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The status of the European Stonechat *(Saxicola rubicola)* in Hungary: a review

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Abstract In 2021, the European Stonechat *(Saxicola rubicola)* became the 'Bird of The Year' in Hungary, which makes it very timely to summarise our knowledge about the status of the species in the country and to emphasise conservation priorities.

In Hungary, the species is a common, widespread breeder of dry roadside grass strips, abandoned ploughlands, bushy slopes and vineyards. It is most likely in the arid habitats between the Danube and Tisza and east of the Tisza. The largest populations also breed in these regions. The Hungarian population was estimated at 195,000–210,000 pairs between 2014 and 2018. The breeding population halved between 1999 and 2018, though it was stable until 2004. The breeding period begins from late March to early April and lasts until the end of July. First males arrive in February, and spring migration peaks in early March. Autumn migration peaks in late September. Few may overwinter. The number of birds ringed in Hungary since 1951 is 13,484, of which 1,401 were juvenile. Three birds ringed in Hungary were found abroad (Italy 2, Greece 1), and two specimens marked abroad (Croatia, Italy) were found in Hungary. The oldest bird was recaptured 1679 days after its ringing day in Hungary. The average body mass of juveniles increased significantly for both sexes by an average of 0.9 grams over 22 years. In the case of all age and sex groups, the average body mass increased during the autumn season. The average wing length of adults also increased during the autumn. The arrival time of either sex during spring migration did not change significantly between 1999 and 2020. In the case of all age and sex groups, the arrival time shifted later in autumn migration. The European Stonechat belongs to the red list category Near Threatened in Hungary.

Keywords: trend of populations, ringing results, migration, dispersion, conservation

Összefoglalás A cigánycsukot *(Saxicola rubicola)* 2021-ben „Az év madarának" választotta a Magyar Madártani és Természetvédelmi Egyesület. Ezen alkalomból jelen írás összefoglalja a cigánycsuk földrajzi elterjedésére, állománynagyságára, fészkelésére, vonulására, valamint természetvédelmi helyzetére vonatkozó lényeges ismereteket, kitekintve más országok fontosabb kutatási eredményeire is.

Magyarországon száraz út menti gyepsávok, felhagyott szántók, cserjés lejtők és szőlőültetvények gyakori, elterjedt fészkelője. Leggyakrabban a Duna–Tisza közén, valamint a Tiszántúl száraz élőhelyein fordul elő, legnagyobb fészkelő állományai is ezeken a vidékeken találhatók. A hazai populációt 2014–2018 között 195 000–210 000 párra becsülték. A költő madarak száma 1999–2018 között felére csökkent, bár 2004-ig lényegesen nem változott. A költési időszak március végén, április elején kezdődik és július végéig tart. Az első hímek februárban érkeznek, a tavaszi vonulás március elején tetőzik. Az őszi vonulás csúcsidőszaka szeptember vége. Kis számban át is telelhetnek. Magyarországon 1951 óta a gyűrűzött madarak száma 13 484, ebből 1401 fióka. Három Magyarországon gyűrűzött madár került meg külföldön (Olaszország 2, Görögország 1), kettő külföldön jelöltet (Horvátország, Olaszország) pedig Magyarországon fogtak vissza. A legidősebb madár 1679 nappal a gyűrűzés után került ismét kézre. A fiatal madarak átlagos testtömege mindkét nem esetében jelentősen, átlagosan 0,9 grammal nőtt a vizsgált 22 év során. Mindkét kor- és ivarcsoport esetében a madarak átlagos testtömege nőtt az őszi vonulás során. Az adult madarak átlagos szárnyhossza is nőtt az őszi vonulási időszak vége felé. 1999–2020 között a hímek és a tojók érkezési ideje sem változott lényegesen a tavaszi vonulási időszakban. Valamennyi kor- és ivarcsoport érkezési ideje az őszi vonulás során későbbre tolódott. Hazánkban a cigánycsuk a veszélyeztetettséghez közeli vörös listás kategóriába tartozik.

Kulcsszavak: költőpopuláció változása, madárgyűrűzési eredmények, vonulás, diszperzió, természetvédelem

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Introduction

The European Stonechat has been generally considered conspecific with the Siberian Stonechat and African Stonechat, lumped together as Common Stonechat *Saxicola torquatus* (Cramp & Simmons 1988, Collar 2005). Using mtDNA cytochrome *b* sequences and nuclear DNA microsatellite fingerprinting evidence strongly supported the separation of Northern Eurasian taxa of the *S. torquatus* into distinct species (European Stonechat – *S. rubicola*, Siberian Stonechat – *S. maurus* and Stejneger's Stonechat – *S. stejnegeri*) (Urquhart & Bowley 2002, Wink *et al.* 2002, Zink *et al.* 2009, Opaev *et al.* 2018, Gill *et al*. 2021). The *S. rubicola* has two races: *S. r. rubicola* and *S. r. hibernans.* In Hungary, the *S. r. rubicola* race occurs (Hadarics & Zalai 2008).

The European Stonechat was the "Bird of The Year" in Hungary in 2021, which provides us with an exceptional opportunity to summarise the information about the distribution, population size, breeding ecology, migration and the nature conservation status in Hungary. Though this review focuses on the characteristics of the Hungarian population but also summarises the major research results from other parts of Europe as well.

Geographical distribution

The European Stonechat is a widespread species in Europe but less common in the northern part of its range (nearly absents from Fennoscandinavia, the Baltic States, Belorussia, southern and north Russia) (Small 2005, Hornman 2020). The breeding area of the *S. r. rubicola* race is from Northwest Africa, West, Central and South Europe to Southwest Russia and North Turkey, *S. r. hibernans* race is from Ireland, Britain, Northwest France to West Iberia and Southwest Norway (Helm *et al.* 2006, Gill *et al*. 2021). Over the past two decades, its distribution area at its northern border (Denmark, north Poland and Lithuania) has grown markedly (Hornman 2020).

Habitats

European Stonechats are usually found in lowlands, from sea level up to 400–500 m. In smaller numbers, it breeds up to 700–800 m in central and eastern Europe, exceptionally occurring up to 1,850 m (Italian Alps) or 2,230 m (Greece). This species prefers open natural or extensively cultivated areas with perches used as hunting and singing posts. It breeds in heathlands, moorlands, coastal dunes and rough grasslands with scattered small shrubs and bramble, open gorse, tussocks or heather, grassy hillsides, bush-studded pastures, roadsides and railway margins, and vineyards. Also prefers low manufactured structures such as fence lines, stone walls, and electric wires (Lardelli & Molnár 1997, Urquhart & Bowley 2002, Collar 2020).

It is a common nesting species in Hungary. Its most specific breeding sites are dry ditch banks, grass strips along dirt roads, abandoned arable lands with weeds, bushy hillsides, vineyards and afforestation, where there are bushes, paths and stalks protruding from the vegetation (Haraszthy 2019). During the breeding season, rarely, during migration, it occurs more often in wetlands (Gyurácz & Csörgő 2021).

Country-wide standardised data collection in the frame of the Hungarian Bird Atlas project (Szép *et al.* 2021) during 2014−2018 with detailed modelling of the probability of occurrence, relative density and its changing with environmental data showed that nesting is not expected only in the middle mountains covered with contiguous forest and in the area of larger cities (Gyurácz & Csörgő 2021). It is most likely in the arid habitats between the Danube and Tisza and east of the Tisza *(Figure 1)*. Based on the models, its occurrence is

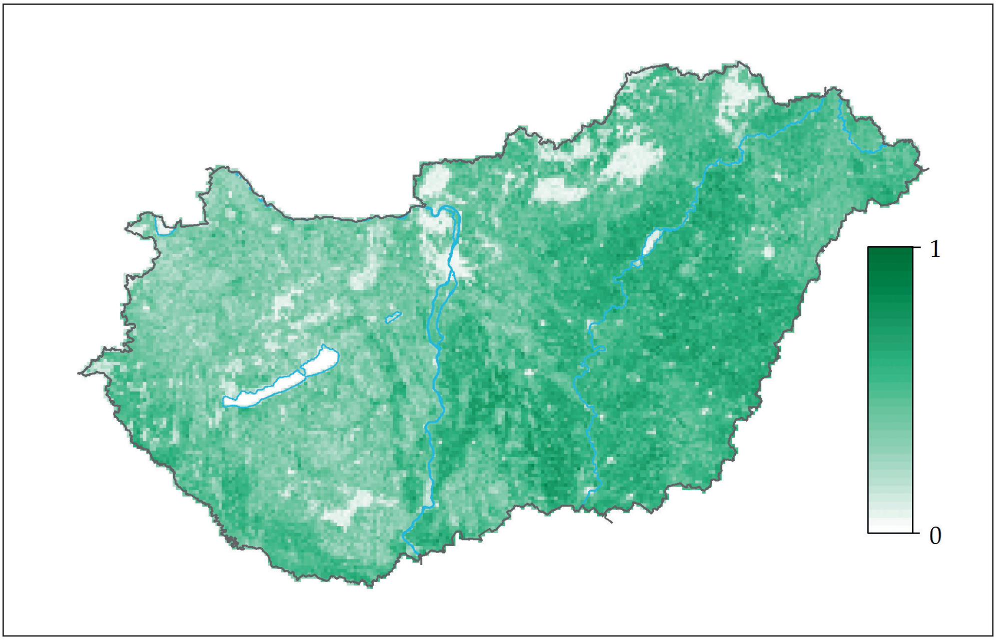


Figure 1. Probability of occurrence in the breeding season in Hungary (Gyurácz & Csörgő 2021)
1. ábra Előfordulási valószínűség a fészkelési időszakban (Gyurácz & Csörgő 2021)

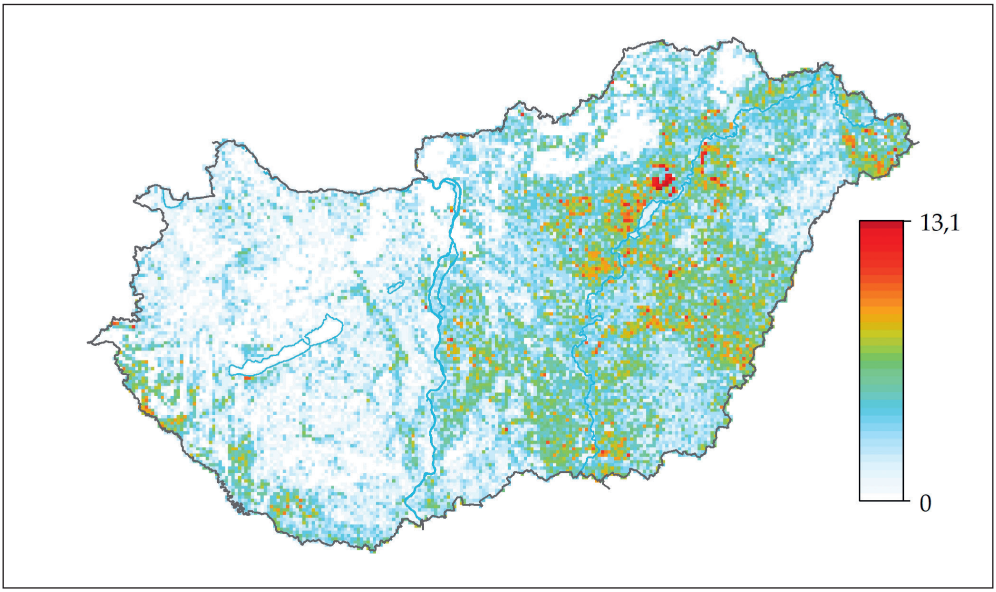


Figure 2. Predicted relative density in the breeding season in Hungary (Gyurácz & Csörgő 2021)
2. ábra Relatív egyedsűrűség a fészkelési időszakban (Gyurácz & Csörgő 2021)

positively correlated to arable lands, a kind of herbaceous vegetation, wetlands rich in soft-stemmed plants, alkaline grasslands, closed grasslands, and dirt roads (Gyurácz & Csörgő 2021). It is significantly less likely to occur in habitats above 200 metres above sea level (Gyurácz & Csörgő 2021).

Based on the modelling of the density of the breeding population, its largest populations are in the northern part of the Danube-Tisza area and east of the Tisza region (Gyurácz & Csörgő 2021) *(Figure 2)*. Its occurrence is positively influenced by paved roads, especially ditches and weeds along public roads (Gyurácz & Csörgő 2021).

Breeding

European Stonechats first breed when they are one year old. Monogamous during the breeding season but do not pair for life (Cramp & Simmons 1988). Interesting that male-female pairs defend territories also in winter quarters, the paired birds change partners regularly and pairs mostly split before leaving the area in spring (Gwinner *et al*. 1994, Rödl 1994, 1999).

Birds adapt flexibly to nesting conditions depending on their condition and remain long when conditions are right (Flinks *et al.* 2008). In Hungary, the first ones arrive in late February, mid-March usually occupies nesting sites, and nest building starts in late March, or early April. Egg-laying of the first clutch is from early April (Haraszthy 2019). Based on the reported observations of probable nesting in the frame of the Hungarian Bird Atlas project, the breeding period begins in mid-March and lasts until the end of July *(Figure 3)* (Gyurácz & Csörgő 2021).

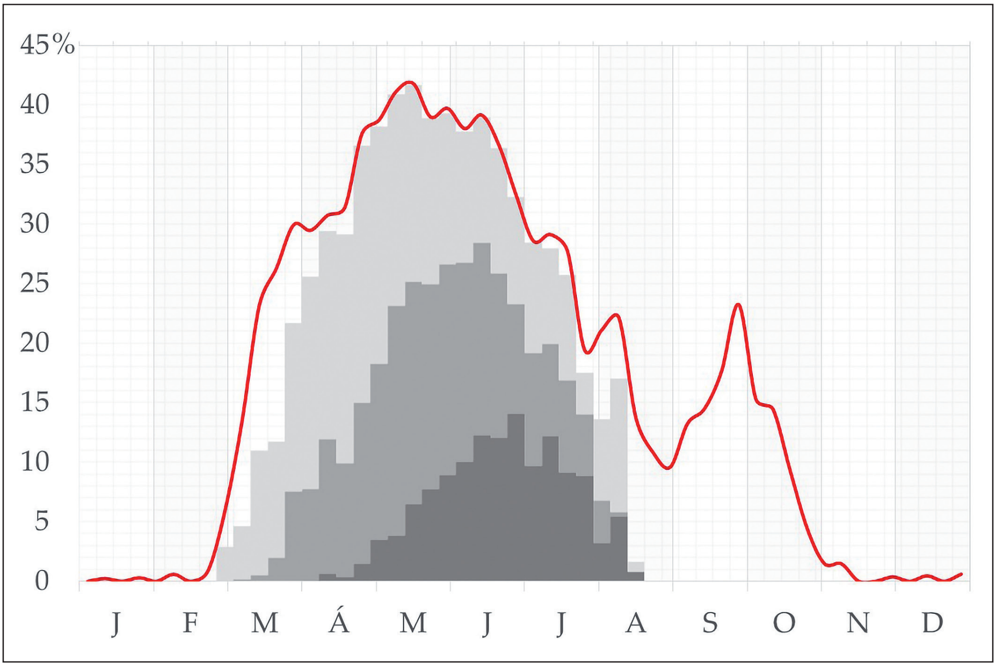


Figure 3. Frequency of observation probability (red line) and breeding evidence (possible – light grey, probable – grey, confirmed – dark grey) (Gyurácz & Csörgő 2021)

3. ábra Észlelési (piros vonal) és fészkelési valószínűség (világos szürke – lehetséges fészkelés, szürke – valószínű fészkelés, sötétszürke – biztos fészkelés) (Gyurácz & Csörgő 2021)

The incubation period is 13−14 days, the mean fledgling period is 13.5 days. Incubation is done by females only, but fledglings are cared for and fed by both parents (Cramp 1998).

According to Raess (2006), European Stonechats lay up to three clutches per season. In Hungary, it usually breeds twice a year, but rarely can third and additional breeding occur (Haraszthy 2019). It has a longer breeding season with the potential for rearing three broods compared to the long-distance migrant Whinchat *(S. rubetra)*. The greater productivity of the European Stonechat might compensate for higher winter mortality (Fuller & Glue 1977).

European Stonechats from Austria usually lay 5.0 to 5.6 eggs per clutch, and the estimated means were 5.09 eggs (Gwinner *et al.* 1995).

In Hungarian egg collections from the early and mid-20th century, clutches consist of 4 (4 cases), 5 (44), 6 (53) or 7 (3) eggs (Haraszthy 2019). Molnár (1986) found 5.4 eggs in 44 nests built on sloping canal banks, 5.4 eggs in 33 nests built on flat areas and 4.3 eggs in holes. The hatching rates were 73, 69, and 51% in the different nest types.

Clutch size decreases from first to second clutches in Slovakia. The clutch size of the third clutches increases again, indicating that only high-quality parents initiate a third seasonal breeding attempt or that a strategy of terminal investment is involved (Raess 2006). A Slovakian breeding pair produces about seven fledglings per season (Raess 2006).

Breeding population

The European population is 5,800,000–9,300,000 pairs (BirdLife International 2021).

In Hungary, Farkas (1958) mentioned it as a widespread but uncommon breeder, and according to Keve (1960, 1984), it was a quite common breeding bird in dry, bushy areas nationwide. The Hungarian population was estimated to be 200,000–400,000 pairs in the 1990s (Magyar *et al.* 1998), 390,000–515,000 pairs between 1999 and 2002 (Hadarics & Zalai 2008), 360,000–434,000 between 2000 and 2012 (Magyar Madártani és Természetvédelmi Egyesület – BirdLife Hungary 2019), 195,000–210,000 pairs between 2014 and 2018, based on the relative population density model (Gyurácz & Csörgő 2021). Modelled relative density is up to 13.1 individuals/km^2 (Gyurácz & Csörgő 2021).

Regional checklists from the last two decades also mention it as a common breeder. It was a common breeder near roads, on pastures, bare hillsides and weedy ditchshores in Nógrád County (northern Hungary) (Drexler 1997). It is a sporadic breeder on grasslands with scattered trees, weedy ditchshores and uncultivated arable lands in Vas County (western Hungary) (Gyurácz & Kóta 2020). A widespread breeder at the Hortobágy (eastern Hungary) and on the surrounding agricultural fields, the maximum density is 0.04–0.07 pairs/ha on abandoned paddy fields (Ecsedi & Kovács 2004).

Modelled relative density in Hungary is lower than the density found in optimal habitats on waste grounds (150–250 pairs/ha) in the Netherlands, 33 pairs/km^2 in Spain (Asturias), 31.5 pairs/km^2 on coastal cliffs in Britain, 8–12 pairs/km^2 in Rheinland grassland in Germany, or 2.8–9.6 pairs/km^2 on inland heathland in Britain (Collar 2020).

Population trends

The European population trend between 1989 and 2013 was stable (EBCC 2015). The EU27 population decreased by no more than 25% in 10 years (BirdLife International 2021), but for example, in Germany, the species has increased over the last two decades (Bairlein *et al.* 2014). The Pan-European Common Bird Monitoring Scheme reported a stable population between 1989 and 2017 (PECBMS 2021).

The population of the UK is probably fluctuating, with no long-term trend. Despite several former declines and range contractions, the population seems to be recovered (Woodward *et al*. 2020).

On the basis of the Hungarian Common Bird Monitoring (MMM, Szép *et al.* 2012), the Hungarian nesting population has a significant decreasing trend between 1999 and 2021 (-52.1%, CI = -59.2% – -43.8%, $P < 0.01$) *(Figure 4)*. It was still relatively stable until 2004, after which it gradually decreased until 2013, by more than 50% compared to 1999. Since 2013, the population index fluctuate and did not show a marked decline *(Figure 4)*. This trend was mainly a characteristic of Transdanubia, the North Central Mountains, and most Danube-Tisza areas. In most of the Trans-Tisza region, in Jászság and Borsodi-Mezőség, the number of nesting pairs was stable or slightly increasing (Gyurácz & Csörgő 2021) *(Figure 5)*.

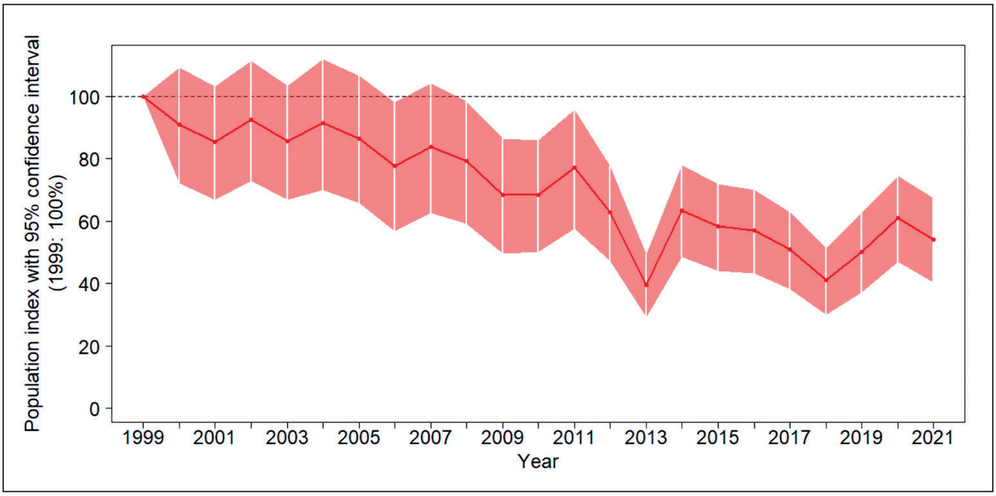


Figure 4. Population trend in the breeding season between 1999 and 2021 on the base of MMM data
4 ábra Állományváltozás a költési időszakban 1999–2021 között az MMM adatok alapján

Regional data are available from the Hortobágy region, where the species was only a passage migrant formerly. First breeding was found in 1976. Since 1984, the species gained ground in the national park area and became a common breeder by the late 1990s. According to the authors, the weeding of pastures due to decreasing grazing intensity helped spread the species countrywide (Ecsedi & Kovács 2004).

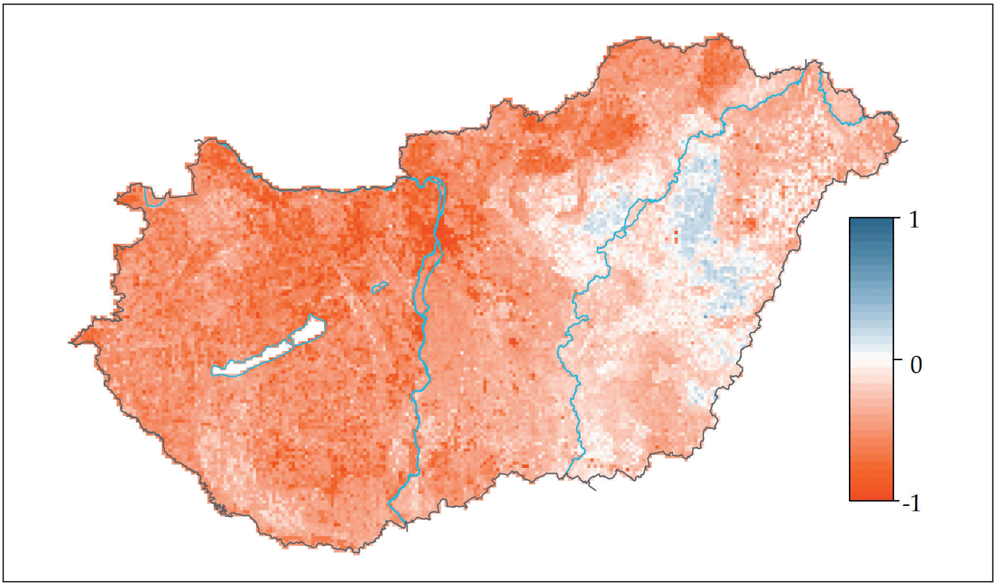


Figure 5. Trend index map in the breeding season in Hungary for the 1999–2018 period (Gyurácz & Csörgő 2021)
5. ábra Az állományváltozási index térképe a költési időszakban az 1999–2018 időszakban (Gyurácz & Csörgő 2021)

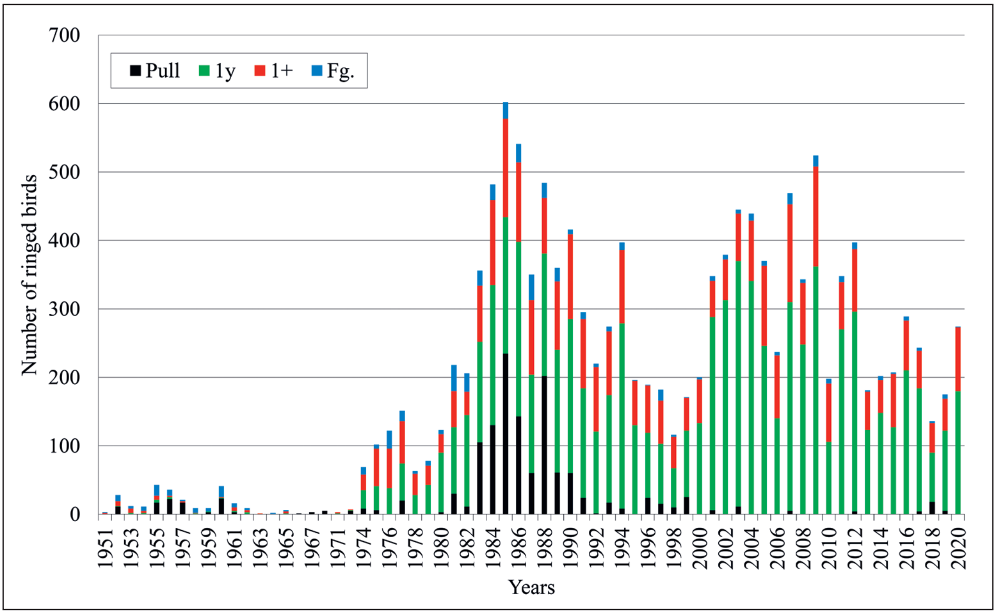
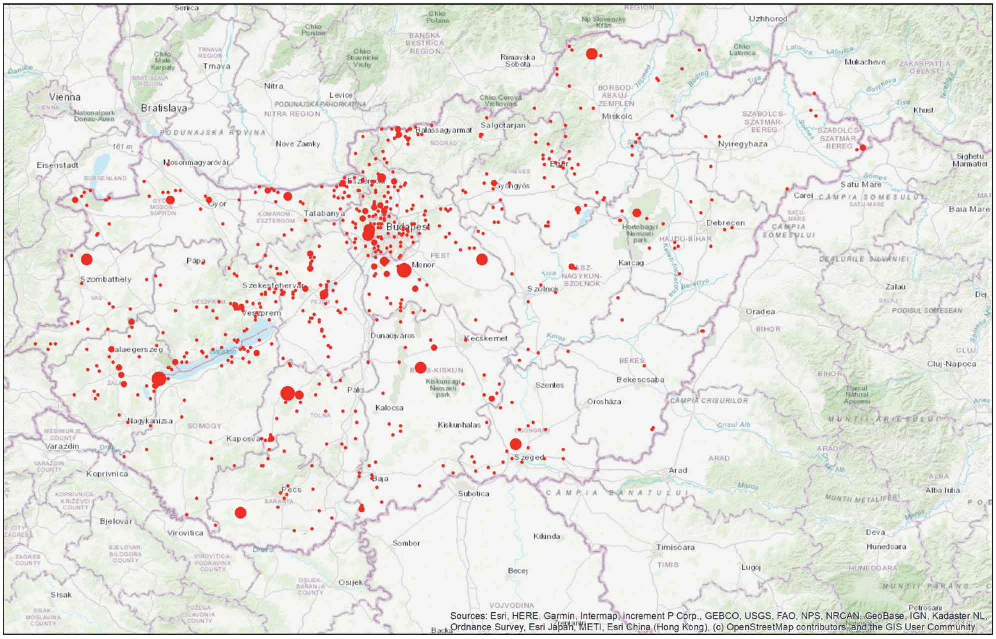


Figure 6. The number of European Stonechats ringed annually in Hungary

6. ábra Az évente gyűrűzött cigánycsukok száma Magyarországon



Sources: Esri, HERE, Garmin, Intermap, increment P Corp., GEBCO, USGS, FAO, NPS, NRCAN, GeoBase, IGN, Kadaster NL, Ordnance Survey, Esri Japan, METI, Esri China (Hong Kong), (c) OpenStreetMap contributors, and the GIS User Community

Figure 7. The locality of ringing sites of European Stonechats in Hungary (the symbols are proportional to the number of ringed birds at that location)

7. ábra A cigánycsuk gyűrűzési helyei Magyarországon (a szimbólumok arányosak az adott helyen jelölt madarak számával)

The population decline was smaller in areas characterised by high relative density. The larger population decrease occurred mainly in the areas where the sand content of the soil is higher (e.g. Kiskunság) and in the hills and mountains (Gyurácz & Csörgő 2021) *(Figure 5)*.

Hungarian ringing results

The intensive ringing of the species began at the time of the formation of the MME (BirdLife Hungary) – and within that with the establishment of the ringing department. It peaked in the mid-1980s, reaching a low point in the late 1990s. It was followed by another wave, which peaked in the mid-2000s, followed by another decline *(Figure 4, 6)*.

There have been ringing all over the country. Most birds were ringed in the Actio Hungarica (network of bird ringing stations) sites *(Figure 7)*.

The number of birds ringed in Hungary between 1951 and 2021 is 13,484, of which 1,401 were juvenile. Of these, 655 specimens (4.86%) were re-found in 826 cases at the place of marking or within 5 km (0–5 km). From a greater distance (> 5 km), only 15 birds (0.11%) with 17 cases were recaptured. Despite the small number of recoveries, it has high site fidelity (databank of BirdLife Hungary Bird Ringing Centre).

Three birds ringed in Hungary were found abroad (Italy 2, Greece 1), and two specimens marked abroad (Croatia, Italy) were recaptured in Hungary. The maximum distance between the ringing and the recapture sites is 1,125 km (Hungary-Italy) *(Figure 8)*.

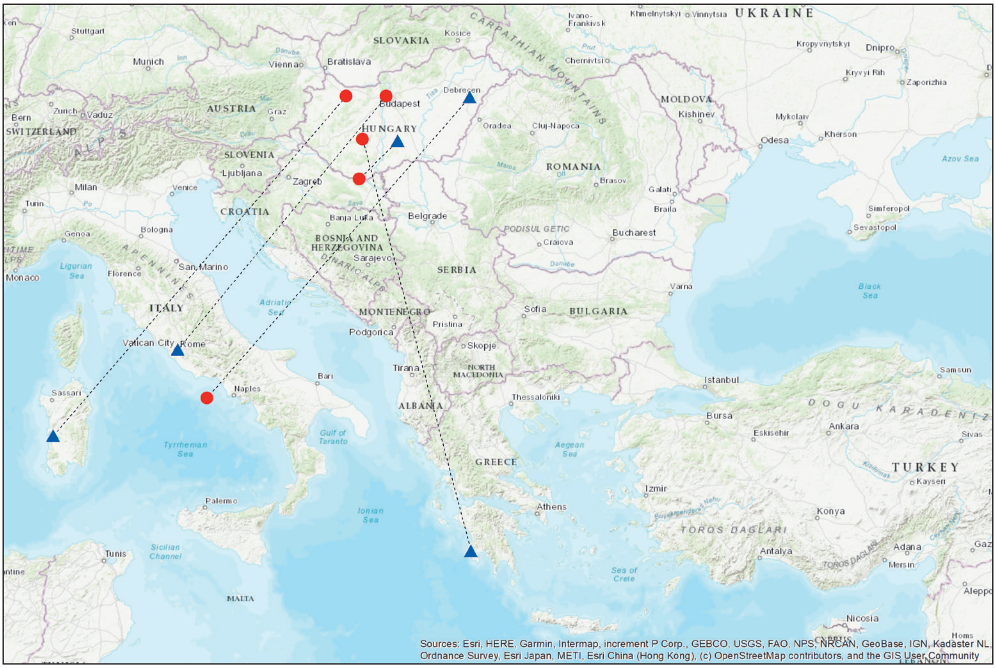

Sources: Esri, HERE, Garmin, Intermap, increment P Corp., GEBCO, USGS, FAO, NPS, NRCAN, GeoBase, IGN, Kadaster NL, Ordnance Survey, Esri Japan, METI, Esri China (Hong Kong), (c) OpenStreetMap contributors, and the GIS User Community

Figure 8. Foreign recoveries related to Hungary of European Stonechats (red – ringing data, blue – recovery data)

8. ábra A cigánycsuk külföldi vonatkozású visszafogásai (piros: gyűrűzési adat, kék: megkerülési adat)

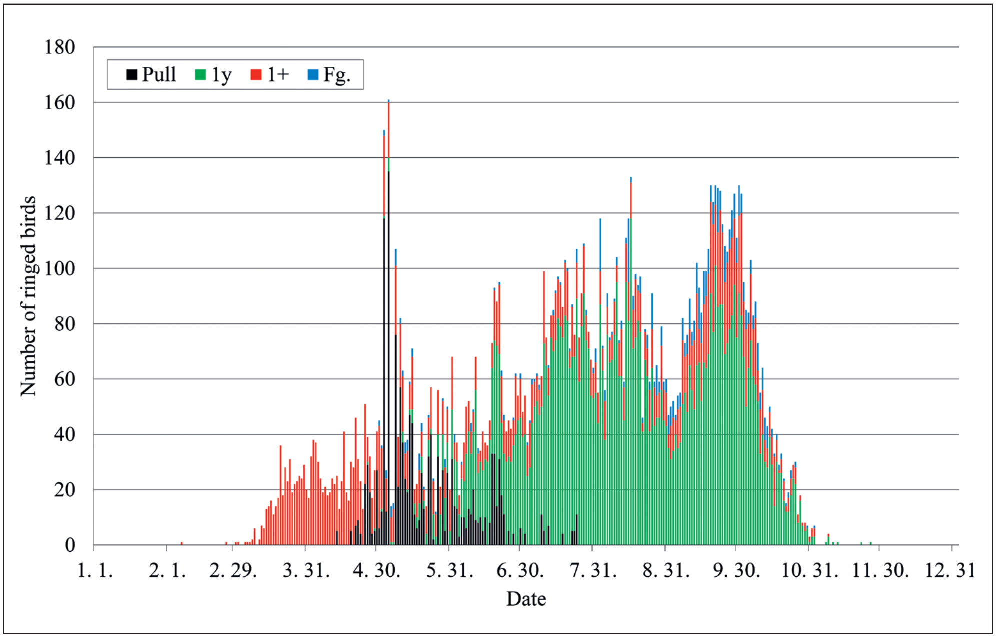


Figure 9. Daily captures of European Stonechats
9. ábra A cigánycsuk napi fogási mintázata

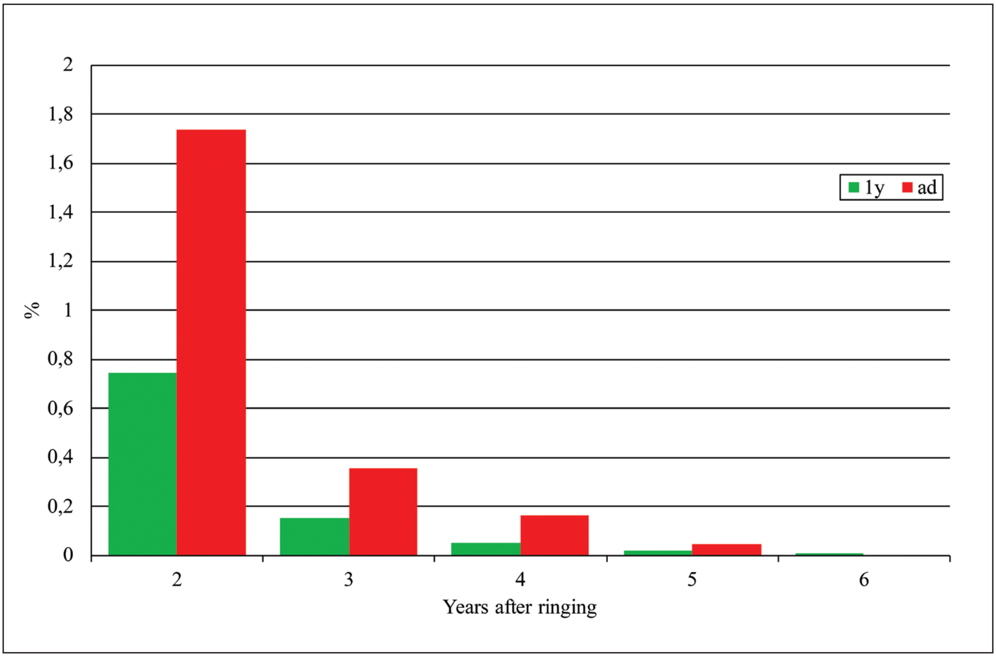


Figure 10. Recapture rate of European Stonechats in the year following their capture
10. ábra A visszafogott cigánycsukok aránya a gyűrűzést követő években

The catch pattern outlines the spring migration, dispersion, and autumn migration periods *(Figure 9)*.

According to return rates of banded birds, adult European Stonechats in Western Europe show annual mortality rates of 64% and 69% at most (Glutz von Blotzheim & Bauer 1988). In Western Europe, the overall proportion of 0.56% of birds was recovered (746 of 135,599). Recovery was lowest in Germany (0.42%) and highest in the Netherlands (1.30%) (Helm *et al.* 2006). In Hungary, 0.75% of the first-year birds (1y) were recaptured in the year following their capture, and the recapture rate declined rapidly in subsequent years. The recapture rate of adults (1+) in the year following their capture is almost two and a half times (1.74%) that of juveniles (1y), but it is also declining sharply from the second year after their capture *(Figure 10)*.

The oldest (8 years 10 months) European ringed bird was found dead in Germany (Fransson *et al.* 2017). Hungary's most prolonged period between ringing and recapture was 1679 days (4 years 7 months 6 days). The specimen was ringed as a first-year (1y) bird.

Morphological measurements

Wing length means (mm) of *S. r. rubicola* race in European countries, Netherland, Belgium, Central France and northern Italy: male 66.4±1.31 (64.0–68.0), female 65.4±1.33 (63.0–68.0), northwestern Africa: male 67.7±1.47 (66.0–70.0), female 67.2±1.20 (65.0–68.0), southern Balkan and Turkey: male 64.8±1.44 (63.0–69.0), female 63.9±1.24 (62.0–66.0) (Cramp 1998), Ukraine: 65.3±1.2 (59.0–68.0) (Opaev *et al.* 2018), Hungary: male 66.1±1.6, (1y male 66.14±1.55, 1+ male 65.94±1.86), female 64.9±1.6 (1y female 65.00±1.54, 1+ female 64.74±1.70).

In northwestern Germany between 1990 and 2012, the wing lengths were increasing and tail lengths mostly decreased (Salewski *et al.* 2014).

The biometric data was collected in Hungary in different locations from 1999 to 2020. We used the records of 2,745 European Stonechats (76 males and 89 females in spring, 461 adult males and 237 adult females, 1,086 juvenile males and 557 juvenile females in autumn). We present data for spring, breeding and autumn migratory seasons separately; birds caught after the 60th and before the 100th day of the year were considered to be spring migrants and birds caught after the 230th and before the 310th day

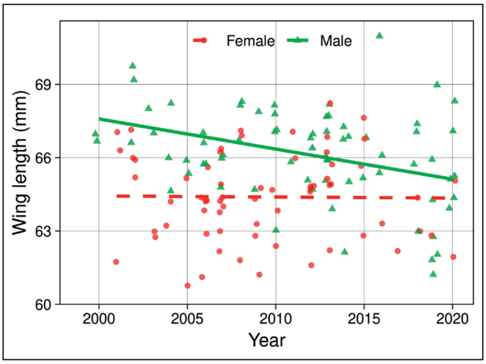


Figure 11. Wing length of male and female European Stonechats during the spring season between 1999 and 2020. The continuous line highlights the significant decrease, while the dashed line shows the non-significant change of average wing length over the years

11. ábra A hím és tojó cigánycsuk egyedek szárnyhossza a tavaszi időszakban 1999–2020 között. A folytonos vonal a szignifikáns átlagos szárnyhossz csökkenést, a szaggatott a nem szignifikáns változást mutatja az évek során of the year were considered to be autumn migrants. The time interval of the spring and autumn migration waves was determined on the basis of the daily catch numbers. First calendar year birds (juveniles) were distinguished from adults (Svensson 1992). All birds were measured according to the same methodology. The wing length was measured with 1 mm accuracy, using a ruler in the case of birds where feather abrasion was low. The body mass was measured with 0.1 g accuracy. Linear mixed-effects (LME) models were used to detect the changes in the timing of migration and morphology. Sex, age and their interaction were included in the models. We put the year as a random factor in all models. We gave 95% confidence intervals (CI) for the estimated changes based on the models. We used the 'nlme version 3.1-152' R package (Pinherio *et al.* 2021). We added a small amount of noise to the figures' data to prevent overplotting. We set the significance level to 0.05. All statistical analyses were done in R 4.1.1 (R Core Team 2021).

