak: sivatagos terület, Mediterráneum, éjszakai ragadozómadarak, köpetelemzés, trofikus ökológia

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Introduction

As top predators, owls play important roles in ecosystem functioning and provide valuable ecological services worldwide (Donazar *et al.* 2016). For conservation purposes, profound knowledge on their trophic ecology is needed. In particular, it is essential to investigate how diet varies among local populations depending on local conditions and prey availability, as this may help assess how owls adapt to environmental changes.

The Little Owl (*Athene noctua*) is a widespread owl species that occurs in different habitats across the Palearctic, preying on a diverse pool of small vertebrates and invertebrates (del Hoyo *et al.* 1992, Nieuwenhuyse *et al.* 2008). Although its diet has been the subject of many investigations in different regions of its geographical range in Europe (e.g. Capizzi & Luiselli 1995, Fattorini *et al.* 1999, Goutner & Alivizatos 2003, Romanowski *et al.* 2013, Hámori *et al.* 2017, 2019), Asia (e.g. Shao & Liu 2008, Zhao 2011), the Middle East (e.g. Al-Melhim *et al.* 1997, Obuch & Krištín 2004, Charter *et al.* 2006) and North Africa (e.g. Alaya & Noura 2007, Chenchouni 2014), this subject still attracts the interest of ornithologists who continue to provide new data (e.g. Kolendrianou *et al.* 2022, Fallahi Akhlamad *et al.* 2023). These studies contribute greatly to the identification and understanding of the spatial patterns of the trophic ecology in this species. In this context, a north-south dietary gradient has been highlighted, with a decrease in small mammals and an increase in insects from continental Europe towards the Mediterranean (del Hoyo *et al.* 1992, Nieuwenhuyse *et al.* 2008). Although an insect-based diet has been accepted as a typically Mediterranean dietary trait, strong empirical evidence from southern Mediterranean populations is still lacking. Indeed, there is a huge asymmetry between the northern and southern borders of the Mediterranean concerning the knowledge on Little Owl diet because of incomparable research efforts dedicated to this subject.

In this paper, we provide detailed data on the diet of Little Owls inhabiting an arid area along the Gulf of Gabès, in southeast Tunisia. Our main goals were to describe the diversity of consumed prey and to assess whether southern Tunisian Little Owls had a typical Mediterranean diet.

Materials and Methods

Data collection

The study area is an arid desert plain close to Gabès city (33°53'N-10°05'E), in southeast Tunisia. The landscape corresponds to a mixture of sparsely-vegetated steppes, farmlands, including oases and dry fruit tree groves, and urban areas. In this area, the Little Owl is a common resident breeder that often uses piles of stone abandoned by people as roosting and nesting sites (Saada *et al.* 2024).

In October and November 2022, we collected Little Owl pellets in thirty territories that we previously located (Figure 1). Pellets were searched for around piles of stones assembled by people and used by Little Owls as daytime roosting and nesting sites. Collected pellets were

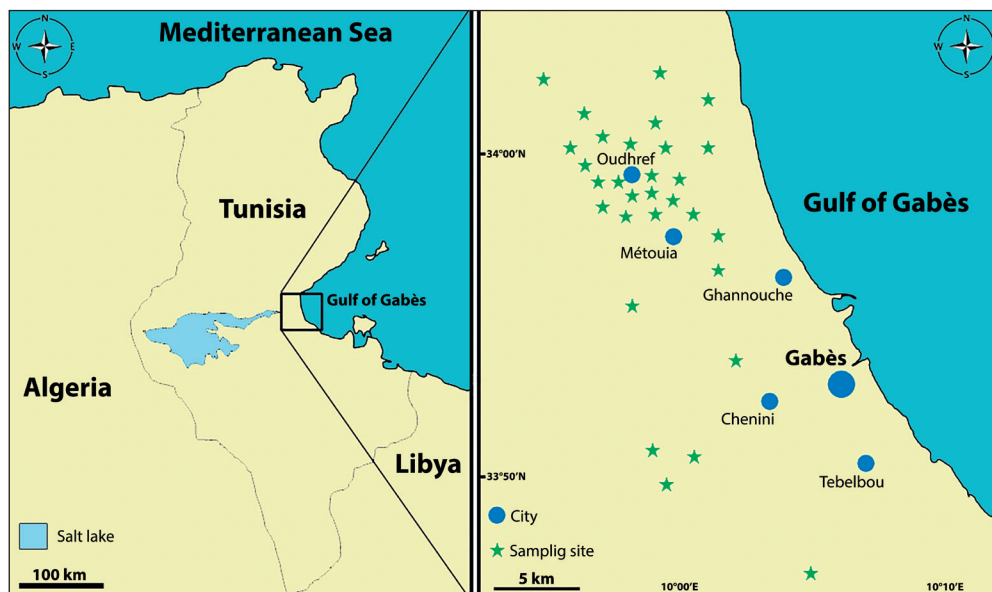


Figure 1. Map of the study area showing the locations of sampled Little Owl territories
 1. ábra A vizsgált terület térképe, amelyen látható a mintavételi helyek elhelyezkedése

brought to the laboratory where they were soaked separately in water for at least 30 minutes. Prey remains, including bones, hair and feathers from preyed vertebrates, and chitinous pieces (jaws, mandibles, heads and chelicerae) from invertebrates, were separated and dried on filter paper. Subsequently, the remains were carefully observed using a stereo-microscope for prey identification and count following Romanowski *et al.* (2013). We tried to identify the prey to the species level by referring to the guides and reference works available in the Zoology laboratory at the Faculty of Biology and Environmental Sciences, Cardinal Stefan Wyszyński University, Warsaw, Poland. For each pellet, the pieces belonging to each category of prey were collected together and counted so that the minimum number of individuals was retained (Rey-Rodríguez *et al.* 2019).

Abundance data were converted into biomass using body weight data available in the literature, in particular Tobias *et al.* (2022) for birds, Amor *et al.* (2010) for amphibians, Murariu and Chișamera (2007) for small mammals, Jelaska *et al.* (2011), Heatwole and Muir (1989) and Byers (2015) for insects, and Zaafour *et al.* (2014) for gastropoda. With regard to scorpions and solifuges, body weight data were not available in the literature, so we calculated the mean weight of individuals that we caught on the field. In each group, the average body weight of the most commonly recorded species was used for converting prey numbers into biomass. In summary, the following conversion coefficients were used: scorpions and solifuges (3 g), Coleoptera (0.28 g), ants (0.007 g), wasps (1 g), Orthoptera (0.77 g), Dermaptera (0.27 g), Gastropoda (3 g), amphibians (30 g), birds (30 g), small mammals (18 g). We emphasize here that these conversion coefficients were determined for use in the particular case of our study system, and we recognize that they may lack precision.

Data analyses

For a given prey class i , we calculated the frequency of occurrence (1) in the pellets (i.e., the number of pellets where that prey class was recorded by the total number of pellets sampled), and (2) in the studied territories (i.e. the number of territories where that prey class was recorded in the diet by the total number of territories sampled). We also determined its contribution to the Little Owl's diet, by calculating its frequency (p_i) relative to the total prey ingested. Calculations were made in terms of abundance and biomass, and the obtained frequencies were subsequently used to assess the diet breadth, using the Levin's index: $B = 1/Sp_i^2$ (Magurran 2004). Values were standardized on a scale of 0 (specialist diet) to 1 (generalist diet), by calculating evenness: $E = (B-1)/(N-1)$, with N being the number of prey classes (Magurran 2004).

Furthermore, we drew abundance-frequency and biomass-frequency curves by ranking the prey classes from the most to the least important based on either abundance or biomass along the x -axis, and their cumulative frequencies on the y -axis (Magurran 2004). The differences between the two curves were then assessed by means of a Kolmogorov–Smirnov two-sample test, using the `npair1way` procedure in SAS (SAS Institute 2008). Finally, the prey classes were ranked according to their importance in the diet by using a modified formula of the Index of Relative Importance (Martin *et al.* 1996): $IRI = (N+W) \times F$, with N , W and F are the frequencies (expressed in %) of abundance, biomass and occurrence (relative to total pellets sampled), respectively.

Results

From the 621 pellets analyzed, we recorded 4,267 prey items belonging to 10 main categories (Table 1). All prey categories were commonly ingested, as each of them occurred in the diet of at least 40% of the territories sampled (Table 2). The smallest prey identified were ants, weighing approximately 7 mg, while the largest prey was the Sand Rat (*Psmammomys obseus*) with a weight exceeding 100 g. Based on abundance data, coleopterans were the most often caught prey, as they accounted for 69% of identified prey items (Table 2), leading to a low value of the Levin's index evenness ($E = 0.112$).

Among coleopterans, the specimens belonging to the Tenebrionidae family were the most commonly consumed, occurring in 91% (563/621) of pellets and accounting for 23% (685/2957) of the coleopterans ingested (16% of identified prey; 685/4267).

However, using biomass data, the diet seemed more evenly distributed across the identified prey categories (Table 2; $E = 0.521$). Actually, vertebrates, whose contribution did not exceed 5% of recorded prey, accounted for 63% of the biomass ingested, while the remaining 37% of prey biomass was almost equally shared between insects and arachnids (Table 2). This distribution is also visible when looking to the abundance and biomass curves (Figure 2). Indeed, there was a tendency of the biomass curve to be below and steeper than the abundance curve, but this difference was still non-significant (Kolmogorov–Smirnov two-sample test: $D = 0.400$; $P = 0.401$). Finally, when data on occurrence, abundance and biomass were

Table 1. List of prey taxa recorded in the Little Owl diet and their frequency of occurrence in pellets

1. táblázat A kuvik táplálék-összetételében azonosított zsákmányok listája és előfordulásuk gyakorisága a köpetekben

Class	Order	Family	Species	Pellets (%)
Arachnida	Scorpiones	Scorpionidae	<i>Scorpio maurus</i>	12 (2%)
		Buthidae	<i>Buthus tunetanus</i>	112 (18%)
			<i>Androctonus australis</i>	20 (3%)
	Solifugae	Galeodidae	Undetermined	124 (20%)
Insecta	Coleoptera	Tenebrionidae	<i>Pimelia</i> sp.	413 (67%)
			<i>Blaps</i> sp.	96 (15%)
			<i>Akis</i> sp.	49 (8%)
			<i>Sepedium</i> sp.	4 (1%)
			Undetermined	138 (22%)
		Scarabaeidae	<i>Scarabaeus sacer</i>	7 (1%)
			Undetermined	78 (13%)
		Carabidae	Undetermined	32 (5%)
		Curculionidae	Undetermined	57 (9%)
		Buprestidae	Undetermined	2 (<1%)
		Undetermined	Undetermined	103 (17%)
	Hymenoptera	Formicidae	<i>Messor arenarius</i>	28 (5%)
		Vespidae	Undetermined	19 (3%)
	Orthoptera	Acrididae	<i>Heteracris annulosa</i>	5 (1%)
			Undetermined	236 (38%)
	Dermaptera	Labiduridae	<i>Labidura riparia</i>	4 (1%)
		Forficulidae	Undetermined	55 (9%)
Gastropoda	Stylommatophora	Helicidae	Undetermined	17 (3%)
Amphibia	Anura	Ranidae	<i>Pelophylax aharicus</i>	14 (2%)
		Undetermined	Undetermined	12 (2%)
Aves	Passeriformes	Alaudidae	<i>Galerida cristata</i>	1 (<1%)
		Sylviidae	<i>Curruca</i> sp.	2 (<1%)
		Passeridae	<i>Passer domesticus</i>	1 (<1%)
		Undetermined	Undetermined	27 (4%)
Mammalia	Insectivora	Soricidae	<i>Crocidura</i> sp.	7 (1%)
	Rodentia	Muridae	<i>Mus musculus</i>	30 (5%)
			<i>Mus spretus</i>	35 (6%)
			<i>Apodemus sylvaticus</i>	2 (<1%)
			<i>Meriones shawi</i>	9 (1%)
			<i>Psammomys obesus</i>	3 (<1%)
			<i>Gerbillus simoni</i>	3 (<1%)
			Undetermined	41 (7%)

Table 2. The occurrence frequency and relative proportion, the abundance and biomass frequency and their relative proportion, as well as the Index of Relative Importance (IRI) of the prey categories identified in the food of the Little Owl

2. táblázat A kúvik táplálékában azonosított zsákmánykategóriák előfordulásának frekvenciája és gyakorisága, abundancia és biomassza frekvenciájuk és ezek relatív aránya, valamint a relatív fontossági index (IRI)