While we found a 2.6 (95% CI: 0.7–4.4) mm average decrease in the wing length of males during spring migration (P = 0.0067), the wing length of females did not change significantly (P = 0.7214) *(Figure 11).*

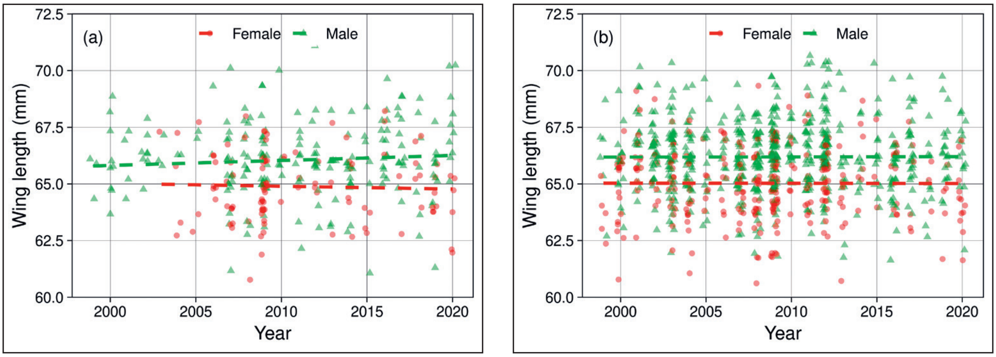


Figure 12a and b Wing length of adult (a) and juvenile (b) male and female European Stonechats between 1999 and 2020 in the autumn season. The dashed lines highlight the non-significant change of average wing length over the years

12a és b ábra Az öreg (a) és fiatal (b) hím és tojó szárnyhosszak az őszi vonulási időszakban 1999–2020 között. A szaggatott vonalak mutatják, hogy a változás nem szignifikáns az átlagos szárnyhossz esetén az évek során

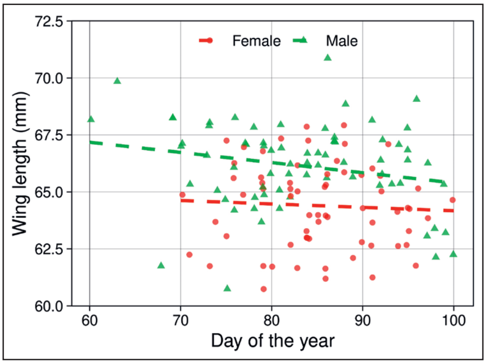


Figure 13. Wing length of male and female European Stonechats during the spring season (Day 1: 1st of January). The continuous line highlights the significant, and the dashed line highlights the non-significant decrease in average wing length over season

13. ábra A hím és tojó cigánycsuk egyedek szárnyhossza tavaszi időszakban. A folytonos vonal a szignifikáns, a szaggatott a nem szignifikáns átlagos szárnyhossz csökkenést mutatja a szezon során

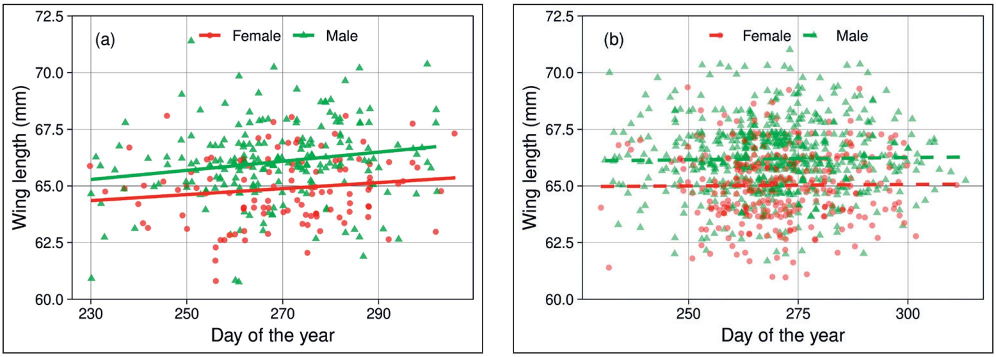


Figure 14a and b Wing length of adult (a) and juvenile (b) male and female European Stonechats during the autumn season (Day 1: 1st of January). The continuous lines highlights the significant, the dashed lines the non-significant increase of average wing length over season

14a és b ábra Az öreg (a) és fiatal (b) hím és tojó szárnyhosszak az őszi vonulási időszakban. A folytonos vonalak a szignifikáns, a szaggatott vonalak a nem szignifikáns átlagos szárnyhossz növekedést mutatják a szezon során

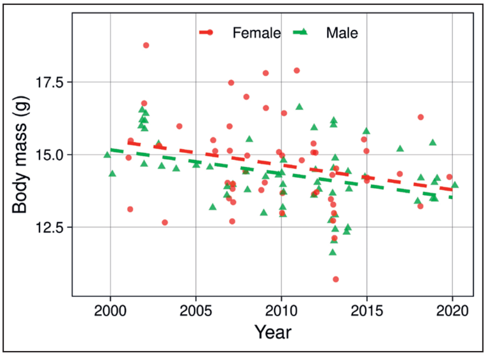


Figure 15. Body mass of male and female European Stonechats during the spring season between 1999 and 2020. The dashed lines highlight the non-significant decrease of over the years

15. ábra A hím és tojó cigánycsuk egyedek testtömege a tavaszi időszakban 1999–2020 között. A szaggatott vonalak a nem szignifikáns változást mutatják az évek során

The autumn wing length did not change significantly (P = 0.2681) during the 22 years in any age and sex group. The males have longer wings on average. The mean difference is 1.2 (95% CI: 1–1.4) mm *(Figure 12)*.

In spring, no significant changes in wing length were observed in any of the sexes (males: P = 0.1139, females: P = 0.6355) *(Figure 13)*.

In the case of adults, the average wing length increased by 1.6 (95% CI: 0.6–2.6) mm during the autumn (P = 0.0019). In the case of juveniles, the wing length did not change on average (P = 0.6879) *(Figure 14)*.

Body mass means of *S. r. rubicola* race in European countries varies between 13.7–16.5g (Cramp & Simmons 1988). In Hungary the mean body mass of males 14.4±1.3 g (n = 1172) and females 14.4±1.4 g (n = 591).

In spring, the body mass did not change significantly (P = 0.0816). There was no significant (P = 0.1048) difference between the sexes *(Figure 15)*.

There were no significant changes in the body mass of adults during the autumn season during the years. There is also a non-significant difference between the age groups.

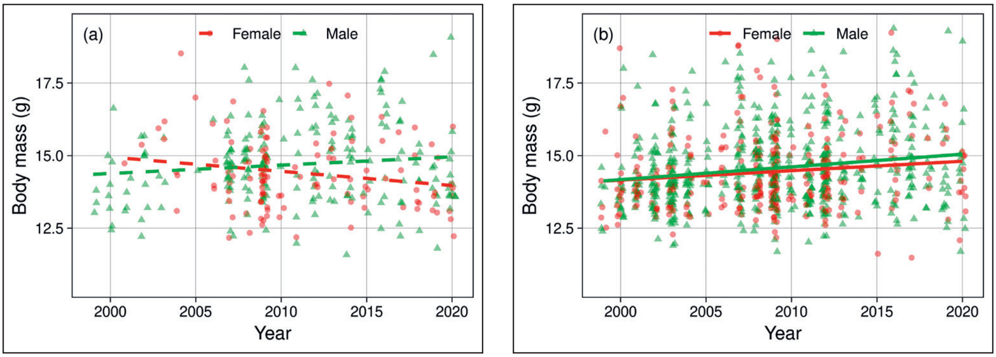


Figure 16a and b Body mass of male and female adult (a) and juvenile (b) European Stonechats during the autumn season between 1999 and 2020. The solid lines highlight the significant increase, the dashed lines the non-significant change over the years

16a és b ábra Az öreg (a) és fiatal (b) hím és tojó cigánycsuk egyedek testtömege az őszi időszakban 1999– 2020 között. A folytonos vonalak a szignifikáns növekedést, a szaggatott vonalak pedig a nem szignifikáns trendet mutatják az évek során

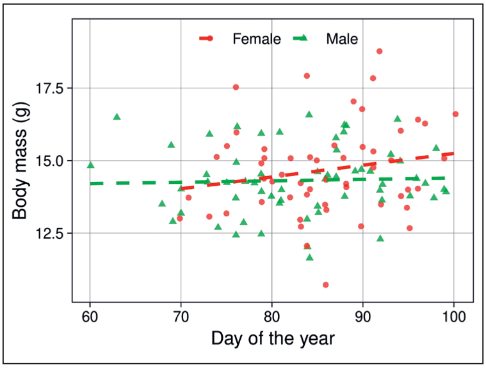


Figure 17. Body mass of males and females during the spring season. The dashed lines highlight the non-significant changes in average body mass over season

17. ábra A cigánycsuk hím és tojó egyedek testtömege a tavaszi időszakban. A szaggatott vonalak a nem szignifikáns átlagos testtömeg változást mutatják a szezon során

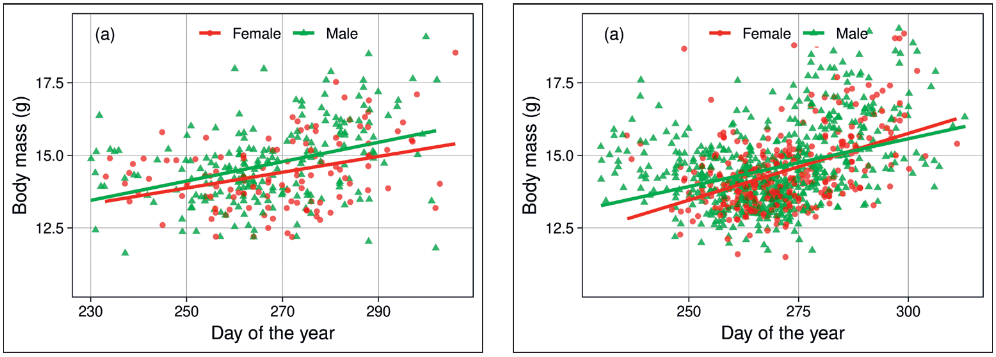


Figure 18a and b Body mass of male and female adult (a) and juvenile (b) European Stonechats during the autumn season. The solid lines highlight the significant increase during the season

18a és b ábra Az öreg (a) és fiatal (b) hím és tojó cigánycsuk egyedek testtömege az őszi időszakban. A folytonos vonalak a szignifikáns növekedést mutatják a szezon során

The average body mass in the juvenile group increased significantly for both sexes (P = 0.001) by an average of 0.9 (95% CI: 0.4–1.3) grams over 22 years *(Figure 16)*.

There was no significant change in the average body mass during the spring season (P = 0.3967) *(Figure 17)*.

In the case of all age and sex groups, the average body mass increased during the autumn season (P < 0.0001). In the case of adult females: by 2.5 (95% CI: 1.3–3.7 grams, in the case of adult males: by 3.2 (95% CI: 2.2–4.1) grams, in the case of juvenile females: 4.1 (95% CI: 3.2–5.1) grams, in the case of juvenile males: by 3 (95% CI: 2.4–3.6) grams *(Figure 18)*.

Timing of migration

In northwestern Germany, based on first and last sightings, males tended to arrive slightly earlier (5 days) and depart slightly later (4 days) than females between 1991 and 2005 (Flinks *et al.* 2008).

In Hungary, the timing of either sex during spring migration did not change significantly (P = 0.56), and the difference in timing between the sexes was also not significant (P = 0.1392) *(Figure 19)*.

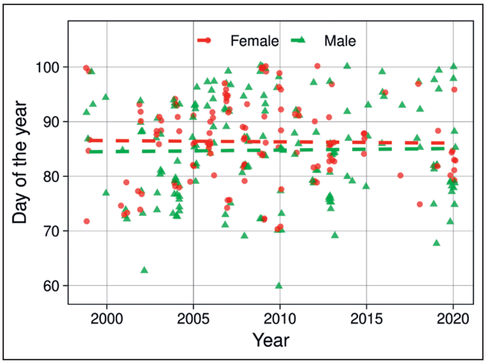


Figure 19. Spring arrival times of male and female European Stonechats between 1999 and 2020. The dashed lines highlight that the change was not significant over the years

19. ábra A cigánycsuk hím és tojó egyedek tavaszi érkezési ideje 1999–2020 között. A szaggatott vonalak a nem szignifikáns trendeket mutatják

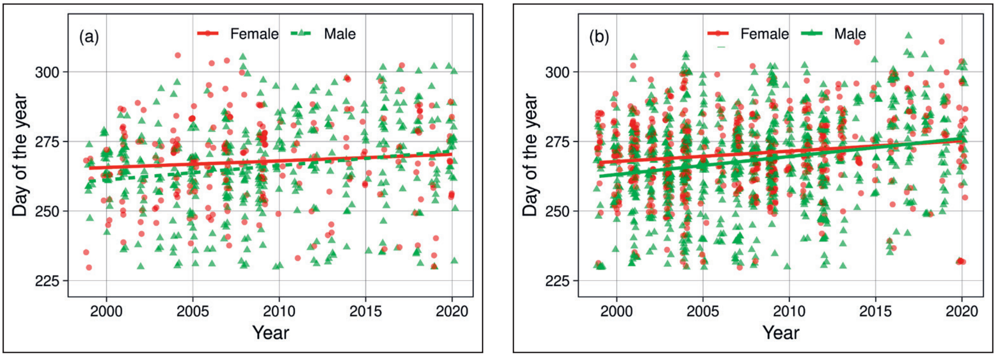


Figure 20a and b Autumn arrival times of adult (a) and juvenile (b) male and female European Stonechats between 1999 and 2020. The solid lines highlight the significant delay in timing over the years

20a és b ábra Az öreg (a) és fiatal (b) hím és tojó cigánycsuk egyedek őszi érkezési ideje 1999–2020 között. A folytonos vonalak a szignifikáns későbbre tolódást mutatják

The shift between age groups during the autumn migration is significant, juveniles migrate on average 3.2 (95% CI: 1.8–4.6) days later ($P < 0.0001$).

In the case of females in both age groups, the migration shifted on average 8.1 (95% CI: 3.3–13) days later ($P = 0.001$). In the case of males, the migration shifted on average 13.8 (95% CI: 9.9–17.8) days later in both age groups ($P < 0.0001$). The change was significantly ($P = 0.0265$) greater in males *(Figure 20).*

Migration and dispersion

European Stonechats are predominantly migratory, but partial migration and summer movements seem surprisingly plastic. Mediterranean populations are apparently resident. Breeders of the northernmost parts of Europe winter primarily in the British Isles, the Mediterranean and oases of the Sahara (Cramp & Simmons 1988, Urquhart & Bowley 2002, Small 2005). Benelux Stonechats were fully migratory, just as their German conspecifics. British Stonechats were partially migrants; according to conservative estimates, almost half (42.1%) of the British Stonechats migrated. Stonechats belonging to the subspecies *S. r. rubicola* are considered resident in South Europe and obligate short-distance migrants in central and East Europe, the *S. r. hibernans* is thought to be a partial migrant. They were classified as 'basically sedentary' in Britain (Thomson 1956, Van Hecke 1965b, Johnson 1971, Lack 1986), although regular movements to North Africa were observed (Zink 1973). During winter, the range of both subspecies extends southwards as far as the desert fringes in North Africa (Callion 2002). Northern birds often move to mild, predominantly coastal regions, resulting in graded winter densities (Helm *et al.* 2006). The European Stonechats from East Europe are short-distance migrants that travel about 1,500 km. Ringing recovery data suggests that many of them pass the Mediterranean Sea on their way (Raess 2006). This species is an obligate short-distance migrant in Central Europe, including Hungary (Flink *et al.* 2008, Magyar 2009).

Directions of Benelux and German Stonechats were mostly oriented towards southwest and northeast, those of British Stonechats towards southeast and northwest. The vector of British migrants was outside the confidence limits of Benelux and German populations, which in turn oriented in almost identical directions (British 158.6°, Benelux 204.6°, German 203.6°. Central and East European Stonechats moved in southwesterly directions, except for a Hungarian nestling captured in Greece in November 1987. For a closer analysis of North African winter quarters, sample sizes were small, suggesting more easterly wintering of East European Stonechats (Helm *et al.* 2006).

In Italy, the peak in ringing numbers is recorded in March (Spina & Volponi 2009). In Northeast Ukraine (Kharkiv Region), the mean date of the arrival of the Stonechat between 1993 and 2008 was 27th March (13th March – 6th April; the median was 31st March). Though most early spring records were of presumably lone males (60%), 38% of the first records in spring were those of already paired birds, and the difference between arrival dates of both sexes was non-significant. Apparent protandry was not found. The mean interval between the first sightings of male and female Stonechats was circa two days (Banik 2019). Studies of Zugunruhe – the 'migratory restlessness' behaviour of captive birds – protandry in spring, although Stonechats winter in heterosexual pairs (Van Doren *et al.* 2017). The first males in Hungary return to their breeding grounds in February. The peak of the spring migration is the first half of March (Molnár 1986, Magyar 2009).

In Germany, the autumn migration is most pronounced from late August to late October and peaks between mid-September and mid-October (Bairlein *et al.* 2014). In the Czech Republic, the departure and autumn migration occur from the second half of September till late October; however, single birds may be observed as late as November and in the first half of December (Pudil & Jelinek 2008). In Italy, the peak in ringing numbers is recorded in September-October (Spina & Volponi 2009). In Malta, migrants arrived in October and November, with about a quarter staying for the winter (Helm *et al.* 2006).

In Hungary, the autumn migration begins at the end of August, the peak period is the last week of September. In October, the majority of the birds leave the country, but some specimens may overwinter in mild winters (Magyar 2009, Gyurácz & Csörgő 2021).

The Mediterranean is an important wintering area for the species (Spina & Volponi 2009). Birds of known origin from all parts of Europe were recorded in North African winter quarters. Local overwintering was only observed in Britain, France, Spain and Italy (Helm *et al.* 2006). Benelux Stonechats were clearly migratory. The origin of wintering birds thus remains obscure (Helm *et al.* 2006). The number of birds overwintering in Belgium is significantly correlated with temperature in the previous winter (Dhondt 1983). The number of wintering birds is increased in Germany (Bairlein *et al.* 2014).

From the birds ringed in Germany, most birds spend the winter in Algeria and Morocco; some may stay in Spain or southern France and more and more are overwintering in Germany (Bairlein *et al.* 2014). From the Czech Republic, the birds have the southwestern direction of migration to wintering grounds in northern Algeria and Tunisia. Single birds are sometimes found in the Czech Republic during winter, mainly in southwestern Slovakia (Pudil & Jelinek 2008).

On average, 27% of the Stonechats banded in one season in Israel returned the following winter (in four seasons) (Rödl 1999), but only 1.95% of c.1,850 ringed birds on Malta for at least one (up to four) additional migration periods. Adults are dominated among returning migrants (Helm *et al.* 2006).

In Hungary, it may overwinter in small numbers; there are few observations from the winter months each year (birding.hu 2022). Observations became regular in the second half of February. The peak of spring migration is the first half of March. Most individuals were found from mid-April to mid-June (probably due to the highest visibility of singing males and feeding parents). Autumn migration starts at the end of August, peaks in late September, and goes down by late October *(Figure 3)* (Gyurácz & Csörgő 2021).

Conservation

The species' population size is extremely large, and the population trend appears to be stable. For these reasons, the species is evaluated as Least Concern. The moderate decrease was mainly due to the intensification of agriculture, the postponement of the harvest time and the cutting of bushes and tree lines in nesting sites (del Hoyo *et al.* 2020, BirdLife International 2021).

In Hungary, it belongs to the red list category Near Threatened (Nagy *et al.* 2019). As a result of weeding due to the decrease in grazing intensity, it became a common nesting ground in the Great Plain by the end of the 20th century. By eliminating the felling of tree lines and shrubs during the breeding period and reducing excessive grazing, the destruction of nestlings can be significantly reduced. Its habitats are also threatened by the spread of certain invasive plants, such as the spread of the goldenrod (*Solidago* spp.), and common milkweed *(Asclepias syriaca)* (Ecsedi 2004, Gyurácz & Csörgő 2021).

The main mortality factors of brood are weather (several days of rain, thunderstorms) and predators (domestic dogs, cats, snakes *Natrix* spp.) (Molnár 1986).

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Higher level taxonomy affects body mass and femur length as predictors for egg size in birds

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Abstract Interpretation of fossil material using comparative anatomy often relies on relationships predicted from data collected from extant species. Some years ago, it was suggested that femur length of birds could be predicted from egg mass but this relationship was counter-intuitive because egg mass is usually related to a measure of body size. The original analysis was also not phylogenetically controlled. This study used the same data to determine phylogenetically controlled relationships for body mass versus egg mass, and egg mass versus femur length. Further analysis showed that order was important in the prediction of egg mass from either body mass or femur length. For some orders, the single regression estimate through all data significantly over-, or under-estimated egg mass. This problem was more pronounced for femur length compared with body mass. Extrapolation of the relationship between femur length and egg mass for large extinct birds seemed to be provide useful data for the Gastornithidae but under-estimated egg mass for other large bird species of a variety of families. Use of equations derived from extant birds to gain insight into the reproductive biology of extinct species needs to be undertaken with great care.

Keywords: bird order, body mass, egg mass, femur length, giant birds, phylogeny

Összefoglalás A fosszilis anyagok összehasonlító anatómiával történő értelmezése gyakran a ma élő fajokról gyűjtött adatokból becsült kapcsolatokra támaszkodik. Néhány évvel ezelőtt azt javasolták, hogy a madarak combcsonthosszát a tojástömeg alapján meg lehet jósolni, de ez az összefüggés ellentétes volt az intuitív hatásokkal, mivel a tojás tömege általában a testmérethez szorosabban kapcsolódik. Az eredeti elemzés szintén nem vette számításba a fajok filogenetikai viszonyait. Ez a tanulmány ugyanazokat az adatokat használta a testtömeg és a tojástömeg, valamint a tojástömeg és a combcsont hosszának filogenetikailag ellenőrzött összefüggéseinek meghatározására. További elemzések kimutatták, hogy a rend (taxonómia) fontos a tojástömeg előrejelzésében akár a testtömeg, akár a combcsont hossza alapján. Egyes rendeknél az összes adaton alapuló egyetlen regressziós becslés jelentősen túl-, vagy alulbecsülte a tojás tömegét. Ez a probléma hangsúlyosabb volt a combcsont hosszára, mint a testtömegre. A combcsont hossza és a tojástömeg közötti összefüggés extrapolálása a kihalt nagytestű madarak esetében hasznos adatnak tűnt, például a Gastornithidae esetében, de alulbecsülte a tojástömeget a különböző családokba tartozó más nagy madárfajok esetében. A kihalt fajok szaporodásbiológiájába való betekintés érdekében a meglévő madarak adataiból származtatható egyenleteket nagy körültekintéssel kell alkalmazni.

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Introduction

Comparative anatomy relies on an understanding of the scaling of morphological features of animals in order to understand evolutionary patterns among taxa. Such relationships can also be useful in the interpretation of fossil specimens of species that are deemed comparable to extant species. For instance, body mass of extinct birds can be predicted from skeletal elements, although whether estimates are perceived as accurate depends on the method employed (Angst & Buffetaut 2017, Deeming & Mayr 2018). Egg mass of extant birds can be estimated with high accuracy from measurements of maximum length and breadth (Hoyt 1979) thereby allowing a reasonable estimate of size to be calculated for fossil eggs of known linear dimensions (Deeming 2006, Deeming & Mayr 2018). An alternative method for calculating egg mass, proposed by Dyke and Kaiser (2010), used a linear relationship between femur length and egg mass in extant birds to predict egg size in the Jurassic Enantiornithine bird *Confuciusornis*. This method has been used subsequently to estimate egg size in a range of extinct giant birds where eggs are unknown (Angst *et al.* 2014, Angst & Buffetaut 2017).

The relationship reported by Dyke and Kaiser (2010) used egg mass as the independent variable to predict femur length, which was seen as a proxy for body size. This was unusual because egg mass would normally be seen as a function of bird size and would be the dependent variable (Deeming 2007), rather than the other way around. Moreover, the relationship reported by Dyke and Kaiser (2010) did not consider different phylogenetic relationships between various species of birds. Higher level taxonomy has little effect on some allometric relations in birds, for instance, hatchling mass and egg mass (Deeming & Birchard 2007). However, for other relationships, e.g. between egg mass and female body mass (Deeming 2007), or between incubation period and egg mass (Deeming *et al.* 2006), there is a significant effect of order. Therefore, the possibility exists that the relationship reported by Dyke and Kaiser (2010), and subsequently used by Angst *et al.* (2014) and Angst and Buffetaut (2017), could be confounded by phylogenetic relationships.

This report details a study where the relationship between femur length (as the independent variable) and egg mass was explored using the data reported by Dyke and Kaiser (2010). In addition, the relationship between egg mass and body was also explored. In these analyses, the effect of order was examined whilst controlling for phylogenetic relationships within each order. It is predicted that whilst female body mass, or femur length, may serve as a good predictor of egg mass, the order of bird will prove very important in determining the value of any relationship that could be used to predict egg size.

Methods

Data for body mass (g), femur length (mm), and initial egg mass (g) reported by Dyke and Kaiser (2010) were available for 137 species (*Table 1*, which also provides order means) from 20 different orders as classified according to del Hoyo (2020). All data were Log_{10} transformed prior to analysis.

Table 1. Sample sizes and descriptive statistics (mean ± SE) for the 20 orders represented in the dataset provided by Dyke and Kaiser (2010) and used in the analysis. SE values are only calculated for orders where there were three or more species represented

1. táblázat Mintaméretek és átlagolt adatok (átlag ± SE) a Dyke és Kaiser (2010) által használt adatkészletben szereplő 20 madárrendre. A hibaértékeket (SE) csak azokra a rendekre számítottuk ki, ahol három vagy több fajra volt adat

	Species	**Female body mass (g)**	**Egg mass (g)**	**Femur length (mm)**
Accipitriformes	5	2199.8 ± 1338.3	85.7 ± 39.9	80.4 ± 13.3
Anseriformes	7	944.2 ± 179.7	57.1 ± 12.7	46.7 ± 4.4
Caprimulgiformes	6	40.3 ± 8.9	4.5 ± 1.2	18.0 ± 2.2
Charadriiformes	19	349.7 ± 68.2	38.4 ± 6.9	35.1 ± 2.3
Columbiformes	2	228.0	10.6	33.2
Coraciiformes	2	84.5	9.9	21.1
Cuculiformes	4	129.5 ± 53.7	9.8 ± 3.6	34.5 ± 6.8
Falconiformes	4	967.5 ±342.7	40.0 ± 10.1	67.8 ± 11.0
Galliformes	9	1242.3 ± 439.1	50.6 ± 18.7	71.0 ± 7.8
Gaviiformes	4	2856.0 ± 821.7	120.6 ± 19.1	50.6 ± 6.4
Gruiformes	7	2139.0 ± 817.2	78.1 ± 21.4	70.5 ± 17.1
Passeriformes	8	114.3 ± 90.4	10.8 ± 8.6	22.7 ± 6.5
Pelecaniformes	10	2948.1 ± 817.0	84.6 ± 14.4	93.6 ± 6.7
Piciformes	6	56.0 ± 8.1	3.9 ± 0.5	23.4 ± 2.2
Podicipediformes	3	587.7 ± 233.1	27.4 ± 5.5	36.9 ± 5.0
Procellariiformes	10	1753.5 ± 710.3	151.1 ± 48.9	49.3 ± 10.0
Sphenisciformes	3	4020.0 ± 567.2	110.7 ± 5.0	76.4 ± 1.4
Strigiformes	6	579.3 ± 189.7	37.4 ± 6.3	63.4 ± 8.1
Struthioniformes	7	23309.0 ± 9001.0	566.7 ± 170.4	166.7 ± 33.2
Suliformes	15	1514.4 ± 168.4	57.5 ± 3.9	52.4 ± 2.7
Total	**137**	**2323 ± 612**	**82.0 ± 13.7**	**56.8 ± 3.6**

Phylogenetically controlled general linear modelling (pglm) was performed in R version 4.1.0 (R Core Development Team 2020) using the packages ape (Paradis & Schliep 2019), mvtnorm (Genz *et al.* 2021), and MASS (Venables & Ripley 2002) using code provided by Carl Soulsbury (pers. comm.). A phylogenetic tree of the species in the dataset was produced based on a Hackett backbone using birdtree.org (Jetz *et al.* 2014). The first model provided a phylogenetically controlled linear regression estimate for all eggs, the slope of which was then tested against an appropriate isometric slope using a one-sample t-test (Bailey 1981). Residuals for each species were calculated for egg mass predicted from the regression equation and standardised by expressing them as a percentage of the reported egg mass. Order means for these standardised residuals were tested against a mean of zero using a one-sample t-test. A second phylogenetically controlled analysis of covariance (ANCOVA) model tested for the effect of order on egg mass whilst controlling for the covariate (either female body mass, or femur length, respectively), and included an interaction term between order and the covariate.

Results

Female body mass and egg mass

The phylogenetically controlled relationship between female body mass (FBM) and egg mass (EM) was:

$$\text{LogEM} = 0.642 * \text{LogFBM} - 0.176.$$

This was highly significant ($F_{1,135} = 552.2$, $P < 0.0001$) *(Figure 1)*, had an R^2 value of 0.804 and the phylogenetic signal was very high ($\lambda = 0.997$). The slope exhibited significant negative allometry (tested against an isometric slope of 1.0, $t_{136} = -13.1$, $P < 0.001$). Although there were positive relationships between female body mass and egg mass in all orders, the calculated regression line did not fit the values very well, especially for birds of a body mass of 50 g or less *(Figure 1)*. The different relationships for orders meant that, for instance, eggs of a 500 g bird could exhibit a four-fold difference in mass.

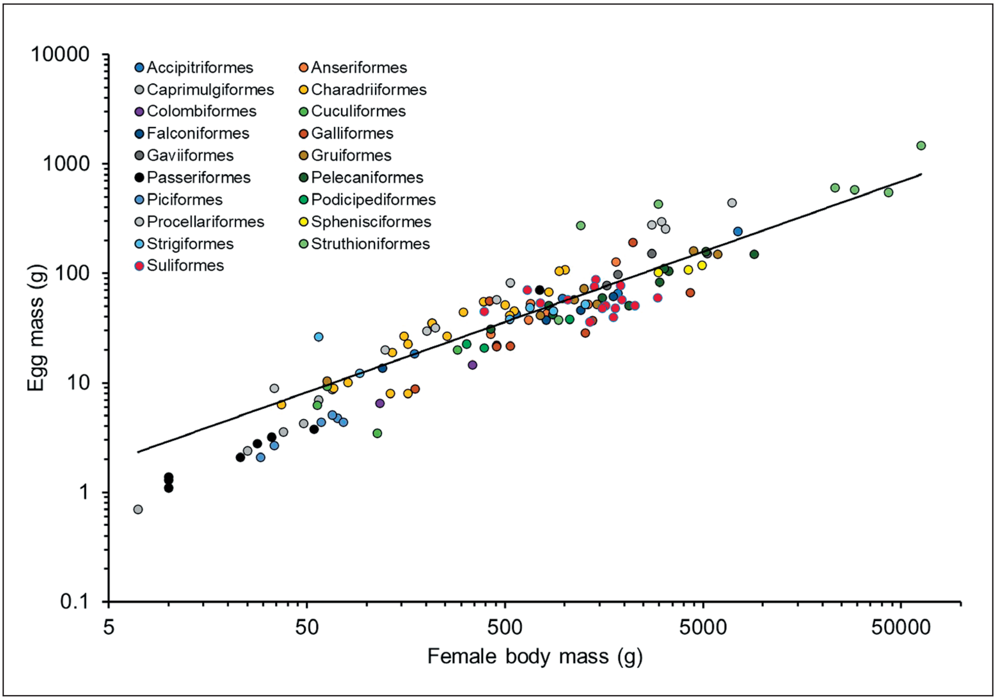

Figure 1. Relationship between female body mass and egg mass for 20 different avian orders. Trendline is generated by a phylogenetically controlled regression in R for all species irrespective of order. The colours correspond to the colour of the symbols indicating each type of bird

1. ábra A tojó testtömege és a tojástömeg kapcsolata 20 különböző madárrend esetében. A trendvonal egy filogenetikailag kontrollált regresszió illeszkedését mutatja a felhasznált madárfajok adataira. A színek megfelelnek az egyes madárfajokat jelző szimbólumok színének

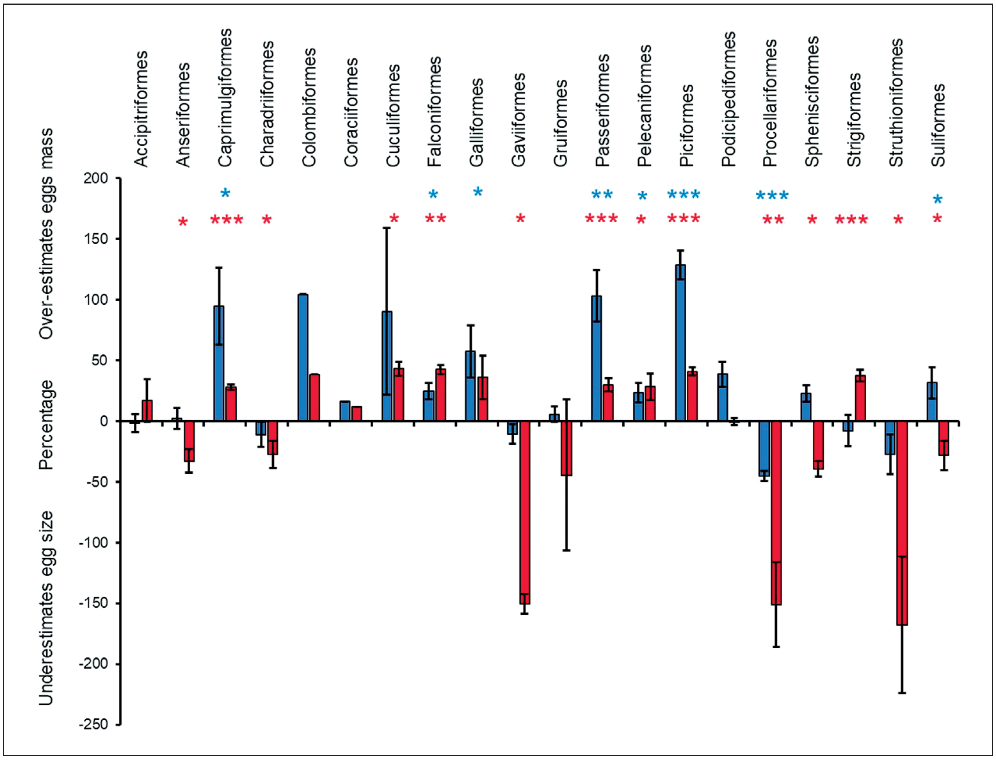

Figure 2. Mean (±SE) for standardised residuals for egg mass. Using phylogenetically controlled regression equations for the relationship with female body mass (blue bars) or femur length (red bars), egg mass was predicted for each species and expressed as a percentage of the actual egg mass for that species. Data were from Dyke and Kaiser (2010). Asterisks represent significant departures from a mean of zero for each order as determined by one-way t-tests; blue symbols indicate significance for egg mass predicted from female body mass and red asterisks indicate significance for egg mass predicted from femur length

2. ábra A tojástömeg standardizált reziduálisainak átlaga (±SE). A tojástömeget minden fajra megbecsültük, és az adott fajra vonatkozó tényleges tojástömeg százalékában fejeztük ki filogenetikailag kontrollált regressziós egyenletek segítségével: kék oszlopok – a tojó testtömegének, piros oszlopok – a combcsont hosszának felhasználásával. Az adatok Dyke és Kaiser (2010) tanulmányából származnak. A csillagok a nulla átlagtól való szignifikáns eltéréseket jelölik, az egyoldalú t-próbák alapján; kékkel a tojó testtömeg alapján előrejelzett tojástömeg jelentőségét, pirossal pedig a combcsont hosszából előrejelzett tojástömeg jelentőségét jelöltük

Representatives of individual orders were not uniformly distributed relative to the regression line. For instance, birds of the Charadriiformes and Procellariiformes were routinely above the line and those of the Piciformes below the line. Mean standardised residuals showed that using the single regression relationship generally over-estimated egg mass, which for several orders was a significant departure from zero *(Figure 2)*. By contrast, for the Procellariiformes the regression line significantly under-estimated egg mass *(Figure 2)*.

A phylogenetically corrected ANCOVA produced a significant interaction between LogFBM and the order ($F_{19,97} = 2.53$, $P = 0.0016$). Therefore, the relationship between female body mass and egg mass was different for at least some orders, e.g. Suliformes. Within the same model there was a significant relationship between LogFBM and LogEM ($F_{1,97} = 2107.81$, $P < 0.0001$) and a significant effect of order ($F_{19,97} = 6.85$, $P < 0.00001$). This model explained over 95% of the variance in egg mass (R^2 value = 0.959) but the phylogenetic signal was very low (λ-value < 0.0001).

Femur length and egg mass

For femur length (FL), the phylogenetically controlled relationship with egg mass was:

$$\text{LogEM} = 1.505 * \text{LogFL} - 0.928,$$

which was highly significant ($F_{1,135} = 303.4$, $P < 0.00001$) *(Figure 3)*, had an R^2 value of 0.692 and a very high phylogenetic signal ($\lambda > 0.999$). This relationship exhibited significant negative allometry (test against an isometric slope of 3.0, $t_{136} = -17.3$, $P < 0.001$). However, the calculated regression line did not fit the values very well especially for birds with a femur length below 20 mm and for a femur length of 50 mm there was a 5-6-fold range in egg sizes *(Figure 3)*.

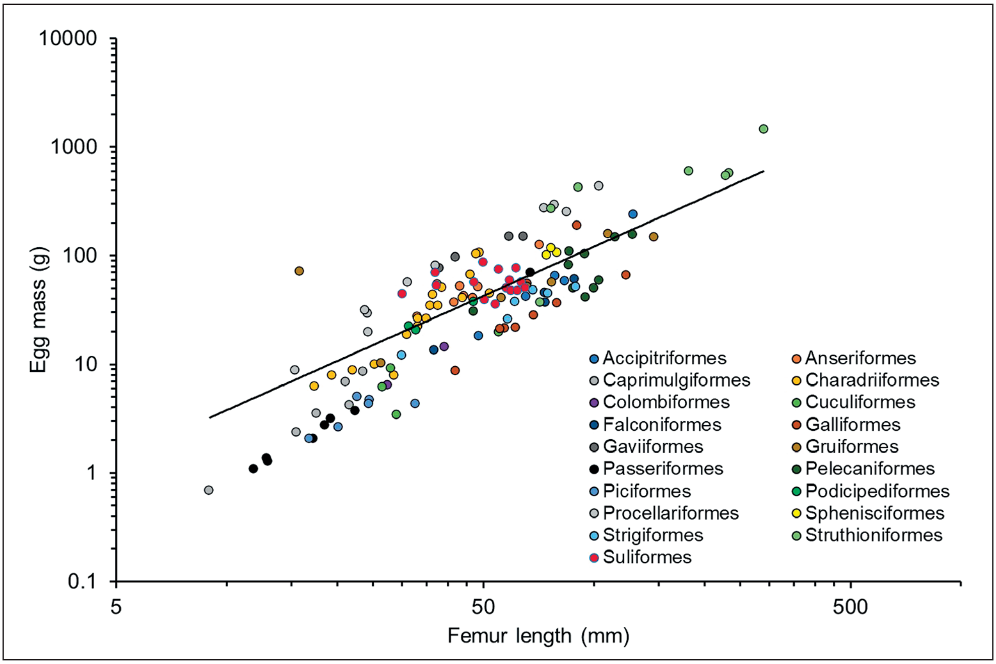

Figure 3. Relationship between femur length and egg mass for 20 different avian orders. Trendline is generated by a phylogenetically controlled regression in R for all species irrespective of order. The colours correspond to the colour of the symbols indicating each type of bird

3. ábra Összefüggés a combcsont hossza és a tojástömeg között 20 különböző madárrendnél. A trendvonal egy filogenetikailag kontrollált regresszió illeszkedését mutatja a felhasznált madárfajok adataira. A színek megfelelnek az egyes madárfajokat jelző szimbólumok színének

Representatives of individual orders were not uniformly distributed relative to the phylogenetically controlled regression line with birds of the Procellariiformes were routinely above the line and those of the Passeriformes below the line *(Figure 3)*. Mean standardised residuals showed that using the single regression relationship did not provide a good prediction of egg mass *(Figure 2)*. For seven of the orders the mean standardised residuals showed that predicted egg mass was significantly over-estimated and for six of the orders it was significantly under-estimated *(Figure 2)*.

The phylogenetically corrected ANCOVA revealed a significant interaction between LogFL and order ($F_{19,97} = 3.41$, $P < 0.0001$) because there were significantly different relationships between femur length and egg mass for the Gruiformes and Suliformes. Within the same model there was a significant relationship between LogFL and LogEM ($F_{1,97} = 1344.06$, $P < 0.00001$) and a significant effect of order ($F_{19,97} = 16.44$, $P < 0.00001$). This model had an R^2 value of 0.947 and the λ-value was very low (< 0.0001).