Prey category	Occurrence/territories	Occurrence/pellets	Abundance	Biomass	IRI	
	(%)	(%)	(%)	(%)	Value	Rank
Coleoptera	30 (100%)	599 (96%)	2957 (69%)	827.96 (12%)	7861.07	1
Mammalia	24 (80%)	125 (20%)	139 (3%)	2502.00 (37%)	814.25	2
Orthoptera	29 (97%)	240 (39%)	452 (11%)	348.04 (5%)	608.72	3
Scorpiones	26 (87%)	135 (22%)	223 (5%)	669.00 (10%)	329.66	4
Solifugae	25 (83%)	119 (19%)	182 (4%)	546.00 (8%)	237.16	5
Aves	14 (47%)	30 (5%)	31 (1%)	930.00 (14%)	70.30	6
Amphibia	12 (40%)	24 (4%)	26 (1%)	780.00 (12%)	47.17	7
Dermaptera	17 (57%)	53 (9%)	152 (4%)	41.65 (1%)	35.64	8
Hymenoptera	18 (60%)	45 (7%)	88 (2%)	19.46 (<1%)	17.02	9
Gastropoda	12 (40%)	17 (3%)	17 (<1%)	51.00 (1%)	3.16	10
Total	-	-	4267 (100%)	6715.10 (100%)	-	-

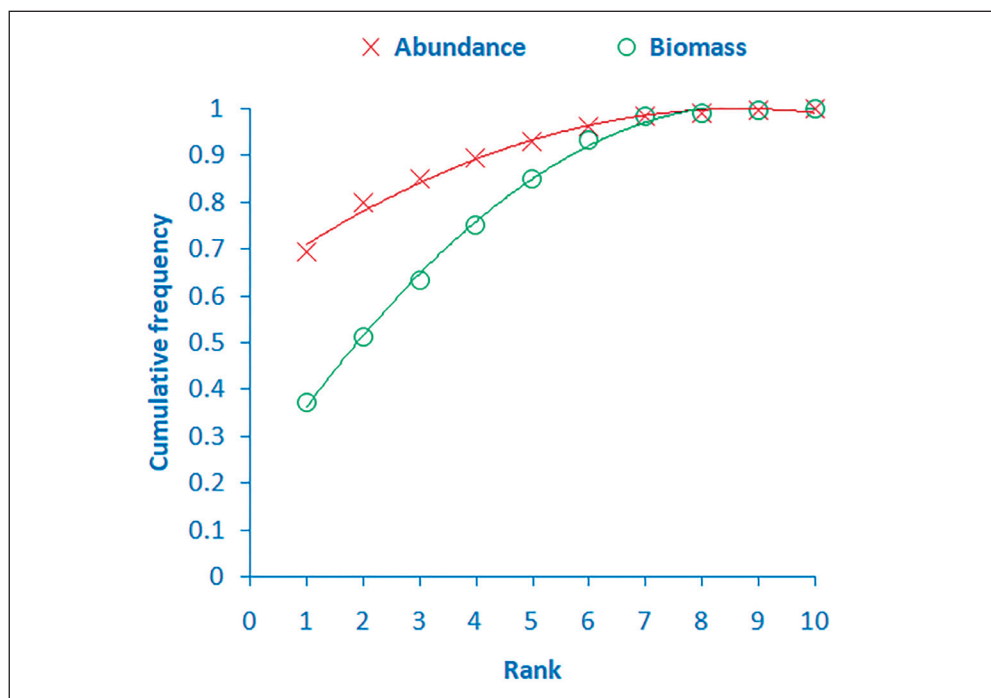


Figure 2. Abundance and biomass curves for prey taxa recorded in Little Owl pellets

2. ábra A kúvik köpetekből kimutatott zsákmány taxonok abundancia és biomassza görbéje

combined together, through the calculation of the IRI, there was a strong evidence of a coleopteran-based diet in our studied Little Owl population. Indeed, coleopterans were ranked first, far ahead of mammals, orthopterans, scorpions and solifugae (*Table 2*). The other classes were clearly much less important (*Table 2*).

Discussion

Using pellet data, we aimed to describe the diversity of prey consumed by Little Owls inhabiting an arid southern Tunisian area. We identified ten common prey categories that contributed differently to the Little Owl's diet. We also noticed that the captured prey covered a wide range of sizes, from ants (around 7 mg) to sand rats whose weight exceeds 100 g. We admit here that we were actually surprised by the observation of sand rat skulls in some pellets. This rodent is normally large for a small owl species whose mammalian prey are often shrews and other small rodents (e.g. Capizzi & Luiselli 1995, Goutner & Alivizatos 2003, Kolendrianou *et al.* 2022). However, the consumption of rodents exceeding 100 g, such as the Black Rat (*Rattus rattus*) has also been reported by some authors (e.g. Charter *et al.* 2006). It is also remarkable that amphibians were part of the diet of Little Owls in our desert area. In fact, frogs were only eaten in some territories near wadis (river valleys) and waterholes where these prey were available.

When prey number and biomass were taken into account together, we found that small prey (i.e., insects and arachnids) were caught in higher numbers, providing significant energy intake, compared to the largest prey (i.e. vertebrates). This suggests that Little Owls have directed their hunting efforts to small prey that are likely more easily catchable than the largest prey available, in accordance with the optimal foraging theory (Stephens & Krebs 1986). Among insects, coleopterans had the most important contribution to the diet of the Little Owls studied, as they were detected in all the pellets sampled and alone accounted for 69% of the items identified and 12% of the total biomass ingested. Although five families of coleopterans were identified, the specimens belonging to Tenebrionidae were the most commonly consumed. This finding is in accordance with those reported in other Mediterranean populations. For example, Fattorini *et al.* (1999) highlighted beetles, including Tenebrionids, as major prey for Little Owls in the Rome urban area. Angelici *et al.* (1997) also noted that the most ingested beetles by Little Owls on the island of Astipalaia (Greece), during summer, were Tenebrionidae (8.80% of the total number of prey eaten). Orthopterans also provided significant diet items, consistent with what was observed in other Mediterranean populations. In particular, the study of Obuch and Kristín (2004) showed that grasshoppers and crickets took a significant place (5.7–16.4%) in the diet of Little Owls in Egypt, Syria and Iran. Orthopterans have also been reported as the main prey of Little Owls inhabiting wetlands in northeastern Greece (Goutner & Alivizatos 2003).

Our results also revealed arachnids as important prey of Little Owls in our arid study area. Indeed, scorpions and solifuges represented together almost 10% of identified prey and accounted for 18% of the biomass ingested. This importance of arachnids actually seems to be a characteristic dietary trait of Little Owls in south Mediterranean deserts,

where arachnids may indeed represent a main component of the community of nocturnal invertebrates. For example, Obuch and Kristín (2004) reported that solifuges and scorpions represented significant components (up to 11%) of the diet of Little Owls in some arid areas in Egypt, Syria and Iran. Overall, these findings suggest that the Little Owl may provide an important public health service to humans living in the desert of south Mediterranean areas, where scorpionism is a major health concern (Chippaux & Goyffon 2008), especially as this nocturnal raptor has synanthropic habits (Saada *et al.* 2024).

In conclusion, our study shows the Mediterranean character of the diet of Little Owls inhabiting the desert plain along the Gulf of Gabès in southern Tunisia. We also note a remarkable enrichment of arachnids, most likely associated with their high availability in this desert area. Overall, these findings stress that the Little Owl has a great capacity of adapting its diet to the availability of potential prey, highlighting the great food opportunism of this species.

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Communal nesting ecology of Little Tern (*Sternula albifrons*) and Small Pratincole (*Glareola lactea*) in two seasonally emerged riverine islands in Bangladesh

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Abstract The nesting ecology of two wetland-dependent bird species, the Little Tern (*Sternula albifrons*) and the Small Pratincole (*Glareola lactea*) were studied in two seasonally emerged riverine islands, locally named as charlands in Padma River of Bangladesh. A total of 269 and 299 nests of Little Tern and Small Pratincole were found in the breeding season of 2022. We monitored 70 nests of the two species from two charlands, and the nesting success was 74.28% and 78.57%, respectively. No correlation was found between nesting success of both species in relation to clutch size, nest height, nest depth and presence of vegetation near nest. We found strong positive correlation between nesting success and nest distance to river for Little Tern. Nesting success was higher if the Little Tern's nests were located far from the river channel. However, no such trends were observed for Small Pratincole in the study area. The main factor affecting Little Tern nesting success was nest predation by avian predator (n = 6) and tropical storms (n = 3). On the contrary, eight Small Pratincole nests were predated by avian predators and four nests were lost due to storm. However, in case of 12 deserted nests (with unhatched eggs) of both species, no apparent reasons were identified. The study suggests that these charlands, if protected from human interferences, can provide great breeding support for colonial ground nesting waterbirds.

Keywords: waterbirds, shorebirds, nesting ecology, colonial breeding, wetlands

Összefoglalás Két, vizes élőhelyekhez kötődő madárfaj, a kis csér (*Sternula albifrons*) és a kis székicsér (*Glareola lactea*) fészkelési ökológiáját vizsgáltuk a bangladesi Padma folyón szezonálisan kialakuló folyóparti szigeteken (Majherchar és Bidirpur), amelyeket helyileg charland-nak neveznek. A 2022-es költési szezonban a kis csérnek 269, míg a kis székicsérnek 299 fészket találtunk meg, amelyek közül mindkét faj esetében hetvenet vizsgáltunk meg részletesen. A költési siker előbbi faj esetében 74.28%, utóbbi esetében 78.57% volt. Nem találtunk összefüggést a két faj költési sikere és a fészkek mérete, a fészkek magassága, a fészkek mélysége és a fészkek közelében lévő növényzet jelenléte között. A kis csér esetében erős pozitív korrelációt találtunk a költési siker és a fészkek folyótól való távolsága között: minél távolabb volt a fészkek a folyótól, annál nagyobb volt a költési siker. A kis székicsérre ez nem volt jellemző. A kis csér költési sikerét leginkább a más madárfaj általi predáció (n = 6) és a trópusi viharok (n = 3) befolyásolták. A kis székicsér esetében nyolc fészkek madarak, négy fészkek pedig viharok miatt pusztult el. 12, bezáptult tojásokkal rendelkező fészekenél nem találtunk meg a költés sikertelenségének okát. Eredményeink arra mutatnak rá, hogy ezek a szigetek, ha védve vannak az emberi beavatkozástól, kiváló fészkelőhelyet nyújthatnak a telepesen talajon fészkelő vízimadarak számára.

Kulcsszavak: vízimadarak, partimadarak, fészkelési ökológia, telepes fészkelés, vizes élőhelyek

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Introduction

Waterbirds are mostly habitat specific and respond quickly to environmental changes. Such changes can influence behaviour and life history traits of their populations and communities (Ma *et al.* 2010, Tavares *et al.* 2015). Waterbird populations have been experiencing declining trends all over the world, due to various anthropogenic threats like habitat loss and alteration, disturbance, hunting, climate change, along with reproductive failure and invasion of domestic and feral animals (BirdLife International 2017, Ramachandran *et al.* 2017, Keller *et al.* 2020, Datta 2022, Marshall *et al.* 2022). Due to their specialized breeding ecology and habitat requirements, ground nesting waterbirds, such as plovers, lapwings, terns, thickknees, and pratincoles are highly vulnerable to animal predation, habitat loss, hydrologic fluctuations, and disturbance from humans and domestic animals (Sanders & Maloney 2002, Chace & Walsh 2006). Understanding local ecological and environmental changes of waterbird habitats can fill the knowledge gaps regarding understudied species and provide crucial information for their conservation.

Two globally widespread, least concern waterbird species, the Little Tern (*Sternula albifrons*) and Small Pratincole (*Glareola lacteal*) inhabit inland wetlands, shallow coastal, marine intertidal and artificial (marine and freshwater) systems (del Hoyo *et al.* 1996, BirdLife International 2016, 2019). The Little Tern has a wide distribution range across the East Asian-Australasian Flyway (EAAF) and the East Atlantic Flyway (EAF) (Jang *et al.* 2015). The species breeds throughout most of Europe, as well as in some locations on the coast and inland in parts of Africa, western, central and eastern Asia, and also in northern Australasia (Oro *et al.* 2004, BirdLife International 2019). It has a declining population trend (BirdLife International 2019). Its habitat specific colonial breeding aspects and nesting ecology is well studied in Europe, where the terns mostly use coastal mudflats, salt marsh, and artificial habitats as breeding sites (Goutner 1990, Catry *et al.* 2004, Medeiros *et al.* 2007, Scarton 2008, Eason *et al.* 2012, Medeiros *et al.* 2012, Ramos *et al.* 2013, Pakanen *et al.* 2014). However, we still do not fully understand the nesting site strategies of Little Tern in seasonally emerged sandy islands and influence of associated local ecological and environmental factors in Bangladesh.

The Small Pratincole occurs in Afghanistan and Pakistan, through India to south China, south and south-east to Southeast Asia, where it is uncommon to locally common resident in Myanmar, Thailand, Cambodia and Lao PDR (Maclean & Kirwan 2020). The species prefers inland wetlands such as rivers, streams, creeks, lakes, estuaries, and marine intertidal habitats such as salt marshes with emergent grasses (del Hoyo *et al.* 1996, Grimmett *et al.* 2016). Current population trend is unknown according to Wetlands International (BirdLife International 2016). Small Pratincoles usually breed on gravel or sandy banks near rivers and lakes. However, detailed breeding ecology of the Small Pratincole such as how they share their breeding sites with other ground nesting species in a seasonally emerged island is under-studied.

In this study, our objectives were to investigate nesting ecology of the two species in two seasonally accreted riverine islands and to examine the influences of other ecological and environmental factors on their nesting success.

Materials and Methods

Study area

Fieldwork was carried out during the breeding season of 2022 (March–June) in several riverine islands (locally called “chars”) of Padma River, the distributary of Ganges under Rajshahi Division in Bangladesh. Although we found several chars as breeding ground for our targeted species, we concentrated on two riverine islands namely Majherchar and Bidirpur char, due to huge crowd of breeding pairs and nests, respectively. Padma riverine islands are nationally and globally important habitat for migratory waterfowl and waders including Painted Stork (*Mycteria leucocephala*), Black Stork (*Ciconia nigra*), Asian Openbill (*Anastomus oscitans*), Asian Woollyneck Stork (*Ciconia episcopus*) and Indian Skimmer (*Rynchops albicollis*) (Thompson 2021).

The Majherchar breeding colony had a comparatively large mass of land and covering an area of 135 ha (Figure 1). This is a seasonally emerged char where vegetation is dominated mostly by grass species. The site is totally devoid of human interferences and the nearest human habitation is situated about 0.45 km apart in Bangladeshi part and 5 km apart from India. No cattle grazing or agricultural activities were observed in this char, however, fishermen sometimes anchored their boats nearby and use the area for resting and pastime activities.

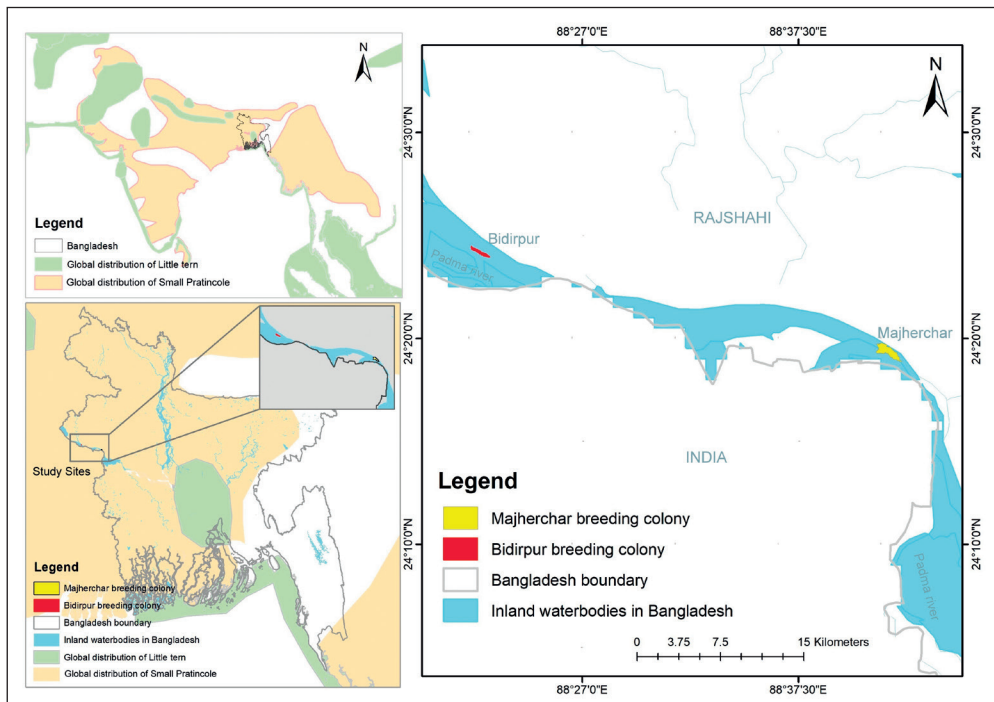


Figure 1. Breeding sites of Little Tern and Small Pratincole in Rajshahi, Bangladesh

1. ábra A kis csér és a kis székicsér fészkelési helyei a vizsgálati területeken Bangladesben

About 33 km upstream is located Char Bidirpur, which is comparatively a small char of about 55 ha area (*Figure 1*). This site is also free from anthropogenic activities and nearest human habitation is situated about 1.1 km apart in Bangladeshi part and 4.65 km apart from Indian part. No significant plant species were observed other than some scattered grassy vegetation. The study sites were sandy and barren, with sporadic patches of Madras carpet (*Grangea maderaspatana*), Bermuda grass (*Cynodon* sp.), cocklebur (*Xanthium strumarium*), wild sugarcane (*Saccharum spontaneum*), common purslane (*Portulaca oleracea*) and white water fire (*Bergia capensis*).

Data collection and analysis

We surveyed both chars through systematic searches along their perimeters and interior along transects spread out 5 m apart and parallel to one another. When a nest was encountered, we marked the nests using small and individually numbered bamboo sticks placed close to the nest. Most of the nests were monitored from the laying of the first egg to the hatching of the last egg by regular visits, on average every four days. For each nest, we measured nest placement (distance between nest and nest distance to shoreline) and nest diameters. We also recorded clutch size and determined laying date (Mayfield 1975). A nest was considered successful if at least one egg hatched, otherwise the nest was considered to have failed. When the same number of eggs remained in a nest after the expected date of hatching it was considered deserted (Bensaci 2014). The length and width of each egg was measured to the nearest 0.05 mm using callipers. Nest fates were determined by observations of chicks, nest age, parental behaviour, and signs of predator (Feral Dog *Canis lupus familiaris*, Golden Jackal *Canis aureus*, Feral Cat *Felis catus*, crow species *Corvus* spp.) and presence of eggshell fragments (Mabee 1997).

Factors affecting nesting success of Little Tern and Small Pratincole were measured by binomial general linear models. All other relevant descriptive statistical analyses and figures were computed in R version 4.2.1 (R Core Team 2022). A critical $p < 0.05$ was used and results are reported as mean \pm standard deviation (SD), unless otherwise specified. A breeding colony map was prepared in ArcMap (version 10.3).

Results

Distribution and abundance of Little Tern nests

A total of 269 nests of Little Tern were recorded from both study sites and 70 nests were monitored till fledging of chicks (*Table 1*). Nests were constructed as shallow depressions in the sand with diameters of 7.5–13 cm (9.8 ± 1.3 , $n = 70$) and depths of 1.5–3 cm (2.02 ± 0.4 , $n = 70$). Nests were located 8–25 m (15.64 ± 4.25 , $n=70$) from water. The average distance to the nearest active nest was 5.26 ± 0.46 m ($n = 70$).

Table 1. Breeding and nesting sites information on Little Tern and Small Pratincole
1. táblázat Adatok a kis csér és a kis székicsér költéséről a vizsgálati területeken

Name of the breeding colony	Species	Area size	Distance (km)			Total nest found	Nests Monitored	Nest fate		Nesting Success (%)
			River Bank	Human habitation				Predated	Deserted	
				Bangladesh	India					
Majherchar	Little Tern	135 ha	0.33	0.45	5	144	45	4	8	73.33
Bidirpur		55 ha	0.97	1.10	4.65	125	25	2	4	76
Majherchar	Small Pratincole	135 ha	0.33	0.45	5	178	40	5	4	77.5
Bidirpur		55 ha	0.97	1.10	4.65	121	30	3	3	80

Clutch size and egg dimensions of Little Tern

Clutch size varied from one to three eggs (2.77 ± 0.48 eggs, $n = 194$). 56 clutches had three eggs, 12 clutches had two eggs, two clutches had only one egg. Egg length and breadth varied between 27.62–40.92 mm (30.88 ± 2.67 , $n = 194$) and 20.86–24.46 mm (22.78 ± 0.85 , $n = 194$), respectively.

Breeding phenology of Little Tern

The first nests were found on 11 March 2022 in the Majherchar and on 14 March 2022 in the Bidirpur char. Egg-laying was observed from 17 March to 16 May (61 days) in Majherchar, and from 18 March – 19 May (63 days) in Bidirpur (Figure 2). Hatching was observed

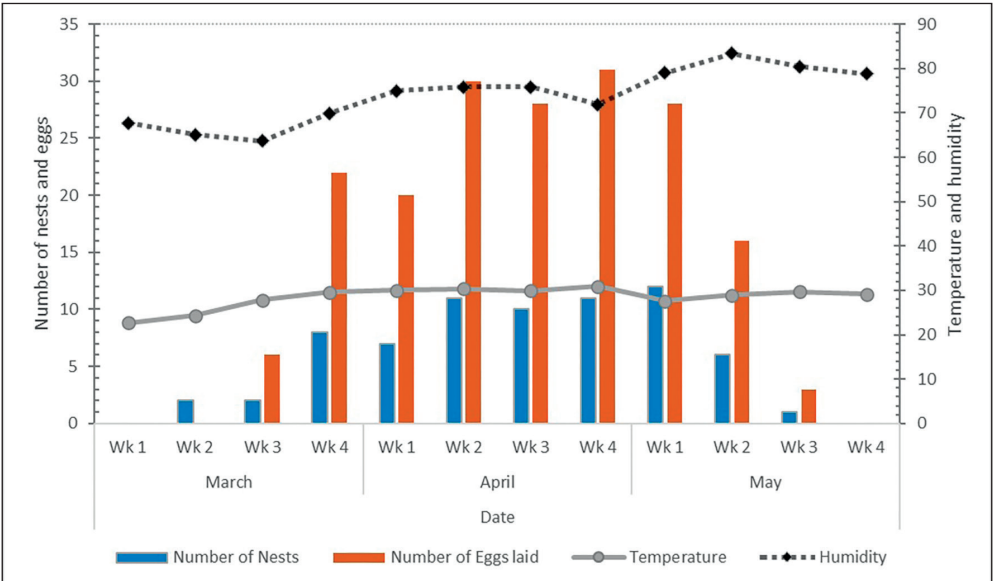


Figure 2. Nesting and egg laying phenology of Little Tern in relation to temperature and humidity in study areas

2. ábra A kis csér fészkelésének és tojásrakásának időzítése a vizsgálati területeken a hőmérséklet és a páratartalom függvényében

from 31 March – 22 May (53 days) in Majherchar and from 1 April – 21 May (51 days) in Bidirpur site. Incubation periods ranged from 15–17 days (16.07 ± 0.8 days). During the breeding season, the mean atmospheric temperature and relative humidity varied from 21.5–30.9 °C (28.5 ± 1.9 °C) and 57–95% ($73.9 \pm 6.5\%$), respectively. *Table 1* and *Figure 2* outlined detailed breeding information on Little Tern from the two study sites.

Distribution and abundance of Small Pratincole nests

A total of 299 nests of Small Pratincole were recorded from both study sites and 70 nests were monitored till fledging (*Table 1*). Nests were found in shallow depressions in the sand with diameters of 6.5–11.5 cm (9.02 ± 1.2 , $n = 70$) and depths of 1–3 cm (1.5 ± 0.97 , $n = 70$). Small Pratincole nests were located 7–26 m (14.45 ± 4.27 , $n = 70$) from water. The average distance to the nearest active nest was 6.14 ± 0.65 m ($n = 70$).

Clutch size and egg dimensions of Small Pratincole

Clutch size varied from one to three eggs (2.77 ± 0.48 eggs, $n = 146$). 14 clutches had three eggs, 48 clutches had two eggs, eight clutches had only one egg. Egg length and breadth varied between 25.16–28.27 mm (26.54 ± 0.82 , $n = 146$) and 19.56–23.45 mm (21.65 ± 0.76 , $n = 146$), respectively.