Discussion

The results of the analysis showed that although egg mass could be predicted from body mass or femur length with a seemingly good level of accuracy, higher level taxonomy, i.e. order, significantly affected these relationships. Female body mass was a better predictor of egg mass than femur length. Therefore, as expected using a single regression relationship to predict egg mass in birds from either body mass, or femur length, introduces a considerable degree of uncertainty because it will depend on the order of bird involved.

It is unclear why Dyke and Kaiser (2010) decided that an analysis to predict body size from egg size was appropriate. The variation in egg size in extant species of different orders means that it is rather unreliable as a method for predicting egg size. Dyke and Kaiser (2010) were interested in understanding the reproductive biology of the Mesozoic enantiornithine bird *Confuciusornis* and were predicting egg size on the basis of the breadth of an egg that could fit through a pelvic canal defined by the fused pubic symphysis. The egg breadth was 17 mm, the eggs were considered as being round and the reported egg mass was 8.2 g. However, using the methodology of Hoyt (1979), i.e., egg mass = $0.548*LB^2$ (L = egg length, B = egg breadth), which Dyke and Kaiser (2010) supposedly used, suggests that such an egg would be 2.7 g. By contrast, Deeming and Mayr (2018) used Hoyt's methodology and for a specimen of *Confuciusornis*, with a pelvic canal width of 14.9 mm, assuming an elongated egg (Deeming & Ruta 2014), the calculated egg mass was 3.5 g. It is unclear why Dyke and Kaiser (2010) had such a high estimate for egg mass. For any of these estimates, egg size in *Confuciusornis*, which has a femur length of 47 mm, it would seem that Dyke and Kaiser (2010) were correct in concluding that egg size would be very small. Deeming and Mayr (2018) concluded that a range of twenty species of Mesozoic birds laid small eggs for their body mass.

Angst *et al.* (2014) used the data published by Dyke and Kaiser (2010) to determine femur length for an egg attributed to the extinct giant bird *Ornitholithus arcuatus* (Gastornithidae), which was calculated to have a mass of 1400 g based on eggshell characteristics. The predicted femur length was 282 mm, which was comparable to other femurs attributed to

Gastornis (Gastornithidae). However, using the phylogenetically controlled relationship reported here for all eggs, a femur attributed to *Gastornis geiselensis* measuring 333 mm (Mayr & Smith 2019) would have a predicted egg mass of 738 g and the range of egg mass of 630–760 g is predicted for other *Gastornis* femurs measuring between 300–340 mm (Angst *et al.* 2014). These estimates are around half the predicted egg mass of 1400 g based on estimated dimensions (Angst *et al.* 2014), which rather undermines the use of egg mass to calculate femur length.

This disparity in egg size estimates is also observed when egg mass is predicted from the femurs of other giant flightless birds. For the Aepyornithiformes of Madagascar, femurs of *Aepyornis* species range from 320 to 430 mm, and for *Mullerornis* species they range from 221 to 246 mm (Angst & Buffetaut 2017). Using the phylogenetically controlled equation reported here, egg masses were estimated as 695–1084 g, and 398–468 g, for *Aepyornis* and *Mullerornis*, respectively. Such estimates are a fraction of the egg mass of 8,000 g predicted from egg dimensions (Angst & Buffetaut 2017). Using the Dyke and Kaiser (2010) relationship the predicted femur length from this egg mass is 618 mm, some 50–100% bigger than fossil bones. For moas (Dinornithiformes) of New Zealand, femur lengths range from 200 to 352 mm (Angst & Buffetaut 2017), which gives predicted egg masses of 342–802 g. By contrast, estimates of egg size from their linear dimensions range from 594 to 4167 g. The largest egg mass would overestimate femur length at 460 mm. Similarly, for *Dromornis stirtoni* (Dromornithidae, Anseriformes) of Australia, has a femur length of 428 mm (Angst & Buffetaut 2017), which predicts an egg mass of 1077 g but thickness of the eggshell predicts that egg mass is over 12,000 g (Murray & Vickers-Rich 2004). Using this egg mass, femur size is predicted at over 740 mm.

These comparisons suggest that for most extinct species the Dyke and Kaiser (2010 relationship does not provide realistic estimates of femur size. Moreover, extrapolation of the phylogenetically controlled relationship between egg mass and femur length does not provide reasonable estimates for egg size. Similar issues are to be expected when attempting to predict egg mass from body mass because Deeming (2007) showed that this relationship varies between orders, a conclusion supported by the analysis reported here. A further problem in predicted egg size in extinct species of giant bird is the estimation of their body mass. For example, estimates for body mass for *Dromornis stirtoni* vary from 440 to 786 kg, and for *Aepyornis maximus* they vary from 243 to 747 kg, depending on the study concerned (Angst & Buffetaut 2017). Whichever equation is used to predict egg mass will be affected by this variation in body mass.

These problems are confounded when the effect of order is taken into account. A single regression estimate would potentially over-estimate or under-estimate egg mass for extant bird species. Extrapolation of the equation to cover the size of larger extinct species could exacerbate the problem. It is possible that a more targeted approach may yield more realistic results. Most extinct giant birds are not closely assigned to extant orders (Angst & Buffetaut 2017), and extreme large size in modern birds is only associated with the Struthioniformes. Therefore, it is difficult to select a subset of extant bird species for which the relationship between body mass or a skeletal measure would be used to predict egg mass in extant species. One exception could be the Dromornithidae, which are attributed to the Anseriformes. It would be interesting to explore how well phylogenetically controlled relationships for a measure of body size in waterfowl would be able to predict egg size to match those values suggested for the Dromornithidae.

This study has demonstrated that use of single relationships for anatomical features, even if they are phylogenetically controlled, can affect the calculated estimates of egg size of extinct species. Different higher level taxonomic classification is important in attempting to predict egg mass from body mass or femur length. Although superficially attractive, the relationship provided by Dyke and Kaiser (2010) is not particularly useful in predicting the size of birds or eggs. Extrapolation of relationships derived from extant bird species should always be undertaken with great care and interpretation should always be conservative.

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Trends of avian locomotion in water – an overview of swimming styles

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Abstract Adaptation to an aquatic lifestyle occurred in the evolution of several primarily terrestrial clades of tetrapods. Among these lineages, aquatic birds' adaptations differ in many ways from other secondarily aquatic vertebrates. As a consequence of the evolution of flight, birds with swimming and diving abilities represent unique locomotion skills and complex anatomical solutions. Here we attempt to overview some of the main aspects of avian locomotion in water and highlight the diversity of their aquatic habits and locomotion types, with the best-known extinct and extant examples. The main features that can distinguish the different groups among these swimmers and divers are their different techniques to overcome buoyancy, the transformation of wings or hind limbs into aquatic propulsive organs, and their swimming techniques besides the presence or absence of the flying and/or terrestrial abilities. Understanding how the musculoskeletal system of aquatic birds evolved to face the requirements of moving in various environments with different physical characteristics provides a good opportunity to get a better view of convergent and divergent evolution.

Keywords: locomotion, swimming, diving, aquatic birds, ecomorphology, evolution

Összefoglalás A vízi életmódhoz való alkalmazkodás többször kialakult az elsődlegesen szárazföldi tetrapodák evolúciója során. Ezen leszármazási vonalak között a vízimadarak alkalmazkodása sok tekintetben különbözik a többi másodlagosan vízi életmódra áttért gerincesétől. A repülés evolúciójának következményeként az úszási és merülési képességekkel rendelkező madarak mozgásformái és komplex anatómiai adaptációi egyedülállóak. Jelen tanulmányban kísérletet teszünk a madarak főbb vízi mozgásformáinak áttekintésére, különös tekintettel mozgástípusaik sokféleségére, példaként említve a legismertebb kihalt és recens képviselőiket. A főbb jellemzők, amelyek megkülönböztethetik egymástól a különböző úszó és merülő csoportokat, azok a víz felhajtóerejének leküzdésére kialakult technikák, a szárnyak vagy a hátsó végtagok vízi meghajtószervekké való átalakulása, valamint az egyes úszásmódok mellett a repülés és/vagy a szárazföldön való mozgás képességének megléte vagy hiánya. Annak megértése, hogy a vízi életmódhoz alkalmazkodott madarak izom- és csontszerkezete hogyan fejlődött úgy, hogy az megfeleljen az eltérő fizikai jellemzőkkel rendelkező közegben való mozgáshoz, kiváló lehetőséget nyújt arra, hogy átfogóbb képet alkothassunk a konvergens és divergens evolúcióról.

Kulcsszavak: mozgás, úszás, merülés, vízimadarak, ökomorfológia, evolúció

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Introduction

During the history of Earth, several primarily terrestrial lineage of tetrapods have returned to the water and adapted on different levels to live in an aquatic environment (Kelley & Pyenson 2015, Motani & Vermeij 2021). Some of these transitions could be connected to the colonisation of aquatic environments during the recovery after devastating mass extinctions, however, most of these events are scattered through geological time and probably were driven by the available food sources and high productivity of aquatic habitats (Vermeij & Motani 2018). The aquatic adaptation of different clades to similar physical parameters and challenges has resulted in similar anatomical solutions and thus, convergent forms in many cases (Lindgren *et al.* 2010, Motani & Vermeij 2021, Gutarra & Rahman 2022). The waters of the Mesozoic were ruled by several clades of marine reptiles (such as sauropterygians, ichthyopterygians, mosasaurs, etc.), several of which survived to this day (like crocodiles, sea turtles, etc.) (Bardet 1994, Bardet *et al.* 2014). From the beginning of the Cenozoic, the radiation of aquatic mammals (such as pinnipeds, cetaceans, sirenians, etc.) has begun (Uhen 2007, Kelley & Pyenson 2015, Motani & Vermeij 2021). Nowadays, aquatic birds represent important components of both marine and freshwater ecosystems. The first avialans obviously adapted to specialised (foot-propelled) swimming and diving (Hesperornithiformes) evolved and colonised the marine and freshwater ecosystems back in the Cretaceous (Marsh 1880, Chiappe & Witmer 2002, Galton & Martin 2002, Bell & Chiappe 2015). Although most of the main groups of modern birds evolved in the Middle to Late Cretaceous, after the Cretaceous-Paleogene extinction birds diversified drastically and conquered the empty ecological space and environments (Moen & Morlon 2014, Brusatte *et al.* 2015). Through the following era numerous species appeared, including early anatids (Zelenkov 2020), grebes (Kurochkin 1976, Zelenkov 2015), loons (Storer 1956, Mayr 2004), rails (García-R *et al.* 2014), alcids (Konyukhov 2002, Smith & Clarke 2015) and penguins (Jadwiszczak 2009, Ksepka & Ando 2011).

Waterfowl show different levels of specialisation ranging from shorebirds to highly aquatic forms (Ashmole 1971). Due to the evolution of powered flight, the rigidity of their body enhanced, and their fore- and hind limbs became suitable for the variant types of movements, and this dual (initially aerial and terrestrial) locomotor system led to different ecomorphological innovations (Gatesy & Dial 1996, Kristoffersen 2001). Their morphologies and locomotion types are exceedingly diverse: they can use their wings or legs (or rarely both) as the main source of thrust production in different ways (Storer 1945). The best swimmers have lost their ability to fly due to the transformation of their wings into flippers (e.g. penguins) (Schreiweis 1982, Louw 1992, Watanabe *et al*. 2021), or sometimes even their ability to effectively move on land because of their highly modified hind limbs (e.g. grebes and loons) (Shufeldt 1898, Clifton *et al*. 2018). However, besides the highly specialised swimmers, many examples of ‘amphibious flyers’ (e.g. alcids) with the ability to swim and dive, fly, and sometimes also to walk effectively are known (Storer 1945, Raikow *et al*. 1988, Bribiesca-Contreras *et al*. 2021). Because air and water represent highly different physical characteristics (density, viscosity, etc.) and because of the presence of distinct mechanical loads during movements in different environments, the musculoskeletal system of birds with aquatic habits must deal with various constraints, which required trade-offs and novel solutions in their morpho-functional adaptations (Raikow *et al.* 1988, Habib 2010, Fish 2016, Bribiesca-Contreras *et al.* 2021).

Among the secondarily aquatic vertebrate taxa, birds' adaptation is unique and in certain aspects deviates from the usual tendencies seen in other tetrapods, since they represent not just a transition between land and water, but a shift from an aerial life to an aquatic (Fish 2016, Fernández *et al.* 2020, Motani & Vermeij 2021). Given the special body plan and evolutionary path of birds among secondarily aquatic vertebrates, explaining their adaptation and functional trade-offs requires understanding their behaviour and locomotor techniques.

Here we intend to sum up and discuss the diverse locomotor and foraging techniques of birds adapted to swimming and diving, in order to form a basis for further comparative works. To highlight and introduce the different trends of avian locomotion in water, we provide summary illustrations based on the available literature.

Aquatic bird locomotor groups

With the intention of highlighting the different trends in avian aquatic locomotion and to describe the most distinguishable locomotor groups, with certain ecological and/or anatomical characteristics, previous authors proposed classifications. These concepts mostly followed the same (and most unambiguous) separation of groups with different propulsive organs (e.g. foot and/or wing) (Townsend 1909, Storer 1945), however, the approach was slightly different. While the concept of Ashmole (1971) emphasised the ecological connections and behaviour besides swimming style, Fish (2016) linked the locomotor groups to evolutionary stages to outline the possible steps of the transition from an aerial to a predominantly aquatic type of movement. From a third perspective, Clifton *et al.* (2018) introduced the terms 'lesser diving' and 'highly diving' for foot-propelled swimming birds based on anatomical and consequent behavioural traits. An additional important viewpoint would be for categorising swimming birds is to take into account their propulsive efficiency, which could also be in connection with their anatomical adaptations (Johansson & Norberg 2001, Johansson & Norberg 2003).

In this work, we attempt to summarize these notions to establish a background for further comparative studies. However, knowing the complexity of the behaviour of birds, we must keep in mind, that every kind of approach could contain outliers and examples which are difficult to classify, moreover the borders and distinctions between different locomotor groups in some cases can be blurry. Another difficulty worth mentioning is that while widespread taxa, and those that are easily accessible in zoos and collections, are frequently studied (such as grebes or cormorants) (Schmid *et al.* 1995, Johansson & Norberg 2001, White *et al.* 2008, Clifton *et al.* 2018), relatively little is known about other rare birds' aquatic behaviour (like finfoots) (Olson 2003, Shepherd 2006, Nurza *et al.* 2017). Following Ashmole (1971), the term 'diving' herein refers to the movement which starts with a resting state on the water-air interface, while 'plunging' is an action that begins from the air. We do not separate taxa from marine environments from those of freshwater ecosystems.

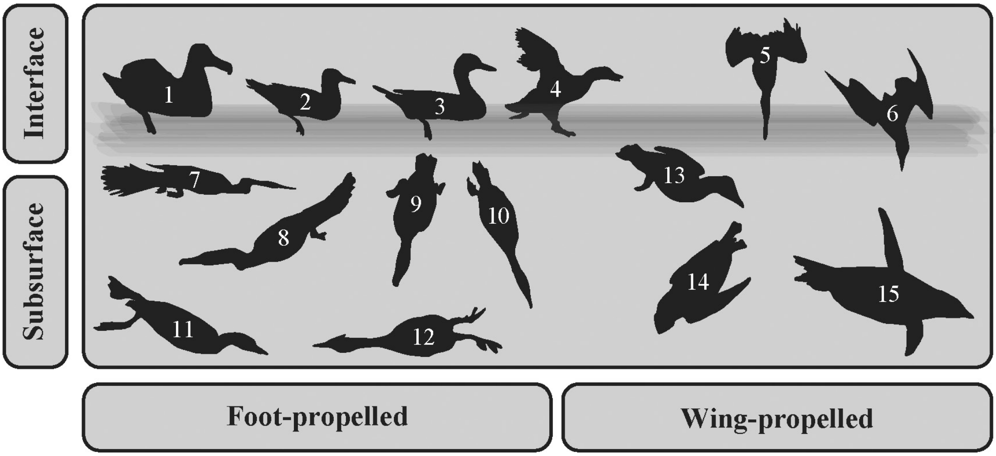

Figure 1. Aquatic bird locomotor groups based on Ashmole (1971), Kristoffersen (2001), Gough *et al.* (2015), Fish (2016), and Clifton *et al.* (2018): (**1–3**) surface swimmers; (**4**) 'skittering'; (**5–6**) plunge-divers; (**7–10**) lesser diving foot-propelled birds; (**11–12**) highly diving foot-propelled birds; (**13–14**) asymmetrical subaqueous flyers; (**15**) symmetrical subaqueous flyers. Examples: (**1**) Wandering Albatross *(Diomedea exulans)*; (**2**) European Herring Gull *(Larus argentatus)*; (**3**) Mallard *(Anas platyrhynchos)*; (**4**) Common Eider *(Somateria mollissima)*; (**5**) Great White Pelican *(Pelecanus onocrotalus)*; (**6**) Northern Gannet (*Morus bassanus*); (**7**) American Darter *(Anhinga anhinga)*; (**8**) Great Cormorant *(Phalacrocorax carbo)*; (**9**) Magellanic Flightless Steamer Duck *(Tachyeres pteneres)*; (**10**) Common Merganser *(Mergus merganser)*; (**11**) Common Loon *(Gavia immer)*; (**12**) Great Crested Grebe *(Podiceps cristatus)*; (**13**) Long-Tailed Duck *(Clangula hyemalis)*; (**14**) Atlantic Puffin *(Fratercula arctica)*; (**15**) King Penguin *(Aptenodytes patagonicus)*

1. ábra Vízimadár mozgási csoportok Ashmole (1971), Kristoffersen (2001), Gough *et al.* (2015), Fish (2016) és Clifton *et al.* (2018) nyomán: (**1–3**) felszíni úszók; (**4**)'skittering'; (**5–6**) becsapódó-merülők; (**7–10**) kevéssé specializált lábbal merülő madarak; (**11–12**) erősen specializált lábbal merülő madarak; (**13–14**) aszimmetrikus víz alatti repülést végzők; (**15**) szimmetrikus víz alatti repülést végzők. Példák: (**1**) vándoralbatrosz *(Diomedea exulans)*; (**2**) ezüstsirály *(Larus argentatus)*; (**3**) tőkés réce *(Anas platyrhynchos)*; (**4**) pehelyréce *(Somateria mollissima)*; (**5**) rózsás gödény *(Pelecanus onocrotalus)*; (**6**) szula *(Morus bassanus)*; (**7**) amerikai kígyónyakúmadár *(Anhinga anhinga)*; (**8**) nagy kárókatona *(Phalacrocorax carbo)*; (**9**) óriás gőzhajóréce *(Tachyeres pteneres)*; (**10**) nagy bukó *(Mergus merganser)*; (**11**) jeges búvár *(Gavia immer)*; (**12**) búbos vöcsök *(Podiceps cristatus)*; (**13**) jegesréce *(Clangula hyemalis)*; (**14**) lunda *(Fratercula arctica)*; (**15**) királypingvin *(Aptenodytes patagonicus)*

The distinction of different locomotor groups *(Figure 1)* is based on exclusive behaviours and anatomical characteristics that are missing from other clusters. For example, even though cormorants use paddling while swimming on the surface, such as non-diver surface swimmers (Johnsgard 1987), the diving performance divides them from the latter group (Zeffer & Norberg 2003). Similarly, although some of the volant advanced wing-propelled birds occasionally plunge from the air (Ashmole 1971), their ability to dive into considerable depth directly from the surface using only their limbs (Ryan & Nel 1999, Low *et al.* 2015) differentiates them from plunge-divers, that are less adapted and most of all utilise momentum to dive and overcome buoyancy (Ashmole 1971).

Physical problematics of diving

The ecological shift from a terrestrial (and aerial) mode of life to an aquatic, requires changes in morphology, physiology, and behaviour. Regarding locomotion, among others, drag reduction, buoyancy control, the increase of stability, and locomotor performance are the main trajectories of evolution (Fish 2016, Houssaye & Fish 2016, Gutarra & Rahman 2022). Aquatic movements mostly depend on the foraging style but include other territorial, escape, or comfort behaviours (Livezey & Humphrey 1982, Johnsgard 1987), and the extent of adaptation varies in accordance with these. Waterbirds are known to utilise aquatic resources at a wide range, depending on their ecology, from surface swimming and dabbling to reaching extreme depths, while consuming planktonic food sources, employing benthic foraging, or performing pursuit diving (Ashmole 1971). Swimming and diving ability could also contribute to the vertical and horizontal distribution of habitats between different taxa (Pöysä 1983a, 1983b, Hustler 1991, Pecsics *et al*. 2017). However, since all of the known aquatic bird taxa (both extant and extinct) continued laying eggs, the reproduction requires some sort of terrestrial abilities even from the most capable divers and swimmers that depend solely on seafood (Shufeldt 1898, Willener *et al*. 2016).

As a result of the different constraints that affect the body in water, significant changes could undergo in morphology, as well as in the inner structure of semi-aquatic and aquatic taxa's bones (Zeffer *et al*. 2003, Habib & Ruff 2008, Hinić-Frlog & Motani 2010). The structural changes could be related either in connection with the adaptation to a buoyant environment (Mendoza & Tambussi 2015, Houssaye *et al*. 2016), but also can be connected to the increased bone strength in elements that are exposed to the strains of swimming (Habib & Ruff 2008, Habib 2010). Besides the anatomical changes caused by the physical constraints of the aquatic mode of life, the heritage of the flying ancestry also had left its mark even on the body plan of most aquatic waterfowl. Among others, the streamlined body shape, developed sternal keel, and powerful pectoral and pelvic musculature (Ostrom 1976, Heers & Dial 2012) provided optimal 'starting point' to develop aquatic abilities and due to the rigid body, the production of thrust relies on the limbs (appendicular locomotion) (Gatesy & Dial 1996). However, in order to provide the proper amount of thrust, to propel the body in a denser environment, and to overcome drag forces both submerged and on the water surface, the structure of fore- and hind limbs shows certain adaptations: such as smaller wings (Raikow *et al*. 1988) or flatter and rigid hydrofoil-like flippers (Louw 1992), webbed or lobbed feet (Johansson & Norberg 2003, Tokita *et al*. 2020). In order to support the streamlined body shape, some of the most aquatic forms, such as loons, grebes, and penguins incorporated to some extent, the hind limbs into the abdominal skin (Kaiser 2011, Clifton *et al*. 2018). The best pursuit divers must be able not just to dive effectively, but also to take sudden turns and use a locomotor technique that provides them enough propulsive force (Clifton & Biewener 2018).

During wing-propelled diving, thrust (the forward component of force) is generated by the lift forces – which are the sum of forces that act perpendicular to the motion direction – of wing strokes (Clark & Bemis 1979). According to Lovvorn and Liggins (2002), this type of thrust production is energetically much more beneficial, than foot-propulsion. Ducks during dives use drag-based (forces that oppose the relative motion of the object in fluid) propulsion to hold their position close to the bottom (Ribak *et al*. 2010). And while previously it was known that paddling is clearly drag-based, according to Johansson and Norberg (2003), at least the later phases of power strokes of the (delta-shaped) webbed feet of cormorants and ducks provide hydrodynamic lift forces. The feet of grebes and loons continuously generate hydrodynamic lift due to the direction of movement and position (Johansson & Norberg 2001, Clifton & Biewener 2018).

Semi-aquatic and aquatic taxa must face buoyancy, the force that is produced by the displaced mass of water and acts opposite to the body weight. Buoyancy is neutral when these two forces are equal (the body is floating), and negative when it causes the animal to sink. Positive buoyancy moves the body upwards in the water column. Penetrating under the surface is problematic for the highly buoyant volant taxa, which must exert force, or gain extra momentum in order to counteract the upward-directed force of water, dive, and reach their foraging depth (Hustler 1991, Eliason *et al*. 2020). The most common way for aquatic vertebrates to decrease the effect of the buoyant force is to increase the density of the body. One solution can be to increase the mass of the skeletal elements with changes in bone microanatomical features (Chinsamy *et al*. 1998, Wilson & Chin 2014, Cerda *et al*. 2015, Houssaye *et al*. 2016). A partially, or fully vettable feather coat could also provide a solution (such as in the case of cormorants and anhingas), however, in this case, the birds could be exposed to the negative effects of weather while they dry their feathers (Grémillet *et al*. 2005, Ryan 2007). The buoyant force decreases with increasing depth and hydrostatic pressure, and since the effect of buoyancy during shallow dives is maximal, shallow diving requires more energy compared to the dives of those species that descend deeper (Kato *et al*. 2006). For example, in shallow water, both penguins and cormorants are positively buoyant, but in addition, this can help in ascending and supports the bird to reach the water surface (Ribak *et al*. 2005, Kato *et al*. 2006). Deep divers descend beyond the depth where buoyancy is minimal or neutral and can spend most of their energy searching and capturing prey (Lovvorn *et al*. 2001), while shallow divers remain in the zone of maximal buoyancy, and must work against it during the entire dive sequence (Kato *et al*. 2006).

Surface swimmers

Birds with aquatic habits (both surface swimmers and divers) typically use their feet to propel themselves on the air-water interface *(Figure 2)* (Ashmole 1971, Ancel *et al.* 2000), between plunges and dive sequences, except for penguins that mainly move with the rowing motion of their flipper-like wings *(Figure 3)* (Butler 2000). Aquatic taxa generally represent a broad, boat-shaped body that provides them stability while floating (Stolpe 1932, Bovy 2002).

The category 'surface swimmer' stands for those buoyant birds that are mostly restricted to the surface and do not necessarily show such drastic anatomical adaptations in connection with their aquatic lifestyle that would affect their terrestrial or aerial abilities, like in more derived foot-propelled aquatic groups (Clifton *et al.* 2018b). On occasion, they are able to partially or fully submerge, however they are not capable of performing deep or long-time diving (for example, albatrosses, gulls, dabbling ducks) (Ashmole 1971, Raikow 1973, Butler *&* Jones 1982). Dabbling birds for example mostly submerge only with the front part of their body (Johnsgard 2010), and their oily waterproof feather coat helps them to forage upside down. Some of them have good terrestrial (e.g. walking or even running) capabilities (Provini *et al.* 2012). The propulsion is provided by the alternate strokes of the feet (Gough *et al*. 2015, Fish 2016), and the paddling movements are supported by lobed or webbed feet, which evolved convergently in several taxa with different phylogenetic background (Tokita *et al.* 2020). Besides progress and resting, eating from the surface could be the advantage of this low-speed movement: such as grasping organisms, or even filtering plankton (Ashmole 1971).

Skittering (special locomotor technique)

Some of the birds that propel themselves by alternate paddling on the water surface apply special temporary locomotor techniques (in short: 'skittering') in order to avoid wave drag and excel the speed-limiting effect of waves generated by the surface swimming (hull speed) without flying (Gough *et al.* 2015). A peculiar way of locomotion is the so-called 'steaming', which is sometimes also mentioned as 'wing-paddling' (Fish 2016). Steaming, as a regular behaviour is mostly cited in connection with large, heavy anatids, especially the steamer ducks (*Tachyeres* spp.) and eider ducks (*Somateria* spp.), however other birds, like alcids, also apply this movement occasionally to a lesser extent (Livezey & Humphrey 1986, Gough *et al.* 2015). During steaming, these waterfowl increase their body angle and gain extra acceleration from the power strokes of the slightly flexed wings combined with foot-paddling. However, this rapid oaring is restricted to just a limited time interval and used only in those cases when the birds have to escape insecure situations or need acceleration for other temporary reasons (Livezey & Humphrey 1986, Gough *et al*. 2015). Steaming is particularly interesting in the case of the Magellanic Flightless Steamer Duck *(Tachyeres pteneres)*, which birds' wing size does not allow the take-off, however, the strong and otherwise functional wings are regularly used as paddlers on the surface (Livezey & Humphrey 1986, Fulton *et al.* 2012).

Besides steaming, another temporary water-surface related speed accelerator behaviour is the so-called 'paddle-assisted flying', which was studied in more depth by the observation of the Common Eider *(Somateria mollissima)* by Gough *et al*. (2015): unlike steaming, in this case, the body stands out completely from the water, the bird paddles against the surface while flapping the wings, to overcome wave drag. A similar series of movements is used by other aquatic avian taxa as well, in order to gain acceleration prior to the take-off (Norberg & Norberg 1971).

Plunge-divers

Plunge-divers are those volant birds that are generally more buoyant and less adapted for aquatic locomotion than other, more derived avian divers. In order to penetrate below the surface and reach the underwater prey, they use the momentum gained from the controlled fall *(Figure 3)* (Weiss *et al.* 2014, Chang *et al.* 2016). The anatomy of the neck, the structure of the beak, and the cranial shape help plunge-divers to strike and dive into the water with the minimal chance of damage and injuries (Sharker *et al.* 2019). Some of them, such as kingfishers, terns, gulls, pelicans, and tropicbirds dive only to a very small depth, not considerably deeper than their own length (surface-plungers), however, others, like gannets and boobies are able to penetrate for several meters below the surface (deep-plungers) (Ashmole 1971). After the end of the descent, these deep-plungers use foot-propulsion and mainly asymmetrical subaqueous flight (wing-propulsion) for a short period, prior they ascend back to the surface, where they employ paddling with their (mostly) webbed hind feet (Lee & Reddish 1981, Corre 1997).

Foot-propelled divers

Among avian groups, foot-propelled diving occurred in different lineages, such as in ducks, snakebirds (anhingas) and cormorants, and in the highly specialised grebes and loons *(Figure 2)*. The more advanced foot-propelled divers are characterised by an elongated body, caudally positioned hind limbs, with powerful pelvic and leg musculature (Kristoffersen 2001, Clifton *et al.* 2018). The majority of them are still capable to fly. Flightlessness is known among foot-propelled divers – with no claim of being exhaustive – in the case of the Cretaceous Hesperornithiformes birds with highly reduced wing bones (Marsh 1880, Martin & Tate 1976), and for several anatids, such as the Magellanic Flightless Steamer Duck (Livezey & Humphrey 1986), moreover for the Galapagos Cormorant *(Phalacrocorax harrisi)* (Shufeldt 1915), and several flightless grebes, like the Titicaca Grebe *(Rollandia microptera)*, the Junin Grebe *(Podiceps taczanowskii)*, and the recently extinct Atitlán Grebe *(Podilymbus gigas)* (Livezey 1989). However, it must be noted that the loss of flight and the shrinking of wing sizes (at least in the better known Cenozoic taxa) were not primarily related to the modification of their wings to propulsive organs (Livezey & Humphrey 1986, Livezey 1989). Those taxa, that have lost their flying capabilities only recently (e.g. some thousand years ago) are sometimes mentioned as 'neoflightless' (Habib & Ruff 2008).

Among foot-propelled divers, following Clifton *et al.* (2018) a distinction can be made between those taxa that retained much of their terrestrial abilities and able to move effectively on a substrate (like diving, and sea ducks, cormorants), and also sometimes to climb (anhingas) (lesser diving foot-propelled birds), and those that have lost much of their walking skills due to their anatomical adaptations – especially in their abdominal region, pelvic girdle and on the legs – (loons, grebes) and are rather clumsy on land (highly diving foot-propelled birds).

On the water surface, both groups apply alternate paddling (Ancel *et al.* 2000) and during the descent and horizontal underwater swimming, much of the propulsion is provided by the strong hind feet (Johnsgard 1987). Usually, both the lesser and highly diving taxa hold their wings close to the body, and strike with them only rarely, for example during sudden turns (Townsend 1909, Owre 1967), or the phase of submergence from the surface (Kelso 1922), or in the case of some anatids, the wings are held partly extended and used as a diving plane (Townsend 1909, Brooks 1945).

Among ducks, several different techniques are known. For example, Lesser Scaups *(Aythya affinis)*, Ruddy Ducks *(Oxyura jamaicensis)* and Mergansers (*Mergus* spp.) dive with simultaneous (symmetrical) leg strokes (White 1957, Tome & Wrubleski 1988). Besides simultaneous symmetrical strokes, Canvasbacks *(Aythya valisineria)* occasionally dive with alternate (asymmetrical) strokes (Tome & Wrubleski 1988), and their wings are always folded such as in *Mergus* spp. (Townsend 1909, Brooks 1945), and in the case of all the above mentioned ducks (Tome & Wrubleski 1988). Steamer Ducks (*Tachyeres* spp.) also dive with alternate strokes of their feet (Livezey & Humphrey 1982b).

The diving phenomenon was deeply studied in the case of cormorants (Schmid *et al.* 1995, Ancel *et al*. 2000, Ribak *et al*. 2005) however, information related to the slower swimmer anhingas are more controversial (Owre 1967), but according to Audubon (1838), anhingas can dive both with alternate and symmetrical paddling. Cormorants are pursuit hunters, diving in greater depths, while anhingas dive only in shallow waters, and as ambush fish-eaters lurking slowly under the surface (Hustler 1991). The feather coat of cormorants is more water and cold resistant therefore these birds are able to forage in colder waters (Grémiller *et al.* 2005). The feather coat of anhingas is getting completely wet during dives, and cannot hold back air bubbles under the feathers (Ryan 2007). The density of their skeletal elements are somewhat higher, compared to cormorant species (Owre 1967), while the pneumatization of the body and the size of the air sacks are not well developed, therefore, combined with their fully wettable feather coat, they need less energy to act against buoyancy during their shallow dives (Owre 1967, Hustler 1991). In addition, this type of feather coat is a limiting factor for their distribution since they prefer only warm waters (Ryan 2007). Despite that anhingas' attributes do not make them fast underwater hunters, they present efficient climbing abilities (Owre 1967, Diederle 2016). Cormorants use their feet paddling backwards, then every power stroke is followed by a gliding phase when their hind limbs are held in an extended position (Ribak *et al.* 2005). During paddling, the feet are positioned ventrally (Ribak *et al.* 2005). This simultaneous symmetrical striking is an energetically more optimal swimming method, which produces greater levels of (in this case lift-based) thrust than the asymmetrical paddling (Johansson & Norberg 2001, Ribak *et al.* 2005).

Grebes and loons almost never leave the water, and just rarely show erect posture standing on hind limbs (Shufeldt 1898). They are rather clumsy on land, since the proximal hind limbs are integrated into the abdominal skin (Clifton *et al.* 2018). These birds during their underwater pursuit dives employ simultaneous (symmetrical) strokes with their laterally positioned feet – without the use of the wings (Townsend 1909) – which provides them powerful lift-based thrust (Johansson & Norberg 2001, 2003, Ribak *et al.* 2005, Clifton & Biewener 2018). Under the surface, both grebes and loons synchronously move their feet laterally backwards and extend the intertarsal ankle joint, then in the following phase the feet are moving forward with closed digits. The movement of the feet are more lateral and not ventral like in cormorants (Johansson & Norberg 2001, Clifton & Biewener 2018). Their swim includes periodic power and recovery strokes, without a gliding phase following the power stroke (Ribak *et al.* 2005, Clifton *et al.* 2018). Rotation is limited at the hip joint, and is present only at the knee to some extent, supported by the high cnemial crest, which

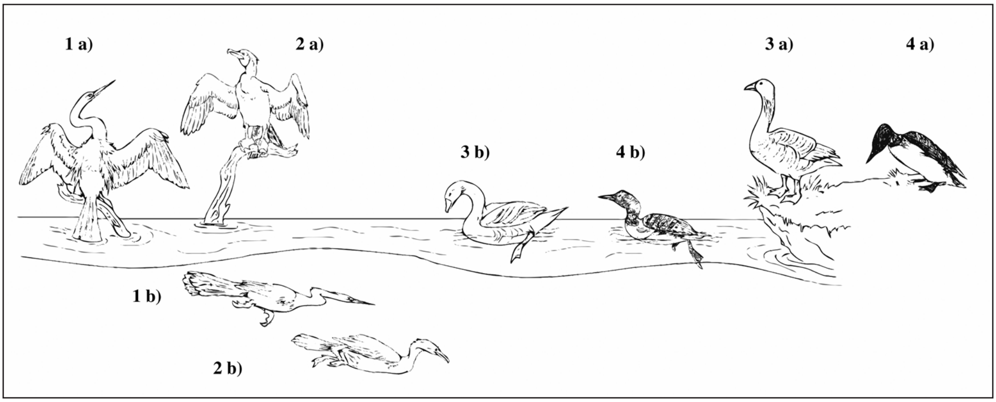

Figure 2. Foot-propelled birds with examples for swimming and terrestrial locomotion abilities
(**1**) American darter *(Anhinga anhinga)* (**a**: drying its wings, **b**: underwater swimming);
(**2**) Great Cormorant *(Phalacrocorax carbo)* (**a**: drying its wings, **b**: underwater swimming);
(**3**) Greylag Goose *(Anser anser)* (**a**: standing, **b**: dabbling);
(**4**) Common Loon *(Gavia immer)* (**a**: unable to stand, **b**: swimming on surface)

2. ábra Úszás közben lábukat használó madarak példákkal az úszási és szárazföldi mozgási képességeik tekintetében
(**1**) amerikai kígyónyakúmadár *(Anhinga anhinga)* (**a**: szárnyait szárítja, **b**: felszín alatt úszik);
(**2**) nagy kárókatona *(Phalacrocorax carbo)* (**a**: szárnyait szárítja, **b**: felszín alatt úszik);
(**3**) nyári lúd *(Anser anser)* (**a**: áll, **b**: vízfelszínről táplálkozik);
(**4**) jeges búvár *(Gavia immer)* (**a**: nem képes állni, **b**: a vízfelszínen úszik)

limits the flexion and extension of the knee, but permits long-axis rotation of the *tibiotarsus* (Clifton & Biewener 2018, Clifton *et al.* 2018).

The reconstructed anatomy and appearance of the extinct Hesperornithiformes are similar to grebes and loons (Zinoviev 2011), however, in some features they also resemble to cormorants and probably these similarities make them an example for mosaic evolution (Bell *et al*. 2017).

Wing-propelled divers

Among diving birds, a wide spectrum of uses of wings exists, and in contrast to foot-propulsion, wing-propelled techniques are known only from the Cenozoic fossil record (Kristoffersen 2001). One of the most extreme morphological and structural responses among birds for locomotion-related mechanical loadings is present in the forelimbs of wing-propelled divers (Habib & Ruff 2008, Serrano *et al*. 2020), thus this behavior *(Figure 3)* represents a strong selective pressure (Smith *et al*. 2021).

Several anatids – such as the White-winged Scoter *(Melanitta deglandi)* and Surf Scoter *(Melanitta perspicillata)* – extend the wings partially during and following the submergence but do not use them for propulsion (Brooks 1945). In contrast, some authors mentioned that Long-tailed Ducks (or Oldsquaw) *(Clangula hyemalis)* apply wing-propulsion for diving and horizontal progression (Townsend 1909, Snell 1985), however, understanding this phenomenon probably needs further observations. The only group of songbirds, that

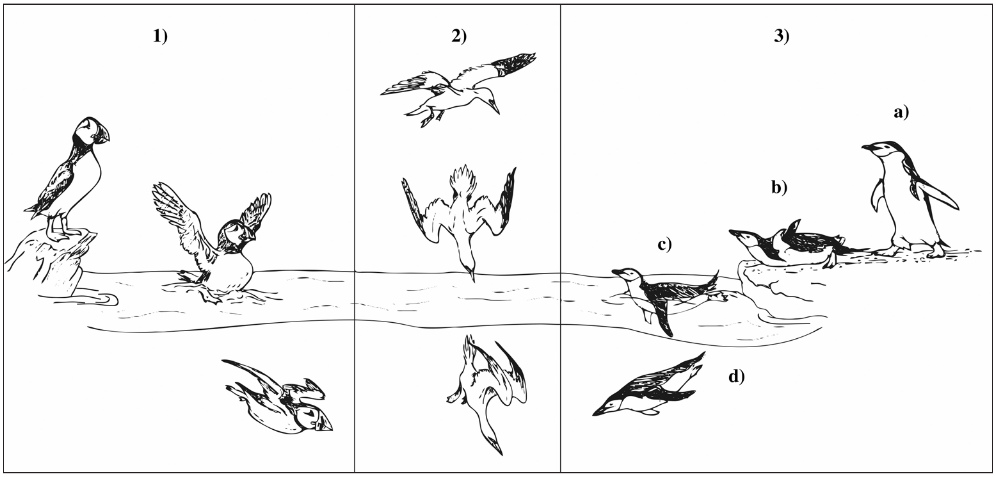

Figure 3. Wing-propelled birds with examples for flying, aquatic and terrestrial locomotion abilities (**1**) Atlantic Puffin *(Fratercula arctica)*; (**2**) Northern Gannet *(Morus bassanus)*; (**3**) Chinstrap Penguin *(Pygoscelis antarcticus)* (**a**: walking, **b**: tobogganing, **c**: swimming on surface, **d**: diving)

3. ábra Úszás közben szárnyukkal evező madarak példákkal a repülési, vízi és szárazföldi mozgási képességeik tekintetében (**1**) lunda *(Fratercula arctica)*; (**2**) szula *(Morus bassanus)*; (**3**) kantáros pingvin *(Pygoscelis antarcticus)* (**a**: sétál, **b**: hason csúszik, **c**: a felszínen úszik, **d**: merül)

are adapted to wing-propulsion are the dippers (Cinclinidae), which progress underwater and search for prey with the thrust produced by the partially opened wings (Goodge 1959). While dippers do share some convergent features (e.g. related to musculature and the feathers) with other wing-propelled divers, the osteological modifications are subordinate (Smith *et al*. 2021).

Following Fish (2016) those taxa that perform underwater flight using wing-propulsion can be grouped regarding their aerial skills. During wing-propulsion, thrust is based on the lift forces produced by the wing strokes (Clark & Bemis 1979). The wings of active flyers (such as Cinclinidae, Alcidae, and genus *Pelecanoides*) must be functional not just in the air but in the water, they employ asymmetrical subaqueous flight (Fish 2016). In their case, the production of thrust is larger during downstrokes, while the upstrokes provide a smaller amount of thrust, however, this movement could counteract buoyancy (Spring 1971, Johnsgard 1987, Johansson & Aldrin 2002). Contrary, in the case of the symmetrical subaqueous flight of flightless wing-propelled birds (such as Sphenisciformes), an important amount of thrust is being produced during both the down- and upstrokes of wings (Clark & Bemis 1979, Lovvorn & Liggins 2002). In addition to wings, certain groups, like alcids use leg strokes to some extent (see the video footage in Enstipp *et al*. 2018), probably to counteract buoyancy (Townsend 1909, Johnsgard 1987).

The wing anatomy of flightless taxa shows flattened bones, with more prominent epiphyses, a drastically decreased mobility of the distal wing joints, together with the wing elevator/retractor muscles becoming more emphasized (Schreiweis 1982, Louw 1992, Watanabe *et al*. 2021). While the changes in pectoral and wing-elevator musculature can be already experienced in volant wing-propelled taxa (Bribiesca-Contreras *et al.* 2021), the most drastic anatomical alterations probably happen after the loss of flight. According to Raikow *et al.* (1988), the stiffening of joints and reduction of wing elements' mobility cannot be observed among those wing-propelled birds that are still active flyers. Despite that birds that perform asymmetrical subaqueous flight exemplify an adaptation to move both in the water and air (Kovacs *et al.* 2000).

Besides penguins, other (now-extinct) flightless wing-propelled taxa are known, such as the Plotopteridae from the North Pacific Ocean (Eocene to early Miocene), that group highly resembles penguins in many aspects of their adaptations (Dyke *et al*. 2011, Mayr 2020). Flightlessness appeared at least two times among auks (Pan-Alcidae): flightless stem-group alcids (Mancallinae) are known from the Miocene to Pleistocene of the Pacific coasts of North America and Japan (Smith 2011, Watanabe *et al*. 2021), while among crown-group alcids Great Auks represented this evolution in the North Atlantic, at least from the Pliocene until the 19th century when their last species *Pinguinus impennis* became extinct due to human activity (Fuller 1999, Watanabe *et al*. 2021).

Future perspectives

In an attempt to deeper understand the complexity of aquatic birds' swimming habits, we must take into account not just the comparative anatomical descriptions (Schreiweis 1982, Louw 1992, Clifton *et al.* 2018, Watanabe *et al.* 2021), but the newer advanced methods and modeling techniques. However, besides the significance of the latter sources, the importance of traditional field observations must not be forgotten (Brooks 1945, Shepherd 2006).

While numerous studies discussed the locomotion skills of some of the best foot- or wing-propelled swimmer and diver taxa – like grebes (Johansson & Norberg 2001), auks (Johansson & Aldrin 2002), and penguins (Clark & Bemis 1979) – relatively little attention has been paid to reliably record (with photo documentation or video recording) and to publish the various swimming habits of less adapted aquatic birds. It would be particularly remarkable to gain more data about the different uses of wings of diving ducks, not only in nature, but also in an observation tank. A possible direction for future works could be the application of three-dimensional methods or tomographic tools in order to better understand the biomechanics of how the skeleton adapts to the movements in different habitat with different physical characteristics (Habib & Ruff 2008, Habib 2010, Eliason *et al*. 2020). These studies would be interesting not just for the comparison of extant taxa with fossil remains, but also to find novel concepts for bioinspired materials and design or even robotics.

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Presentation of so far undetermined bird remains from the Pliocene of Beremend 26 and Csarnóta 2 and 4 (Baranya county, South Hungary)

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Abstract The authors have defined at the order, subfamily, family or genus level the very fragmentary and small-size bird bone material from the three Pliocene-age sites in southern Hungary (Beremend 26, Csarnóta 2 and 4), which is in the collection of the Museum of the Hungarian Institute of Geology and Geophysics. The non-catalogued bone fragments remaining from the already examined material were identified. The number of taxa identified is 26, of which one species is new to science. The new species (*Pliogallus csarnotanus* n. sp.) belongs to a hitherto disputed genus, which is thus recognised through the newly defined material. Of the rest of the material, only *Paleocryptonix hungaricus* Jánossy, 1991 and *Glaucidium baranensis* Kessler, 2010 have been identified to species level, the *Gallinula, Porzana, Merops, Garrulus, Nucifraga* finds to genus level, while the other 18 taxa have been identified only to subfamily or family level (Perdicinae, Columbidae, Alaudidae, Hirundinidae, Panuridae, Paridae, Sittidae, Certhiidae, Muscicapidae, Turdidae, Sylviidae, Motacillidae, Prunellidae, Laniidae, Sturnidae and Fringillidae), or only to order level (Charadriiformes, Coraciiformes).

Keywords: Beremend, Csarnóta, Pliocene, Hungary, birds, fossils

Összefoglalás Jelen tanulmányban három pliocén korú (Beremend 26, Csarnóta 2 és 4) dél-magyarországi lelőhelyről származó, a Magyar Geológiai és Geofizikai Intézet múzeumának gyűjteményében található igen töredékes és apró méretű madárcsont anyagot dolgozták fel és határozták meg a szerzők legalább rend, alcsalád, család vagy genus szintig. A már vizsgált anyagokból visszamaradt, nem katalogizált csonttöredékek meghatározásával jelentősen bővült ismeretünk. Az azonosított taxonok száma 26, amelyekből egy faj új a tudomány számára. Az új faj (*Pliogallus csarnotanus* n. sp.) egy eddig vitatott genusba tartozik, amely ezáltal elismerésre kerül, az újonnan meghatározott anyag révén. Az anyag többi részéből csak a *Paleocryptonix hungaricus* Jánossy, 1991 és a *Glaucidium baranensis* Kessler, 2010 fajokat sikerült azonosítani, a *Gallinula*, *Porzana*, *Merops*, *Garrulus*, *Nucifraga* leleteket genusz szintig, míg a többi 18 taxont csak alcsalád vagy család szintig (Perdicinae, Columbidae, Alaudidae, Hirundinidae, Panuridae, Paridae, Sittidae, Certhiidae, Muscicapidae, Turdidae, Sylviidae, Motacillidae, Prunellidae, Laniidae, Sturnidae és Fringillidae), vagy csak rend szintig (Charadriiformes, Coraciiformes).

Kulcsszavak: Beremend, Csarnóta, Pliocén, Magyarország, madarak, fosszíliák

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Introduction

A number of Pliocene-age vertebrate remains have been recovered from the splits of low mountain ranges built up from Mesozoic limestone in the southernmost part of Hungary. These have been collected, studied and published since the mid-19th century, including bird material. The undetermined, mostly very small and fragmentary bones from the Hungarian Institute of Geology and Geophysics (HIGG) collection of the Csarnóta 2 material (uncatalogued) and the „Beszélő Kövek Alapitvány" Harkány (BKAH) collection of Beremend 26 and Csarnóta 4 (uncatalogued) have been identified and are reported in this paper. As most of the material is highly fragmentary and not previously identified by Kessler J. (Kessler 2013a, 2013b), the present study could only identify a few bones or bone fragments to species level, the rest could only be identified to order, family or genus level. They can be compared with the characters and sizes of the species already reported from the sites and their affiliation can be assumed.

Beremend sites: about 9 km south of the Villány Mountains and the village of Villány lies the flat, loess-covered, 174 m high Szőlő Hill by Beremend, which is made of Lower Cretaceous limestone (Nagyharsány Limestone Formation). The limestone has been quarried for more than a hundred years, and almost year after year new karst cavities and splits with bone have been discovered in the quarry. Already in 1847, Salamon János Petényi, accompanied by Ágoston Kubinyi, had collected from Beremend (sites 1–3), and thanks to the excavations of Miklós Kretzoi, which began in 1956, the number of sites had increased to ten. Dénes Jánossy continued the excavations with Endre Krolopp (sites 11–17) from 1973. László Pongrácz has then followed the discovery of new sites as a result of quarrying (sites 17–39). The age of the sites covers a large time interval. Some of the vertebrate faunas of the investigated sites belong to the Pliocene MN Zones 15–16, the Csarnóta and Beremend biochronological units. Their age is between 3.2 to 1.8 million years old and follow the chronological order (Csarnóta 2 and 4 at the same time as Beremend 26) with interruptions, moving from older to younger (Jánossy 1979a, 1979b, Kordos 1992, 2001, Kessler 2009a, 2009b, 2010, 2013a, 2013b, Pongrácz oral communication).

Beremend (26): the Beremend 26 site is the 100 m (deepest) mining level of a 30 m wide and 20–25 m high red-clay fissure filled with lithic crushed rock, from which HIGG has carried out a detailed geological investigation („base section"). The site was largely demolished after the collection, but László Pongrácz continued to collect from the site as long as it was possible. The mammal fauna was processed by László Kordos (2001a). The bird material identified and published are: *Podiceps* sp. (*ruficollis* size), *Egretta* sp. (*garzetta* size), *Accipiter* sp. (*nisus* size), *Falco tinnunculus atavus* Jánossy, 1972; *Falco* sp. (*peregrinus* size) *Tetrao praeurogallus* Jánossy, 1969; *Tetrao partium* Kretzoi, 1962; *Gallus beremendensis* Jánossy, 1976; *Francolinus capeki* Lambrecht, 1933; *Palaeocryptonix hungaricus* Jánossy, 1991; *Perdix perdix jurcsak* Kretzoi, 1962; *Rallus polgardensis* Jánossy, 1991; *Miorallus major* Milne-Edwards, 1869–1871; *Porzana* sp. (*Porzana porzana* size), *Otis kalmani* Jánossy, 1972; *O. lambrechti* Kretzoi, 1941; *Chlidonias* sp., *Tringa* sp. (*glareola – ochropus* size), *Columba* sp., *Glaucidium baranensis* Kessler, 2010; *Athene noctua veta* Jánossy, 1992; *Strix intermedia* Jánossy, 1972; *Picus pliocaenicus* Kessler, 2013; *Dendrocopos praemedius* Jánossy, 1974; *Melanocorypha minor* Kessler, 2013; *Galerida pannonica* Kessler, 2013; *Lullula parva* Kessler, 2013; *Lullula minuscula* Kessler, 2013; *Delichon major* Kessler, 2013; *Parus robustus* Kessler, 2013; *Parus medius* Kessler, 2013; *Sitta villanyensis* Kessler, 2013; *Luscinia pliocaenica* Kessler, 2013; *Phoenicurus baranensis* Kessler, 2013; *Oenanthe pongraczi* Kessler, 2013; *Saxicola baranensis* Kessler, 2013; *Saxicola magna* Kessler, 2013; *Erithacus minor* Kessler, 2013; *Monticola pongraczi* Kessler, 2013; *Turdus major* Kessler, 2013; *Turdus medius* Kessler, 2013; *Turdus praeminor* Kessler, 2013; *Oriolus beremendensis* Kessler, 2013; *Acrocephalus kretzoii* Kessler, 2013; *Sylvia pusilla* Kessler, 2013; *Locustella magna* Kessler, 2013; *Locustella janossyi* Kessler, 2013; *Regulus plioceanicus* Kessler, 2013; *Motacilla minor* Kessler, 2013; *Motacilla robusta* Kessler, 2013; *Bombycilla kubinyii* Kessler, 2013; *Prunella kormosi* Kessler, 2013; *Lanius major* Kessler, 2013; *Lanius intermedius* Kessler, 2013; *Sturnus pliocaenicus* Kessler, 2013; *Sturnus baranensis* Kessler, 2013; *Passer pannonicus* Kessler, 2013; *Coccothraustes major* Kessler, 2013; *Loxia csarnotanus* Kessler, 2013; *Emberiza gaspariki* Kessler, 2013 (Kessler 2009a, 2009b, 2010, 2013a, 2013b).

Csarnóta (2, 4): in the western part of the Villány Hills, on the border of the village of Csarnóta, on the flat top of Cserhegy. Bone material found in the red clay columns filling the crevices of abandoned quarries was first collected by Tivadar Kormos (according to information from M. Pálfy) from 1910 to 1930 (marking the site as „upper quarries"), then Miklós Kretzoi and Dénes Jánossy collected here regularly. Of the four sites, sites 2 and 4 also provided bird material. The latter is not quite the same age as the other three. Site 1 was destroyed following the construction of the road. The species list of site 2 is as follows: *Tetrao praeurogallus* Jánossy, 1969; *Francolinus capeki* Lambrecht, 1933; *Rallus aquaticus*, *Gallinago veterior, Cuculus csarnotanus* Kessler, 2010; *Bubo bubo, Aegolius* sp., *Hirundo* sp., *Garrulus glandarius*, *Pyrrhocorax graculus vetus* Kretzoi, 1962; *Sitta* sp., *Turdus viscivorus*, *Turdoides borealis* Jánossy, 1979 (Kretzoi, 1962, Jánossy 1976a, 1976b, 1977, 1979a, 1979b, 1979c). The remaining and undetermined material has been identified by J. Kessler over the years and the following species have been defined: *Podiceps csarnotanus* Kessler, 2009; *Anas albae* Jánossy, 1979; *Falco tinnunculus atavus* Jánossy, 1972; *Palaeortyx brevipes* Milne-Edwards, 1869; *Gallus beremendensis* Jánossy, 1976; *Tetrao praeurogallus* Jánossy, 1969; *Tetrao partium* Kretzoi, 1962; *Francolinus capeki* Lambrecht, 1933; *Otis kalmani* Jánossy, 1980; *Rallicrex polgardensis* Jánossy, 1991; *Porzana kretzoii* Kessler, 2009; *Rallus aquaticus*, *Gallinago veterior* Jánossy, 1979; *Cuculus csarnotanus* Jánossy, 1979; *Bubo bubo*, *Aegolius* sp., *Glaucidium baranensis* Kessler, 2009; *Athene noctua veta* Jánossy, 1992; *Apus baranensis* Jánossy, 1977; *Pyrrhocorax graculus vetus* Kretzoi 1962; *Corvus harkanyensis* Kessler, 2009; *Miocorvus larteti* Milne-Edwards, 1871; *Pica pica major* Jánossy, 1979; *Turdoides borealis* Jánossy, 1979; Passeriformes indet, Aves indet (Kessler 2009a, 2009b, 2010a), while from Csarnóta 4 (collected by L. Pongrácz): *Tetrao partium* (by Kessler 2009a). Following these new determinations, the following new extinct songbird species have been identified and described: *Galerida pannonica* Kessler, 2013; *Lullula parva* Kessler, 2013; *Hirundo major* Kessler, 2013; *Delichon pusillus* Kessler, 2013; *Aegithalos congruis* Kessler, 2013; *Parus robustus* Kessler, 2013; *Parus parvulus* Kessler, 2013; *Sitta pusilla* Kessler, 2013; *Certhia immensa* Kessler, 2013; *Saxicola baranensis*

Kessler, 2013; *Saxicola parva* Kessler, 2013; *Phoenicurus erikai* Kessler, 2013; *Oenanthe pongraczi* Kessler, 2013; *Turdus major* Kessler, 2013; *Turdus medius* Kessler, 2013; *Turdus praeminor* Kessler, 2013; *Cettia kalmani* Kessler, 2013; *Acrocephalus kretzoii* Kessler, 2013; *Acrocephalus kordosi* Kessler, 2013; *Sylvia pusilla* Kessler, 2013; *Locustella janossyi* Kessler, 2013; *Phylloscopus pliocaenicus* Kessler, 2013; *Anthus baranensis* Kessler, 2013; *Cinclus minor* Kessler, 2013; *Prunella kormosi* Kessler, 2013; *Lanius hungaricus* Kessler, 2013; *Passer minusculus* Kessler, 2013; *Carduelis parvulus* Kessler, 2013; *Carduelis medius* Kessler, 2013; *Pyrrhula minor* Kessler, 2013; *Fringilla petényii* Kessler, 2013; *Loxia csarnotanus* Kessler, 2013; *Pinicola kubinyii* Kessler, 2013; *Emberiza media* Kessler, 2013; *Emberiza parva* Kessler, 2013 (Kessler 2013a, 2013b).

Abrevations: Q1-Q2-Lower Pleistocene; Q3 (Q3/I-Q3/II)-Middle Pleistocene; Q4 (Q4/I-Q4/II)-Upper Pleistocene; †-extinct/fossil species-subspecies. A-total lengths; B-partial lengths; C-breadth of proximal epiphysis; C1-partial breadth of proximal epiphysis; D-thickness of proximal epiphysis; E-breadth of diaphysis; E1-partial breadth of diaphysis; F-breadth of distal epiphysis; G-thickness of distal epiphysis; H-height of distal epiphysis. Hungarian Institute of Geology and Geophysics (HIGG); „Beszélő Kövek Alapítvány” Harkány (BKAH)
(Note: Q3/I = Upper Bihar stage; Q3/II = Pilis stage – Solymar substage; Q4/I= Pilis stage – Szanto substage; Q4/II = Holocen)

Anatomical terminology: after Lambrecht (1933), Baumel *et al.* (1979), Gilbert *et al.* (1981), Kessler (2013a)

Method of measurement: after von den Driesch (1976), Kessler (2013b)

Systematics

Ord. Galliformes (Temminck, 1820)
Fam. Phasianidae (Vigors, 1825)
Pliogallus Gaillard, 1939
Pliogallus csarnotanus n. sp. *(Figure 1/1–4, 2/5–6)*

Locus typicus and stratum typicum: Csarnóta 4 (Hungary), Pliocene (MN15)

Material: *carpometacarpus* (holotype), 4 *phalanga pedis*, 2 *phalanx ungualis* (paratypes).

Dimensions (in mm): *carpometacarpus*: A-33.47; B-29.88; C-11.51; D-7.15; E-9.63; E1-3.99; F-6.81; G-3.72; *phalanga pedis* (4): A-7.78, 14, 18; *phalanx ungualis* (2): A-10.61 and 12.03.

Derivatio nominis: „*csarnotanus*” from the name of locality.

Diagnosis of species: A big Phasianidae species, bigger than *Gallus beremendensis* identified by D. Jánossy in 1997 from the Early Pleistocene of Beremend 5. The *metacarpus* differs from the recens *Gallus gallus domesticus* in the following:
– the proximal edge of the *trochlea carpalis* less outstanding *(Figure 1/1a)*;
– the *processus extensorius* is narrowed and the end is curved upwards *(Figure 1/1b)*;

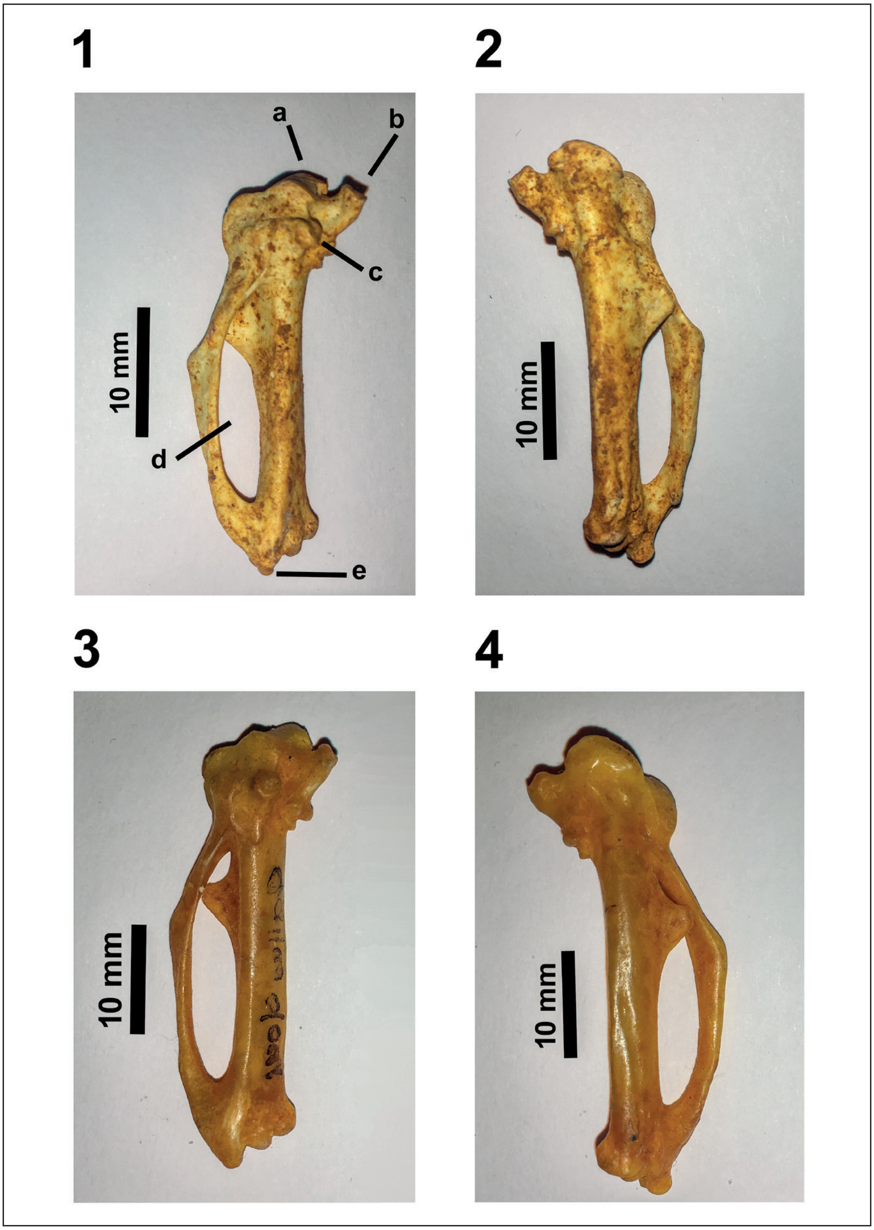

Figure 1. 1. *Pliogallus csarnotanus* n. sp. – Csarnóta 4. *carpometacarpus* (left side, ventral aspect); 2. *Pliogallus csarnotanus* n. sp. – Csarnóta 4. *carpometacarpus* (left side, dorsal aspect); 3. *Gallus gallus domesticus* (L. 1758) extant (left side, ventral aspect); 4. *Gallus gallus domesticus* (L. 1758) extant (left side, dorsal aspect); a – proximal edge of the *trochlea carpalis*; b – the end of the *processus extensorius*; c – the form of *processus pisiformis*; d – the form of *spatium intermetacarpalis*; e – the protruding end of *os metacarpale majus*

1. ábra 1. *Pliogallus csarnotanus* n. sp. – Csarnóta 4, kézközépcsont (baloldali, ventrális nézet); 2. *Pliogallus csarnotanus* n. sp. – Csarnóta 4, kézközépcsont (baloldali, dorzális nézet); 3. *Gallus gallus domesticus* (L. 1758) recens kézközépcsont (baloldali, ventrális nézet); 4. *Gallus gallus domesticus* (L. 1758) recens kézközépcsont (baloldali, dorzális nézet); a – *trochlea carpalis* proximális éle; b – a *processus extensorius* hegye; c – a *processus pisiformis* alakja; d – a *spatium intermetacarpalis* alakja; e – *os metacarpale majus* kiugró vége

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– the *processus pisiformis* is more prominent *(Figure 1/1c)*;
– the *spatium intermetacarpalis* is narrower *(Figure 1/1d)*;
– the protruding end of *os metacarpale majus* is longer *(Figure 1/1e)*.

Description: robust skeletal part of the same size as the extant Domestic Hen (*Gallus gallus domesticus* L. 1758), differing in a few morphological features. This is presumably related to its better flight abilities. The character of the claw bone indicates clinging to wood.

The genus was established in 1939 by C. Gaillard on the basis of two *tarsometatarsus* from Csarnóta, Hungary, describing the species as *Pliogallus crassipes* and *P. kormosi*. In 1975, D. Jánossy examined the material in the collection of the University of Lyon and concluded that the bones of extant hen *tarsometatarsus* were treated with chemicals (Jánossy 1976b, Mlíkovskỳ 2002). In the early 2000s, L. Pongrácz re-excavated Csarnóta 4 and found some bird bones, which he gave to J. Kessler for identification. Among them were a *carpometacarpus* and some foot toes and claw bones. They show typical chicken characteristics, both morphologically and in size. They are yellowish brown in colour, which is quite typical of the fossil remains of the site. We do not take a position on Jánossy's opinion of the original material but accept the genus name with the new species name. The genus is not known from sites in Europe outside the Carpathian Basin.

Palaeocryptonix **(Depéret, 1892)**

Palaeocryptonix hungaricus **Jánossy, 1991 (syn: *Eurobambusicola turolicus* Zelenkov, 2016)** *(Figure 2/1–3)*

Site and era: Beremend 26 (Hungary). Pliocene (MN15)

Material: damaged *coracoideum*, distal fragment of *tibiotarsus* (Beremend 26)

Dimensions (in mm): *coracoideum*: B-ab. 23.50, C-5.11, E-2.89; *tibiotarsus*: E-2.37, F-5.13, G-5.41.

Figure 2. 1. *Palaeocryptonix hungaricus* (Jánossy, 1991) – Beremened 26, *coracoideum* (right side, dorsal surface); 2. *Palaeocryptonix hungaricus* (Jánossy, 1991) – Beremend 26. *coracoideum* (right side, medial aspect); 3. *Palaeocryptonix hungaricus* (Jánossy,1991) – Csarnóta 2. *tibiotarsus* (fragment distal, right side, cranial aspect); 4. Perdicidae indet. – Csarnóta 2. *coracoideum* (fragment proximal, right side, dorsal surface); 5. *Pliogallus csarnotanus* n. sp. – Csarnóta 4. *carpometacarpus* (left side, ventral aspect); 6. *Pliogallus csarnotanus* n. sp. – Csarnóta 4. *phalanx ungualis* (lateral aspect); 7. *Porzana* sp. – Csarnóta 2. *humerus* (fragment proximal, right side, caudal surface); 8. *Porzana* sp. – Csarnóta 2. *humerus* (fragment proximal, right side, cranial surface); 9. *Galinulla* sp. – Csarnóta 2. *carpometacarpus* (fragment distal, right side, ventral aspect); 10–12. *Glaucidium baranensis* (Kessler, 2010) – Csarnóta 2. *phalanges ungualis* (lateral aspect)

2. ábra 1. *Palaeocryptonix hungaricus* (Jánossy, 1991) – Beremened 26. hollócsőrcsont (jobboldali, dorzális nézet); 2. *Palaeocryptonix hungaricus* (Jánossy, 1991) – Beremend 26. hollócsőrcsont (jobboldali, mediális nézet); 3. *Palaeocryptonix hungaricus* (Jánossy, 1991) – Csarnóta 2. lábszárcsont (jobboldali, disztális töredék, dorzális nézet); 4. Perdicidae indet. – Csarnóta 2. hollócsőrcsont (jobboldali, proximális töredék, mediális nézet); 5. *Pliogallus csarnotanus* n. sp. – Csarnóta 4. kézközépcsont (baloldali, ventrális nézet); 6. *Pliogallus csarnotanus* n. sp. – Csarnóta 4. karomcsont (oldalnézet); 7. *Porzana* sp. – Csarnóta 2. felkarcsont (jobboldali, proximális töredék, caudális nézet); 8. *Porzana* sp. – Csarnóta 2. felkarcsont (jobboldali, proximális töredék, craniális nézet); 9. *Galinulla* sp. – Csarnóta 2. hollócsőrcsont (jobboldali, disztális töredék, ventrális nézet); 10–12. *Glaucidium baranensis* (Kessler, 2010) – Csarnóta 2. karomcsontok (oldalnézet)

A species of larger quail-sized hen, quite common in the Late Miocene and Early Pliocene sites of the Carpathian Basin. The only almost complete skeleton in Hungary was also provided by this species in the Upper Miocene from Northern Hungary (Rátka, in the Encsi private museum from Tállya). Mlíkovskỳ assigns the genus and the species to *Alectoris donnezani* (Depéret, 1892), while in 2016, N. Zelenkov establishes a new genus and species *Eurobambusicola turolicus* based on material in the collection of the Museum of the HIGG. For our part, both attempts are met with scepticism (Jánossy 1991, Mlíkovskỳ 2002, Kessler 2009b, 2013a, Zelenkov 2016).

Subfam. Perdicinae (Horsfield, 1821)

Perdicinae gen. *et* sp. indet. *(Figure 2/4)*

Site and era: Csarnóta 2 (Hungary), Pliocene (MN15)

Material: proximal fragment of *coracoideum*

Dimension (in mm): C-ab. 3.10

Description: the very fragmentary *coracoidum* remains undoubtedly belong to the Phasianidae family, but nothing more can be determined. It is probably from a quail-sized species.

Rallidae (Vigors, 1825)

***Gallinula* (Brisson, 1860)**

***Gallinula* sp.** *(Figure 2/9)*

Site and era: Csarnóta 2 (Hungary), Pliocene (MN15)

Material: distal fragment of *carpometacarpus*

Dimensions (in mm): E-4.69, E1-2.64, F-3.60, G-3.16

The fragmentary material does not allow a species identification, as only one fossil species from the European Neogene is known in the literature: the species *Gallinula balcanica* Boev, 1999 from the Late Pliocene of Bulgaria (Varshets, MN17), identified from an *ulna* (Boev 1999, Mlíkovskỳ 2002). The presence of the genus in Csarnóta is of considerable value.

***Porzana* (Vieillot, 1826)**

***Porzana* sp.** *(Figure 2/7–8)*

Site and era: Csarnóta 2 (Hungary), Pliocene (MN15)

Material: proximal fragment of *humerus*

Dimension (in mm): C-5.53

Its features suggest the crakes, and its size the smaller species of the genus. Crakes are very rarely found in the European Neogene. Three of the oldest species are known from the Middle and the Late Miocene and the Early Pliocene in Hungary: *Porzana matraensis* Kessler, 2009; *P. kretzoii* Kessler, 2009; *P. estramosi* Jánossy, 1979 (from Mátraszőlős, Polgárdi and Osztramos). From European sites outside the Carpathian Basin, species of the genus have been reported only from the Late Pliocene – Early Pleistocene sites in Spain (Mallorca), Bulgaria (Varshets) and Czech Republic (Stránská skála) (Jánossy 1979b, Mlíkovskỳ 2002, Kessler 2009b).

Ord. Charadriiformes (Huxley, 1867)
Charadriiformes fam, gen. *et* sp. indet. *(Figure 3/9)*
Site and era: Csarnóta 2 (Hungary), Pliocene (MN15)
Material: proximal fragment of *humerus*
Dimension (in mm): C-6.20

A remain from a small species (*Charadrius – Calidris* size), which is compatible with the order but not suitable for a closer taxonomic classification because of its worn markings.

Representatives of this order are not very common in the European Neogene. From the Carpathian Basin, *Gallinago veterior* Jánossy, 1979 (Polgárdi, MN13; Csarnóta 2, MN15); *Tringa* sp. (Polgárdi, MN13; Beremend 26, MN15); *Scolopax baranensis* Jánossy, 1979 (Csarnóta 2, MN15); *Charadrius lambrechti* Kessler, 2009 (Polgárdi, MN13); *Calidris janossyi* Kessler, 2009 (Polgárdi, MN13); *Chlidonias* sp. (Beremend 26, MN15) are known (Jánossy 1979a, Mlíkovskỳ 2002, Kessler 2009b).

As the *humerus* is not included in the listed finds, no comparison is possible.

Ord. Columbiformes (Latham, 1790)
Fam. Columbidae (Illiger, 1811)
Columbidae sp. indet. *(Figure 3/1)*
Site and era: Csarnóta 2 (Hungary), Pliocene (MN15)
Material: distal fragment of *humerus*
Dimensions (in mm): F-6.67; G-4.37

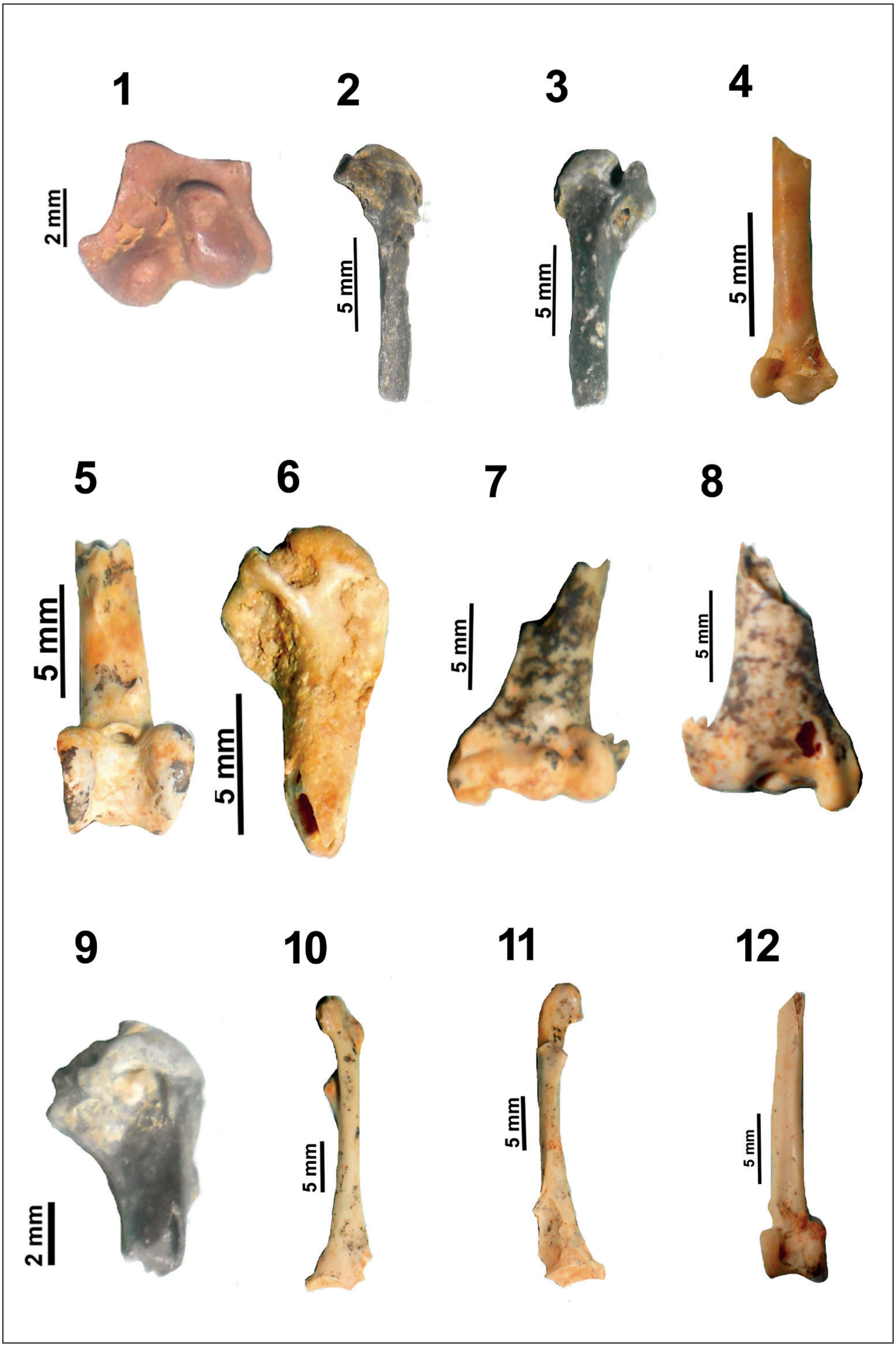
A remain from a small (dove-sized) species of the family.

The pigeons are known only at the level of the recurrent genus from the Neogene of Europe: *Columba* sp. from Bulgaria (Varshets, M17) and from Croatia (Sandalja, MN17) (Mlíkovskỳ 2002). From the Carpathian Basin, it has been identified to the family level from the Middle Miocene of Mátraszőlős 2 (MN7–8) on the basis of a distal fragment of a *tibiotarsus* (Kessler & Hír 2012a).

Ord. Strigiformes (Wagler, 1830)
Fam. Strigidae (Vigors, 1825)
***Glaucidium* (Boie, 1826)**
***Glaucidium baranensis* Kessler, 2010** *(Figure 2/10–12)*
Site and era: Csarnóta 2 (Hungary), Pliocene (MN15)
Material: 3 *phalanx ungualis*
Dimensions (in mm): A- 6.65, 7.99 and 9.52; C-3.35, 3.50 and 4.35

A species of owl that largely matches the size and character of the extant European Pygmy Owl, which may have been the ancestor of the extant species in Europe and thus, in the Carpathian Basin. From the Early Pliocene of Csarnóta 2 and Beremend 26, the distal end of a *humerus*, the sternal fragment of a *coracoideum* (Csarnóta 2, MN15) and the proximal half of a *humerus* (Beremend 26, MN15) have been described (Kessler 2010).

Considering that the claw bones are from a small owl, it is reasonable to assume that they are the remains of fossil Pygmy Owls (*Glaucidium passerinum* (L.1758)) described from the same site.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Jánossy (1974) reports another find of *Glaucidium* sp. from the Late Pliocene of Poland (Rebielice, MN16), otherwise only the extant species is known in fossil material from the Early Pleistocene onwards. The genus is known from only one Late Pliocene site in Florida (USA) (Inglis, Cytrus County, Florida) with one described species: *G. explorator* Emslie, 1998, with numerous skeletal parts but only a fragmentary proximal *humerus*, corresponding to the size of the extant *G. brasilianum*. A *tibiotarsus* is also reported from the same material, with dimensions similar to those of extant *G. minutissimum* but defined only to genus level. An extinct species described from the Pleistocene of the Bahamas as *G. dickinsoni* Brodkorb, 1959, by a *tibiotarsus*, was subsequently (Olson 1985) synonymised with the extant species *Speotyto cunicularia* (Molina, 1782). The ancestor of the Pygmy Owl was already present in the Eocene. It has been described from the Middle Eocene of Germany (Geiseltal and Messel) from several *humerus* specimens under the name *Eoglaucidium pallas* Fischer, 1987 (Fischer 1987, Mlíkovskỳ 2002).

Ord. Coraciiformes (Forbes, 1884)
Fam. Meropidae (Vigors, 1825)
***Merops* (Linnaeus, 1758)**
***Merops* sp.** *(Figure 3/6)*
Site and era: Csarnóta 2 (Hungary), Pliocene (MN15)
Material: proximal fragment of *humerus*
Dimensions (in mm): C-5.55; E-1.64

Figure 3. 1. Columbidae gen. *et* sp. indet. – Csarnóta 2. *humerus* (fragment distal, left side, caudal surface); 2. Coraciiformes fam, gen. *et* sp. indet. – Csarnóta 2. *humerus* (fragment proximal, left side, caudal surface); 3. Coraciiformes fam, gen. *et* sp. indet. – Csarnóta 2. *humerus* (fragment proximal, left side, cranial surface); 4. Coraciiformes fam, gen. *et* sp. indet. – Csarnóta 2. *ulna* (fragment distal, left side, dorsal aspect); 5. Coraciiformes fam, gen. *et* sp. indet. – Csarnóta 2. *tibiotarsus* (fragment distal, right side, cranial aspect); 6. Meropidae gen. *et* sp. indet. – Csarnóta 2. *humerus* (fragment proximal, right side, caudal surface); 7. *Nucifraga* sp. – Beremend 26. *humerus* (fragment distal, left side, caudal surface); 8. *Nucifraga* sp. – Beremend 26. *humerus* (fragment distal, left side, cranial surface); 9. Charadriiformes fam, gen. *et* sp. indet. – Csarnóta 2. *humerus* (fragment proximal, right side, caudal surface); 10 *Garrulus* sp. – Beremend 26. *coracoideum* (right side, medial aspect); 11. *Garrulus* sp. – Beremend 26. *coracoideum* (right side, dorsal surface); 12. *Garrulus* sp. – Csarnóta 2. *tibiotarsus* (distal fragment, right side, cranial aspect)

3. ábra 1. Columbidae gen. *et* sp. indet. – Csarnóta 2. felkarcsont (baloldali, disztális töredék, caudális nézet); 2. Coraciiformes fam, gen. *et* sp. indet. – Csarnóta 2. felkarcsont (baloldali, proximális töredék, caudális nézet); 3. Coraciiformes fam, gen. *et* sp. indet. – Csarnóta 2. felkarcsont (baloldali, proximális töredék, craniális nézet); 4. Coraciiformes fam, gen. *et* sp. indet. – Csarnóta 2. singcsont (baloldali, disztális töredék, dorzális nézet); 5. Coraciiformes fam, gen. *et* sp. indet. – Csarnóta 2. lábszárcsont (jobboldali, disztális töredék, craniális nézet); 6. Meropidae gen. *et* sp. indet. – Csarnóta 2. felkarcsont (jobboldali, proximális töredék, caudális nézet); 7. *Nucifraga* sp. – Beremend 26. felkarcsont (baloldali, disztális töredék, caudális nézet); 8. *Nucifraga* sp. – Beremend 26. felkarcsont (baloldali, disztális töredék, craniális nézet); 9. Charadriiformes fam, gen. *et* sp. indet. – Csarnóta 2. felkarcsont (jobboldali, disztális töredék, caudális nézet); 10. *Garrulus* sp. – Beremend 26. hollócsőrcsont (jobboldali, mediális nézet); 11. *Garrulus* sp. – Beremend 26. hollócsőrcsont (jobboldali, dorzális nézet); 12. *Garrulus* sp. – Csarnóta 2. lábszárcsont (jobboldali, disztális töredék, craniális nézet)

The genus is not known from the Tertiary period of Europe. From the Carpathian Basin, it is known only in the Middle Miocene of Croatia (Radoboj, MN7) (von Meyer 1865, Mlíkovskỳ 1997); in the Late Miocene of Hungary (Rudabánya, MN9) (Kessler 2010a) and in the Early Pleistocene of Romania (Betfia 9) (Gál 2002). In addition, it is known from Europe only from the Late Pleistocene of France (Combe Grenal, Salpétre a Pompignan, Q4/I) (Mourer-Chauviré 1975, Tyrberg 1998).