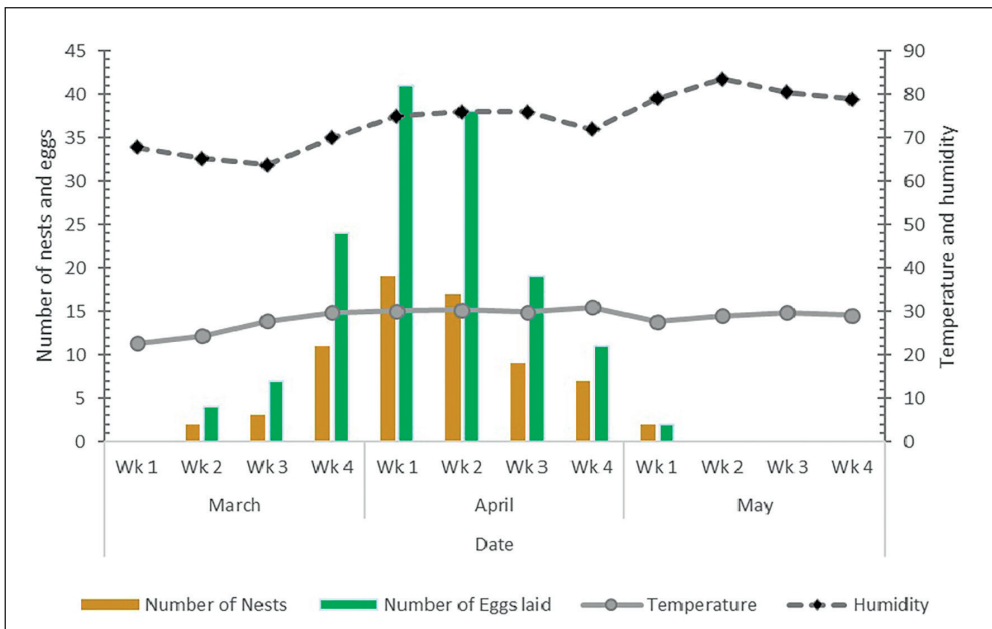


Figure 3. Nesting and egg laying phenology of Small Pratincole in relation to temperature and humidity in study areas

3. ábra A kis székicsér fészkelésének és tojásrakásának időzítése a vizsgálati területeken a hőmérséklet és a páratartalom függvényében

Breeding phenology of Small Pratincole

The first breeding pair of Small Pratincole was seen on 4 March 2022 in Majherchar and on 5 March 2022 in Bidirpur. The first nests were found on 11 March and 14 March, respectively. Egg-laying was observed from 11 March to 4 May 2022 (55 days) in Majherchar, and from 14 March to 3 May 2022 (51 days) in Bidirpur (Figure 3). Hatching was observed from 24 March to 17 May (55 days) in Majherchar and from 26 March to 15 May (51 days) in Bidirpur. Incubation periods ranged from 13–15 days (13.64 ± 2.75). Table 1 and Figure 3 outlined detailed breeding information on Small Pratincole from the two study sites.

Nest survival for Little Tern and Small Pratincole

Among the 70 monitored nests of Little Tern in two riverine chars, at least one chick from 52 nests successfully hatched, yielding a nesting success of 74.28%. Both the sites showed almost similar trends in terms of nest survival (Table 1). The main factors affecting nesting success were nest predation by avian predators (Figure 5c) ($n = 6$) (House Crow *Corvus splendens*, Black Kite *Milvus migrans*, and *Circus* sp.) and storms ($n = 3$). In case of nine deserted nests (with unhatched eggs), no apparent reasons were identified. No correlation was found for nesting success of Little Tern regarding clutch size, nest height, nest depth and presence of vegetation near nests (Table 2). Although, strong positive correlation was found between nesting success and nest distance to river for Little Tern (Figure 4).

Overall breeding success of Small Pratincole was 78.57%. No significant differences were observed in terms of nesting success in two distant islands (Table 1). Eight Pratincole nests were predated by avian predators and four nests were lost due to storm. However, in case of three deserted nests (with unhatched eggs) we failed to find any cues. No influence of clutch size, nest height, nest depth, nest proximity to water and presence of vegetation near nests were found in favor of Small Pratincole's nesting success (Table 2).

Table 2. Binomial general linear models showing parameters affecting nesting success of Little Tern and Small Pratincole in the study area

2. táblázat A kis csér és a kis székicsér költési sikerét befolyásoló tényezők a binomiális általános lineáris modellekkel végzett számítások alapján

Model parameters	Coefficients		Z value	Pr(> z)
	Estimated	Standard Error		
Nesting success of Little Tern				
Distance to river	0.160	0.080	1.995	0.040
Nest height	0.193	0.906	-0.213	0.831
Clutch size	-0.220	0.673	-0.328	0.743
Nesting success of Small Pratincole				
Distance to river	-0.073	0.080	-0.914	0.361
Nest height	-0.066	0.350	-0.190	0.850
Clutch size	0.335	0.623	0.538	0.591
Presence of vegetation near nest	0.422	1.878	0.225	0.151

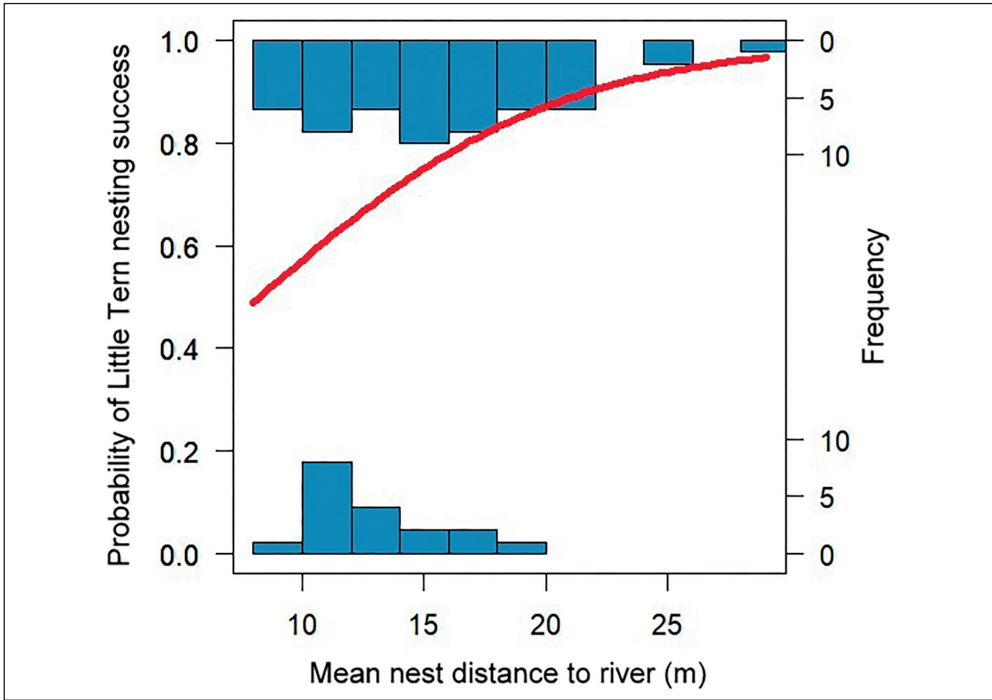


Figure 4. Predicted probability of Little Tern nesting success in relation to nest distance to river

4. ábra A kis csér költési sikerének feltételezett valószínűsége a folyótól való távolság függvényében

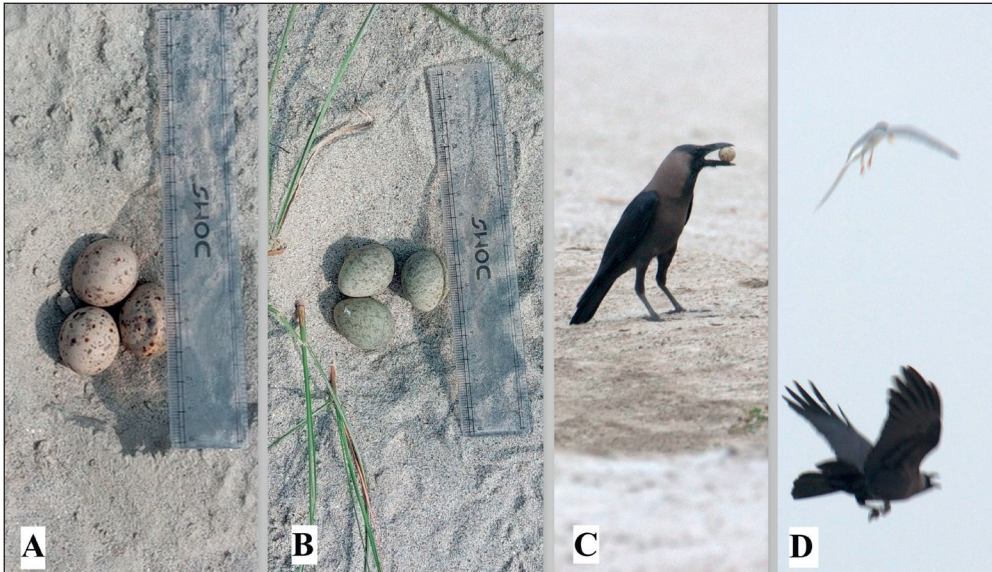


Figure 5. (a) Eggs of Little Tern (b) eggs of Small Pratincole (c) Little Tern egg predation by House Crow and (d) nest defense by Little Tern

5. ábra (a) kis csér tojásai (b) kis székicsér tojásai (c) indiai varjú kis csér tojásával a szájában (d) fész-
két védőmező kis csér

Discussion

Colonial nesting of Little Tern and Small Pratincole with other Charadriiforms (including Collared Pratincole *Glareola pratincola*) is common and observed in many other countries (Fasola & Canova 1992, Fernandes & Besten 2013, Kiss *et al.* 2018). In this study, we confirm it for the Ganges-Padma River floodplain in Rajshahi district, Bangladesh. Nest building, egg laying and hatching of Little Tern and Small Pratincole occurred from March to May in both study sites with a peak during April. Both study species completed the breeding activities concurrently and no inter specific aggression behaviour was observed. On the contrary, both species defend their nests from avian predators.

Little Tern and Small Pratincole collective nesting success was similar (74.28% and 78.57%) in the study sites. We did not find any mammalian predator, cattle and feral dogs or cats, which probably increase the breeding success of the two studied species. Medeiros *et al.* (2007) found that predation and anthropogenic disturbance play a vital role in limiting breeding success. In case of Little Tern, nests near water have less probability to success as nests located close proximity to water channel had a higher chance of flooding (Claassen *et al.* 2018). We suspected that about 10 nests of both species were abandoned due to heavy rain and storms. Similar nest flooding was observed as an important limiting factor for nesting success in Little Tern and Small Pratincole (Goutner 1990, Pakanen 2014, Debata 2019).

The study sites are only two of the many riverine islands in the Ganges-Padma River floodplain, which are periodically transformed in shape by silt deposition and erosion. These islands become completely submerged during monsoon. Previous studies found many nationally and globally threatened bird species from these areas (Chowdhury *et al.* 2014, Thompson 2021). However, cattle grazing, sand extraction, hunting during winter season, extensive fishing, conversion of grassland into croplands are some prevailing threats (Chowdhury *et al.* 2014, Datta 2022, Jahan *et al.* 2022), which must be managed for better protection of waterbird and grassland bird species. We further recommend conducting additional in-depth surveys to learn more about the diversity of fish and other waterbird prey species, as well as their influence on breeding biology.

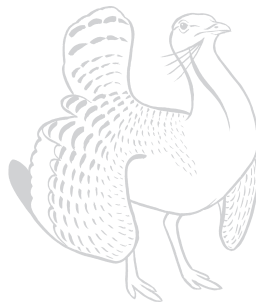
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The recolonization of the Collared Pratincole (*Glareola pratincola*) in its natural steppe habitats in Hungary – an overview of the species' grassland activities between 1995 and 2024

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Abstract The Collared Pratincole (*Glareola pratincola*) is an endangered shorebird in Hungary, which disappeared as a character breeding species from its sodic grassland habitats during the mid-1990s. Therefore, it was a positive experience to witness the return of this species to its natural habitats, which we observed and documented in 2022 and 2024, and suggested in 2023. During the 2022 observation, one of the three pairs nesting in the Alsó-Szúnyog-pusztá reached the chick-rearing stage. We have summarised twenty-one observations of the species in its natural habitat during the breeding season between 2013 and 2024. Based on our data, Collared Pratincole has been confirmed to breed three times in Hungary in its natural habitats, with five additional instances of assumed breeding between 1995 and 2021. In 2022, the species bred in sodic grassland (*Camphorosmetum annuae*) grazed with livestock. Pratincoles nested in colonies mixed with Kentish Plovers (*Charadrius alexandrinus*) and other shorebirds, such as Northern Lapwings (*Vanellus vanellus*), Black-tailed Godwits (*Limosa limosa*), and Common Redshanks (*Tringa totanus*).