Coraciiformes fam, gen *et* sp. indet. *(Figure 3/2–5)*
Site and era: Csarnóta 2 (Hungary), Pliocene (MN15)
Material: proximal and distal fragment of *humerus*; distal fragment of *tibiotarsus*
Dimensions (in mm): *humerus*: C-5.18; E-1.77 and 1.80; F-3.20; *tibiotarsus*: E-1.94, F-3.23, G-2.95

Because of their conservation status, only their membership of the order can be established.

The distal fragment of *humerus* is a bone from a species with typical Coraciiform features, but differing in both size and morphology from species of known European families.

Ord. Passeriformes (Linnaeus, 1758)
Fam. Corvidae (Vigors, 1825)
***Garrulus* (Vieillot, 1816)**
***Garrulus* sp.** *(Figure 3/10–12)*
Site and era: Beremend 26 (Hungary), Pliocene (MN15)
Material: damaged distal fragment of *coracoideum*, distal fragment of *tibiotarsus*
Dimensions (in mm): *coracoideum*: B-about 28.08, C-5.50, E-1.75; *tibiotarsus*: E-5.48, F-5.40, G-5.06

The extant species of jay (*Garrulus glandarius* (L. 1758)) and the earliest record of the genus is from the Late Pliocene of France (Mountoussé) (Clot *et al*. 1976a, 1976b). It is much younger than the Beremend find. The other dates are from the Pleistocene (UK, Austria, Czech Republic, Croatia, Romania, Germany, etc.) (Tyrberg 1998, Mlíkovskỳ 2002, Kessler 2013b, 2020).

***Nucifraga* (Vieillot, 1816)**
***Nucifraga* sp.** *(Figure 3/7–8)*
Site and era: Beremend 26 (Hungary), Pliocene (MN15)
Material: distal fragment of *humerus*
Dimensions (in mm): F-9.06, G-4.34

The earliest European records of the modern species (*Nucifraga caryocatactes* (L. 1758)) is from the Late Pliocene of Bulgaria (Varshets) and of Spain (S'Onix in Mallorca) and from the Early Pleistocene of Czech Republic (Stránská skála) (Soondar *et al*. 1995, Mlíkovskỳ 1995, Boev 2000, Kessler 2020).

Fam. Alaudidae (Vigors, 1825)
Alaudidae gen. *et* sp. indet. *(Figure 4/1–3)*
Site and era: Csarnóta 2 (Hungary), Pliocene (MN15)
Material: proximal fragment of *coracoidum*; 5 distal fragment of *ulna*; 2 distal fragment of *tibiotarsus*; distal fragment of *tarsometatarsus.*

Dimensions (in mm): *coracoideum*: C-2,50, D-2.50, E-2.00; *ulna*: E-1.40–2.00, F-2.20 and 3.50; *tibiotarsus*: E-1.70–1.90; *tarsometatarsus*: E-1.70, F-2.10

The family is very well represented by fossil species in the Neogene and Quaternary of the Carpathian Basin: *Galerida cserhatensis* Kessler *et* Hír, 2012 (Litke, MN5); *Praealauda hevesensis* Kessler *et* Hír, 2012 (Felsőtárkány, MN7–8); *Lullula neogradensis* Kessler *et* Hír, 2012 (Mátraszőlős 1, MN7–8); *Alauda tivadari* Kessler, 2013; *Lullula minor* Kessler, 2013; *Calandrella gali* Kessler, 2013 (Polgárdi, MN13); *Lullula parva* Kessler, 2013; *Galerida pannonica* Kessler, 2013 (Csarnóta 2, MN15); *Lullula minuscula* Kessler, 2013; *Lullula parva* Kessler, 2013; *Galerida pannonica* Kessler, 2013; *Melanocorypha minor* Kessler, 2013 (Beremend 26, MN15, Kessler 2020).

From areas outside the Carpathian Basin: the *Galerida* genus was reported outside the Carpathian Basin from the Upper Pliocene of Bulgaria (Varshets, MN17) as *Galerida bulgarica* Boev, 2012 (Boev 2012), the *Melanocorypha* genus was reported from the Upper Miocene and Upper Pliocene of Bulgaria: *Melanocorypha serdicensis* Boev, 2012 (Hrabarsko) and *Melanocorypha donchevi* Boev, 2012 (Varshets) (Boev 2012), the *Alauda* genus was reported from the Upper Pliocene of Bulgaria (Varshets, MN17) as *Alauda xerarvensis* Boev, 2012 (Boev 1996, 2012), the *Lulullа* genus was reported from the Late Miocene of Bulgaria (Chrabarsko) as *Lullula* sp. (Boev 2000), and from the Late Pliocene-Early Pleistocene as *Lullula slivnicensis* Boev, 2012 (Slivnica, MN17) and *L. balcanica* Boev, 2012 (Varshets, MN18) based on other skeletal types (Boev 1996, 2012), the *Eremophila* genus was reported from the Late Miocene of Bulgaria Chrabarsko as *Lullula* sp. (Boev 2000), and from the Late Pliocene – Early Pleistocene as *Lullula slivnicensis* Boev, 2012 (Slivnica, MN17) and *L. balcanica* Boev, 2012 (Varshets, MN18) based on other skeletal types (Boev 1996, 2012). Several Pliocene finds are reported as extant species from the Czech Republic, France, Spain and Russia (Kessler 2020).

Fam. Hirundinidae (Vigors, 1825)

Hirundinidae gen. *et* sp. indet. *(Figure 4/4)*

Site and era: MN15: Csarnóta 2 (Hungary)

Material: distal, fragment of *ulna*; proximal fragment of *carpometacarpus*; *phalanx ungualis*

Dimensions (in mm): *ulna*: E-2.06, F-3.36, G-2.56; *carpometacarpus*: C-4.32; *phalanx ungualis*: A-3.44

The earliest described representatives of this family from the Carpathian Basin are from the late Miocene of Polgárdi (MN13): *Hirundo gracilis* Kessler, 2013; *Delichon polgardiensis* Kessler, 2013; *Riparia minor* Kessler, 2013; and from the Pliocene of Csarnóta 2: *Hirundo major* Kessler, 2013; *Delichon pusillus* Kessler, 2013; and from Beremend 26; *Delichon major* Kessler, 2013 (Kessler 2020).

From the area outside the Carpathian Basin they are known only from the Early Pleistocene through modern species.

Fam. Panuridae (Des Murs, 1860)

Panuridae gen. *et* sp. indet. *(Figure 4/5)*

Site and era: MN15: Csarnóta 2 (Hungary)

Material: distal fragment of *ulna*

Dimension (in mm): F-3.41

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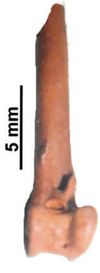

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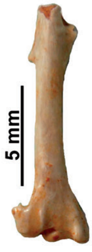

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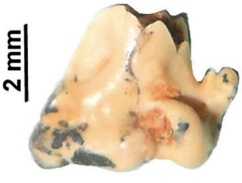

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It is the only fossil representative of the family in Europe. Modern species from European sites outside the Carpathian Basin have been reported from the Late Pleistocene of Germany (Scythenloch-Bayern) (Tyrberg 1998).

Fam. Paridae (Vogors, 1825)

Paridae gen. *et* sp. indet.

Site and era: MN15: Csarnóta 2 (Hungary)

Material: distal fragment of *ulna*

Dimension (in mm): F-2.00

From the Carpathian Basin, the family is earliest known from the Late Miocene of Polgárdi (MN13) as *Aegithalos gaspariki* Kessler, 2013; and from the Pliocene (MN15) from Csarnóta 2 as *Aegithalos congruis* Kessler, 2013; *Parus robustus* Kessler, 2013; and *Parus parvulus* Kessler, 2013; and from Beremend 26 as *Parus medius* Kessler, 2013 (Kessler 2020).

Figure 4. 1. Alaudidae gen. *et* sp. indet. – Csarnóta 2. *coracoideum* (fragment proximal, left side, medial aspect); 2. Alaudidae gen. *et* sp. indet. – Csarnóta 2. *ulna* (fragment distal, left side, dorsal aspect); 3. Alaudidae gen. *et* sp. indet. – Csarnóta 2. *tibiotarsus* (fragment distal, left side, cranial aspect); 4. Hirundinidae gen. *et* sp. indet – Csarnóta 2. *carpometacarpus* (fragment, proximal, right side, ventral aspect); 5. Panuridae gen. *et* sp. indet. – Csarnóta 2. *ulna* (fragment distal, right side, dorsal aspect); 6. Muscicapidae gen. *et* sp. indet. – Csarnóta 2. *coracoideum* (fragment proximal, right side, dorsal aspect); 7. Muscicapidae gen. *et* sp. indet. – Csarnóta 2. *humerus* (fragment distal, right side, cranial surface); 8. Muscicapidae gen. *et* sp. indet. – Csarnóta 2. *tarsometatarsus* (fragment distal, left side, dorsal aspect); 9. Turdidae gen. *et* sp. indet. – Csarnóta 2. *humerus* (fragment distal, left side, cranial surface); 10. Turdidae gen. *et* sp. indet. – Csarnóta 2. *tibiotarsus* (fragment distal, right side, cranial surface); 11. Sylviidae gen. *et* sp. indet. – Csarnóta 2. *humerus* (fragment distal, right side, cranial surface); 12. Motacillidae gen. *et* sp. indet. – Csarnóta 2. *humerus* (fragment proximal, right side, caudal surface); 13. Motacillidae gen. *et* sp. indet. – Csarnóta 2. *ulna* (fragment distal, right side, dorsal aspect); 14. Sittidae gen. *et* sp. indet. – Csarnóta 2. *humerus* (left side, caudal surface); 15. Laniidae gen. *et* sp. indet. – Csarnóta 2. *humerus* (fragment distal, left side, cranial surface); 16. Fringillidae gen. *et* sp. indet. – Csarnóta 2. *coracoideum* (fragment proximal, rights ide, dorsal surface)

4. ábra 1. Alaudidae gen. *et* sp. indet. – Csarnóta 2. hollócsőrcsont (baloldali, proximális töredék, dorzális nézet); 2. Alaudidae gen. *et* sp. indet. – Csarnóta 2. singcsont (baloldali, disztális töredék, dorzális nézet); 3. Alaudidae gen. *et* sp. indet. – Csarnóta 2. singcsont (baloldali, disztális töredék, craniális nézet); 4. Hirundinidae gen. *et* sp. indet – Csarnóta 2. kézközépcsont (jobboldali, proximális töredék, ventrális nézet); 5. Panuridae gen. *et* sp. indet. – Csarnóta 2, singcsont (jobboldali, disztális töredék, dorzális nézet); 6. Muscicapidae gen. *et* sp. indet. – Csarnóta 2. hollócsőrcsont (jobboldali, proximális töredék, dorzális nézet); 7. Muscicapidae gen. *et* sp. indet. – Csarnóta 2. felkarcsont (jobboldali, disztális töredék, craniális nézet); 8. Muscicapidae gen. *et* sp. indet. – Csarnóta 2. csüd (jobboldali, disztális töredék, dorzális nézet); 9. Turdidae gen. *et* sp. indet. – Csarnóta 2. felkarcsont (baloldali, disztális töredék, craniális nézet); 10. Turdidae gen. *et* sp. indet. – Csarnóta 2. lábszárcsont (jobboldali, disztális töredék, craniális nézet); 11. Sylviidae gen. *et* sp. indet. – Csarnóta 2. felkarcsont (jobboldali, disztális töredék, craniális nézet); 12. Motacillidae gen. *et* sp. indet. – Csarnóta 2. felkarcsont (jobboldali, proximális töredék, caudális nézet); 13. Motacillidae gen. *et* sp. indet. – Csarnóta 2. singcsont (jobboldali, disztális töredék, dorzális nézet); 14. Sittidae gen. *et* sp. indet. – Csarnóta 2. felkarcsont (jobboldali, caudális nézet); 15. Laniidae gen. *et* sp. indet. – Csarnóta 2. felkarcsont (baloldali, disztális töredék, caudális nézet); 16. Fringillidae gen. *et* sp. indet. – Csarnóta 2. hollócsőrcsont (jobboldali, proximális töredék, dorzális nézet)

The family is known outside the Carpathian Basin only from the Late Pliocene of Bulgaria (Varshets, MN17) as *Parus* sp. (Boev 2000, Kessler 2020).

Fam. Sittidae (Bonaparte, 1831)

Sittidae gen. *et* sp. indet. *(Figure 4/14)*

Site and era: MN15: Csarnóta 2 (Hungary)

Material: damaged distal of *humerus*

Dimensions (in mm): A-ap.19.30, C-4.08, E-1.35

From the Carpathian Basin, the representative of the family is known only from the Late Miocene of Polgárdi (MN13), as *Sitta gracilis* Kessler, 2013; then from the Pliocene of Csarnóta 2 (MN15) as *Sitta pusilla* Kessler, 2013; and the modern species from several Pleistocene sites (Kessler 2020).

The family is known outside the Carpathian Basin only from the Early Pliocene (MN16) of Poland (Rebielice Królowskie I.) as *Sitta* sp. (Jánossy 1974) and from the Late Pliocene of Bulgaria (Varshets, MN17) (Boev 1996, 2000, Kessler 2020), and the modern species as *Sitta europaea* from the Pleistocene.

Fam. Certhiidae (Vigors, 1825)

Certhiidae gen. *et* sp. indet.

Site and era: MN15: Csarnóta 2 (Hungary)

Material: distal fragment of *tarsometatarsus*

Dimensions (in mm): E-1.00, F-1.70

On the distal epiphysis of the third finger trochlea (*trochlea metatarsi* III) of the snout is uniquely deep, which is unique to this genus.

The genus and the family are known from the late Miocene in the Carpathian Basin, from which *Certhia janossyi* Kessler *et* Hír, 2012 was described from Rudabánya (MN9), while *Certhia immensa* Kessler, 2012 was described from Pliocene of Csarnóta 2 (Kessler 2020). The modern species is reported from the Pleistocene in Europe.

Fam. Muscicapidae (Vigors, 1825)

Muscicapidae gen. *et* sp. indet. *(Figure 4/6–8)*

Site and era: MN15: Csarnóta 2 (Hungary)

Material: proximal fragment of *coracoideum*, distal fragment of *humerus*, distal fragment of *ulna*, distal fragment of *tarsometatarsus*, 4 *phalanx ungualis*

Dimensions (in mm): *coracoideum*: C-2.40, D-2.00, E-0.80; *humerus*: E-1.30, F-3.80; *ulna*: F-3.21; *tarsometatarsus*: E-1.20, F-2.50; *phalanx ungualis*: A-4.25 and 5.28

Their earliest representatives are known from the Early Miocene from sites in the Carpathian Basin. Thus, the *Luscinia praeluscinia* Kessler *et* Hír, 2012 (Litke, MN5), *Luscinia jurcsaki* Kessler *et* Venczel, 2011 (Kőalja/Subpiatra, Romania, MN6), *Muscicapa leganyii* Kessler *et* Hír, 2012 (Felsőtárkány, Felnémet, MN7–8), *Erithacus horusitskyi* Kessler *et* Hír, 2012 (Mátraszőlős 1, MN7–8), then a *Muscicapa miklosi* Kessler, 2013; *Luscinia denesi* Kessler, 2013; *Saxicola lambrechti* Kessler, 2013; *Oenanthe kormosi* Kessler, 2013 (Polgárdi, MN13), *Muscicapa petényii* Kessler, 2013; *Erithacus minor* Kessler, 2013; *Luscinia pliocaenica* Kessler, 2013; *Saxicola magna* Kessler, 2013; *Monticola pongraczi* Kessler, 2013; *Phoenicurus baranensis* Kessler, 2013 (Beremend 26, MN15); *Saxicola baranensis* Kessler, 2013; *S. parva* Kessler, 2013; *Phoenicurus* *erikai* Kessler, 2013; *Oenanthe pongraczi* Kessler, 2013 (Csarnóta 2, MN15) (Kessler 2020).

The fossil representatives of the family are not known outside the Carpathian Basin. The modern species in Europa is known only from Early Pleistocene (Austria, Bulgaria, Croatia, Cyprus, Czech Republic, France, Germany, Italy, Poland, Spain (Tyrberg 1998)).

Fam. Turdidae (Rafinesque, 1815)

Turdidae gen. *et* sp. indet. *(Figure 4/9–10)*

Site and era: MN15: Beremend 26, Csarnóta 2 (Hungary)

Material: Beremend: distal fragment of *humerus*; distal fragment of *tibiotarsus*; Csarnóta: distal fragment of *tibiotarsus*; 3 *phalanx ungualis*

Dimensions (in mm): Beremend: *humerus*: F-6.40, G-2.66; *tibiotarsus*: E-1.96, F-4.05, G-3.86; Csarnóta 2: *tibiotarsus*: E-1.67, F-3.57, G-3.48; *phalanx ungualis*: A-6.10 and 7.60, C-3.15 and 3.54.

The earliest Carpathian Basin thrush finds date from the Middle Miocene, such as the: *Turdicus matraensis* Kessler *et* Hír, 2012 (Mátraszőlős 3, MN7–8), then a *Turdicus pannonicus* Kessler, 2013; *Turdus miocaenicus* Kessler, 2013 (Polgárdi, MN13); *Turdus major* Kessler, 2013; *T. medius* Kessler, 2013; *T. praeminor* Kessler, 2019 (Csarnóta, MN15), *Turdicus tenuis* Kretzoi, 1962 (Betfia-Romania, Early Pleistocene) (Kessler 2020).

The family is known outside of the Carpathian Basin from the Middle Miocene of Romania (Credinta, MN8) as *Turdus* sp. (Gál & Kessler 2006), while from the Late Pliocene from of Poland (Rebielice Królowskie I.), (Jánossy 1974), Bulgaria (Varshets), (Boev 1996, 2000), Croatia (Sandalja I.) (V. Malez-Bacic 1979, Kessler 2020).

Fam. Sylviidae (Vigors, 1825)

Sylviidae gen. *et* sp. indet. *(Figure 4/11)*

Site and era: MN15: Beremend 26 (Hungary).

Material: Beremend 26: damaged distal fragment of *humerus*; Csarnóta 2: distal fragment of *ulna*

Dimensions (in mm): Beremend: *humerus*: A-ap. 15.20, E-1.62, F-3.81, G-1.78 and F-3.15, G-1.72; Csarnóta 2: *ulna*: E-2.06, F-3.36, G-2.56

The family is richly represented by fossil finds in the Neogene of the Carpathian Basin. Earliest reports are: *Sylvia* sp. (Kőalja/Subpiatra, Romania, MN6), *Phylloscopus miocaenicus* Kessler *et* Hír, 2012 (Felsőtárkány, MN7–8), *Acrocephalus major* Kessler, 2013; *Acrocephalus minor* Kessler, 2013; *Cettia janossyi* Kessler, 2013; *Hippolais veterior* Kessler, 2013; *Sylvia intermedia* Kessler, 2013; *Locustella kordosi* Kessler, 2013; *Phylloscopus venczeli* Kessler, 2013 (Polgárdi, MN13); *Acrocephalus kretzoii* Kessler, 2013; *Acrocephalus kordosi* Kessler, 2013; *Cettia kalmani* Kessler, 2013; *Sylvia pusilla* Kessler, 2013; *Locustella janossyi* Kessler, 2013; *Phylloscopus pliocaenicus* Kessler, 2013 (Csarnóta 2, MN15); *Sylvia pusilla* Kessler, 2013; *Locustella janossyi* Kessler, 2013; *Locustella magna* Kessler, 2013; *Regulus plioceanicus* Kessler, 2013 (Beremend 26, MN15) (Kessler 2020).

Outside of Carpathian Basin the family is known from the Late Pliocene from Bulgaria (Varshets, MN17) as *Phylloscopus* sp. (Boev 1996, 2000); and as *Regulus bulgaricus* Boev, 1999 (Boev 1999). The modern species is known in Europa from Early Pleistocene (Kessler 2020).

Fam. Motacillidae (Vigors, 1825)

Motacillidae gen. *et* sp. indet. *(Figure 4/12–13)*

Site and era: MN15: Csarnóta 2 (Hungary).

Material: 2 proximal and distal fragment of *humerus*, proximal fragment of *carpometacarpus*, *phalanx ungualis*

Dimensions (in mm): *humerus*: C-5.48, E-2.00, and E-2.14, F-5.47, G-2.48; *carpometacarpus*: C-3.75, E1-1.75; *phalanx ungualis*: A-4.64, C-2.24

The earliest presence of species of the family in the Carpathian Basin is known from the Middle Miocene, as: *Anthus antecedens* Kessler *et* Hír, 2012 (Felsőtárkány, MN7–8); then as: *Anthus híri* Kessler, 2013; *Motacilla intermedia* Kessler, 2013; (Polgárdi, MN13), *Anthus baranensis* Kessler, 2013 (Csarnóta 2, MN15); *Motacilla minor* Kessler, 2013; *M. robusta* Kessler, 2013a, 2013b (Beremend 26, MN15) (Kessler 2020).

Outside the Carpathian Basin, the *Anthus* genus is known from the Upper Pliocene of Poland (Rebielice Królowskie 1, MN16) (Jánossy 1974); Bulgaria (Varshets, MN16, MN17) (Boev 1996, 2000); and the *Motacilla* genus was described from the Upper Pliocene of Bulgaria (Varshets, MN17) by Boev (1996, 2000, Kessler 2020).

Fam. Prunellidae (Richmond, 1907)

Prunellidae gen. *et* sp. indet.

Site and era: MN15: Csarnóta 2 (Hungary).

Material: *phalanx ungualis*

Dimension (in mm): A-4.60

Prunella freudenthali Kessler, 2013 is known from the late Miocene of Polgárdi (MN13); while *Prunella kormosi* Kessler, 2013 is known from the Pliocene of Csarnóta 2 (MN15). The genus is not known outside the Carpathian Basin with fossil species (Kessler 2020).

Fam. Laniidae (Swainson, 1834)

Laniidae gen. *et* sp. indet. *(Figure 4/15)*

Site and era: MN15: Csarnóta 2 (Hungary).

Material: distal fragment of *humerus*, *phalanx ungualis*

Dimensions (in mm): *humerus*: F-4.00; *phalanx ungualis*: A-5.24, C-1.64

The earliest Carpathian Basin record of the family is from the Early Miocene as *Lanius* sp. (Kőalja/Subpiatra, Romania, MN6), then from the Middle Miocene as *Lanius schreteri* Kessler *et* Hír, 2012 (Felsőtárkány, MN7–8), from the Late Miocene as *Lanius capeki* Kessler, 2013 (Polgárdi, MN13), from the Pliocene as *Lanius hungaricus* Kessler, 2013 (Csarnóta 2, MN15); *Lanius major* Kessler, 2013; *L. intermedius* Kessler, 2013 (Beremend 26, MN15); (Kessler 2020).

The family and genus are known outside the Carpathian Basin only from the Late Pliocene of Bulgaria (Varshets, MN17) as *Lanius* sp. (Boev 1996, 2000, Kessler 2020).

Fam. Sturnidae (Vigors, 1825)

Sturnidae gen. *et* sp. indet.

Site and era: MN15: Csarnóta 2 (Hungary).

Material: *phalanx ungualis*

Dimensions (in mm): A-8.27, C-3.47

The family appears first in the Early Miocene in the Carpathian Basin: *Sturnus kretzoii* Kessler *et* Hír, 2012 (Rudabánya, MN9), then in the Late Miocene: *Sturnus brevis* Kessler, 2013 (Polgárdi, MN13), and in the Pliocene: *Sturnus pliocaenicus* Kessler, 2013; *Sturnus baranensis* Kessler, 2013 (Beremend 26, MN15) (Kessler 2020).

The family and genus were described outside the Carpathian Basin as *Sturnus* sp. from the Late-Pliocene and the Early-Pleistocene localities of Bulgaria (Varshets, MN17–MQ1) by Boev (1996, 2000), England (West Runton and Boxgrove, Harrison 1979, Harrison & Stewart 1999) and Czech Republic (Prezletice, Čapek 1917, Jánossy 1983, 1992, Kessler 2020).

Fam. Fringillidae (Leach, 1820)

Fringillidae gen. *et* sp. indet. *(Figure 4/16)*

Site and era: MN15: Csarnóta 2 (Hungary)

Material: 2 proximal fragment of *coracoideum*; *phalanx ungualis*

Dimensions (in mm): *coracoideum*: C-3.10 and 3.50, D-2.70–2.80, E-1.20; *phalanx ungualis*: A-4.59, C-2.80

The family is very well represented by fossil remains in the Carpathian Basin. From the earliest Miocene as: *Carduelis kretzoii* Kessler, 2013; *C. lambrechti* Kessler, 2013; *Pyrrhula gali* Kessler, 2013; *Fringilla kormosi* Kessler, 2013 (Polgárdi, MN13); *Carduelis parvulus* Kessler, 2013; *C. medius* Kessler, 2013; *Pinicola kubinyii* Kessler, 2013; *Pyrrhula minor* Kessler, 2013; *Fringilla petenyii* Kessler, 2013; *Loxia csarnotanus* Kessler, 2013 (Csarnóta 2, MN15); *Coccothraustes major* Kessler, 2013; *Loxia csarnotanus* Kessler, 2013 (Beremend 26, MN15) (Kessler 2020).

The family was described outside of the Carpathian Basin from the Late Pliocene-Early Pleistocene of Bulgaria (Varshets and Cerzenica, MN17–MQ1) by Boev (1996, 2000), Spain (Quibas and S'Onix) by Montoya (1999) and Sondaar *et al*. (1995); France (Mas Ramboult), by Mourer-Chauviré (1995) and Czech Republic (Stránská skála) by Jánossy (1972). The *Coccothraustes* genus was reported with extinct species only from the Upper Pliocene-Early Pleistocene of Bulgaria (Varshets and Slivnita, MN17–Q1) as *Coccothraustes simeonovi* Boev 1998 and *C. balcanicus* Boev, 1998 (Boev 1998). The *Fringilla* genus is known outside of the Carpathian Basin from the Lower Pliocene of Spain (Hostalets de Pierola, MN16) as *Fringilla* sp. (Villalta 1963), from the Late Pliocene-Early Pleistocene of Bulgaria (Varshets, MN17–MQ1), (Boev 1996); Spain (S'Onix-Mallorca) (Sondaar *et al.* 1995) and Ukraine (Tarchankut) (Vojinstvens'kyj 1967) as *F. cf. coelebs* Linnaeus, 1758. (Kessler 2020).

Conclusions

In accordance with the nature of the fossil material, it was possible to identify about half of the bone fragments to at least the order level. Only three taxa were identified to species level, of which one new species (*Pliogallus csarnotanus* n. sp.) and two species already known from the sites (*Palaeocryptonix hungaricus* Jánossy, 1991 and *Glaucidium baranensis*

Kessler, 2010) were identified. Five taxa were identified down to genus level (*Galinulla* sp., *Porzana* sp., *Merops* sp., *Garrulus* sp., *Nucifraga* sp.) while the other 18 were identified only to order or family level.

The three sites in question are close to each other both in age and geographic location. The vast majority of the bones are ochre-yellow, in keeping with the medium, but there are also some very light and almost black specimens. There are few completely intact skeletal parts, and most of these are from the *phalanges pedis* and claw bones.

Beremend 26 is located a few kilometres east of the other two sites, in the vicinity of the Villány Hills, on the Lower Cretaceous limestone of Szőlő Hill, and like the other sites there, its bone material comes from the filling of a karst split. There are currently 39 known former sites from the quarry, which is still in operation, but these sites were destroyed by mining.

The two Csarnóta sites are located in former quarries on the flat surface of Cserhegy, near the namesake settlement. Site 1 was destroyed by road construction and site 3 did not provide bird material.

The faunal assemblage is consistent from a palaeoecological point of view with that reported in previous studies. The bird fauna of both wetlands, open and wooded areas and rock faces is represented in the fossil material.

As the majority of species are small in size, with only Grouse, Hen Harriers and Bustards represented alongside small-medium sized predators, this is an indication of the size of the predators. The remains of Pygmy Owls, Goshawks, Little Owls, Tawny Owls and Eagle Owls also indicate these birds of prey. Of the diurnal predators, only the Sparrow-Hawk, which is also a prey species, is included in the faunistics.

In conclusion, the Pliocene finds from Beremend and Csarnóta reflect the bird life of the southern edge of the Carpathian Basin.

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“Leucism resulting in xanthochroism” – A report on colour aberration in Coppersmith Barbet *Psilopogon haemacephalus* from Asia

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Abstract Plumage colouration is important for birds as it helps them in camouflage, mate selection, social signalling and various other physiological and behavioural processes. The most common pigments responsible for colouration are melanins and carotenoids. In a few individuals, colouration is disrupted due to various causes. The most common colour aberrations found in birds are leucism, albinism, melanism, carotenism, schizochroism and dilution whereas xanthochroism is a lesser-known phenomenon. In this article, five records of colour aberrant Coppersmith Barbet *Psilopogon haemocephalus* are reported. The authors along with the help of citizen scientists observed four individuals with disruptions in plumage colouration from different areas of West Bengal and Assam, India and one from Rajsahi, Bangladesh. Due to the lack of melanins, the birds were mostly yellowish and whitish in colouration with or without some normally coloured feathers on the head and wings. The carotenoid deposition was unaffected in the case of the observed adults and juveniles. These records can be cited as xanthochroistic individuals resulting from leucism. This is the first record of such colour aberration for Coppersmith Barbet from the whole of its distribution range.

Keywords: xanthochroism, carotenoids, melanin, Megalaimadae, India, Bangladesh

Összefoglalás A tollazat színezetének fontos szerepe van a madaraknál, mivel segíti őket a rejtőzködésben, a párválasztásban, a szociális kommunikációban és számos egyéb élettani, viselkedési folyamatban. A színezetért felelős leggyakoribb pigmentek a melaninok és a karotinoidok. Némely egyednél a színezet különböző okok miatt eltér a normálistól. A madaraknál előforduló színezeti aberrációk közt a legáltalánosabb a leucizmus, az albinizmus, a melanizmus, a karotenizmus, a skizokroizmus és a fakulás, míg a xantokroizmus egy kevésbé ismert jelenség. Tanulmányunkban a sárgatorkú bajszika *Psilopogon haemocephalus* színezetére vonatkozóan mutatunk be öt aberráns esetet. A szerzők „közösségi tudomány” (citizen science) segítségével négy egyedet figyeltek meg Nyugat-Bengália és Assam különböző területein, valamint egyet a bangladesi Rajsahiban. A melaninok hiánya következtében a madarak sárgás és fehéres színűek voltak, a fejükön és a szárnyaikon néhány normális színű tollal vagy anélkül. A karotinoid depozíciója a megfigyelt felnőtt és fiatal egyedek esetében nem változott. Ezek az esetek a leucizmusból eredő xantokroizmusnak tekinthetők. Ez a sárgatorkú bajszika első ilyen típusú színaberrációs észlelése a faj teljes elterjedési területén.

Kulcsszavak: xantokroizmus, karotinoidok, melanin, Megalaimadae, India, Banglades

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Introduction

Plumage colouration is of great importance for birds as it helps them in camouflage (Barragán-Farías *et al.* 2019), mate selection (Hill *et al.* 1999), social signalling (Senar 2006) and various other physiological and behavioural processes. The plumage colourations in birds are mainly formed by two distinct mechanisms, structures and pigments of which some colours are produced by combining different pigments whereas some are produced by the combination of structure and pigment (Laczi *et al.* 2019). All types of colourations are a result of both gene expression and environmental effects (Venizelos & Benetti 1999, Mills & Patterson 2009).

Occasionally, the normal plumage colouration is disrupted which results in new, aberrant phenotypes (van Grouw 2013). Plumage abnormalities are mainly caused by an abnormal distribution of pigments (present mainly in the feathers), chemical changes of the pigments, changes in the microstructure of feathers (Harrison 1985), environmental or dietary factors and also genetic mutations (Dorst 1971, Gonçalves Jr. *et al.* 2008). The most extreme variations of colour aberrations occur in those individuals who show noticeable reductions or increase in the normal colour causing pigments like the melanins or the carotenoids (Smith 2016). According to van Grouw *et al.* (2021) the different forms of colour aberrations noticed in case of birds include leucism (complete lack of both melanin pigments from some or all of the skin due to a neural crest disorder along with melanised eyes, pink bill and feet or normally coloured bill or feet), progressive greying (complete lack of both melanin pigments from some or all of the skin due to loss of melanins with growing age along with normally coloured eyes, pink bill and feet or normally coloured bill or feet), albinism (complete lack of both melanins in feathers, eyes and skin due to the absence of tyrosinase enzyme in the cells along with red eyes and pink feet and bill), brown (qualitative reduction of eumelanin only due to incomplete synthesis of eumelanin along with the original black colour turning into brown and the plumage bleaches rapidly in the sunlight), ino (reduction of melanin and qualitative reduction of the remaining melanins resulted due to incomplete synthesis of both the types along with original black colour turning into pale brown to dark brown and original reddish or yellowish hardly observed to be slightly paler), dilution (abnormal clumped deposition of both the melanins or only the eumelanin in feathers along with original black turning into bluish or silvery grey and original reddish or yellowish brown turning into buff /cream or remains unaffected), melanism (abnormal production and distribution of melanins resulting in increase of black and reddish brown colouration). There are mixed views regarding the proportion of occurrence of colour aberrations in wild birds. According to Gonçalves Jr. *et al.* 2008, plumage aberrations are not rare in wild birds, but the published documents and reports greatly underrate the proportion of occurrence. In another study conducted on Collared Flycatchers *Ficedula albicollis* by Laczi *et al.* 2020 for over four decades and with more than 40,000 birds monitored, albinism was seen only once which suggests that colour aberrations are quite rare in natural populations. In India, reports on the bird colour aberrations like leucism, melanism and albinism are not that uncommon, examples have been reported for Large Grey Babbler *Argya malcolmi* (Taher 2020), Red-necked Falcon *Falco chicquera* (Adaki *et al.* 2020), Spotted Owlet *Athene brama* (Pande *et* *al.* 2005), Changeable Hawk-Eagle *Nisaetus cirrhatus* (Parashar & Sharma 2010), Lesser Sand Plover *Charadrius mongolus* (Sathiyaselvam 2003), Indian Peafowl *Pavo cristatus* (Yadav & Arigela 2020), Ashy-crowned Sparrow Lark *Eremopterix griseus* (Pawashe *et al.* 2006) and many others species of birds. Only a few records exist for colour aberrant birds from Bangladesh (Rahman 2018, Islam *et al.* 2020, Sultana *et al.* 2020).

Leucism is a condition with total absence of melanin pigments in either some portions or whole of the plumage and skin due to a neural crest disorder which results in the congenital absence of the melanin cells from some or all of the plumage area. Unlike albinos, the tyrosinase enzyme is generally present in leucistic individuals. Leucism can vary from a few white feathers (partially leucistic) to fully white individuals (total leucistic), but eyes are always has normal colour whereas the beak may or may not be normal (van Grouw 2006, 2021). It is the most common inheritable colour abnormality occurring in birds and is documented in a lot of species in the Indian subcontinent. Examples of leucistic individuals from India includes Jungle Myna *Acridotheres fuscus* (Nandy 2019), Indian Peafowl *Pavo cristatus* (Yadav & Arigela 2020), Kalij Pheasant *Lophura leucomelanos* (Thareja & Thareja 2017), Collared Kingfisher *Todiramphus chloris* (Adhikary & Mondal 2019), Garganey *Anas querquedula* (Karuthedathu *et al.* 2014), Indian Spot-billed Duck *Anas poecilorhyncha* (Raju 2017), House Sparrow *Passer domesticus* (Sankpal *et al.* 2019), Red-crested Pochard *Netta rufina* (Mahajan 2016), Brahminy Starling *Sturnia pagodarum* (Phalke 2020), etc. Leucism is also reported in birds from Bangladesh like in Eurasian Collared Dove *Streptopelia decaocto* (Islam *et al.* 2020), Western Koel *Eudynamys scolopaceus* (Rahman 2018) and partial leucism is observed in Common Redshank *Tringa totanus* (Sultana *et al.* 2020).

Another less common and lesser-known colour aberration found in birds is 'xanthochroism' (also known as 'xanthochromism' and xanthism') which occurs rarely in the case of wild birds. Though it is commonly found in caged birds like in some varieties of birds belong to the families Psittacidae and Fringillidae (Smith 1966). Xanthochroism is generally defined as a genetic or diet-induced abnormal condition that affects the carotenoid pigment expression, resulting in the red colouration to be replaced by yellow (Gómez *et al.* 2013). Gross (1965) defined this phenomenon as an abnormal colouration of the plumage, possibly related to food or some pathological conditions where yellow replaces the normal plumage colouration. Additionally, Schnell and Caldwell (1966) proposed that the definition of xanthochroism should not only include the conditions where excessive yellow pigments are present and they replace the red pigments, but also, the situations where the yellow colour is revealed due to the lack of melanin pigments. However, Hailman (1984) found xanthochroism to have no stable meaning and is used to name at least five different conditions starting from original simple yellow colouration, via any type of abnormal yellow colouration due to various factors, to yellow colouration that appears due to the loss of melanin. Recently, some articles refer only to the expression of abnormal orange-red-yellow colourations as xanthochroism (Quigley 2017, Stagličić 2019). Reports of xanthochroism is noticed from different parts of the world namely North America, South America, Africa, Australia but no such report is available till now from Asia. Records of this less commonly known colour aberration are noted in the case of Cape May Warbler *Setophaga tigrina* (Schnell

& Caldwell 1966), Yellow-faced Grassquit *Tiaris olivaceus* (Smith 1966), House Finch *Haemorhous mexicanus* (Barragán-Farías *et al.* 2019), Vermilion Flycatcher *Pyrocephalus rubinus* (Gómez *et al.* 2013, Smith 2016), Norfolk Robin *Pteroica multicolor*, Flame Robin *Pteroica phoenicea* (Wall 1966) and Northern Cardinal *Cardinalis cardinalis* (McGraw *et al.* 2003).

Colour aberration in the case of barbets is also noted but no such report is available for Coppersmith Barbet *Psilopogon haemacephalus* from India and Bangladesh, as well as from its other distribution zones. Thus, we report evidences of colour aberrations noted in the case of Coppersmith Barbet for the first time from India and Bangladesh.