Keywords: shorebird, sodic grassland, grazing management, nesting behaviour

Összefoglalás A székicsér (*Glareola pratincola*) egy kritikusan veszélyeztetett partimadár-faj Magyarországon amely, mint sziki fészkelő karakterfaj eltűnt természetes élőhelyeiről az 1990-es évek közepén. Ebben a tanulmányban a faj szikes pusztai élőhelyeire történő visszatérését dokumentáltuk, hiszen az eredeti szikes pusztai környezetben hosszú idő eltelte után 2022-ben jelent meg és fészkelte sikeresen Magyarországon. Feltehetően 2023-ban is fészkelte a faj gyepeken, de 2024-ben a gyepi költését újra sikerült bizonyítani. A 2022-es megfigyelés során az Alsó-Szúnyog pusztán fészkelő maximum három párból legalább egy pár eljutott a fiókanevelési időszakig. Adataink alapján 2013 és 2024 között huszonegyszer észleltek a székicsért kóborlóként a költési időszakban gyepeken, továbbá három bizonyított és öt feltételezett költése ismert 1995 és 2021 között. Az Apaj határában fekvő Alsó-Szúnyog-pusztán bizonyított fészkelése magyar szürke szarvasmarhakkal illetve bivalyokkal legeltetett vakszikes területen (*Camphorosmetum annuae*) történt, ahol többek között széki lilékkel (*Charadrius alexandrinus*), bibicékkel (*Vanellus vanellus*), nagy godákkal (*Limosa limosa*) és piros-lábú cánkókkal (*Tringa totanus*) vegyes kolóniában költöttek.

Kulcsszavak: partimadár, szikes pusztá, legeltetéses élőhelykezelés, fészkelési viselkedés

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Introduction

The Collared Pratincole (*Glareola pratincola*) is an insectivorous shorebird species that typically breeds in open, flat sodic areas interspersed with shallow wetlands and low vegetation in its surroundings (Cramp & Simmons 1983). The global population is presumed to be stable (BirdLife International 2021); however, European populations have shown a declining trend in recent decades (Lokhman *et al.* 2020). Within Europe, its populations are isolated and population declines were observed in both western and eastern regions (Lokhman *et al.* 2020). Threat factors include the degradation and loss of traditional steppe areas and wetland habitats. Due to these effects, the species has appeared to breed in alternative habitats, such as agricultural areas, similarly to many other shorebirds – a phenomenon increasingly observed over the past decades (Calvo 1994, Lokhman *et al.* 2020).

In Hungary, the Collared Pratincole is a critically endangered breeding shorebird, with a fluctuating population trend, though due to direct nest protection efforts, the number of breeding pairs increased significantly from 30–31 pairs to 78–86 pairs during the past twelve years (Kiss 2021, Kiss *et al.* 2024). Currently, two breeding populations are known within the Carpathian Basin, which are located in the cultivated fields near the sodic steppes of the Kiskunság region, and the agricultural habitats close to the rice fields in the Nagykunság, in the middle of the Great Hungarian Plain (Kiss *et al.* 2024). Pairs nesting in intensively cultivated farmlands are particularly vulnerable to direct and indirect impacts of agricultural practices, such as disturbance, trampling, ploughing, adverse weather conditions, and increased nest predation (Kiss *et al.* 2018). In the early 20th century, the previously stable population began to decline significantly following the changes that occurred after the Second World War, including the rapid decrease of grazing livestock. Traditionally, the Collared Pratincole was a characteristic and unique nesting species in sodic grasslands grazed by cattle and sheep, typically near soda pans, marshes, and fishponds, which served as its typical feeding, and roosting places (Sterbetz 1974). The last stable breeding populations of the Collared Pratincole disappeared from the Southern Hortobágy in the mid-1990s (Kovács & Kapocsi 2004). In addition to disappearing from their traditional nesting sites, nesting pairs began to appear sporadically in various arable lands with sparse vegetation, typically in fallow fields. This phenomenon was documented by several ornithologists working in Hungary in the 1970s, 1980s, 1990s, and 2000s. Such nesting records were also known from Békési-sík, Hortobágy, Nagykunság, and Kiskunság (Haraszthy 2019). One of the last confirmed nesting events with parenting chicks in a natural habitat was recorded in 2000 (Pigniczki 2000).

The aim of the study is 1) to describe the recolonization of the species to its natural sodic habitat based on observations in the last three years; 2) to summarise the sightings at former breeding sites during the breeding seasons of Collared Pratincole between 2013 and 2024; and 3) to aggregate probable, suspected, and confirmed breeding data from grassland between 1995 and 2021.

Material and Methods

Collared Pratincole observation data in the former nesting sites were collected between 2013 and 2024 in Hungary, and these traditional breeding sites were documented by Sterbetz (1974) between the early 1900s and early 1970s. Further data were collected from the databases of Hortobágy National Park Directorate and Hortobágy Environmental Association, supplemented by additional records from the database of www.birding.hu and Gábor Kovács personal observations (Kovács 2017). Observations were filtered to the regional nesting period of Pratincoles, specifically between 1 April and 30 June, and include records on wandering individuals as well as typical breeding behaviours such as courtship activities, nest building, and incubation, if there are no other observations on breeding success.

Nesting events were confirmed via the observation of typical behavioural traits associated with the current breeding stage using binoculars and spotting scopes from a hotspot point (Kiss *et al.* 2018), supplemented by field methods developed by Székely *et al.* (2006) for documenting behavioural cues.

Results

Observations in the traditional nesting sites between 2013 and 2024

Based on available data, we collated twenty-one observations of Collared Pratincoles from grasslands during the nesting seasons in four counties (*Table 1, Figure 1*). Among these observations, courtship activities were noted in three instances, although these did not continue with nesting at the observed locations.

Suspected and confirmed breeding attempts between 1995 and 2021

Between 1995 and 2021, eight breeding events were previously recorded (*Table 2, Figure 2*). These data typically reflect the recolonization of nesting at traditional sites between the 1970s and 1990s. The data from Kiskunság can be considered verified nesting, as the nests were found and/or young chicks were observed with the parents. In the cases from Hortobágy, nests were not found after 1992, however, the observation in 2001 was documented as a colony. In other cases, based on the ageing condition of the juveniles, they could have come from nearby nesting sites in Nagy-kunság.

Observations in 2022

- May 18, 2022: Four Collared Pratincoles displayed intense courtship behaviour at the Alsó-Szúnyog-pusztá flooding area. The site appeared highly suitable for potential resettlement, prompting its monitoring initiation (Á. Takács).
- Between May 20 and June 1, 2022: Fluctuating numbers of three to six individuals were observed in the area, showing varying levels of nesting activities (Cs. Pigniczki; Á. Takács).

Table 1. Observations of Collared Pratincoles at former grassland breeding sites during the nesting period between 2013 and 2024

1. táblázat Adatok a székicsérek előfordulásairól az egykori gyepi költőhelyeken 2013 és 2024 között, a fészkelési időszakokban

Date	Location	Number of individuals	Breeding behaviour	Observers
29.04.2014 – 19.06.2014	Apaj Alsó-Szúnyog-pusztá	2	No	N. Kovács, Z. Cserna, Zs. Semperger, Á. Kiss
28.06.2014 – 28.06.2014	Sándorfalva, Fertő (fishponds)	3	No	O. Kiss, Zs. Kiss, B. Csibrány
09.06.2015	Hortobágy, Hortobágy-fishponds	1	No	D. Balla
28.06.2015 – 30.06.2015	Hortobágy, Hortobágy-fishponds	2	No	T. Emri, M. Barna Zöld, J. Tar
14.05.2016	Nagyhegyes, Kis-Álomzug	1	No	A. Szilágyi
17.05.2016 – 20.05.2016	Balmazújváros, Magdolna-pusztá	1	No	J. Tar, Z. Ecsedi, D. Balla, J. Kecskés <i>et al.</i>
24.05.2016	Sándorfalva, Fertő (fishponds)	1	No	Á. Bede
22.04.2018	Nagyhegyes, Nagy-Álomzug	1	No	S. Konyhás
22.05.2018	Hortobágy, Hortobágy-fishponds	1	No	G. Kovács
16.05.2020	Hortobágy, Akadémiai-libanevelő	2	Yes, displaying male and female	J. Katona, A. Szilágyi, P. Gyüre, M. Nagy, J. Oláh
26.04.2021	Apaj, Alsó-Szúnyog-pusztá	2	No	Zs. Paráda
26.06.2021	Kardoskút, Fehér-tó	1	No	E. Kovács
11.05.2023	Kunmadaras, Kunmadarasi-pusztá	1	No	S. Borza, S. Ujfalusi
14.05.2023 – 24.05.2023	Hortobágy, Vincze-fenék	1	No	J. Tar, T. Zalai, T. Emri, M. Barna Zöld
01.05.2024	Hortobágy, Fényes-fishponds	1	No	M. Nagy, E. Tóth
02.05.2024	Akasztó, Miklapusztá	1	No	T. Bárdos, A. Liker, L. Kocsis
12.05.2024	Hortobágy, Hármaspusztá	1	No	M. Nagy, E. Tóth
12.05.2024 – 26.05.2024	Karcag, Kecskeri-pusztá	6	Yes, displaying pairs (3)	Á. Kiss
27.05.2024	Pusztaszer, Vesszős-szék	1	No	Cs. Mészáros
27.05.2024	Mórahalom, Nagy-Széksós-tó	2	Yes, displaying pair	Cs. Mészáros, B. Tokody
17.06.2024	Kunmadaras, Kunmadarasi-pusztá	3	No	S. Borza

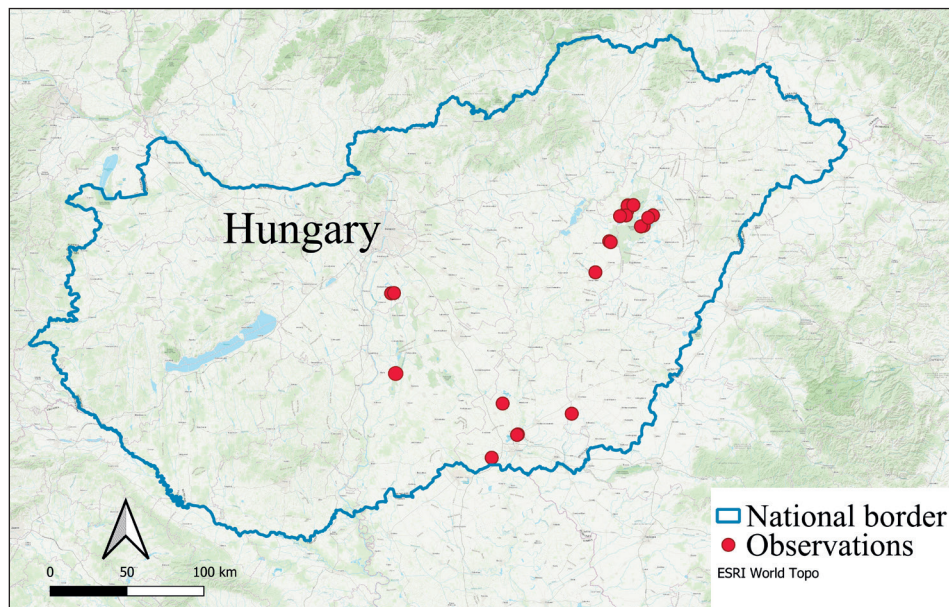


Figure 1. Data of Collared Pratincoles at traditional breeding sites during the nesting period between 2013 and 2024

1. ábra Költési időszakban mutatott székicsér-előfordulások az egykori költőhelyeken 2013 és 2024 között

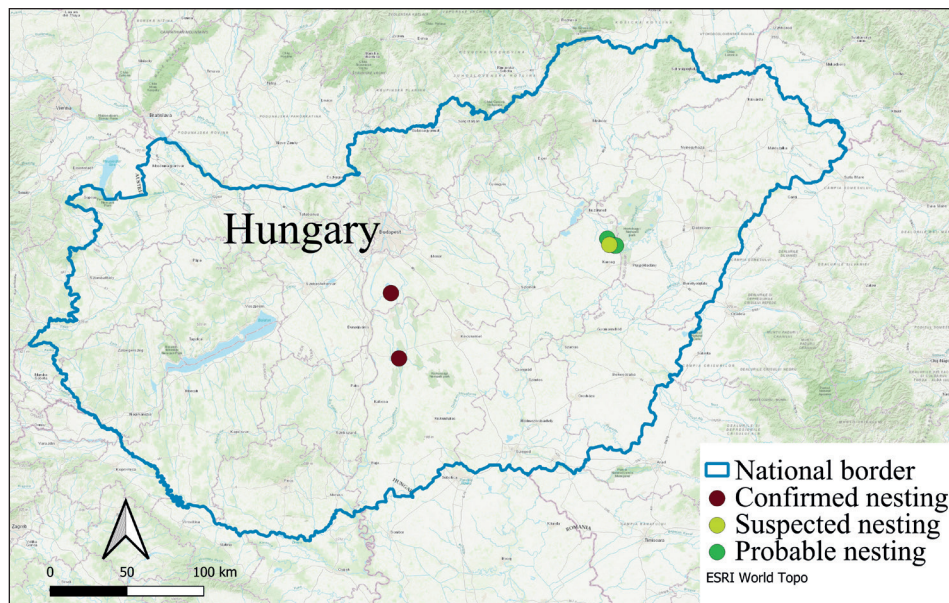


Figure 2. Probable, suspected and confirmed breeding data of Collared Pratincoles in grasslands, Hungary between 1995 and 2021

2. ábra Valószínű, feltételezett és bizonyított székicsér-fészkelések gyepeken, Magyarországon 1995 és 2021 között

Table 2. Assumed and verified nesting performances of Collared Pratincoles in grassland habitats in Hungary between 1995 and 2021

2. táblázat Feltételezett és igazolt gyepi székcicsér fészkelések 1995 és 2021 között Magyarországon

Date	Location	Habitat (Plant community)	Confirmed nesting (no. of pairs)	Details	Observer
1997	Apaj, Alsó-Szúnyog- puszta	Sodic grassland <i>Camphorosmetum annuae</i> , <i>Puccinellietum limosae</i>	Yes (10–12)	Found nests	A. Széll
09. 09. 1998	Kunmadaras, Ecsezug	Sodic grassland <i>Camphorosmetum annuae</i> <i>Puccinellietum limosae</i>	No (1)	Two chicks (unknown age of chicks) with two parents, nest not found	G. Kovács
24. 06. 2000	Fülöpszállás, Kelemen-szék	Soda pan	Yes (1)	Two freshly hatched chicks with parents	Cs. Pigniczki (Pigniczki 2000)
2001	Kunmadaras, Csíkos-hát	Sodic grassland <i>Camphorosmetum annuae</i>	No (4–5)	Alarming behaviour, no found nests	G. Kovács
2004	Apaj, Alsó- Szúnyog-pusztá	Sodic grassland, <i>Puccinellietum limosae</i>	Yes (8–10)	Nesting, nests with eggs were flooded by water after a heavy rain	Cs. Pigniczki, A. Széll
07.08. 2004	Kunmadaras, Forrás-fenék	Sodic grassland <i>Camphorosmetum annuae</i>	No (3)	Alarming behaviour, nest not found	G. Kovács
10. 08. 2005	Karcag, Ecsezug	Sodic grassland <i>Camphorosmetum annuae</i> , <i>Puccinellietum limosae</i>	No (1)	One chick (unknown age of chicks) with two parents, nest not found	G. Kovács
16. 07. 2013	Karcag, Ecsezug	Sodic grassland <i>Camphorosmetum annuae</i> , <i>Puccinellietum limosae</i>	No (3)	Adults with freshly fledged juveniles	G. Kovács, S. Borza

- June 1, 2022: In the late afternoon and early evening, three adult Collared Pratincoles were present. At 18:15, three individuals aggressively attacked a Northern Lapwing. Twenty minutes later, two individuals rested on the shore of the floodplain. At 19:15, one individual flew approximately 300 metres to a barren area, exhibited typical nesting behaviour, including checking and adjusting eggs, and assumed a brooding posture (Á. Kiss, P. Ócsai, Cs. Pigniczki, Á. Takács).
- June 18, 2022: No Collared Pratincole sightings were reported in the area (Cs. Pigniczki).
- June 28, 2022: Three individuals displayed typical nesting behaviour for the species, chasing away European Magpies (*Pica pica*). Later, they returned to roughly the same spots on the ground and remained vigilant (Á. Kiss, M. Lóránt, Á. Takács) (Figure 3).
- July 22, 2022: Two pairs were observed feeding; at one pair, Á. Takács spotted a nestling estimated to be no more than one week old.



Figure 3. A sentinel adult Collared Pratincole at Alsó-Szúnyog-pusztá in 2022 (Photo: Ádám Kiss)
 3. ábra A fészkelőhelyen őrködő székicsér Alsó-Szúnyog-pusztán 2022-ben (Fotó: Kiss Ádám)

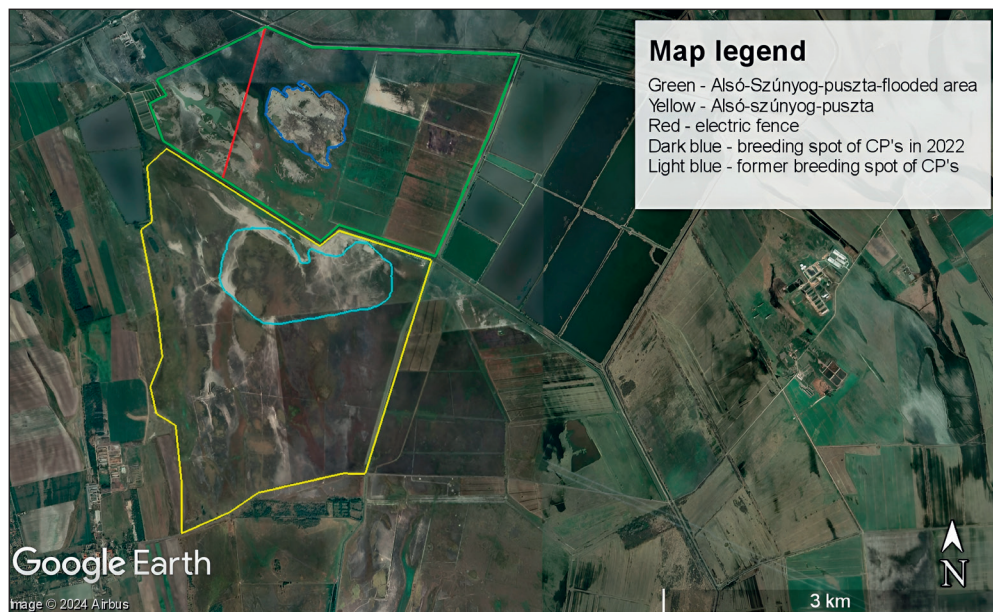


Figure 4. Distribution map of the breeding area of Apaj in 2022, important characteristics bordered by different coloured lines

4. ábra Az apaji fészkelőhely és jellemző szerkezeti elemei 2022-ben



Figure 5. Structure of the alkaline grassland as a breeding site of Collared Pratincoles in 2022 (Photo: Ádám Kiss)

5. ábra A székicsér gyepterületi fészkelőhelye 2022-ben (Fotó: Kiss Ádám)

Based on our observations, it is likely that between late May and early June in 2022, a minimum of one, and a maximum of three pairs of Collared Pratincoles began nesting under natural conditions on a grassland site. First clutches probably failed, however 1–2 pairs in the replacement clutches nested successfully.

This site covers approximately 329 hectares and has been managed through extensive grazing by local farmers under the professional supervision of the Kiskunság National Park Directorate. In 2022, the area was grazed by Hungarian grey cattle and water buffalo, enclosed by electric fencing (Figure 4). This high-level grazing management served as habitat maintenance. The area was flooded by freshwater during the winter. Within the flooding zone, the livestock created a highly mosaic habitat structure through grazing and trampling, benefiting shorebirds with varying habitat requirements.

The Collared Pratincoles and Kentish Plovers (*Charadrius alexandrinus*) selected a barren, approximately 1.6 ha area in the central part of the grassland that sparsely covered with halophile vegetation, and the surface of the ground was poached because of the trampling of the cattles (Figure 4). This white, sodic patch had a few larger areas with manure crust and scattered dung. This specific microhabitat was ideal for Pratincoles due to its unique structure and vegetation (Figure 5).

In Alsó-Szúnyog-pusztá, the area's conservation status has been consistently excellent over the years, indicated by the presence of numerous valuable and rare shorebirds as important character species nesting alongside Collared Pratincoles. These include numerous

pairs of Northern Lapwing (*Vanellus vanellus*), 2–4 pairs of Kentish Plover, and a few pairs of Black-tailed Godwit (*Limosa limosa*), and Common Redshank (*Tringa totanus*), all confirmed as co-nesting shorebird species.

Visitors were closed out, thus direct and indirect human disturbance was eliminated. Trampling by grazing livestock in the flooded area likely destroyed some shorebird nests. Predation by mammalian and avian predators could have affected the nesting success of Collared Pratincoles and co-nesting shorebird species at the flooding site. We observed Red Foxes (*Vulpes vulpes*), Western Marsh Harriers (*Circus aeruginosus*), European Magpies, and Hooded Crows (*Corvus cornix*) in the area. Fox sightings occurred once, while Harriers were regularly present in small numbers (1–2 individuals). Hooded Crows occasionally flew alone above the area, however, did not establish a sustained presence due to the lack of suitable roosting sites. It is important to note that the medium-voltage power line in the area was removed in 2019, previously, the pylons and wires were both used as perching spots by predators, including Hooded Crows and other avian predators.

In 2022, intensive rainfall likely did not cause nest failures as no significant precipitation events happened that year. However, in 2023 and 2024, the area was affected by larger amount of precipitation during spring and summer.

Observations in 2023

- One adult was observed on 29th April (Á. Takács).
- Between May 6 and July 3, five additional observations were recorded from the area, typically involving 1–4 individuals, either on the ground or feeding above the site, reported by observers and uploaded to the www.birding.hu database.

Observations in 2024

- Two displaying adults were observed on the site on 2nd May (Á. Takács). Cs. Pigniczki likely found and recorded an alarming, plus an incubating pair on 25 May. These birds attacked flying predators, too. The two pairs moved to the former, traditional breeding site. A few days later, nesting could no longer be confirmed as the breeding bird was not present, the cause of nest failure remained unknown.
- On June 3, T. Sápi observed a mating pair at Kelemen-szék near Fülöpszállás.
- On 5 June, Cs. Pigniczki also observed the same pair at Zab-halmi-szék, they arrived from the south, probably from Kelemen-szék and moved forward to the north while hunting above the area.

Discussion

Here, we presented the recolonization events of the Collared Pratincole in sodic grasslands habitats in Hungary in 2022–2024. Pratincoles are also nested in similar, low-vegetation and rarely covered nesting sites with huge bare areas on the ground, which

have historically characterized as the typical habitat of this species. This type of sodic grassland environment with this character plant community structure grazed mainly by cattle was similar to that was documented by Szabó (1980) in the Hortobágy. In southwest Spain, Collared Pratincoles chose breeding sites on dry mud spots close to marshlands, which were covered by cow pats in higher densities (Villanueva 1993). In 2022, Hungarian grey cattles often rested close to the breeding sites during the midday hours, leading to a higher density of cow pats.

The persistent emergence of Collared Pratincole nesting in natural habitats provides hope for conservation efforts, as the species' habitat selection likely involves a complex interplay of known and unknown factors. Indeed, it is a well-known fact in management processes that the grazing level of the area needs to be high. A similar habitat management practise was previously observed at the Kunmadarasi-pusztas area, in the Hortobágy by Gábor Kovács, moreover in Doñana National Park, Spain, where the breeding habitats around the salt lakes were grazed by a mixed flock of cattle and horses before the breeding season in 2018 (Á. Kiss *pers. obs.*). The high level of grazing in the habitat likely increased the local insect density, making potential prey more easily available in the area, however, we did not investigate this at this time. Hopefully, in the coming years, the species will reappear in previous traditional steppe and soda pan habitats, especially following successful nesting events by resident pairs during the preceding years. If management conditions remain favourable and the presence and density of co-nesting shorebird species show similar patterns, Collared Pratincoles could potentially become regular breeders in the natural habitats of the Kiskunság region.

In recent years, grazing management changes have also taken place in various parts of the Hortobágy region, suggesting that the species may also appear there before and during the breeding season, with individuals or pairs exploring the area. The species regularly visited the wetlands and "pusztas" of Southern Hortobágy during the nesting period and immediately after, which assumes that the ecological conditions for the species were favourable at that time. Though less frequent, spring and early summer sightings indicate that individuals potentially wandering through the region or those starting their breeding season later or after failed attempts may return to former nesting sites if suitable feeding conditions persist, possibly staying longer. Moreover, after the end of the breeding season, individuals from the current population in Nagykunság also reach their former steppe nesting sites. This was evidenced by a colour-ringed individual marked as a chick near to Nagyrét (Kisújszállás) in 2022, and observed as a juvenile by S. Borza close to Nagyiván.

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Chewing lice (Phtiraptera) on the Bonelli's Eagle *Aquila fasciata* in the Iberian Peninsula (Spain)

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Abstract Bonelli's Eagle is an endangered species with little information on its ectoparasites. We studied their chewing lice (Phthiraptera: Amblycera, Ischnocera) infestations by visually searching the plumage of 64 wild and captive specimens. Three different species were identified: *Degeeriella* sp., *Colpocephalum impressum* and *Laemobothrion maximum*. This is the first case of *Laemobothrion maximum* on Bonelli's Eagle described. We also observe its haematophagous behavior. Due to the low prevalence of infestations, no alterations in the host health are described.