Observation

This article is aimed to document a total of five observations of colour aberrant Coppersmith Barber from different locations of India and Bangladesh *(Figure 1)*. On 21 November 2018, while birding at Arjun Bandh area (23.54°N, 87.32°E) of Durgapur, Paschim Bardhaman district, West Bengal, India; the authors observed an abnormally coloured bird flying from one tree to the other. We took photographs of that abnormally coloured individual *(Figure 2A)*. The bird was identified as Coppersmith Barbet on the basis of its small size, shape, beak structure, flight patterns and call. The individual was not normal in colouration like that of the typical Coppersmith Barbet *(Figure 3)*. The size of the photographed specimen resembles a Coppersmith Barbet with similar red (or crimson) marking at forehead (including fore-crown) and breast. Overall colouration of the specimen shows pale yellowish-green colouration, which is nearly similar to the facial yellow

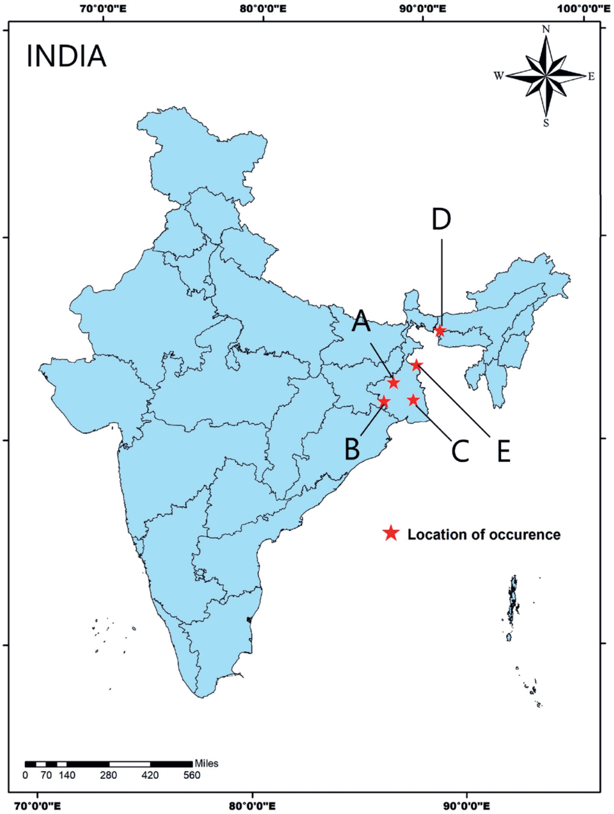

Figure 1. Relative location of the observations of the colour aberrant Coppersmith Barbets in the map of India. A: Durgapur, West Bengal, India; B: Belpahari, West Bengal, India; C: Bally, West Bengal, India; D: Lakhimari, Assam, India; and E: Rajshahi, Bangladesh

1. ábra A színaberráns sárgatorkú bajszika észleléseinek viszonylagos elhelyezkedése India térképén. A: Durgapur, West-Bengal, India; B: Belpahari, West-Bengal, India; C: Bally, West-Bengal, India; D: Lakhimari, Assam, India; és E: Rajshahi, Banglades

Figure 2. Photographs of colour aberrant Coppersmith Barbets recorded from India & Bangladesh. A: The adult with colour aberration photographed in Durgapur, West Bengal, India (Photo courtesy: Subhajit Roy); B: The adult with colour aberration photographed in Belpahari, West Bengal, India (Photo courtesy: Avik Dutta); C: The juvenile with colour aberration photographed in Bally, West Bengal, India rescued by Mana dada (Photo courtesy: Sriparna Dinda) [Source: https://www.facebook.com/groups/AskidsofIndianBirds/posts/1466737543464947/]; D: The adult with colour aberration photographed in Lakhimari, Assam, India (Photo courtesy: Tahijul Ali) [Source: https://www.facebook.com/groups/sundaywatch.org/posts/1488728064603351]; E: The colour aberrant adult photographed from Rajshahi, Bangladesh (Photo courtesy: Maroof Rana) [Source: https://www.facebook.com/groups/AskidsofIndianBirds/posts/2218427248295969/]

2. ábra Az Indiában és Bangladesben észlelt színaberráns sárgatorkú bajszikák fényképei. A: A Durgapurban (West-Bengál, India) fényképezett színaberráns adult (fotó: Subhajit Roy); B: A Belpahariban (West-Bengal, India) fényképezett színaberráns adult (fotó: Avik Dutta); C: A Ballyban (West-Bengal, India) fényképezett színaberráns fiatal (fotó: Sriparna Dinda) [Forrás: https://www.facebook.com/groups/AskidsofIndianBirds/posts/1466737543464947/]; D: A Lakhimariban (Assam, India) fényképezett színaberráns adult (fotó: Tahijul Ali) [Forrás: https://www.facebook.com/groups/sundaywatch.org/posts/1488728064603351]; E: A Rajshahiban (Banglades) fényképezett színaberráns adult (fotó: Maroof Rana) [Forrás: https://www.facebook.com/groups/AskidsofIndianBirds/posts/2218427248295969/]

Figure 3. Normal coloured individuals of Coppersmith Barbet. A: An adult (Photo courtesy: Sagar Adhurya); B: A juvenile (Photo courtesy: Saikat Adhurya) [Source: https://bdwb.wildwingsindia.in/share.php?id=B6VeBiPiMePshax4]

3. ábra Normális színezetű sárgatorkú bajszika. A: Adult (Fotó: Sagar Adhurya); B: Fiatal (fotó: Saikat Adhurya) [Source: https://bdwb.wildwingsindia.in/share.php?id=B6VeBiPiMePshax4]

colouration of a normal Coppersmith Barbet. The yellowish breast-line flanking below the red breast-line is as similar in colouration like a normal specimen. The most striking difference is the complete replacement of black facial pattern (continuous black line in normal specimen extending lore, eye stripe, moustachial stripe, hind crown and edges of ear coverts) with the white colouration. The breast is more or less uniform white instead of presence of broad olive-green streaking. The bill is pale whitish instead of blackish in normal. The leg is showing normal colouration. Though black colour is absent from the body, tail tip and tips of primaries strikingly showing expression of black colouration. These characters clearly indicate that the observed specimen was a colour aberrant individual.

After that on 10 November 2020, the fifth author observed another colour aberrant individual of the same species from Belpahari, Jhargram district in West Bengal (22.63°N, 86.76°E). The individual was identified as an adult based on the red colour patches on its crown and breast. Also, this individual shows exactly the same features as the first except it has some normal green coloured feathers in its wings and a little on its head. Additionally, it has some blue coloured feathers on its wings but the overall colouration on the body looks very similar to the other individuals *(Figure 2B)*.

Apart from our observations, three more similar kinds of colour aberrant individuals were recorded from India and Bangladesh. Sriparna Dinda and Mana Dada also rescued and photographed a third individual of the same species with similar condition in Bally, Howrah district, West Bengal (22.65°N, 88.34°E) on 20 July 2019 posted it in Ask Ids of Indian Birds Facebook group for its identification (Patra 2019) *(Figure 2C)*. The identification was confirmed as a juvenile [refers to young bird having different colouration than adult, not 'juvenal' plumage (Bostwick 2016)] of Coppersmith Barbet having abnormal yellow colouration. This individual shows exactly the same features as the first except it has no red coloured patches in its crown and breast and also the overall colouration is more yellowish than the other two individuals. The juvenile normal Coppersmith Barbet also shows no red marking in its body *(Figure 3B)*. So, this bird was identified as a juvenile colour aberrant specimen.

Tahijul Ali photographed another individual of Coppersmith Barbet with abnormal yellow colouration from Lakhimari Village, Dhubri district, Assam, India (26.02°N, 89.97°E) on 5 January 2019 (Patra 2019) *(Figure 2D)*. The individual shows similar colouration as the second individual except the absence of blue-coloured wing feathers. So, this individual was identified as an adult.

Maroof Rana also photographed an individual of the same species with aberrant colouration from Rajsahi, Bangladesh (24.37°N, 88.60°E) on 27 November 2021 (Rana 2021) *(Figure 2E)*. This individual is also an adult as it has red colour patches on its crown and breast and for the rest of the body it shows similar kind of colouration like that of the second individual.

Discussion

Coppersmith Barbet is a widespread resident across Asia, occurring in the mainland of Asia starting from Pakistan and Sri Lanka in the west to south China and Indochina in the east and south into the Malay Peninsula. It can also be found in the islands of Sumatra, Java and Bali (Dickinson *et al.* 1991). It generally prefers woodlands, grooves and forests but can also be found in gardens and orchards of the urban areas where there is plenty of fruiting trees. They mainly take fruits (especially figs, *Ficus* spp.) as their main food resource, but sometimes also take insects (Short & Horne 2001, Grimmett *et al.* 2011). Two colour morphs are known for this species across their distribution zone. One is the red colour morph in which the face and throat are mostly red coloured alongside the red forehead and a red patch below the throat and its distribution is restricted only to Java, Bali and some Philippine islands. Another one is the yellow colour morph where the face and the throat is yellow coloured alongside the red forehead and a red patch below the throat and is found over most of its distribution zone (den Tex & Leonard 2014).

In the case of birds, black, grey, rufous and brown colours are associated with the two varieties of melanins (eumelanin and pheomelanin) whereas yellow, red and orange colours are produced by the carotenoids (Olson & Owens 1998, Jawor & Breitwisch 2003). The greenish tone in the plumage of many bird results from either the combination of these two classes of pigments or from the combination of pigment and structural properties of the feathers (Dyck 1976, Guay *et al.* 2012). So the normal green colour in the plumage of both adult and juvenile Coppersmith Barbet is probably the product of the conjugated effects of melanins and carotenoids. The red colour on the forehead and chest (present only in adults and absent in juveniles) and the yellow colour on the face and throat in both adults and juveniles are the results of carotenoid pigmentation. However, the black coloured facial pattern and the dark beak are the results of melanin pigmentation. Though the facial and throat colour varies between the two kinds of colour morphs, but the variation in plumage colouration exhibited by the observed individuals is totally different and cannot be found in any of the previous literature.

The specimens in *Figure 2A, 2B, 2D* and *2E* were the adult individuals because of the presence of red forehead and red patch in the chest. The remaining specimen *(Figure 2C)* was a juvenile because it lacks the red colouration on forehead and chest. All the individuals not only lacked the normal olive-green colouration on the body, they also lacked blackish facial colouration, olive green streaks on the abdomen and dark coloured beak. These areas mostly had yellowish green or whitish tones, as did much of the body. The changes were such that the colour patterns were hardly recognizable for a Coppersmith Barbet. This can be due to the fact that melanin pigments are absent in those individuals. The olive-green streaking pattern on the underside of the body was also absent and an off-whitish colouration replaced it in all the observed individuals. Again, in the facial regions, the blackish pattern is replaced by white and the beak also appeared paler in comparison to a normal individual. But the colours caused by the carotenoid pigments in the case of a normal individual (red in forehead and chest and yellow in the face and throat) are present in their normal position in these individuals. Thus, the noticeable absence of the green colour from the plumage, unmasking of the yellow colouration, absence of black colour from the face and beak and the appearance of the red colouration in their normal positions without any changes supports the absence of the melanin in these individuals. As the birds simply lack one or the other or both the types of melanins on its plumage, and they consistently fail to produce it, certainly qualifies them as leucistic following the definitions of Guay *et al.* (2012).

But according to the definitions of xanthochroism provided by Gross (1965) and Schnell and Caldwell (1966), these reports certainly qualify also as an example of xanthrochroism too, as here the yellow colour replaces the original plumage colouration and is unmasked due to the lack of melanin pigments. Thus, we can say that it is an example of leucism resulting in xanthochroism in case of Coppersmith Barbet.

The birds having this kind of colour aberrations face a lot of hardship in the wild. As these birds are more conspicuous than the normal individuals, the risk of predation could be greatly increased. These abnormal conditions could also make them more susceptible to feather weakening, which can also hamper the flight (Harrison 1985). Additionally, it is also reported that these colour aberrant individuals, in occasional cases, may not be accepted by its potential mating partner and are harassed by its conspecifics (Smith & Rios 2017). Though we could only note its weak flight pattern but could not study whether the other conditions are also true for the individuals observed by us. So, a more detailed and prolonged study on the colour aberrant individuals in India is sure to unveil various new information about them.

Colour aberrations is known from only two varieties of barbets, namely in Black-collared Barbets (albinism, Blaker 1980; xanthochroism, Davies & Symes 2012) and Great Barbets (albinism and xanthochroism, Short & Horne 2001), but no report is available for colour aberrations in the Coppersmith Barbet from any part of its distribution zone including India and Bangladesh. Thus, the present report represents the first record of colour aberration in the case of Coppersmith Barbets from the India and Bangladesh as well from its whole distribution zone.

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The current status of Chukar (*Alectoris chukar* J. E. Gray, 1830) in Armenia

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Abstract The Chukar *Alectoris chukar* is one of the most popular game birds of Armenia, but there is no governmental monitoring programme for that exploited species. We carried out national surveys in 2003–2019 and estimate the current occupied range of the Chukar in Armenia as 5,082 km^2 (17% of the country). In 2019, we estimate Chukar population size at 12,472 breeding pairs (95% CL: 10,266–14,677). Its population trend shows a moderate decline underlying strong annual fluctuations. In Armenia, the Chukar occurs on 20 public hunting lands covering 2,414 km^2 in total. Its abundance on these public hunting lands is estimated to 5,558 breeding pairs in 2019 (95% CL: 3,656–7,460). Surveys of the seven Hunters' Unions of Armenia found that there are 10,000 to 20,000 active hunters. The number of hunting permits issued annually increased ten-fold between 2016 and 2019, exceeding the capacity of the public hunting lands in 2019. A hunter survey found that each Chukar hunter shot on average (± SD) 5.88±3.05 birds in the 2018/19 season, which extrapolates to 17,052–34,104 shot specimens of Chukar (at least 46% of autumn numbers). Current hunting management practice is thus unsustainable and we provide recommendations for sustainable approaches that should replace it urgently.

Keywords: game birds, distribution, population, hunting, management

Összefoglalás Bár a csukár *(Alectoris chukar)* Örményország egyik legismertebb vadászható madárfaja, jelenleg nincs állami állomány-felmérési protokoll erre a fajra. Országos felméréseket végeztünk 2003–2019 között, ami alapján 5082 km^2-re (az ország 17%-ára) becsültük a csukár jelenlegi elterjedési területét. 2019-ben a faj örmény állományának méretét 12 472 költőpárra becsültük (95% CL: 10 266–14 677). A faj állománytendenciája mérsékelt csökkenést mutat, erős éves ingadozással. Örményországban a csukár 20 állami vadászterületen fordul elő, összesen 2414 km^2-en. Egyedszámát ezeken a vadászterületeken 2019-ben 5 558 költőpárra becsültük (95% CL: 3 656–7 460). A hét örmény vadászszövetség felmérései szerint 10 000–20 000 aktív vadász került regisztrálásra. Az évente kiadott vadászati engedélyek száma 2016–2019 között a tízszeresére nőtt, meghaladva a 2019-es közcélú vadászterületek kapacitását. Egy vadászati kutatás megállapította, hogy egy csukárvadász átlagosan (± SD) 5,88±3,05 madarat lőtt ki a 2018/2019-es időszakban, ami alapján a valós szám 17 052–34 104 között mozog az extrapoláció alapján, ami az őszi számok legalább 46%-a. A jelenlegi vadászati gyakorlat ezért fenntarthatatlan, és így javaslatokat teszünk olyan fenntartható megközelítésekre, amelyek ezt sürgősen felváltják.

Kulcsszavak: vadászható madárfaj, elterjedés, populáció, vadászat, állománykezelés

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Introduction

Armenia is a relatively small (29,743 km^2), landlocked mountainous country situated in South Caucasus between the Black and Caspian Seas, within the Western Palearctic. Elevation varies from 375 to 4,090 m above sea level, determining a variety of climatic conditions and thus creating many different landscapes, including semi-desert, juniper woodland, deciduous forest, mountain steppe, and a sub-alpine zone. Much of the terrain is steep, often containing rocky outcrops (Aghababyan *et al.* 2015). Among such a variety of landscapes, the Chukar *Alectoris chukar* inhabits quite a large range including semi-deserts, juniper woodlands, and arid mountain steppes from 400 to about 2,000 m above sea level (Adamian & Klem 1999). The Chukar is the only *Alectoris* partridge species in Armenia, which is inhabited by the subspecies *A. c. kurdestanica* (Cramp & Perrins 1977, Collar 2019, Christensen 2020). The species is classified as Least Concern in the IUCN Global Red List owing to its extremely large range and apparently stable population trend (BirdLife International 2019). It was, however, considered Near Threatened in Europe because of moderately rapid population declines in its European breeding range (BirdLife International 2015). The latest IUCN European Red List assessment revised it to Least Concern (BirdLife International 2021), and put the Armenian Chukar population at 3,600–5,200 pairs in 2013–2018, down from 5,000–12,000 in 2002–2012 (BirdLife International 2015) and from 6,500–14,000 in 1997–2002 (BirdLife International 2004). These estimates were based on expert opinion rather than systematic surveys.

In Armenia, the species is a game bird for which the hunting season (usually between September and January), daily bag limit (typically two birds per person per day) and number of permits are set annually (Ministry of Nature Protection 2016, 2017, 2018, Ministry of Environment 2019). There is, however, no government monitoring scheme providing a scientific basis for these decisions. In such a situation, it is possible that the number of hunting permits sold could exceed the number that allow hunting to be sustainable.

Separately, within the context of a Europe-wide initiative to monitor bird populations using skilled volunteers, we have regularly and systematically collected data on Chukar abundance across Armenia since 2003. We hereby seek to provide a scientifically rigorous assessment of the species' distribution, population size and trends in Armenia, identify threats, and analyse existing management practices. Such information will provide a basis for future hunting management and be crucial in informing a review of the procedure for issuing shooting permits.

Material and Methods

Chukar data collection

Early Chukar observations in Armenia recorded in the literature were collated and summarized in Adamian and Klem (1999). Systematic data collection on the Chukar started in 2003 using volunteer birdwatchers and professional ornithologists. The standard European

Monitoring Grid with a 10×10 km mesh was applied to Armenia, dividing the territory of the republic into 374 squares. The counted squares were of two kinds: “systematic” ones that, once counting started on them, were systematically counted every subsequent year, and “opportunistic” ones, where counts were carried out when the opportunity arose. In total, in the period of 2003–2019, 325 squares were visited at least once during that period, including 147 squares with systematic data collection *(Figure 1)*. The remaining 49 squares were not visited because four covered the waters of Lake Sevan, and the other 45 overlapped the country’s international border in militarily sensitive areas with restricted access; such squares covered an Armenian land area of 162 km^2, representing only 0.5% of the total area of Armenia. The surveyed squares relevant to Chukar are summarized in *Table 1*. The initial survey in 2003 comprised 38 systematic and 16 opportunistic squares; by 2019,

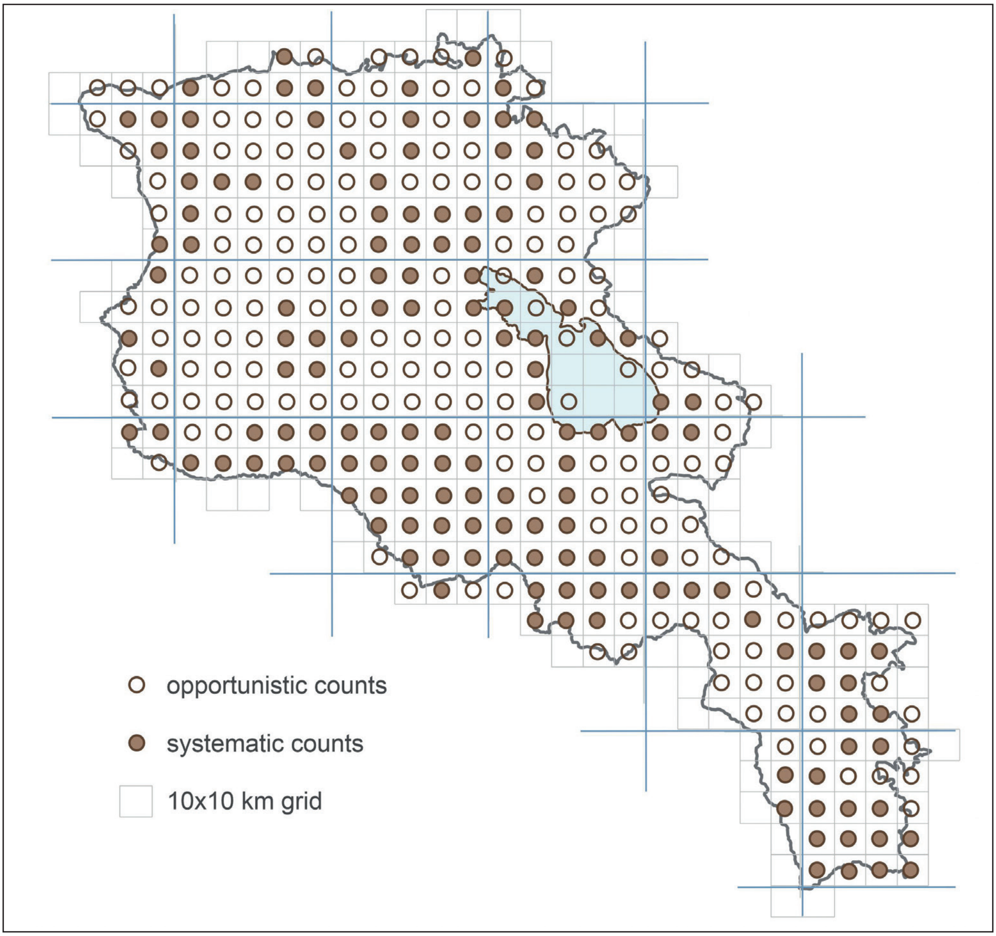

Figure 1. Squares (on a 10×10-km grid) surveyed for Chukars in Armenia either systematically (annually after first count) or opportunistically (at least in one year) over the period 2003–2019.

1. ábra A 2003–2019-es időszakban szisztematikusan (évente az első számlálás után) vagy opportunista módon (legalább egy éven belül) örményországi csukárokra felmért négyzetek (10×10 km-es négyzetekben)

Table 1. Number and type of squares surveyed in Armenia each year in the period 2003–2019, according to whether or not Chukars were recorded

1. táblázat A felmért négyzetek száma és típusa Örményországban 2003–2019 között

Survey type	Chukars recorded	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019
Systematic	Yes	38	38	38	41	41	45	45	45	45	45	51	58	59	59	59	59	59
Systematic	No	34	34	34	40	40	46	46	46	46	46	48	48	54	54	54	54	54
Opportunistic	Yes	16	14	18	17	20	23	25	28	26	22	24	32	38	41	41	39	41
Opportunistic	No	20	18	17	21	30	32	22	29	33	30	25	36	45	42	51	32	38
Total surveyed		54	52	56	58	61	68	70	73	71	67	75	90	97	100	100	98	100

21 of the squares surveyed opportunistically in earlier years had been integrated into the systematic component, and a further 41 squares had been surveyed opportunistically, giving 100 squares surveyed in total.

In each square, data on Chukars were obtained from two different sources: (1) incidental observations and (2) standardized counts (data collected according to standard methodology). Both types of data may be used to create species distribution maps, and data collected by the second method may be used for estimating population densities and trends.

1. Incidental observations were provided by birdwatchers and accepted as long as they conformed to minimum data requirements: accurate species identification, observation date, geographical coordinates, name of nearest locality (human settlement, mountain, historical site, etc.), breeding code (based on the bird's behaviour, indicating how likely the bird is breeding in the surveyed area – Voříšek *et al.* 2008), observer name and contact details. The observations often have additional information, e.g. time, observation duration, number of people in the group, etc. Since it was not always possible to record the precise geographical coordinates on the spot, the information was sometimes provided at the level of the 10×10-km square. Most of the data was accumulated in the platform Observation.org, however, data from iNaturalist, eBird, and GBIF were also considered in cases where the identification of the species was supported by photos or voice recordings.

2. Standardized spring counts (counts done following a predefined standard protocol) can be conducted by both ornithologists and skilled birdwatchers. Counts were carried out during a fixed period of 1 or 2 hours, when an observer slowly walked along a transect route counting all the birds within 100 m either side of the transect (hence in a strip 200 m wide). As far as possible, surveys were done at the time of the day when birds were most active (as a rule, early morning) in favorable weather conditions, such as absence of rain and weak wind (below Beaufort Force 3). The best season for Chukar counts was considered to be the period between 1 April and 20 May. Nevertheless, data collected in March and in June were used as well. The standardized counts required more detailed data collection than incidental observations: number of individuals observed or heard, observation date, geographical coordinates of the beginning and end of the route, type of habitat (semi-desert, juniper woodland, mountain steppe, lower subalpine zone, semi-urban), start and end times of the count, individual-specific breeding codes, observer name and contact details. The number of routes in one 10×10 km square varied from one

to three, depending on how many habitat types were present in a square. Each route was dedicated to one type of habitat only. We tried to keep the same routes for the standardized counts and to survey them every year, whenever possible. However, in the period 2013–2017, when the number of volunteer counters increased thanks to the fieldwork required for the European Breeding Bird Atlas 2 (Zbiden *et al.* 2020), many new standardized counts were created from atlas routes. All data were collated at the end of each counting season, entered into a database and checked.

Hunting data collection

To gather information on hunting pressure on the Chukar, we conducted surveys of the heads of seven Hunters' Unions and their hunter members (keeping the hunters' survey confidential to reduce the risk of false reporting). We tried to keep the numbers per Hunters' Union roughly equal (maximum difference was 10 hunters). The survey was conducted in spring 2019, after the end of the 2018–2019 hunting season, by sending out over 800 questionnaires to randomly selected hunters. A total of 486 responses were received to the mailing, and a further 14 responses obtained following personal requests, giving a total of 500. The following questions were included in the questionnaire: (1) Do you hunt? (2) Do you ever hunt Chukar? (3) How often do you hunt Chukar (almost every year; not frequently; rarely)? (4) How many Chukars do you hunt annually? As the Chukar is the only *Alectoris* species in Armenia, and the only other partridge species is the very different Grey Partridge *Perdix perdix*, there was little risk of hunters confusing Chukars with another species.

We also interviewed staff at the State Inspectorate for Nature Protection and Mineral Resources. These interviews were conducted with eight inspectors from Ararat, Armavir, Vayots Dzor, and Syunik Provinces and were less structured. The main questions that were relevant here were related to the ability of the inspectors to monitor the number of shot birds, to detect poaching outside the legal hunting season, and to detect poaching in protected areas.

Data analysis

Chukar distributional range was determined at the 10×10-km square level. A given square was deemed occupied if the species had been recorded in any one of the 17 years 2003–2019 through incidental observation or standardized count. To compare the change in distribution from before 2003 with that during 2003–2017, we also digitized all the previous records summarized in Adamian and Klem (1999). Mapping was implemented using the software package ArcGIS 10.0 (Environmental Systems Research Institute, Inc.).

Chukar density was estimated for each transect route by dividing the recorded number by the area surveyed, obtained as the transect length multiplied by the strip width of 200 m (Bart 2005). With a single distance band within which birds were recorded, we were unable to correct for detectability, so the density estimates that we obtained were minimum estimates. The density values were then averaged across transects, overall and by habitat type. We compared the rates of change between habitats using analysis of covariance applied to log-transformed density, using year as covariate and habitat as factor. The total size of the Armenian Chukar breeding population (pairs) in each of the years 2003–2019 was estimated using post-stratification by habitat type (Lehtonen & Pahkinen 2004): it was calculated as half of the sum of the number estimated in each habitat type, obtained by multiplying the habitat-specific density by the area of habitat within the occupied range. The annual estimates were averaged over the periods 2003–2012 and 2013–2018 for comparison with BirdLife International (2015, 2021) values. For the most recent estimates in 2018 and 2019, we calculated 95% confidence limits as total size ± 1.96 SE, where SE (standard error) was the square root of the sum of the squared SEs of the habitat-specific numbers, themselves obtained by multiplying the SE of the habitat-specific densities by the habitat-specific areas. A similar calculation, restricted to the public hunting lands within the occupied range, provided estimates of the number of Chukar pairs on public hunting lands. The same was done for non-hunted lands. In the absence of data on the age of hunted birds, the number of Chukars present in the autumn was approximated as three times the number of pairs, using a standard approach from French hunters based on the average ratio of young birds to adult females for the closely related Red-legged Partridge (Pasquet 2006). The approximation was carried out for 2018 (corresponding to the start of the 2018/2019 hunting season, which was the season before the hunter survey and hence most suitable for comparison with the hunting bag estimated from the survey) and 2019 (for comparison with the number of 2019 hunting licenses).

To calculate population trends, we used transects with multi-year data series and processed the data (density values per transect and year) using TRIM 3.54 software (Van Strien *et al.* 2004). In total, there were 1,062 data values analysed, and their number increased from 44 in 2003 to 82 starting from 2015, resulting in 332 missing counts in the whole period 2003–2019. We calculated a population index using log-linear Poisson regression, and applying a time effect model; the indices are calculated relative to 2003, which is given a value of 100. TRIM also provides an estimate of overall trend in the form of the average annual rate of change r and its standard error $SE(r)$ across the full span of years (Pannekoek & van Strien 2005). To assess the importance of the trend, van Strien *et al.* (2001) recommended considering both its magnitude and statistical significance according to five categories: substantial decline or increase (confidence interval lies below -20% or above 20% respectively in a 20-year period), non-substantial decline or increase (confidence interval lies above -20% or below 20% respectively and excludes zero), decline or increase (confidence interval includes -20% or 20% respectively and excludes zero), stable (confidence interval lies above -20%, below 20% and includes zero) and poorly known (confidence interval includes both zero and one or both of -20% and 20%). The annual rate of change was converted to a 20-year change ρ by calculating $\rho = r^{19}$, with standard error $SE(\rho) = SE(r)\ (19\ r^{18})$ and 95% confidence interval $\rho \pm 1.96\ SE(\rho)$ (Hensler 1985, van Strien *et al.* 2001).

Results

Chukar distribution, population size and trend in Armenia

During the surveys of 2003–2019, the Chukar was recorded in relatively large areas of the Central, Southern, South-eastern and North-eastern regions of the country *(Figure 2)*. The species was recorded at elevations ranging from 400 to 2,500 m above sea level. The habitats occupied by Chukar included semi-deserts (1,684 km^2), juniper woodlands (1,065 km^2), mountain steppes (2,233 km^2), lower parts of the subalpine zone (100 km^2), and semi-urban areas (124 km^2). Its presence was typically associated with rocky slopes and cliff formations *(Figure 3)*. The total area occupied by Chukars in Armenia is estimated at 5,206 km^2, representing 17% of the country.

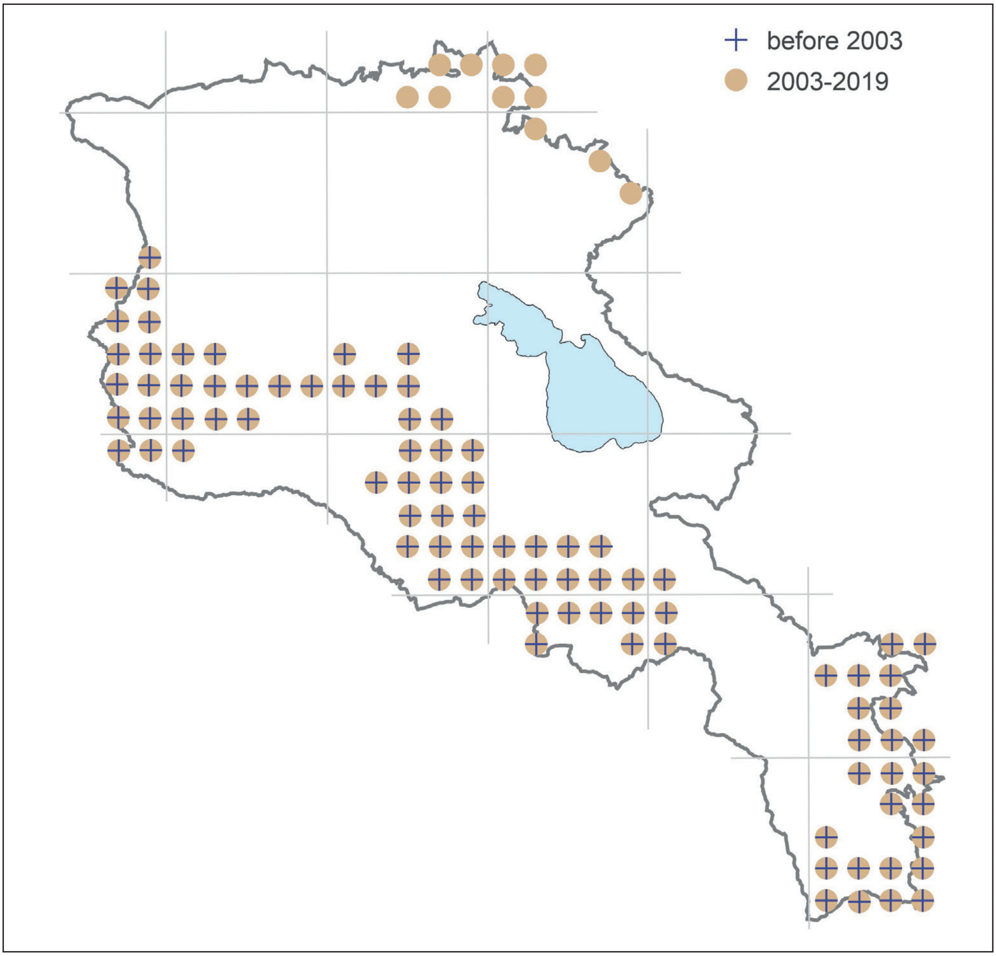

Figure 2. Distribution of the Chukar in Armenia based on a 10×10-km square grid, before and after 2003
2. ábra A csukárok elosztása Örményországban 10x10 km-es négyzetekben, 2003 előtt és után

Figure 3. Typical habitat of the Chukar in the Urts mountains of Armenia (Photo by K. Aghababyan)
3. ábra A csukár tipikus élőhelye Örményország Urts-hegységében (Fotó: K. Aghababyan)

The average density of the species during 2003–2019 varied by habitat, being highest in juniper woodlands and the upper zone of semi-deserts *(Table 2)*. However, the average rate of change in density over time differed between habitat types ($F_{4,60} = 6.11$, $P < 0.001$). In juniper woodlands, the rate did not differ significantly from zero (-1.06%, SE = 1.12, $t_{15} = 0.95$, $P = 0.358$); it was negative in mountain steppes (-2.76%, SE = 0.95, $t_{15} = 2.86$, $P = 0.012$) and semi-deserts (-3.22%, SE = 1.04, $t_{15} = 3.05$, $P = 0.008$), and increased in the lower subalpine zone (3.07%, SE = 0.88, $t_{15} = 3.54$, $P = 0.003$). For the BirdLife International (2015) period 2002–2012, we estimated an average breeding population size of 20,077 pairs (range 13,816–26,119). For the BirdLife International (2021) period 2013–2018, the estimate was 17,278 pairs (range 14,530–20,823). For 2019, the size of the Armenian Chukar breeding population was estimated at 12,472 pairs (95% confidence limits 10,266 to 14,677 pairs).

The TRIM model fitted well (goodness of fit: $\chi^2 = 679.19$, df = 1296, $P > 0.999$). From 2003 to 2019, the average annual change in the population index calculated by TRIM was -1.41% (*SE* = 0.24), significantly different from zero ($z = 5.88$, $P < 0.001$). Over 20 years, this equates to a 23.6% decrease (confidence interval -30.6 to -16.7%), qualifying as a decline *(Figure 4)*. The trend shows strong fluctuations over time *(Figure 4)*.

Table 2. Average breeding density (birds/km2) of the Chukar in Armenia in five different habitat types, and estimated breeding population size (pairs). NA = Not available

2. táblázat A csukár átlagos fészkelési denzitása (egyed/ km2) és becsült populáció nagysága öt különböző élőhelytípusban Örményországban

Year	**Average breeding density per habitat type**											**Breeding population size**
	Juniper woodland (1,065 km^2)		**Lower subalpine zone (100 km^2)**		**Mountain steppe (2,233 km^2)**		**Semi-desert (1,684 km^2)**		**Semi-urban areas (124 km^2)**		**Total counts**	**½ sum across habitats of density x area**
	Average	**(n)**	**Average**	**(n)**	**Average**	**(n)**	**Average**	**(n)**	**Average**	**(n)**		**Total pairs**
2003	13.93	11	3.11	9	7.69	11	11.83	13	NA	0	44	26,119
2004	13.10	11	2.84	9	7.04	11	10.29	13	NA	0	44	23,635
2005	14.27	11	3.11	9	6.94	11	12.07	13	NA	0	44	25,670
2006	12.60	12	3.89	10	4.81	13	7.19	13	NA	0	48	18,334
2007	8.93	12	3.50	10	5.56	13	6.00	13	NA	0	48	16,187
2008	7.93	13	3.63	13	4.35	15	5.40	15	NA	0	56	13,816
2009	8.60	13	3.76	13	4.44	15	6.24	15	NA	0	56	14,982
2010	10.27	13	4.29	13	5.37	15	6.95	15	NA	0	56	17,531
2011	15.27	13	5.08	13	5.83	15	8.02	15	NA	0	56	21,652
2012	12.10	13	5.34	13	7.50	15	9.21	15	NA	0	56	22,843
2013	14.43	15	5.08	16	5.46	19	7.55	18	NA	0	68	20,394
2014	12.10	15	4.95	17	6.02	24	7.55	20	NA	0	76	19,765
2015	15.27	15	6.66	19	5.65	27	7.19	21	NA	0	82	20,823
2016	11.10	15	4.68	19	3.98	27	6.12	21	NA	0	82	15,743
2017	7.93	15	3.63	19	4.72	27	5.76	21	NA	0	82	14,530
2018	10.93	15	4.42	19	4.72	27	6.83	21	2.50	5	87	17,217
2019	8.93	15	4.55	19	3.15	27	4.57	21	2.00	5	87	12,472
Average across years ± *SD*	**11.63±2.55 (*n*=17 years)**		**4.27±0.98 (*n*=17 years)**		**5.48±1.27 (*n*=17 years)**		**7.58±2.14 (*n*=17 years)**		**2.25±0.35 (*n*=2 years)**			**18,924±4,139 (*n*=17 years)**

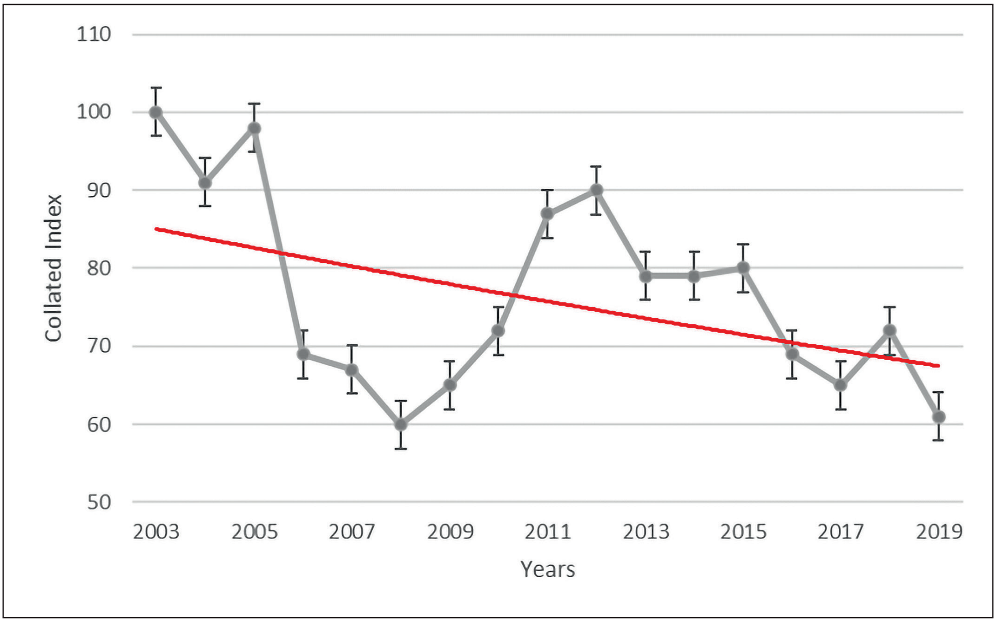

Figure 4. Annual TRIM index (relative to 2003, which is standardized to 100) of Chukar abundance in Armenia during 2003–2019.The black line is the best-fitting curve with a constant rate of change of -1.41% (*SE* = 0.24) per annum

4. ábra A csukár abundanciájának éves TRIM indexe (2003-hoz viszonyítva, amely 100-ra van szabványosítva) Örményországban 2003–2019 között. A fekete vonal az évi változási rátát mutatja (-1,41%, SE = 0,24)

Chukars on hunted and non-hunted areas

In Armenia, Chukars are present on 20 public hunting lands with a total area of 2,414 km^2. Based on areas within them of semi-desert (1,052 km^2), juniper woodland (80 km^2), mountain steppe (1,254 km^2), and subalpine habitats (28 km^2), the total number of breeding pairs of the species within the public hunting lands is estimated at 8,213 (95% confidence limits 6,082 to 10,343) in 2018 *(Table 3)*. Likewise, the total number of breeding Chukar pairs on non-hunted lands in 2018 is estimated at 9,444 (95% confidence limits 7,664–11,224) *(Table 3)*.

Despite apparent differences in density within habitats between hunted and non-hunted areas *(Table 3)*, there was no detectable interaction between habitat and hunted status ($F_{3,78} = 1.25$, $P = 0.297$), and no significant difference in overall density between hunted and non-hunted areas in 2018 ($F_{1,85} = 0.03$, $P = 0.870$) or 2019 ($F_{1,85} = 0.29$, $P = 0.591$).

Over the period 2003–2019, the average annual rate of change in overall Chukar density on hunted areas was negative (-3.44%, SE = 0.77, $t_{15} = 4.47$, $P < 0.001$). On non-hunted areas, it did not differ significantly from zero (-1.00%, SE = 0.89, $t_{15} = 1.12$, $P = 0.280$). The two rates differed significantly ($F_{1,30} = 4.30$, $P = 0.047$). In order to 2018 and 2019 densities to be similar on hunted and non-hunted areas, the starting density on hunted areas needed to be higher than on non-hunted ones, and in fact in the first three years (2003–2005) density averaged 10.74 on hunted areas versus 7.99 on non-hunted areas.