Keywords: feather, ectoparasite, hematophagous, louse, raptor

Összefoglalás A héjasas egy veszélyeztetett faj, amelynek ektoparazitáiról kevés információ áll rendelkezésre. Tolltetű (Phthiraptera: Amblycera, Ischnocera) fertőzéseiket 64 vadon élő, illetve fogságban tartott példány tollazatának vizuális vizsgálatával tanulmányoztuk. Három fajt azonosítottunk: *Degeeriella* sp., *Colpocephalum impressum* és *Laemobothrion maximum*. Ez a *Laemobothrion maximum* általi fertőzés első ismert esete a héjasason. Megfigyeltük az említett faj vérfogyasztását is. A fertőzés alacsony prevalenciája miatt a vizsgált héjasasok egészségi állapotában nem történt változás.

Kulcsszavak: madártoll, ektoparazita, hematofág, tolltetű, ragadozó madarak

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Prevalence of parasites on a bird is conditioned by factors such as sex, age, sociability, mobility or its reproductive status and can have an impact on the physiological, physical and even immunological state of the host (Hamstra & Badyaev 2009, Owen *et al.* 2010, Doña *et al.* 2018, Talabante & Bernal 2022). In cases of seriously threatened birds, knowledge of their parasites can also provide information about their health status at a population level (Talabante *et al.* 2019). Therefore, knowledge of parasite-host relationships can be essential to determine different infective degrees or physiological deficits in their hosts. Despite this importance, little attention has been paid to parasite-host relationships for certain groups of birds.

One of the most diversified groups of parasites are chewing lice (Phthiraptera: Amblycera and Ischnocera suborders), which are mainly present in birds. Although they are considered parasites, numerous species can carry out a mutualistic relationship with their host (Martín-Mateo 2002). However, there are many species that interact with their hosts as parasites and can cause health problems to their hosts.

For our research we use the Bonelli's Eagle (*Aquila fasciata*) as a study model. Bonelli's Eagle is a large-sized eagle which inhabits mountainous areas across the Palearctic, Indo-Malayan and Afrotropical regions in arid and semi-humid habitats (Ferguson-Lees & Christie 2001). Their nests are placed on cliffs or in different tree species (Del Moral & Molina 2018). It is currently considered as endangered in Spain and as of Least Concern worldwide (Real Decreto 139/2011, BirdLife International 2019). It is a monogamous, territorial and solitary species, so the horizontal transmission of ectoparasites is likely to be less frequent than in more social raptors (Talabante & Bernal 2022). Studies on parasites and pathogens in Bonelli's Eagle are scarce and mainly focus on endoparasites (Real *et al.* 2000, Reche *et al.* 2003, Cabezón *et al.* 2011). According Pérez *et al.* (1996) and Martín-Mateo (2002), six lice species have been recorded globally on Bonelli's Eagle, Amblycera suborder: *Kurodaia fulvofasciata* Piaget 1880, *Colpocephalum impressum* Rudow 1866b and *Laemobothrion maximum* Scopoli 1763; Ischnocera suborder: *Degeeriella fulva* Giebel 1874, *D. regalis* Giebel 1866 and *Craspedorrhynchus fasciati* Gallego *et al.* (1987). In Spain, Gallego *et al.* (1987) cited the presence of *Cr. fasciati* in a review of this genus of lice. Cordero del Campillo *et al.* (1994) and Martín-Mateo (2009) cited the presence of *D. fulva* in their respective reviews. Finally, Pérez *et al.* (1996) published the occurrence of *D. regalis* and Martín-Mateo (2002) cited *Co. impressum* (under the synonym name *Aquilogogus impressus*).

Despite the degree of threat of the Bonelli's Eagle and the interest it has aroused in the community over these years, the study of its lice species remains insufficient. The purpose of this study is to review the lice present in the Bonelli's Eagle, providing new record of species not previously cited in Spain. In addition, the possible effect that the chewing lice species detected may have on this threatened species is indicated.

Lice were collected directly from the bird feathers in scientific bird ringing sessions or in wild specimens recently admitted to wildlife recovery centers, from 2022 to 2023, following the method visual examination of Martín-Mateo (1994) and Clayton & Drown (2001). Lice were stored in capped tubes containing 70% ethanol until their identification and were cleaned in 10% KOH, mounted in DMHF (Dimethyl hydantoin formaldehyde) on slides, and identified to species level under a light-microscope and stereo-microscope (Martín-Mateo 1994). The chewing lice species were identified according to Martín-Mateo (2002, 2009). Data on lice infestation on Bonelli's Eagles have been analyzed according to the methodology presented in Reiczigel *et al.* (2019). To avoid biases in these determinations all specimens were identified by the same researchers (CT and IB) and deposited in the authors' collections.

The chewing lice of a total of 64 Bonelli's Eagles have been studied: 42 wild specimens either nestling or adults and 22 captive individuals from recovery centers. Lice have been found in 7 Bonelli's Eagles, implying a prevalence of 10.93%. The presence of three lice has

Table 1. Prevalence of three chewing lice species detected on 64 Bonelli's Eagles from Spain. Confidence intervals (CI 95%) are calculated by Sterne's method (Reiczigel *et al.* 2019)

1. táblázat Három tolltetű faj prevalenciája 64 héjasason Spanyolországban. A konfidencia intervallumokat (CI 95%) Sterne módszerével számoltuk (Reiczigel *et al.* 2019)

Suborder	Species	Prevalence (%)	CI (95%) of prevalence	Mean intensity
Amblycera	<i>Laemobothrion (L.) maximum</i>	1.6	0.1–8.3	9.00
	<i>Colpocephalum impressum</i>	9.4	4.2–19.3	2.33
Ischnocera	<i>Degeeriella</i> sp.	3.1	0.6–10.7	1.50

been determined: *Laemobothrion (L.) maximum* (6 females and 3 males), *Colpocephalum impressum* (8 females and 6 males) and *Degeeriella* sp. Neumann 1906 (3 imaginal females). The prevalence and mean intensity of each species is shown in Table 1. Blood was detected in the digestive system in 2 females out of the 9 *L. maximum* specimen, which indicates hematophagous behavior (Figure 1).

Hematophagy brings possible veterinary and conservation consequences since this host is a seriously threatened species. Although in most cases the presence of a small number of lice in birds do not produce symptoms, when the number of lice is very high, it can cause discomfort to the animal, affect the plumage quality, reduce its ability to reproduce and even affect its survival (Vysakh Mohan *et al.* 2001, Krone & Cooper 2002, Clayton *et al.* 2008).

Ontiveros *et al.* (2008a, 2008b) and Martínez *et al.* (2022) show a lower prevalence of ectoparasites in Bonelli's Eagles that incorporate pine branches in their nests, characterized



Figure 1. Females of *Laemobothrion (L.) maximum* with blood (dark red coloration) in the digestive system

1. ábra *Laemobothrion (L.) maximum* nőstények, emésztőrendszerükben vérrel (sötétvörös elszíneződés)

by a high level of aromatic compounds highly repellent for insects. These authors suggest a higher reproduction rate in couples that have a lower load of ectoparasites. However, the results of both studies focus on diptera and coleoptera, not considering lice species. Furthermore, Ontiveros *et al.* (2008b) suggest that the maintenance of alternative nests in raptors favors the reduction of ectoparasites, but this does not happen every year, so that the nests may be functioning as reservoirs for several seasons.

The presence of *D. fulva* has been previously reported by Cordero del Campillo *et al.* (1994). *D. fulva* is a louse present in other species of diurnal raptors, among others the Bonelli's Eagle (Martín-Mateo 2009). In our study, only females *Degeeriella* specimens were obtained, which has made species-level identification difficult. Although the chaetotaxy of the specimens studied suggests *D. fulva*, since it was not possible to study male genitalia, it prevents the reliable identification of the species. *Co. impressum* (syn. *Aquilogogus impressus* Martín-Mateo 2002) has also been cited in Golden Eagle (*Aquila chrysaetos*) (Martín-Mateo 2002). Both *Degeeriella* and *Colpocephalum* Rudow 1866 are considered common ectoparasites in large raptors. We could not detect *Cr. fasciati* in our sample, though it does appear in the review by Gallego *et al.* (1987).

Our results represent the first record of *L. maximum* in Bonelli's Eagle in Spain. Furthermore, the presence of specimens of *L. maximum* with hematophagous behavior has been described in our study, which is also the first described behavior of this type of ectoparasites on the Bonelli's Eagle.

The hematophagous behavior of lice in birds has been described in several species of Amblycera (Agarwal *et al.* 1983, Saxerna *et al.* 1985). Agarwal *et al.* (1983) describes hematophagy in *Menacanthus eurysternus* Burmeister, 1838 when feeding on the blood of growing feathers, even making injury in the skin of its host.

In diurnal raptors, hematophagous behavior has been indicated for certain species of the genus *Laemobothrion* Nitzsch 1818. This genus currently has four species that parasitize several species of Falconiformes and Accipitriiformes, presenting a worldwide distribution. Three of the four species have been reported in diurnal raptors in Western Europe (Martín-Mateo 2002). All *Laemobothrion* species are large chewing lice and are considered to be primarily keratinophages. Regarding its oral morphology, Pérez *et al.* (1995) cites sharp structures close to the mandible in *L. maximum*, which is related to possible hematophagous behavior. The same authors indicate hematophagous behavior in this species on the Common Buzzard (*Buteo buteo*) (Pérez *et al.* 1994). Other authors have indicated a similar behavior in *Laemobothrion vulturis* Fabricius 1775 in Bearded Vulture (*Gypaetus barbatus*) (Złotorzycka & Danecki 1969, Srivastava 1974). Despite some evidence on the hematophagous behavior of *Laemobothrion* in diurnal raptors, the incidence of this behavior has been poorly studied. For our case study, the record of *L. maximum* is the first of the Bonelli's Eagle in Spain. Furthermore, it is the first documented case of hematophagous behavior in this species of raptor.

Heavy infestation with hematophagous lice has been related in some studies with alteration of fitness and hematological changes in wild and domestic birds. The principal findings that had been observed are related with lower rates of hemoglobin (Hb), total plasma protein, packed cell volume (PCV) or total erythrocytes (Al-Saffar & Al-Mawla

2008, Kloskowski *et al.* 2016, Al-Shaibani *et al.* 2018). In our study, no anomalous values have been found in the Hb value of the specimens with the presence of *L. maximum* with hematophagous behavior. Further studies in raptors are needed to analyze if the presence of those parasites is also related with analytical alterations and affection of the health status of the host.

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First documented case of albinism in the African Chaffinch (*Fringilla spodiogenys*) in Algeria

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Abstract In the wild, it is rare and difficult to see birds with plumage colour abnormalities. In Algeria, despite its vast surface area, sightings of birds with colour aberrations are very rare. The present note concerns the observation of an African Chaffinch albino on 04-10-2022, in Mont Zaccar, Ain Defla (North-West Algeria). In this note, we refer to a bird that has never been observed to be affected by this colour anomaly in this species.

Keywords: Zaccar, Ain Defla, African Chaffinch, albinism, *Fringilla spodiogenys*

Összefoglalás A vadonban csak ritkán fordulnak elő, és nehezen találkozhatunk olyan madarakkal, amelyek tollazata rendellenes színezetű. Algériában, hatalmas területe ellenére, nagyon ritkán lelni színhibás madarakat. Esettanulmányunk egy afrikai pinty albinó egyedének 2022. október 4-én, a Zaccar-hegyen, Ain Deflában (Északnyugat-Algéria) tett megfigyeléséről szól. Tanulmányunkban egy olyan esetet mutatunk be az afrikai pintynél, amit még soha nem figyeltek meg e fajnál.

Kulcsszavak: Zaccar-hegy, Ain Defla, afrikai pinty, albinizmus, *Fringilla spodiogenys*

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Introduction

The Fringillidae is a large family of passerines comprising 240 species in 50 genera (BirdLife International 2024). The genus *Fringilla* includes eight species of finches (Gill *et al.* 2024). Since the early days of ornithology, the chaffinch has been considered as a single species *Fringilla coelebs* with several subspecies, ranging from 10 to 24, depending on the author (Hartert 1912, Gengler 1917, Hartert & Steinbacher 1932). With the advancement of systematics, many authors granted the rank of species to several subspecies (Vaurie 1956, Sudilovskaja 1972, Eck 1975, Krägenow 1981). Subsequently, through genetic studies and analysis of vocal behaviour (Metzmacher 1982, 1995, 2016, Ruelle 1988, Joachim 1996), which have led modern taxonomists to draw up a list of eight species of chaffinches (Gill *et al.* 2024), including one with a North African distribution, *Fringilla spodiogenys*, a sedentary species that generally disperses only over short distances. It is found in North-West Africa, from Morocco in the south to North-West Libya, from the Mediterranean to the Anti-Atlas

in Morocco and the wooded regions of the pre-Saharan Atlas in Algeria (Ottaviani 2008, Mlodinow *et al.* 2024). The African Chaffinch lives in pine forests, particularly the Aleppo pine (*Pinus halepensis*), olive groves (Ruelle 1988) and oak forests, particularly the holm oak (*Quercus ilex*).