Table 3. Habitat areas, habitat-specific Chukar densities (birds/km^2 ± 1 SE), number of counts (in parentheses) and estimated population sizes (pairs, 95% confidence interval in brackets) in hunted and non-hunted parts of the Chukar range in Armenia in 2018 and 2019

3. táblázat A csukár élőhelyek területnagyságai, az élőhely-specifikus egyedűrűség (madár/km^2 ± SE), a felmérések száma (zárójelben) és becsült a populációméretek (párok, zárójelben 95%-os konfidencia intervallum) az örményországi elterjedési terület vadászott és nem vadászott részein 2018-ban és 2019-ben

Hunting status		**Juniper woodland**	**Lower subalpine zone**	**Mountain steppe**	**Semi-desert**	**Semi-urban areas**	**Overall**	**Population size (pairs)**
Hunted	Area (km^2)	80	28	1254	1052	0	2414	
	Density 2018	7.50±0.00 (1)	5.00±0.00 (1)	5.00±0.62 (12)	8.95±1.93 (11)	NA	6.80±0.90	8,213 (6,082-10,343)
	Density 2019	10.00±0.00 (1)	2.50±0.00 (1)	3.33±0.56 (12)	5.77±1.72 (11)	NA	4.60±0.80	5,558 (3,656-7,460)
Not hunted	Area (km^2)	985	72	979	632	124	2792	
	Density 2018	11.18±1.71 (14)	4.39±0.43 (18)	4.50±0.50 (15)	4.50±0.73 (10)	2.50±0.79 (5)	6.77±0.65	9,444 (7,664-11,224)
	Density 2019	8.86±1.17 (14)	4.67±0.45 (18)	3.00±0.75 (15)	3.25±0.75 (10)	2.00±0.94 (5)	5.12±0.52	7,151 (5,729-8,574)
Overall	Area (km^2)	1065	100	2233	1684	124	5206	
	Density 2018	10.93±1.61 (15)	4.42±0.41 (19)	4.72±0.39 (27)	6.83±1.16 (21)	2.50±0.79 (5)	6.61±0.53	17,217 (14,528-19,905)
	Density 2019	8.93±1.09 (15)	4.55±0.44 (19)	3.15±0.43 (27)	4.57±0.99 (21)	2.00±0.94 (5)	4.79±0.43	12,472 (10,266-14,677)

Chukar hunting management

According to the seven Hunters' Unions of Armenia, there are over 50,000 hunters in the country. However, the number of active hunters is reckoned by the Hunters' Unions to lie between 10,000 and 20,000 people. The number of hunting permits issued annually (one permit per Chukar allowed to be shot) increased ten-fold between 2016 and 2019 *(Table 4)*. The table also shows that survey data was used for decision making only in 2017 for the hunting season 2017–2018.

Out of the 500 hunters surveyed in 2019, 146 (29%) of them responded that they hunt Chukar annually. Among those 146 hunters, the number of Chukars shot per person per annum varied from 1 to 15, with an average (± SD) of 5.88±3.05 birds. Extrapolation of that number to a total of 10,000 active hunters, of which 2,900 shoot Chukars, results in an estimated 17,052 Chukars shot per year (95% confidence limits 14,342–19,762), while in case of 20,000 active hunters, the estimate is 34,104 (95% confidence limits 28,685–39,523) Chukars shot per year. These compare with a rough population estimate of 24,639 (95% confidence limits 18,246–31,029) Chukars present in hunted areas in autumn 2018, implying a harvest rate between 46% and 100%. The 19,534 Chukar hunting permits issued in 2018 *(Table 4)* represented 79% of the autumn population size. The number of 30,006 permits issued in 2019 was nearly double an estimated 2019 autumn population size in hunted areas of 16,674 birds.

Interviews with the heads of seven Hunters' Unions established that the hunters obtain hunting permits based on two recommendations from existing hunters and a face-to-face interview. The questions asked at the interview cover weapon safety but do not assess knowledge on game bird species identification, which public lands are open to hunting, which species are Red-listed, which hunting methods are allowed and which ones prohibited, daily bag limits, cases of poaching and the punishments meted out for infractions.

Table 4. Hunting permits issued for Chukar in the hunting seasons of 2016–2019 in Armenia. NA = Not available

4. táblázat Az Örményországban csukárra kiadott vadászati engedélyek száma a 2016–2019-es vadászati szezonban. (NA = nincs adat)

Hunting season	**Number of hunting permits for Chukar**	**Count of adult Chukar individuals in public hunting lands**	**Year of count**	**Comments**
2016-2017	3,040	NA		Number of permits fixed without a population count
2017-2018	2,860	27,610	2017	Count data provided by TSE NGO, used to determine number of permits
2018-2019	19,534	NA		Same count data used as in 2017, calculation method is unknown
2019-2020	30,006	NA		Number of permits fixed without considering the 2017 count

The interview with the State Inspectorate body established that during the last four years there were over 100 cases of poaching of Chukars, defined as hunting during the closed season or away from public hunting lands. In the same period the Inspectorate stated that there were no recorded cases of hunters exceeding daily bag limits. However, the Inspectorate pointed out that the absence of such records is probably because the number of inspections is very low, owing to understaffing within the Inspectorate body and a lack of financial resources allocated for the inspection process. The Inspectorate also noted an absence of cooperation between the Inspectorate and the Hunters' Unions, in contrast to the situation that prevailed in Soviet times (before independence in 1991), when such cooperation was very efficient and hunters volunteered for the inspection process, keeping poaching at a low level.

Discussion

Chukar population status

Adamian and Klem (1999) summarise the historical distribution of the Chukar in Armenia, and state that the species is found in the Central and Southern regions. It does not mention the elevation range, but does note that it inhabits semi-deserts, juniper thickets, and arid mountain steppes. By comparison with the results of our surveys, it therefore appears that since 1995 the Chukar has expanded its latitudinal distribution, occupying the north-eastern regions of the country, as well as its vertical range as it has colonised the lower subalpine zone. Such an altitudinal shift may be related to climate change, which has been documented in Armenia (Ministry of Nature Protection 2015).

Our estimate of 20,077 pairs (range 13,816–26,119) for 2013–2018 is more than twice as high as the BirdLife International (2015) estimate of 5,000–12,000 pairs. Likewise, our estimate of 17,278 pairs (range 14,530–20,823)) for 2013–2018 is four times higher than the BirdLife International (2021) estimate of 3,600–5,200 pairs. This is good news for the status of the Chukar in Armenia as it is more abundant than previously believed. The fluctuations in annual Chukar abundance that we have observed could be related to several possible reasons: variations in hunting pressure (Besnard *et al.* 2010, Mustin *et al.* 2011), outbreaks of disease, such as the ones detected in Red-Legged Partridge *Alectoris rufa* (Millán 2009), weather (Guzmán *et al.* 2020) and changes in food availability for adults or young (e.g. Norman 2008, Potts 2012). Our results suggest indirectly that hunting pressure may be influencing density in that in the two habitat zones that are predominantly non-hunted, juniper woodlands (8% hunted from *Table 3*) and the lower subalpine zone (28% hunted), the average annual rate of change over 2003–2019 either does not differ significantly from zero or is increasing, whereas significant declines were observed for mountain steppe (56% hunted) and semi-desert (62% hunted). At the level of the whole country, the long-term average annual population change of -1.4% per annum has resulted as classifying the trend as a decline.

The conservation status of the Chukar was not evaluated for the latest edition of the Red Book of Animals of Armenia, as it was considered a common game bird (Aghasyan &

Kalashyan 2010). Under IUCN Red List guidelines, the time period over which to assess population change is three generation lengths, which in the case of the Chukar is 11.7 years (from the BirdLife global assessments at http://datazone.birdlife.org). This translates into a decline of 14%, well below the threshold of 30% needed to qualify under Red List criterion Taking into consideration its distributional range and population size, the species does not meet the requirements under Red List criteria B, C or D and should be considered as Least Concern (IUCN Standards and Petitions Committee 2019) in Armenia. Nevertheless, the slow long-term population decline should be viewed as a warning signal, indicating the need for continued monitoring and precautionary measures.

Hunting management

Many of the possible causes of the long-term decline of the Chukar population are ones linked to fluctuations in the natural environment, which are difficult to counteract. However, hunting is a human activity under state control, so this is one area where it is possible to ensure sustainability by improving the way in which hunting is managed. Our data imply a harvest rate in 2018 between 46% and 100%, which is unsustainably high. We draw attention to several management issues that are currently deficient and need to be addressed. First, the lack of game bird monitoring, which leads to uninformed decision-making when deciding how many individual birds may be hunted each year. This is mostly the result of a lack of targeted financial resources and a lack of cooperation between the Ministry of Environment, the Hunters' Unions and the NGOs which are able to organize volunteer-based bird monitoring. Second, there are few checks in the field on the number of shot specimens, meaning that hunters are able to shoot more birds than is decreed in their hunting permits without being caught. Again, the causes are a lack of finances and inadequate cooperation with the Hunters' Unions. In turn, the lack of financial resources exists mainly because the prices paid for the hunting permits are not directly targeted at species monitoring and in-field hunting inspections. Instead, the Department for licenses, permits and compliances (responsible for monitoring) and the State Inspectorate Body (responsible for control) receive an annual budget from the State, which is not enough for proper implementation of game bird monitoring and hunting inspections (State Inspectorate, personal communication).

It could be thought that an alternative to the hunting of wild Chukars is the establishment of game farms where the indigenous Chukars are bred in captivity and, starting from the second generation, are released for shooting. Currently, Armenian law significantly complicates obtaining the necessary licenses for such business ("Zinvors" Hunters' Union, personal communication). In fact, this is likely to be a blessing in disguise, because there are many reports of negative effects of released game birds on wild populations (e.g. reviews in Sokos *et al.* 2008, Mustin *et al.* 2011). Releasing reared Chukars for shooting in Armenia will inevitably increase the hunting pressure on wild Chukars (cf. Potts 1986, Watson *et al.* 2007, Sokos *et al.* 2008, Mustin *et al.* 2011), as well as increasing the risks to wild birds of disease transmission and genetic contamination by increasingly domesticated and maladapted captive-bred birds of potentially dubious genetic stock (Blanco-Aguilar *et al.*

2008, Barbanera *et al.* 2009, 2010, Mustin *et al.* 2011). Moreover, hunters themselves rate wild birds much more highly as quarry than released captive-reared birds (e.g. Delibes-Mateos *et al.* 2014).

Recommendations

As part of a programme to halt the decline in the Armenian Chukar population, we recommend starting with improving the management and control of hunting. In particular, we suggest the following: (1) set up annual monitoring of the Chukar on public hunting lands (along with other game bird species), which should include counts of adult birds and their offspring; (2) develop a mechanism to base decisions concerning the annual number of hunting permits on the monitoring data; (3) develop alternative mechanisms for allocating the funds which are generated from the sale of hunting permits, targeting the revenue towards monitoring the populations of the Chukar and other game species, and towards better control of hunting and poaching on the ground; (4) develop a new State exam for obtaining a hunting licence aimed at having better educated and more responsible hunters; (5) strengthen the current legislation that makes it difficult to allow the releasing of captive-bred Chukars for shooting; (6) consider implementing winter feeding of Chukars (and probably other game birds) on public hunting lands to decrease their mortality during harsh winters. Further elements of the programme should be to study the reasons for annual changes in breeding success, and find out more about the epidemiology of Chukar diseases to better understand the causes of the short-term population fluctuations.

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Sacred green spaces in semi-urban areas sustain more birds than its adjacent areas: A study from lower Gangetic plains, West Bengal, India

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Abstract With rapid urban expansion and flourishing real estate sector, remaining green patches in many sub-urban/peri-urban areas are facing immense threat and/or being destroyed. We therefore, explored the avian abundance and richness of green spaces around temples (sacred sites) and compared them with adjoining green spaces without religious places (control sites). The species richness and abundance in sacred sites (12.16 ± 0.65 species; 25.54 ± 1.176 individuals) was significantly higher than control sites (6.31 ± 0.77 species; 20.04 ± 1.4 individuals). The compositions of avian communities of sacred sites were significantly different and the presence of temple positively influenced the species richness. GLMM also revealed that the species richness was positively influenced by the distance to building and tree cover area and not influenced by distance to road, areas of water body, bare land. Our findings indicate that the green spaces around the sacred places have greater avian diversity in semi-urban areas, and could be prioritized for the conservation of avian diversity. Generating local support could be relatively easier due to traditional, religious and/or cultural belief against tree felling around the places of worship.

Keywords: Avifauna; Species richness, Sacred, green spaces, urbanization

Összefoglalás A gyors városi terjeszkedés és az ingatlanszektor virágzása miatt számos külvárosi és városkörnyéki zöldterületet óriási veszély vagy pusztulás fenyeget. Ezért megvizsgáltuk a templomok (szent helyek) körüli zöldterületek madárvilágát és fajgazdagságát a szomszédos, nem vallási jellegű zöldterületek (kontroll területek) összehasonlításában. A szent helyek fajgazdagsága és a fajok abundanciája (12,16 ± 0,65 faj; 25,54 ± 1,176 egyed) statisztikailag is kimutathatóan magasabb volt, mint a kontroll területeken (6,31 ± 0,77 faj; 20,04 ± 1,4 egyed). A szent helyek madárközösségeinek összetétele szignifikánsan különbözött, és a templom jelenléte pozitívan befolyásolta a fajgazdagságot. A GLMM azt is kimutatta, hogy a fajgazdagságot pozitívan befolyásolta az épületek távolsága és a fával borított terület nagysága, de nem befolyásolta az úttól való távolság, a vízfelület és a kopár terület nagysága. Eredményeink azt mutatják, hogy a szent helyeket körülölelő zöldterületek nagyobb diverzitással rendelkeznek a kevésbé városi területekhez képest, és elsődlegesek lehetnek a sokféleség megőrzése szempontjából. A helyi támogatások megszerzése viszonylag könnyebb lehet a hagyományos, vallási és/vagy kulturális hiedelmek miatt, amelyek ellenzik a fák kivágását a vallási helyek körül.

Kulcsszavak: madárvilág, fajgazdagság, szent, zöldterület, városiasodás

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Introduction

Birds are widely distributed in various ecosystems (Mekonen 2017) and are often abundant in areas with suitable survival conditions (Veech *et al.* 2010). Urban areas across the globe are also inhabited by many species of flora and fauna (McKinney 2008, Shwartz *et al.* 2014), including birds (Echevema & Vassallo 2008, Hu & Cardoso 2009). However, there are marked differences between natural and human dominated ecosystems (Alberti 2005).

Cities and towns across the globe are ever-expanding with explosion in human population (Kumar 2017) leading to large scale destruction of many natural habitats, especially in the peri-urban areas. Such loss, degradation and fragmentation of habitats coupled with environmental changes (McKinney 2006, Bar-Massada *et al.* 2014), destruction of trees (Watson *et al.* 2004), reduction of wilderness areas (Dumont 2012, Di Marco *et al.* 2019) and increase in impervious surfaces (Souza *et al.* 2019), which often threatens the survival of many species thriving in these areas (Marzluff *et al.* 2001, Sol *et al.* 2017). In such a scenario, various native green spaces served as important remnant patches of habitats for wildlife thriving in human-dominated landscapes (McKinney 2002, Shwartz *et al.* 2014) and have received importance for wildlife conservation in many European countries (Clergeau *et al.* 2001). Apart from positively influencing the wellbeing and good health of human citizens (Cohencline *et al.* 2015, Shanahan *et al.* 2015, Botzat *et al.* 2016), green spaces in urban areas are often rich in biodiversity (Shwartz *et al.* 2014) and have long been identified to increase the functional connectivity for the local fauna (Ikin *et al.* 2015). For these reasons, urban green spaces have received importance for wildlife conservation in many western countries (Clergeau *et al.* 2001).

Avian assemblages often serve as useful indicator of the habitat quality and biodiversity of any particular area (Alexandrino *et al.* 2016). Therefore, studies investigating bird species richness and diversity have been carried out in parks and forest remnants (Croci *et al.* 2008, Nielsen *et al.* 2014), cemeteries (Lussenhop 1977, Löki *et al.* 2019) and sacred groves (Kumar & Chhaya 2015) in urban areas. However, with flourishing real estate sector and rapid urbanization in many sub-urban/peri-urban areas, several green spaces have already been converted to built-up areas and many other remaining green patches are facing similar imminent threats, which make conservation of these habitats increasingly challenging for urban planners in the developing nations (Ikin *et al.* 2015, Gopal *et al.* 2018). The conservation biologists need to explore new areas within human dominated landscapes to conserve biodiversity. Religious places are often surrounded by green spaces and are the integral part of almost every sub-urban and urban landscapes. Yet, there is a clear paucity of studies on avian diversity from the green spaces around sacred/religious places from the developing nations with intense population explosion and rapid urbanization, and is practically absent from any semi-urban areas of a highly populated country like India. Therefore, we carried out this study in a semi-urban area (i) to make an assessment of the community composition, species richness and abundance of avifauna in the sacred green spaces compared to the adjoining matrix of non-sacred green spaces; and (ii) explored the relationship of habitat features with the species richness of birds. Our findings highlight the importance of the sacred sites in the semi-urban areas in sustaining avian diversity.

Methods

Data collection

Field surveys were carried out from January 2017 – May 2017 in Ashoknagar and adjoining areas (22.833°N, 88.633°E), which is a semi-urban area located in the lower Gangetic plains, India *(Figure 1)*. We selected three green spaces with temples (hereafter referred as a "sacred site") maintaining a minimum gap of 500 m between them. Temples are the building/architectures used as places of worship, which are usually surrounded by several large, old trees and other greeneries within their boundary. Adjacent to each sacred site, we selected one green space without temple (henceforth "control site") maintaining a minimum gap of 500 m between the sacred sites and their respective adjacent control sites. To assess the abundance and diversity of birds, we adopted belt transect method (Bibby *et al.* 2000). For this purpose, two fixed transects (length 100 m and width 10 m) were placed on each of the study sites keeping a gap of 200 m between adjacent transects to avoid overlapping of data. Thus, a total of 12 belt-transects (2 transects * 3 sacred sites + 2 transects * 3 control sites) were laid in the study area. Each of these transect was traversed on foot, twice in a month from January to May during days with calm weather conditions (without rain and strong wind) and during morning hours (between 07:00 and 09:30), when birds are usually most active. Equal efforts

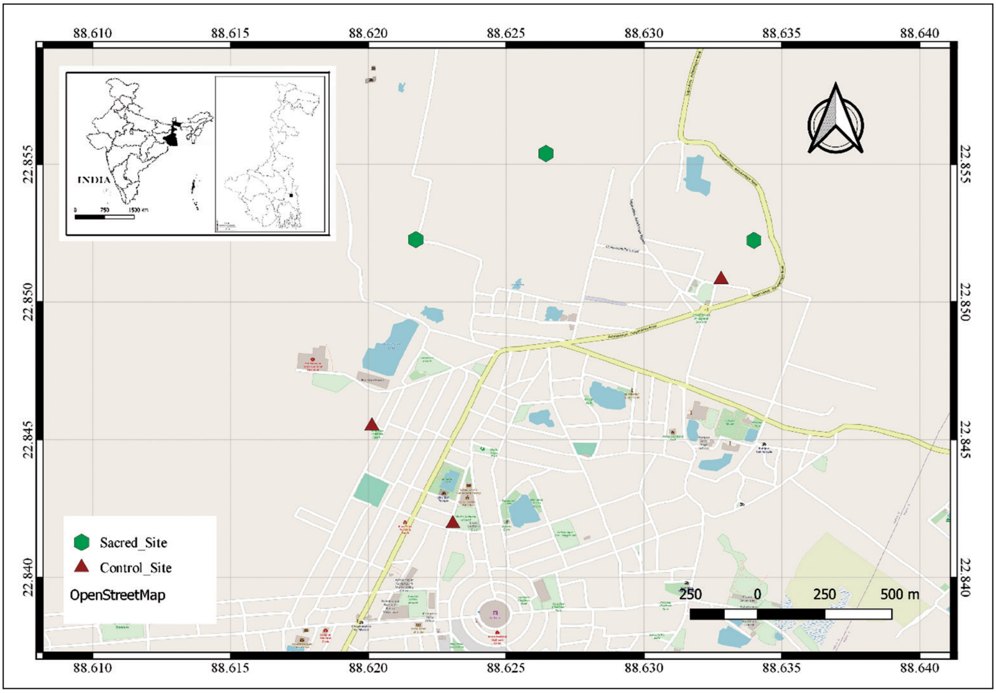

Figure 1. Location of the study area in West Bengal, India (a) and the locations of "sacred sites and control sites" within the study area (b)

1. ábra A vizsgálati terület elhelyezkedése India Nyugat-Bengál tartományában (a) és a felmért szent helyek és kontroll területek feltüntetésével (b) were made in each of the sampling sites and ten observations were made at each transect (i.e. twice in a month for five months). While traversing along the transects, we collected data on the abundance (total number of individual birds) and species richness (total number of avian species recorded) of avifauna that is present between 5 m on either side of transects. Bird were observed either with unaided eyes or with the help of a pair of binoculars (Nikon 8 × 40), identified using bird field guides (Grimmett *et al.* 2011) and photographs were taken with a digital camera (Fuji Finepix S6800) for documentation of the avifauna. Land covers are often considered as important habitat features for birds (Litteral & Shochat 2017). Google Earth satellite images are freely available with high resolution synoptic view of the study area and hence, used for the assessment of land cover features (Hu *et al.* 2013, Barik *et al.* 2021). We calculated the area of tree cover, bare land cover within the belt transect (i.e. 100 m × 10 m) by drawing separate polygons over the outer boundary of each land cover feature and measured the linear distance to nearest road, building and water body using a cloud free Google image in Google Earth Pro platform v7.3.3.7699.

Data analysis

Shapiro-Wilk's tests revealed that the abundance ($W = 0.938$, $df = 45$, $P = 0.02$) and species richness ($W = 0.94$, $df = 45$, $P = 0.02$) were non-normally distributed, over-dispersed and negatively skewed. Non-metric multidimensional scaling (henceforth, NMDS) is considered as a popular statistical method to compare bird communities between study sites (Legendre & Legendre 1998). Thus, we performed NMDS along with two-dimensional stress-plot ($K = 2$) to compare avian assemblages between sacred sites and control sites using Bray-Curtis similarity index. As the data was non-normally distributed, we carried out Kruskal-Wallis tests to evaluate the difference of abundance and species richness of avifauna between sacred sites and control sites (Kruskal & Wallis 1952). Thereafter, we performed Generalized Linear Mixed Model (GLMM) with negative binomial distribution and log link function to examine the effect of landscape variables on species richness of birds (Paun *et al.* 2019). We considered presence of temple, distance to road, building, water body, and areas of tree cover and grassland as fixed factors and study site as a random factor in the model to test their effects on species richness (response variable). All statistical analyses were performed in PAST, SPSS and RStudio software packages. Significance was tested at $P < 0.05$ and data were presented as mean ± standard error.

Results

We recorded 45 species of birds belonging to 9 orders and 25 families during the study *(Table 1)*, of which the order Passeriformes has the highest number of species ($n = 25$). Most of the recorded species were resident (88.88%). NMDS of avian assemblages revealed that the species composition of bird communities in all three control areas overlapped, but were clearly and significantly different from the sacred areas *(Figure 2)*. We found a stress value of the NMDS ≤ 0.2, which indicates good representation of data (Tryjanowski *et al.*

Table 1. The bird species observed in the sacred sites and control sites together with their respective taxonomic positions (order, family), feeding guild (Car – carnivore, Omn – omnivore, Herb – herbivore, Nect – nectarivore, Gran – granivore, Ins – insectivore, Frug – frugivore), Habitat Specialization (G – generalist, S – Specialist) and global population trend (↓ – declining, ↑ – increasing, → – stable, and ? Unknown)

1. táblázat A szent helyeken és a kontroll területeken megfigyelt madárfajok, valamint azok taxonómiai helyzete (rend, család), táplálkozási guildje (Car – ragadozó, Omn – mindenevő, Herb – növényevő, Nect – nektárevő, Gran – nagyevő, Ins – rovarevő, Frug – gyümölcsevő), élőhelyi specializációja (G – generalista, S – specialista) és globális populációs trendja (↓ – csökkenő, ↑ – növekvő, → – stabil ? – ismeretlen)

SL. No.	Order	Family	Common name	Scientific name	Feeding guild	Habitat Specialization	Global Population Trend	Sacred Site	Control site
1	Columbiformes	Columbidae	Spotted Dove	*Streptopelia chinensis*	Gran	G	↑	✓	✓
2	Columbiformes	Columbidae	Yellow-footed Green Pigeon	*Treron phoenicopterus*	Gran	G	↑	✓	-
3	Columbiformes	Columbidae	Rock Dove	*Columba livia*	Gran	G	↓	✓	✓
4	Coraciiformes	Alcedinidae	Stork-billed Kingfisher	*Pelargopsis capensis*	Car	S	↓	✓	-
5	Coraciiformes	Alcedinidae	White-throated Kingfisher	*Halcyon smyrnensis*	Car	G	↑	✓	✓
6	Coraciiformes	Alcedinidae	Common Kingfisher	*Alcedo atthis*	Car	G	?	✓	-
7	Coraciiformes	Meropidae	Green Bee-eater	*Merops orientalis*	Ins	G	↑	✓	
8	Cuculiformes	Cuculidae	Lesser Coucal	*Centropus bengalensis*	Omn	S	↑	✓	✓
9	Cuculiformes	Cuculidae	Asian Koel	*Eudynamys scolopaceus*	Omn	G	→	✓	✓
10	Gruiformes	Rallidae	White-breasted Waterhen	*Amaurornis phoenicurus*	Omn	G	?	✓	-
11	Passeriformes	Dicruridae	Black Drongo	*Dicrurus macrocercus*	Ins	G	?	✓	-
12	Passeriformes	Dicruridae	Bronzed Drongo	*Dicrurus aeneus*	Ins	S	?	✓	-
13	Passeriformes	Aegithinidae	Common Iora	*Aegithina tiphia*	Ins	S	?	✓	-

SL. No.	Order	Family	Common name	Scientific name	Feeding guild	Habitat Specialization	Global Population Trend	Sacred Site	Control site
14	Passeriformes	Cisticolidae	Common Tailorbird	*Orthotomus sutorius*	Ins	G	→	✓	✓
15	Passeriformes	Corvidae	Rufous Treepie	*Dendrocitta vagabunda*	Ins	G	↓	✓	-
16	Passeriformes	Corvidae	House Crow	*Corvus splendens*	Omn	G	→	✓	✓
17	Passeriformes	Corvidae	Large-billed Crow	*Corvus macrorhynchos*	Omn	G	→	✓	✓
18	Passeriformes	Estrildidae	Tricoloured Munia	*Lonchura malacca*	Gran	S	→	✓	-
19	Passeriformes	Laniidae	Brown Shrike	*Lanius cristatus*	Ins	S	↓	✓	-
20	Passeriformes	Leiothrichidae	Jungle Babbler	*Turdoides striata*	Omn	G	→	✓	✓
21	Passeriformes	Monarchidae	Indian Paradise Flycatcher	*Terpsiphone paradise*	Inse	S	→	✓	-
22	Passeriformes	Monarchidae	Brown-breasted Flycatcher	*Muscicapa muttui*	Inse	S	↓	✓	-
23	Passeriformes	Muscicapidae	Oriental Magpie-Robin	*Copsychus saularis*	Inse	G	→	✓	✓
24	Passeriformes	Passeridae	House Sparrow	*Passer domesticus*	Omn	G	↓	✓	✓
25	Passeriformes	Muscicapidae	Red-throated Flycatcher	*Ficedula parva*	Ins	S	→	✓	-
26	Passeriformes	Nectariniidae	Purple Sunbird	*Nectarinia asiatica*	Nect	G	→	✓	-
27	Passeriformes	Nectariniidae	Purple-rumped Sunbird	*Leptocoma zeylonica*	Nect	G	→	✓	-
28	Passeriformes	Oriolidae	Black-hooded Oriole	*Oriolus xanthornus*	Omn	S	→	✓	✓
29	Passeriformes	Pycnonotidae	Red-whiskered Bulbul	*Pycnonotus jocosus*	Omn	G	↓	✓	-

SL. No.	Order	Family	Common name	Scientific name	Feeding guild	Habitat Specialization	Global Population Trend	Sacred Site	Control site
30	Passeriformes	Pycnonotidae	Red-vented Bulbul	*Pycnonotus cafer*	Omn	G	↑	✓	✓
31	Passeriformes	Sturnidae	Asian Pied Starling	*Gracupica contra*	Omn	G	↑	✓	✓
32	Passeriformes	Sturnidae	Chestnut-tailed Starling	*Sturnia malabarica*	Ins	S	?	✓	-
33	Passeriformes	Sturnidae	Common Myna	*Acridotheres tristis*	Omn	G	↑	✓	✓
34	Passeriformes	Sturnidae	Jungle Myna	*Acridotheres fuscus*	Omn	G	↓	✓	✓
35	Passeriformes	Turdidae	Orange-headed Thrush	*Geokichla citrina*	Ins	S	↓	✓	-
36	Pelecaniformes	Ardeidae	Black-crowned Night Heron	*Nycticorax nycticorax*	Car	S	↓	✓	-
37	Pelecaniformes	Ardeidae	Indian Pond Heron	*Ardeola grayii*	Car	G	?	✓	-
38	Pelecaniformes	Ardeidae	Cattle Egret	*Bubulcus ibis*	Car	S	↑	✓	✓
39	Piciformes	Picidae	Streak-throated Woodpecker	*Picus xanthopygaeus*	Ins	S	?	✓	-
40	Piciformes	Picidae	Lesser Golden-backed Woodpecker	*Dinopium benghalense*	Ins	G	→	✓	-
41	Piciformes	Picidae	Fulvous-breasted Woodpecker	*Dendrocopos macei*	Ins	S	→	✓	-
42	Piciformes	Ramphastidae	Blue-throated Barbet	*Psilopogon asiaticus*	Frug	S	→	✓	-
43	Piciformes	Ramphastidae	Coppersmith Barbet	*Psilopogon haemacephalus*	Frug	S	↑	✓	-
44	Psittaciformes	Psittaculidae	Rose-ringed Parakeet	*Psittacula krameri*	Frug	G	↑	✓	-
45	Strigiformes	Strigidae	Brown Hawk Owl	*Ninox scutulata*	Car	S	↓	✓	-

2017). Although, the avian communities in three control sites highly overlapped, yet the birds of three sacred sites did not overlap with each other *(Figure 2)*. Out of all species of birds recorded during this study, 28 species (62.23%) were noticed exclusively on sacred sites, 17 species (37.78%) were found only in the control sites and 17 species (37.78%)

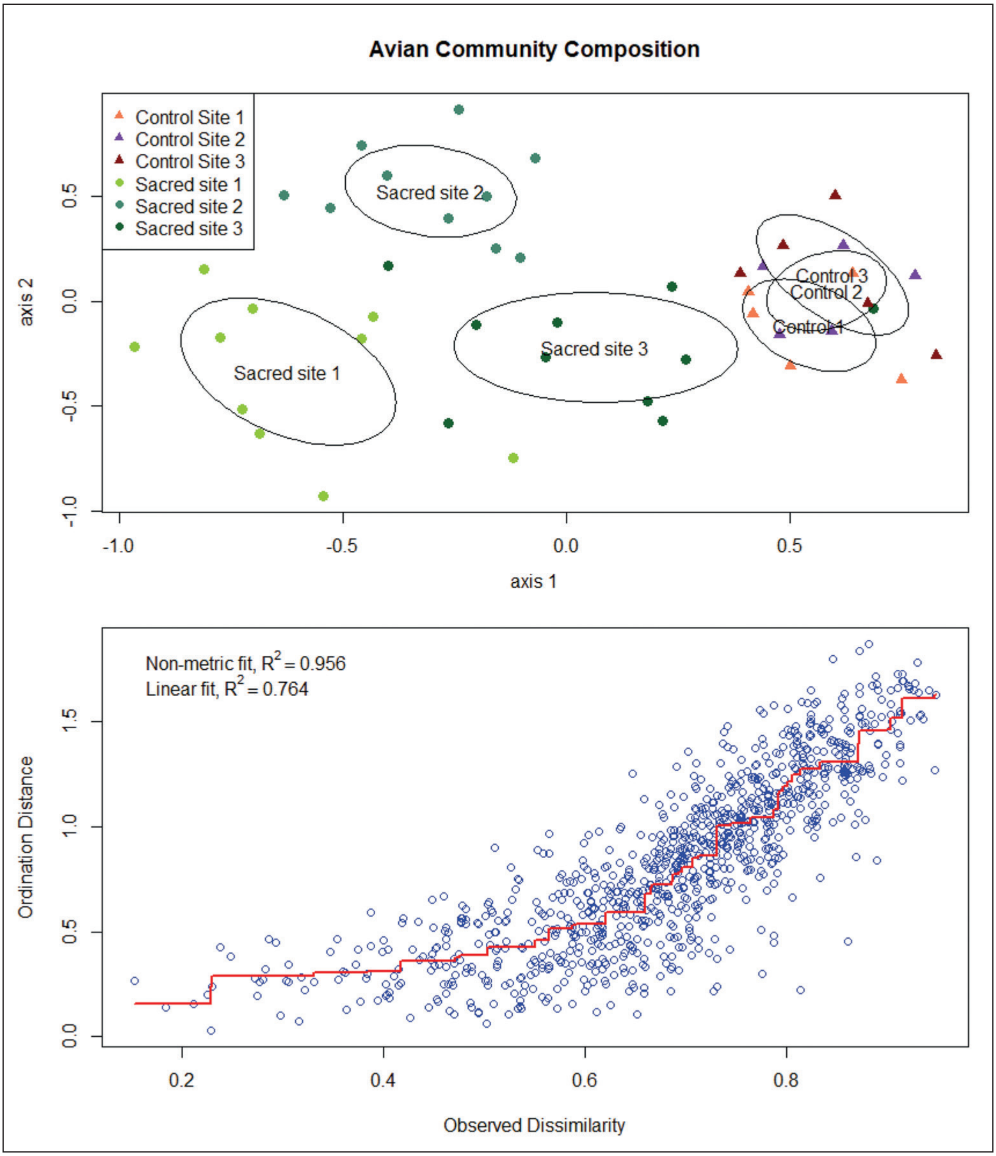

Figure 2. Non-metric multidimensional scaling (NMDS) of avian assemblages sampled in sacred sites and control sites using Bray-Curtis similarity index in the study area along with the stress-plot (2D stress = 0.23)

2. ábra A szent helyeken és a kontroll területeken felmért madárközösségek nem-metrikus többdimenziós skálázása (NMDS) a Bray-Curtis hasonlósági index segítségével, a stressz-diagrammal kiegészítve (2D stressz = 0,23)

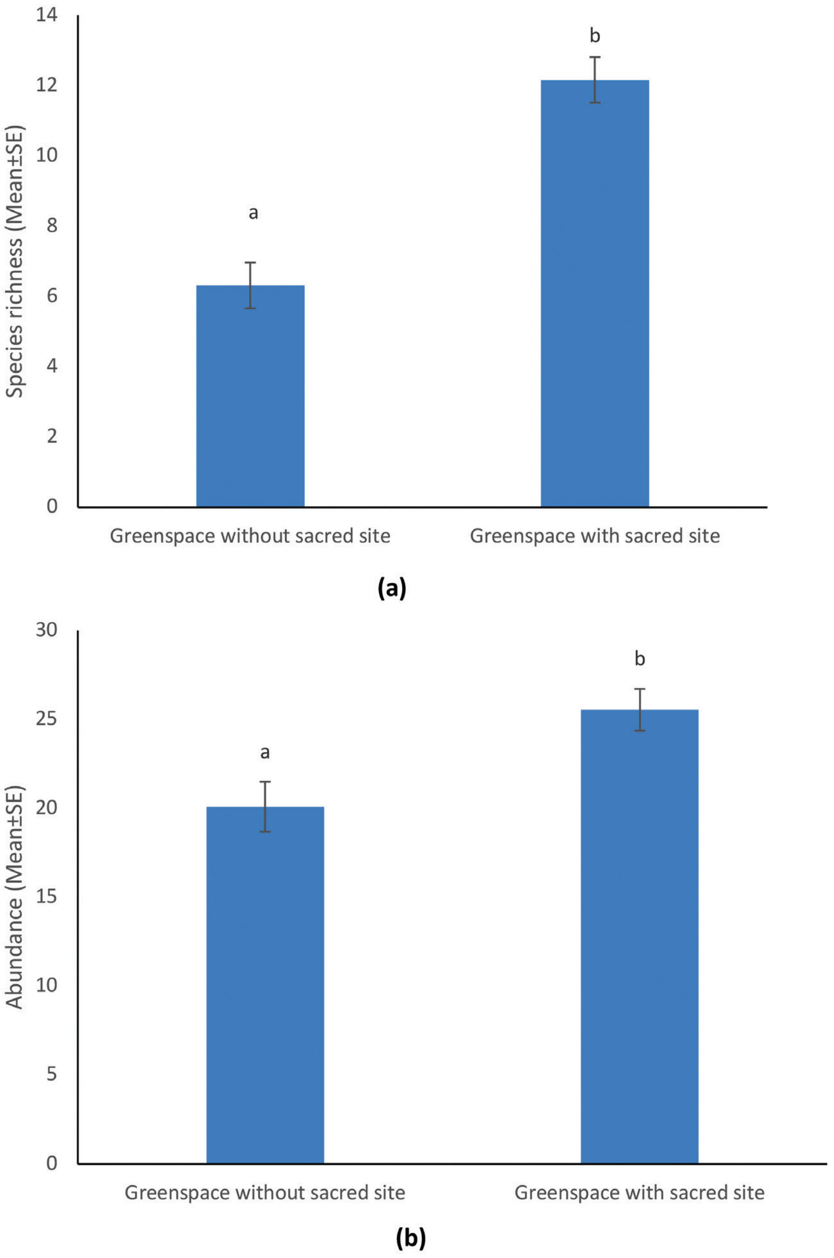

Figure 3. (a) Species richness and (b) Abundance of birds in sacred sites and control sites [Columns with different letters indicate significant difference (Bonferroni post hoc tests P < 0.05). Error bars indicate standard errors (SE) of means]

3. ábra A szent helyeken és a kontroll területeken mért fajgazdagság (a) és madárbőség (b), ahol az oszlopok betűjelei a szignifikáns eltéréseket jelölik (Bonferroni korrekció, P < 0,05), feltüntetve az átlagok standard hibáját is

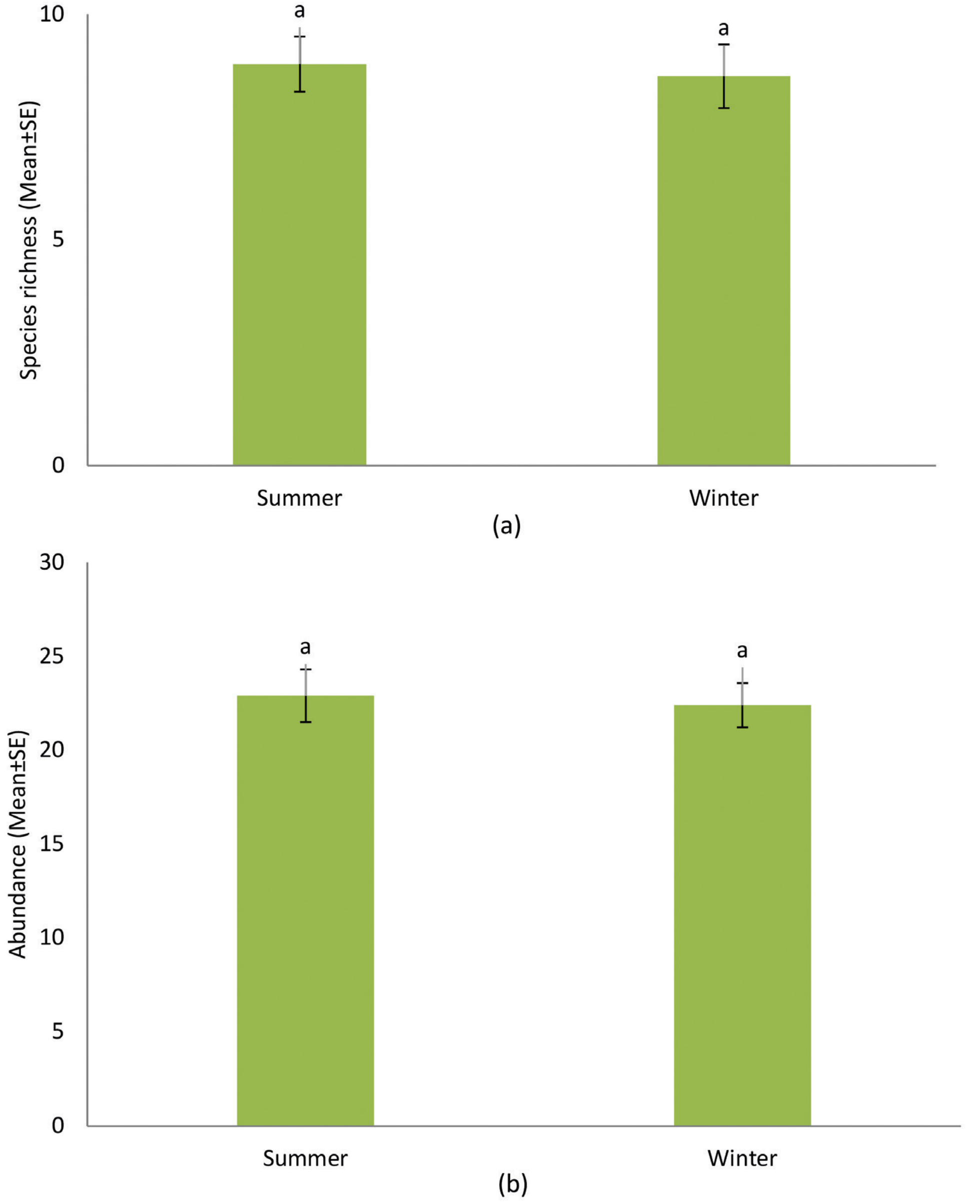

Figure 4. (a) Species richness and (b) abundance of birds during summer and winter [Columns with the same letters are not significantly different from each other (Bonferroni post hoc tests P > 0.05). Error bars indicate standard errors (SE) of means]

4. ábra A nyári és a téli időszakban mért fajgazdagság (a) és madárbőség (b), ahol az oszlopok betűjelei a szignifikáns eltéréseket jelölik (Bonferroni korrekció, P < 0,05), feltüntetve az átlagok standard hibáját is were common in both sacred and control sites. Among the unique species recorded from sites during the present study, 14 were insectivores, 5 were carnivores, 3 were frugivores and others were granivores, omnivores and nectarivores. Out of 45 species encountered in the study sites, 19 (42.3%) species were habitat specialists, of which 17 species were forest dwellers and mainly found only in sacred sites, while 26 (57.7%) were generalist species and were found in both sacred and control sites (SoIB 2020) *(Table 1)*.