The topic of aberrant colouration in birds has been the subject of several studies worldwide (Hanson 1949, Phillips 1954, Fox & Vevers 1960, Warner 1963, Delmotte & Doucet 1981, Bechtel 1991, Thompson *et al.* 1997, Zahn & Rothstein 1999, Brawner *et al.* 2000, van Grouw 2006, van Grouw & De Jong 2009, Frith & Murphy 2012, van Grouw 2012, 2013, Guay *et al.* 2012, Hayashi & Suzuki 2018, Tissier 2020). However, in Algeria, the works that have dealt with this subject are very limited in number and superficial (Chedad *et al.* 2019, Chedad *et al.* 2021, 2022, Haddad 2024), and on the whole, they consist of referring to certain species affected by these colour anomalies without looking deeply into the causes and mechanisms of the effect. This note concerns the subspecies *Fringilla spodiogenys africana*. It is widespread in a variety of environments from the Mediterranean to the northern limits of the Saharan Atlas and from southern Morocco to western Tunisia (Svensson 2015).

Overview of observation and habitat description

During a prospecting trip to the Zaccar mountain, which belongs to the Dahra Range, on 04-10-2022, at around 09:30 on the way to the summit, an albino male African Chaffinch was observed perched on a dying Aleppo pine tree ($36^{\circ}18'26''\text{N}$, $2^{\circ}11'05''\text{E}$), altitude: 765 m (*Figure 1*). The second author immediately took a photo (using Canon EOS 7D) on the individual (*Figure 2a*). This is an albino individual, as the finch has completely lost its

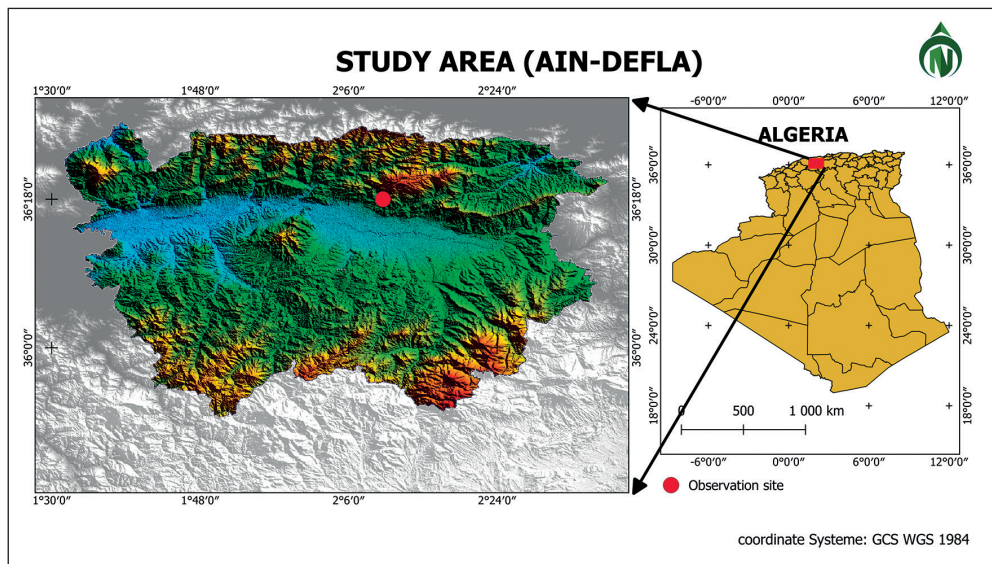


Figure 1. Geolocation of the observation point

1. ábra A megfigyelési pont földrajzi elhelyezkedése

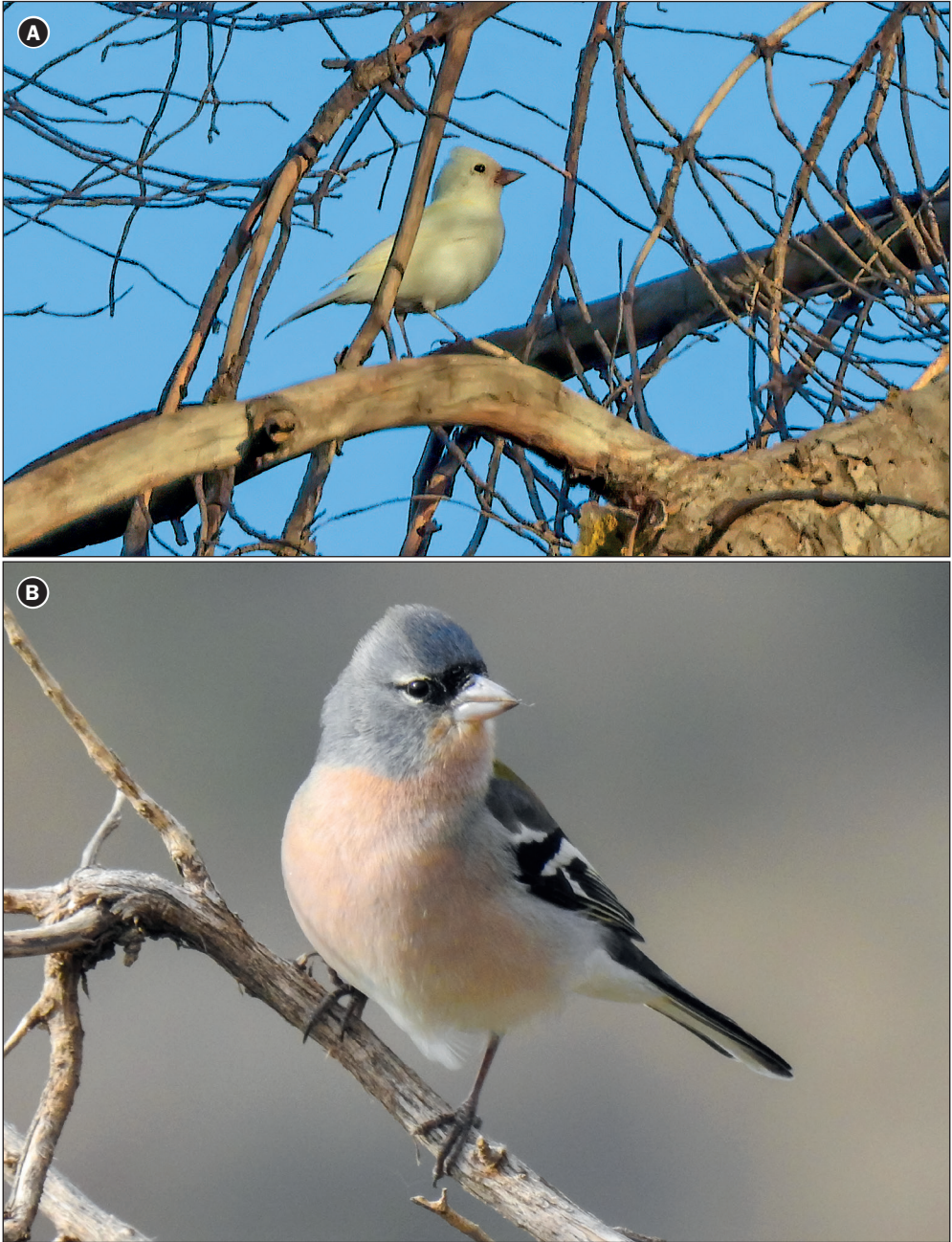


Figure 2. **A:** Albino African Chaffinch in the Zaccar mountains, municipality of Meliana-Ain Defla, Algeria, (04-10-2022). Photo: N.B. **B:** A male of African Chaffinch with natural plumage, municipality of Youssoufia – Tissemsilt, Algeria (13-02-2020). Photo: M.M.

2. ábra **A:** Albinó afrikai pinty a Zaccar-hegységben, Meliana-Ain Defla településen, Algéria (2022. 10. 04.). Fotó: N.B. **B:** természetes tollazatú afrikai pinty hím, Youssoufia – Tissemsilt település, Algéria (2020.02.13.). Fotó: M.M.

colouring and appears entirely white, plus the eyes are red whereas in their natural state they are black, and the beak and legs are devoid of melanin appear whitish-reddish (*Figure 2a*).

The observation zone is located in the north of the province of Ain Defla, approximately 4 km from the town of Meliana and 2 km from the town of Ben Allal (northern Algeria). The area is a forest at the foot of the massif, covered in wild olive, with a few scattered Aleppo pine trees. The soil is of the red fersiallitic type, moderately deep, which explains the high density of vegetation in the area. The province of Ain Defla has a sub-humid climate, with a cold winter (January: 6 °C) and a very hot, humid summer (August: 35 °C). Annual rainfall varies, with an average of around 380 mm/year (Weather Spark 2024).

Discussion

A number of studies have focused on the colour anomalies that occurred in birds in Algeria, notably cases of leucism in *Oenanthe leucopyga* recorded near the Dhayet Oum-Suid naturel wetland, province of Ghardaïa – Saharan Algeria (Chedad *et al.* 2019), albinism in the hybrid sparrow *Passer domesticus* × *P. hispaniolensis* in the region of Ghardaïa (Chedad *et al.* 2021), leucism in *Emberiza sahari* in the city centre of the M'Zab Valley at Ghardaïa (Chedad *et al.* 2022), leucism in *Corvus monedula cirtensis* and *Turdus merula* in the city of Constantine in eastern Algeria (Haddad 2024) and partial leucism in *Lanius excubitor* in Tissemsilt province (Mairif & Hamecha in press). Abroad in Europe, several documented cases of albinos in the European Chaffinch *Fringilla coelebs* were reported by the Association des Naturalistes des Yvelines (2006) in Vésinet, France, the Association Bretagne Vivante (2015) in Finistère, France and Tissier (2020) in Lyon region, France, but for the African Chaffinch, this is the first documented case of this phenomenon in this species in Algeria, including its entire range in North Africa.

These whitish aberrations are classified into different types according to their proximal origin, including leucism, progressive greying, dilution (Guay *et al.* 2012, van Grouw 2021). Albinism resulting from the total absence of melanin pigments not only in feathers, but also in the eyes and skin too, due to the inherited absence of the enzyme tyrosinase in pigment cells leading to defects in melanin synthesis (Fox & Vevers 1960, Hayashi & Suzuki 2018, van Grouw 2021).

The colour and appearance of an albino is determined by the presence or absence of other pigments (e.g. carotenoids/psittacin), so contrary to popular belief, albinos are not necessarily white; they can be creamy or yellowish, golden, pinkish or reddish (van Grouw 2006). There are various causes of the occurrence of abnormal plumage colouration, and many of them, including albinism, can be explained by hereditary abnormalities (Bechtel 1991, van Grouw 2006, 2012, van Grouw & de Jong 2009, Frith & Murphy 2012). Several causes of plumage colour irregularities have non-genetic and congenital origins, such as progressive ageing (Hanson 1949, Warner 1963, Guay *et al.* 2012), diseases like fowl pox (Thompson *et al.* 1997, Zahn & Rothstein 1999), internal and external parasites (Thompson *et al.* 1997, Brawner *et al.* 2000), injuries (Phillips 1954) and nutritional deficiency (van Grouw 2013), and the action of light rays (Delmotte & Doucet 1981).

As a result, individuals with these aberrant colours could have limited probabilities to survive in the wild because they will be more visible, and therefore, more quickly predated (Tissier 2020), unlike natural plumage, which probably allows the animal to blend in with certain elements of the surrounding environment (*Figure 2b*). The same applies to albino predators, which will have less chance of discreetly approaching their prey (Tissier 2020). Some species do not tolerate conspecific individuals that are different, and will be attacked within their group, making it difficult to find a partner to reproduce with (Harris 1983, Withgott & McMahon 1993). In some birds, the white plumage may be a transitory characteristic, specific to a particular phase of their life, and after the subsequent moult, the white feathers are replaced by normal-coloured feathers (Nogueira & Alves 2011).

Conclusion

By reporting this rare observation, we have attempted to shed light on the phenomenon of albinism in the African Chaffinch, which represents a valuable addition for researchers interested in studying of this phenomenon. It is certain that this is not the first case and will not be the last, so in order to increase the chances of encountering these rare species in Algeria, it is necessary to intensify field trips, whether scientific or amateur, and the need to create a coordination network in order to document as many species as possible that are subject to colour aberrations, which is what the Algerian Wildlife Watching Association is seeking to embody.

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