Table 2. Generalized linear mixed model (GLMM) accounting for variation in urban bird species richness in relation to the habitat features Road, Building, Water body, Bare land and Tree cover. Significant variables (at P < 0.05) are in bold

2. táblázat Az általánosított lineáris kevert modell (GLMM) eredményei a madarak városi fajgazdagságának varianciáját magyarázandó változókkal, út, épület, vízfelület, kopár terület és faborítottság. A szignifikáns változók (P < 0,05) félkövérrel kiemelve

Fixed effects	β	SE	z	p
Presence of Temple	**0.65**	0.11	5.87	**4.27e-09**
Tree cover	**0.73**	0.13	5.64	**1.75e-08**
Bare land	0.14	0.1	1.42	0.16
Road	0.32	0.17	1.93	0.06
Building	**0.33**	0.1	3.26	**0.002**
Water body	0.08	0.12	0.71	0.48

The species richness in sacred sites (12.16 ± 0.65 species) was significantly higher (Kruskal-Wallis test: $H = 37.28$, $P < 0.05$) than in control sites (6.3 ± 0.77 species) as shown in *Figure 3a*. Likewise, *Figure 3b* shows that the abundance of birds were also significantly higher (Kruskal-Wallis test: $H = 10.97$, $P < 0.05$) in sacred sites (25.54 ± 1.176 individuals) as compared to their neighboring control sites (20.04 ± 1.403 individuals). But no significant seasonal difference was noticed in the abundance (Kruskal-Wallis test: $H = 0.1$, $P > 0.05$) and species richness (Kruskal-Wallis test: $H = 0.09$, $P > 0.05$) as shown in *Figure 4*.

GLMM revealed that the species richness was positively influenced by presence of temple, tree cover area and distance to building and was not influenced by distance to road, areas of water body, bare land *(Table 2)*. The areas of tree cover within the sacred sites were significantly larger ($H = 13.66$, df = 1, $P < 0.05$) and bare lands were significantly less ($H = 4.918$, df = 1, $P < 0.05$) than the control sites.

Discussion

Sacred groves in inaccessible and remote terrains are rich in biodiversity as they are less disturbed. The sacred sites can also serve as important areas for conservation of biodiversity in semi-urban and rural areas (Devkota 2013, Gopal *et al.* 2018). Urban cemeteries are known to serve as important habitat for many sensitive and threatened species (Čanády & Mošanský 2017). These areas often act as 'habitat islands' for many rare and native species (Morrehouse & Hassen 2004) and have been found to sustain rich biodiversity in many urban areas across the globe (Bhagwat & Rutte 2006, Kowarik *et al*. 2016). Green spaces around the religious places (i.e. sacred green spaces) are often integral component of many urban ecosystems around the globe. We found that the sacred green spaces (sacred sites) in human dominated areas sustained higher species richness of birds than its neighboring green spaces without sacred areas (control sites). During the present study, the species richness of birds did not show any significant seasonal difference between summer and winter, possibly because most of the birds were resident birds and were present throughout the year in our study area in different weather conditions.

Land cover features also give important cues for birds to decide upon whether to use that particular habitat or not (Cody 1981). Greater habitat heterogeneity and complexity of vegetation increase the species richness of avifaunal communities (Lorenzón *et al.* 2016, Mukhopadhyay & Mazumdar 2017, 2019). Importance of any green space to birds is influenced by the density and diversity of trees, number of native flora, as well as the amount of anthropogenic disturbances existing there (Mills *et al.* 1989, Chamberlain *et al.* 2007). We found that the species richness of birds during the present study were positively influenced by the areas of tree cover *(Table 2)*, which is in agreement with earlier studies (Chamberlain *et al.* 2007). Areas with rich vegetation in the green spaces usually sustain greater bird abundance as compared to areas with impoverished vegetation (Chace & Walsh 2006). Increasing tree cover provides crucial resources for the arboreal and forest birds (Ciach & Fröhlich 2017). Not only inside the green spaces, increased amount of vegetation cover in the adjoining urban areas also increased the richness and abundance of the native fauna (Ikin *et al.* 2013, 2015), while the surrounding dense urban matrix leads to the decline in the abundance and diversity of native birds (Canedoli *et al.* 2018). Moreover, buildings in urban areas are often impoverished of suitable resources (Rodewald *et al.* 2011). They also manifest greater associated anthropogenic pressures (Rodewald *et al.* 2011). Few urban exploiters may be found in greater numbers in areas closer to buildings, but most of the urban avoiders tend to avoid buildings (Blair 1996). During the present study, we also found habitat specialists (19 species), including 17 forest dwelling species, in the sacred sites and they clearly avoided the control sites. For these reasons, we found the species richness of birds being higher in areas away from buildings *(Table 2)*, and are therefore, attractive for birds. Human attitude plays very important role in shaping the composition of avifauna in any particular habitat (Reynaud 1995, Borghesio 2008). Sacred green spaces are often associated with various cultural and religious importance to the local people (Rutte 2011) and are considered as 'cultural heritage sites with conservation importance' (Verschuuren *et al.* 2010). The devotees and other people visiting the religious places are in a peaceful and contemplating state of mind, when they usually appreciate the presence of natural surroundings. People out of religious belief also refrain from destroying the flora and fauna in these sacred spaces even in the urban areas. Possibly for this attitude of people many religious places (at least in our study area) are surrounded by large amount of green spaces, with fair number of old trees, which in turn sustain rich avian diversity. We clearly found that the areas of tree cover within the sacred sites were significantly larger than their adjoining sites without temples. Such human attitude and religious belief against tree felling around the places of worship are often very effective in saving trees and other floral components in these areas (Nagendra 2016), which in turn make such sacred green spaces important from the perspective of conservation of avian diversity (McKinney 2002, Shwartz *et al.* 2014).

Our findings emphasize the importance of green spaces around religious places in conserving avian diversity in human dominated landscapes. Particularly in urban/

semi-urban areas in developing countries, which are rapidly losing the green spaces due to infrastructure development, the sacred green spaces often serve as stable, favorable and "keystone habitats" for several forest-dwelling birds and urban avoiders to thrive in the human dominated habitats (Brandt *et al.* 2013, Gopal *et al.* 2018). Hence, we suggest that the green spaces around the religious places in urban and semi-urban areas should be prioritized for the conservation of avian diversity, which might not only be beneficial for the conservation of avian communities, but also other species thriving in those areas. In spite of rapid urbanization, generating people support for conserving such sacred green spaces could also be relatively easier using the traditional religious / spiritual / cultural belief systems of people (Rutte 2011), as compared to many other natural habitats in human dominated areas. Managers and wildlife planners have to realize the importance of such sites from biodiversity conservation perspective. More studies need to be carried out in human dominated areas across the globe to understand the role of these sacred green spaces in conserving avian diversity in particular, and biodiversity parse.

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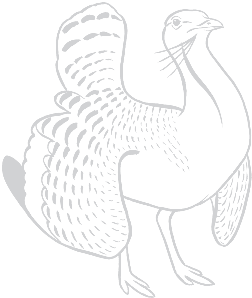

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Distribution and demographic trend of Algerian Nuthatch *(Sitta ledanti)* population of Mount Babor forest in Babor-Tababort National Park

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Summary The Algerian Nuthatch *(Sitta ledanti)* is the only bird species endemic to Algeria and it's a protected species by the Algerian law since 1983. So far, we have no precise information on recent changes in population density as well as the demographic trends of this endangered species in its entire range in the Kabylia of Babors. The population of the species in the Babor-Tababort National Park remains the most abundant, with 275 individuals recorded in 2020. Thirty-eight years after the 1982 census, it multiplied with an annual multiplication rate of (λ) = 1.01 and grew with an annual growth rate of r = 1.36%.The Algerian Nuthatch has been present throughout the Mount Babor forest from 1,300 m altitude to the summit at 2,004 m altitude. The Algerian Nuthatch, in 2020, was more abundant in the mixed cedar forest because this type of forest covers the largest area in Mount Babor. The population of the species in the Mount Babor forest remains isolated that, however, could be connected to the population in the Tababort forest through the setting of an ecological corridor.

Keywords: Algerian Nuthatch, Babor-Tababort National Park, distribution, growth, habitat

Összefoglalás Az Atlasz csuszka *(Sitta ledanti)* Algériában 1983 óta védett, egyetlen endemikus madárfaj. Az Kizárólag a Badors Kabylia régióban fordul elő, de nincsenek pontos információk a populációk nagyságáról és az egyedszámok változásáról. A Babor-Tababort Nemzeti Parkban 2020-ban végzett felmérés szerint az ott élő populációt kb. 275 egyed alkotja. Az 1982-ben (Ledant 1985) végzett felméréshez képest a populáció növekedett, éves átlagos növekedési rátája 1,36% volt. A Babor-hegy erdei élőhelyein az 1300 m tengerszint feletti magasságtól a 2004 m-es csúcsig mindenhol előfordul. A 2020-as felméréskor a legnagyobb számban a legnagyobb kiterjedésű, kevert cédrus erdőkben fordult elő. A Babor-hegyi populáció elszigetelt a többi költőpopulációtól. Fennmaradását segítené, ha ökológiai folyosókkal kapcsolódhatna a Tabort erdőség nagyobb populációihoz.

Kulcsszavak: Atlas csuszka, Babor-Tababort Nemzeti Park, elterjedés, denzitás, populációdinamika, élőhely

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Introduction

Population dynamics is a discipline that has as interest the description and understanding the fluctuations in number of individuals in a given population over space and time. Thus fluctuations result in growth, decline or stability (Besnard & Salles 2000, Kricsfalusy 2012). The population growth potential of vertebrates allows the creation of population reserves, which can only exist with the availability of heterogeneous and favorable habitats (Blondel 2000) that provide refuges in case of rare climatic events such as prolonged drought or extreme cold (Wiens 1977). Demographic analyses of long-term monitoring data allow exploration of a variety of ecological and evolutionary questions, ranging from social behavior (Boulinier *et al.* 1996) to adaptation to spatio-temporal habitat heterogeneity (Blondel 2007). This kind of analysis is also applicable to investigate the impact of global changes and human activities on bird populations (Lebreton 2019) through the management of exploited populations and conservation (Weimerskirch *et al*. 1997, Nichols & Williams 2006). Thus, studies of population dynamics are used in population viability analysis and threat assessment criteria (IUCN 2012).They are also involved in assessing the status of endemic and threatened species (Schemske *et al*. 1994).

The Algerian Nuthatch *Sitta ledanti* is the only endemic bird for Algeria (Isenmann & Moali 2000) and a protected species by the Algerian law since 1983 (J.O.R.A. 1983). Since its discovery in 1975 in the Mount Babor forest situated in the Babor-Tababort National Park (Ledant 1977, Ledant & Jacobs 1977), its numbers continue to increase thanks to new discoveries in new habitats. The Algerian Nuthatch is distributed over twelve different habitats, which are all with in the biogeographic area of Kabylia of Babors (Petite Kabylie) in northeastern Algeria (Chalabi 1989, Bellatrèche & Chalabi 1990, Moulai & Mayache 2018, Haddad & Afoutni 2019, Bougaham *et al*. 2020, Mayache & Moulai 2021, Bougaham *et al*. 2021). Despite the increase of its numbers, the Algerian Nuthatch still remains on the IUCN Redlist with numbers below 1000 individuals (IUCN 2012, Hamitouche & Bougaham 2021). The Algerian Nuthatch population estimations were carried out via censuses in many of its habitats, namely Guerrouch, Babor, Djimla, Tamentout, Larbaâ, El-Djarda, Sendouh, Tloudène, Tazegzeout, Coudia and Tababort (Hamitouche & Bougaham 2021). However, studies on the aspect of the population dynamics of the bird remain anecdotal and less precise. The only work was done in the Mount Babor forest and there is variation in the number of individuals from year to year (Vielliard 1978). Fluctuations in the number of individuals (between 12–70 pairs) from year to year may be related to field sampling effort (Vielliard 1976a, Ledant 1977, Ledant & Jacob 1977, Gatter & Mattes 1979). Apart from the variable sampling efforts, the availability of coniferseeds of the Atlas cedar *Cedrus atlantica* and the Numidian fir *Abies numidica* may also cause differences in nuthatch density.

The exhaustive and general census that took place on the whole Mount Babor forest (Ledant 1985) in 1982 is the reference work for our present study. The demographic changes of this endemic bird must be followed very closely (Vielliard 1978) because of its reduced numbers (Ledant 1977).

In this paper, our main objective is to describe the population dynamics of Algerian Nuthatch in the Mount Babor forest on the basis of: 1) the increased population density after

38 years; 2) the rate of multiplication and 3) the annual growth rate. Finally, the distribution of the species in the Babor forest is defined according to habitat types and altitudinal gradients.

Material and Methods

Study area

The Babor-Tababort National Park is part of the territory of Kabylia of Babors. It spreads over three Wilayas (Districts), namely Bejaia, Jijel and Sétif. It covers an area of 23,656 hectares. It is delimited by the following geographical coordinates: (36.600948°; 36.463018°) * N (5.327397°; 5.583446°) E (J.O.R.A. 2019) *(Figure 1)*.The Mount Babor forest is defined as a core area of the Babor-Tababort National Park (J.O.R.A. 2019). It is located at 45 km east of Bejaia and forms the highest point of Petite Kabylie (2,004 m). The extent of the forest is 1,268 hectares (Ledant 1985). Mount Babor is a 7 km long and 4 km wide extrusion (Duplan 1952). It has got more than 500 plant taxa; including 45 tree species and 24 endemic species, as well as 22 species have a protected status by the Algerian law (J.O.R.A. 2012). Recently, two new orchid species have been reported including *Epipactis microphylla* (Bougaham & Rebbas 2020), as a new orchid for the Algeria flora and *Cephalanthera rubra* (Bougaham *et al.* 2020) as a new site for the species in the Mount Babor forest. The average annual precipitation ranges from 512 mm (1,000 m altitude) to 920 mm (2,004 m altitude) on the

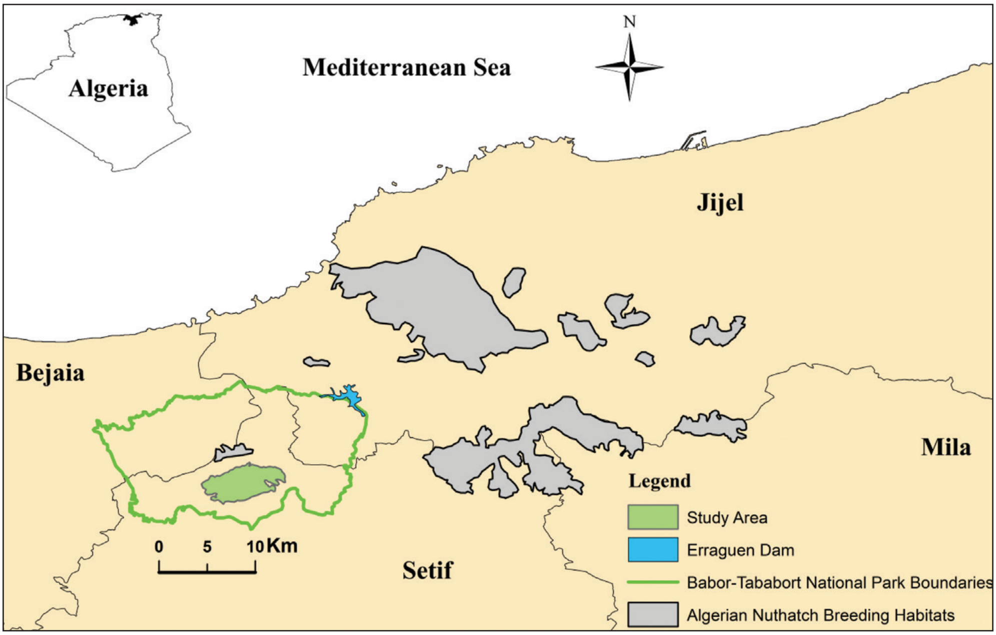

Figure 1. The study area in the Babor-Tababort National Park
1. ábra A vizsgálati terület földrajzi elhelyezkedése a Babor-Tababort Nemzeti Parkban

northern slopes and from 980 mm (1,000 m altitude) to 1,397 mm (2,004 m altitude) on the southern slopes. The average annual temperatures of the coldest month are between 0.0 °C (1000 m altitude) and -5.4 °C (2,004 m altitude) on the north slope and between 3.8 °C (1,000 m altitude) and -3.2 °C (2,004 m altitude) on the south slope (Gharzouli 2007).

The census

To estimate the size of the Algerian Nuthatch population, we used a point count census method suggested by Frochot (2010).The census in the Mount Babor forest was carried out from the beginning of June (04/06/2020) until the beginning of July (11/07/2020). We took the liberty of spreading out our census over time because of the late breeding period of the Algerian Nuthatch in this forest and especially in high altitudes (Vielliard 1976b, Vielliard 1978). The census was carried out during the breeding season, during the mornings (bird activity is intense) and during favorable weather conditions (absence of rain, fog and wind). The census was conducted on the basis of point counts (Blondel 1975, Ledant *et al*. 1985, Bougaham *et al*. 2018, Hamitouche *et al*. 2020). The point counts were practiced along a line transect in aim to cover the entire forest. At each point count, the song of the species is played via a tape recorder in order to stimulate and observe the species. The point counts on the line transects are spaced about 200 to 300 meters apart, except in the case of crossing the ridge line or the no possibility of access. This distance between two point counts was in order to avoid counting the same individual twice, while limiting the risk of not detecting some individuals. At each point count, after the species' song has been played, a 20 minutes period was observed on site. During the 20 minutes, the presence or absence of the species is first noted by auditory and/or visual identification. In case of presence, we noted if it was a pair or a solitary bird. Finally, in each point count, we noted the GPS coordinates, the altitude and the type of dominant forest stand on a radius of 30m.

Mapping and statistics

We used Esri 2014, ArcGis (spatial analysis software) with the ArcMap application v10.2.2 for processing geographic information. The spatial geo-referencing data (GPS coordinates of presence/absence points and altitude) were taken in the field with a GPS Carmin map 76 CSx' model. The distribution map (presence and absence) of the species in the Mount Babor forest was made on the basis of Arcmapv10.2.2 and the spatial geo-referencing data taken in the field. The chi-squared statistical test was carried out under the software R. This test was performed in aim to show relationship of the Algerian Nuthatch numbers in each habitat type, between the two census periods (1982 and 2020).

Distribution according habitat type

The dominant trees were noted in a radius of 30 meters around of each point count in aim to identify the habitat type.

Distribution according to altitudinal classes

Altitudinal classes were calculated via Yule's rule: the number of classes=$2.5\sqrt[4]{n}$, where n corresponds to the total number of point counts performed (n = 240), and the class interval is calculated by the ratio of the difference between the highest and lowest altitude divided over the already calculated classes number (Highest altitude – Lowest altitude/number of classes).

Population dynamics of the Algerian Nuthatch

The numbers obtained after our count of the Algerian Nuthatch in the Mount Babor forest during the breeding season of the year 2020 were compared to those of Ledant (1985) carried out in the same forest during the breeding season of the year 1982. A comparison of numbers according to the habitat type was also made. The typology of our habitats was respected, as far as possible, with respect to the one defined by Ledant (1985). The population dynamics of the species was checked, after a 38-year period, via densities, annual multiplication rate and annual growth rate.

The density and the individuals mean number

The density is the total number of pairs counted divided by the forest total area surface, multiplied by 10 (Hamitouche *et al*. 2020). The individuals mean number per point count is the ratio between the number of individuals counted in each habitat and the number of point counts performed.

Average annual multiplication rate (λ)

The population numbers variation, in a considered time lapse, is given according to the formula of Migot and Linard (1984):

$$\boldsymbol{\lambda = \sqrt[n]{Ef/Ei}}$$

n = number of years between two censuses; Ef = final number; Ei = initial number;
$\lambda = 1$: Population size is stable;
$\lambda > 1$: Population size is increasing;
$\lambda < 1$: Population size is declining.

Annual growth rate (r)

According to Gibbs (2006), the annual growth rate (%) is calculated according to the following function:

$$\boldsymbol{r = \frac{Ln(Nt) - Ln(N0)}{t}}$$

t = number of years between two censuses; Nt = final number; N0 = initial number; Ln = natural logarithm;

r ≃ 0: Population size is stable;
r > 0: Size of the population is increasing;
r< 0: Population size is declining.

Results

Number of pairs and individuals counted

Out of 240 point counts carried out on the whole Mount Babor forest, we counted 133 pairs, nine solitaries and 105 absences. A total of 275 individuals were counted. The number of this population in the Mount Babor forest is the highest compared to that counted in other known forests.

Distribution of the Algerian Nuthatch

The species distribution within the forest follows the same pattern of distribution as that of the different habitat types distinguished. The species distribution map in the Mount Babor forest *(Figure 2)* shows the concentration of numbers was in the forest central parts which are the least degraded. The very degraded and open patches (clearings and edges) were totally avoided by Algerian Nuthatches.

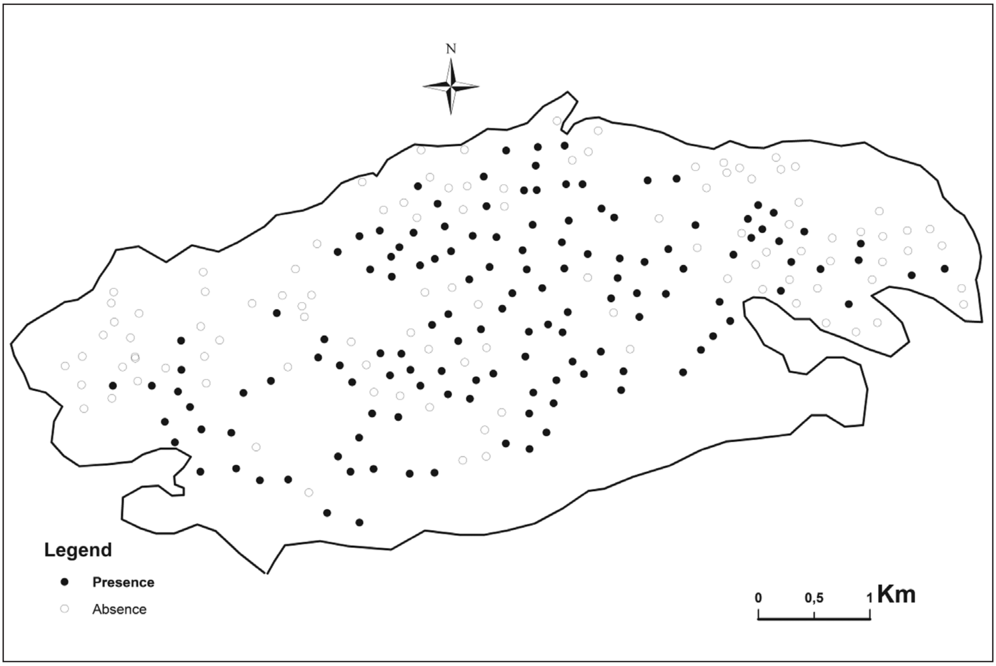

Figure 2. The Algerian Nuthatch distribution map (presence and absence) in the Mount Babor forest
2. ábra Az Atlasz csuszka elterjedési térképe (jelenlét és hiány) a Babor-hegyi erdőben

Table 1. Distribution of the Algerian Nuthatch by habitat type. N: number of point counts
1. táblázat A csuszkák számának megoszlása élőhelytípusok szerint. N: felvételezési pontok száma

Habitat	N	Pairs	Solitaries	Absences	Total individuals number	Mean individuals number
Mixed cedar forest	119	85	4	27	174	1.46
Cedar forest	70	30	5	35	65	0.93
Deciduous oak forest	29	10	0	19	20	0.69
Fir-Deciduous oak forest	10	6	0	4	12	1.20
Summit forest	2	2	0	0	4	2.00
Evergreen oak forest	10	0	0	10	0	0.00
Totals	240	133	9	92	275	1.15

Distribution according to habitat type

The total point counts number was lower than the sum of the points with pairs, solitaries (unpaired birds)and without birds in the mixed Cedar forest, because there were point counts where more than two birds were recorded *(Table 1),* especially in the mixed cedar forest.

The most preferred habitat by the Algerian Nuthatch for reproduction is the mixed cedar forest (85 pairs and four solitaries), followed by the pure cedar forest (30 pairs and five solitaries), the deciduous oak forest with 10 pairs, followed by the fir oak forest with 6 pairs. The summit forest recorded only 2 pairs based on two counts only. In terms of mean individual number, the summit forest is apparently the most abundant (although the number of points is not significant), followed by the mixed cedar and fir-oak forest *(Table 1)*. Finally, on the northern slope (Ubac), the evergreen oak *Quercus ilex* forest, located at an altitude below 1,200 m, was infiltrated in 2020 by some young individuals of zeen oak *Quercus canariensis,* and we even recorded at this altitude small bunches of young and pure zeen oak forest. The Algerian Nuthatch was still absent in this habitat and at this altitude.

Distribution according to altitudinal classes

The number of altitudinal classes was 10, with a class interval of 92.4 m *(Figure 3)*. The first two altitude classes included between 1,079–1,263.8 m were empty of any Algerian Nuthatch presence. The first Algerian Nuthatch pair was noted at 1,326 m altitude, and the first solitary male was noted at 1,357 m altitude. The two altitude classes ([1,818.2–1,910.6] and [1,541–1,633.4]) recorded more occurrences with 52 and 54 individuals, respectively. The other altitudinal classes had relatively high abundances ranging from 12 to 43 individuals. The Algerian Nuthatch has been present throughout the Mount Babor forest from 1,300 m altitude to the summit at 2,004 meters altitude. It should be noted that our census stopped at 1,700 m altitude on the southern slope because of habitat degradation on this exposure below that altitude.

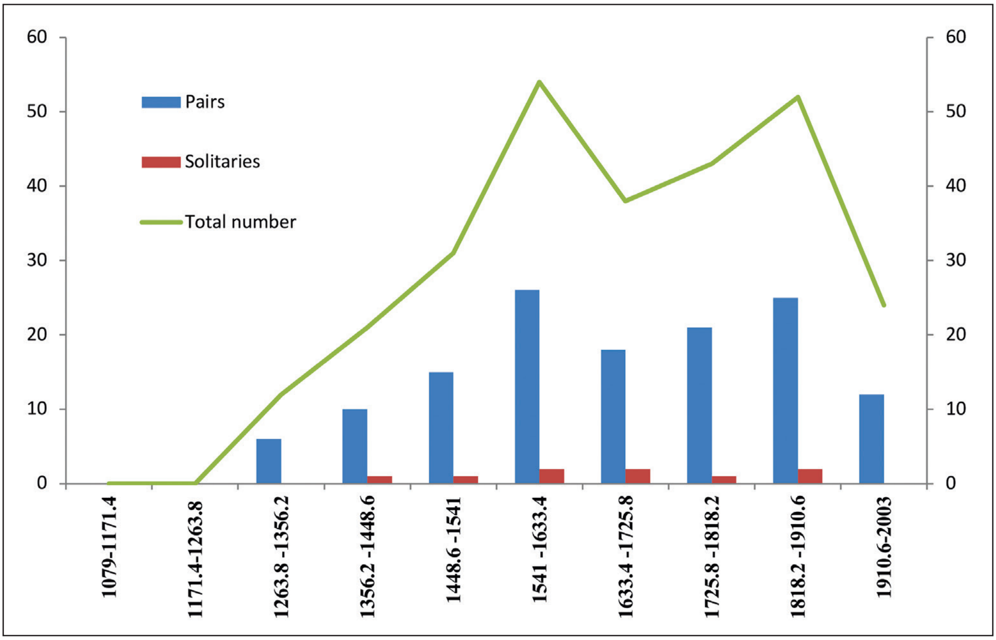

Figure 3. The Algerian Nuthatch distribution according to altitudinal classes in the Mount Babor forest
3. ábra Az Atlasz csuszka eloszlása a Babor-hegyi erdőben a magassági osztályok szerint

Numbers and population growth of the Algerian Nuthatch

The Algerian Nuthatch population total density within this habitat for the year 2020 was estimated to be 1.05 pairs/10 ha and 2.17 individuals/10 ha. The individuals mean number varies according to the type of habitat, with the highest one recorded in the summit forest (2 individuals/point count) and no nuthatches were observed in the evergreen oak forest. The mixed cedar forest and the fir oak forest also contained many individuals, while the Cedar forest and the deciduous oak forest recorded less than one individual per point count *(Table 1).*

The Algerian Nuthatch numbers in the Mount Babor forest increased from 164 individuals in 1982 to 275 individuals in 2020. The annual growth rate and the average multiplication rate calculated gave positive values and greater than 1, which are in the order of 1.36% and 1.01, respectively.

Discussion

The species flourishing in the Mount Babor forest may be due to the absence of species-specific predators of adults (Vielliard 1976b, Bougaham *et al*. 2017). The population size of the Algerian Nuthatch in this area has almost doubled in a 38 year period (from 164 individuals in 1982 to 275 individuals in 2020). In the Guerrouch forest, 91 individuals were counted by Bellatrèche and Chalabi (1990), 60 individuals were counted in the Djimla forest by Bougaham *et al*. (2018), and 187 individuals were estimated in the Tamentout forest by Hamitouche *et al*. (2020). In the other habitats recently discovered, the population size was low (Hamitouche & Bougaham 2021). The Algerian Nuthatch distribution in Mount Babor forest started from 1326 m altitude up to 2004m. In the lower altitudes, the Algerian Nuthatch is still absent below 1200 m (Ledant *et al*. 1985), despite at this altitude the existing evergreen oak forest is dotted with zeen oak forest. This absence may be due to its preference for forests with older trees (Brichetti & Di Capi 1987). The same is true for the Corsican Nuthatch (Tibault *et al*. 2006), which chooses trees with diameters greater than 80 cm due to the greater availability of pine seeds from older trees (Villard *et al*. 2014). In the higher altitudes, we encountered many overlapping nesting territories (the distance between nests was less than 100 m). This observation was made in the mixed cedar forest on the northern slope between the altitudes of 1800 m and 1900 m. This may be due to the abundant food availability offered by a diverse habitat with several plant species (Bougaham *et al*. 2018, Hamitouche *et al*. 2020) and a sedentary lifestyle more marked in the Algerian Nuthatch (Bellatrèche 1994). We used the same habitat typology defined by Ledant (1985) in order to have a more realistic comparison between the population sizes from the two censuses carried out in 1982 and in 2020. The habitat types distinguished by Ledant (1985) are those illustrated in the text and not those in the tables. In the Mount Babor forest, the Algerian Nuthatch has been absent in the evergreen oak forest since 1982. The absence of the species in the Evergreen oak stand in 1982 and 2020 may be due to the fact that this type of stand is the result of regeneration after fires (Ledant 1985). These degraded types of habitat are avoided by the Algerian Nuthatch (Bougaham *et al*. 2018, Hamirouche *et al*. 2020). A Chi-squared test was used to check the density differences between habitat types. Number of nuthatch individuals significantly differed between habitats (Chi-squared test = 251.23, df = 4, P < 2.2e-16). The most frequented habitat in 1982 was the deciduous oak forest, whereas in 2020 it was the mixed cedar forest. The 2020 census showed a higher number than in 1982. Indeed, the annual population growth of the species is 1.36% with a multiplication rate of 1.01. The calculated growth indices indicate a low population demography of the species, as already predicted by Ledant and Jacobs (1977). In addition to nestling predation (Vielliard 1976b, Bougaham *et al.* 2017), the low demography could be due to the low fecundity of the Algerian Nuthatch (Ledant & Jacobs 1977). The number of fledged nestlings is lower than the number of eggs laid per nest (Bougaham *et al.* 2017). The mortality rate is high for this species during the winters because of a single and fluctuating food resource (Vielliard 1978). Its diet consists of coniferous seeds (Vielliard 1978) and the acorns of deciduous trees (Ledant & Jacobs 1977). In the Stenbrohult Forest in southern Sweden, the mortality rate of adult Nuthatches was studied by Nilsson (1982) between 1974 and 1978 via monitoring of adults (at least one-year-old). The study showed that only 47% of ringed adults survived to the next nesting period, when the maximum mortality rate was recorded during winter. For our study area, the average temperature of the three winter months (December, January, February) recorded from 1946 to 2019 is around 10.09 °C (Harris *et al*. 2020). However, the same temperature 37 years before the 1982 census (1946–1982) is 10.01 °C and 37 years after the 1982 census (1983–2019) is around 10.18 °C. This increase in temperature leads to mild winters which, according to Nilsson (1987), will lead to a high survival rate probably due to lower thermoregulation costs and an increase in food availability. The birds` total density calculated in 2020 was 2.17 individuals (10 ha) and 1.05 pairs (10 ha). These figures indicate a slight increase compared to the density found by Ledant *et al*. (1985), which was in the order of 0.65 pairs (10 ha) and 1.29 individuals (10 ha) in 1982. It seems that the densities obtained varied more in direct proportion to the population size of the species than to the change in the area of the Mount Babor forest. On the other hand, the Tamentout forest hosts the Algerian Nuthatch with a density of 0.19 individuals/10 ha (Hamitouche *et al*. 2020), and in the Djimla forest the species is distributed with a density of 0.6 individuals/10 ha (Bougaham *et al*. 2018). The low densities in these habitats may be due to the non-wooded (degraded) habitats in these forests, which the Algerian Nuthatch does not use as breeding sites (Bellatrèche & Chalabi 1990, Bougaham *et al*. 2018).

Threats and conservation

The Mount Babor forest is still among the more preserved forests in Algeria (Ledant 1985). The insecurity that Algeria had observed since the 90s until the end of the 2010s has reduced and limited all anthropogenic origin nuisances (wood cutting, overgrazing, fires and clearing). The non-adventurousness of humans in the Mount Babor forest has contributed to forest safeguarding (living trees) and especially to the standing dead wood safeguarding used by the Algerian Nuthatch as nesting and foraging sites (Bougaham *et al*. 2017, Bougaham *et al*. 2018, Hamitouche *et al*. 2020, Zemouri *et al*. 2021). In spite of the Mount Babor forest having a protected status in an official way, after the foundation of the National Park Babor-Tababort (J.O.R.A. 2019) until now, there is no visible physical and administrative progression on the field. Uncontrolled grazing still remains a threat to the Mount Babor forest, especially after the return of quietness. We have noticed a high number in cattle individuals, which graze the young trees and prevent the regeneration of this forest. The Algerian Nuthatch population in the Mount Babor forest is considered an isolated population (no exchange of individuals with other nearby populations). Due to its isolated location, it can neither receive other individuals from other nearby populations (immigrants) nor allow its individuals to nest elsewhere (migrants) because of the absence of an ecological corridor. It would be interesting to create an ecological passage (corridor) between this forest and the nearest forest of Tababort over a distance of 4 km, to allow the species` individuals to fly between these isolated habitat patches (Bougaham *et al*. 2021).

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Threat status assessment and conservation recommendations for Ibisbill in Kashmir Himalaya

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Abstract The concern that population decline in wild species may lead to disruptions in the ecosystems has triggered numerous ecological studies across the globe. Therefore, monitoring biodiversity plays a key role in identifying priority species for evaluating the effectiveness of conservation measures. Ibisbill *(Ibidorhyncha struthersii)* is a habitat-specialist wader inhabiting high-altitude river rapids with cobbles, boulders and moderate flow of water in Asia. This study aimed to empirically assess the conservation and threat status of Ibisbill in the Kashmir Himalayan region (NW India). The species qualified as Endangered in the study region according to criterion B2 (area of occupancy < 500 km^2) of the IUCN Red List categories and criteria. Moreover, six priority sites for the conservation of Ibisbill were identified during extended field surveys in River Sindh, Kashmir, NW India. Threats faced by the Ibisbill along the priority sites were also recorded. Out of six sites, mining, livestock grazing and vehicle movement was observed at three sites, human interference including tourism activity at five sites and predation was observed at four sites. The results of this study suggest several conservation recommendations which need to be implemented to ensure the long-term persistence of the species.

Keywords: priority sites, disturbance, conservation, threat status, Himalaya, IUCN

Összefoglalás A felismerés, hogy a vadon élő fajok állományainak csökkenése az ökoszisztémák működését negatívan befolyásolhatja, számos ökológiai vizsgálatot eredményez világszerte. A biológiai sokféleség monitorozása ezért kulcsszerepet játszik a természetvédelmi beavatkozások hatékonyságának értékelésére alkalmas prioritás-fajok azonosításában. A kardcsőrű töcs *(Ibidorhyncha struthersii)* kavicsokkal és kisméretű kövekkel szegélyezett és lassan áramló vizű magashegységi folyók élőhely-specialista partimadár-faja Ázsiában. A jelen vizsgálat célja a kardcsőrű töcs természetvédelmi és fenyegetettségi helyzetének adatokon alapuló értékelése volt a Himalája Kasmír régiójában (ÉNY India). A faj az IUCN Vörös Lista kategóriái és kritériumai szerint a B2 kritérium (az előfordulások területe < 500 km^2) alapján a vizsgált régióban Veszélyeztetett helyzetűnek minősíthető. A vizsgálatban az északnyugat-indiai Kasmír régióban található Sindh folyó mentén végzett terepi felmérések során hat fő előfordulási helyet azonosítottunk, ahol felmértük a fajt fenyegető veszélyeztető tényezőket is. A hat helyszín közül a bányászat, legeltetés és gépjármű-közlekedés három helyen, emberi beavatkozás, beleértve a turizmust, öt helyen volt jelen, míg predációt négy helyen figyeltünk meg. A vizsgálat eredményei alapján több természetvédelmi javaslat fogalmazható meg, amelyek megvalósítása szükséges a faj hosszú távú fennmaradásának biztosításához.

Kulcsszavak: prioritás helyszínek, zavarás, természetvédelem, fenyegetettségi státusz, Himalája, Természetvédelmi Világszövetség

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Introduction

Biodiversity, in light of the growing pressure for numerous human activities, is facing a decline and increased risk of extinction globally (Butchart *et al.* 2010, Ceballos *et al.* 2017). Recently, 13% of bird species have been listed as globally threatened with extinction, whilst a further 9% are listed as 'Near Threatened' (BirdLife International 2015). Extinction rates estimated by the International Union for the Conservation of Nature (IUCN) are now 50 to 500 times higher than previous rates calculated from the fossil record (Baillie *et al.* 2004). The concern that population decline in wild species may lead to an imbalance in various ecosystems has triggered numerous ecological studies across the globe. Ecological studies are vital for providing information, prerequisite for initiating conservation action that ultimately help to alleviate the impact of developmental activities on the environment. Therefore, monitoring biodiversity plays a key role in identifying the priority conservation species for evaluating the effectiveness of restoration measures (Kéry & Schmid 2004).

All over the world, birds are known to utilise a plethora of habitats with a bewildering array of foraging strategies. Predominantly, a wide range among them makes use of the aquatic ecosystem as their specific habitat to carry out their life processes (Lovette & Fitzpatrick 2016). Being widely distributed, birds form a major component of biological diversity and are widely recognised as indicators for monitoring ecosystem quality and health of the environment (Gill 2007). Several birds are sensitive and do not tolerate even mild disturbances in the ecosystems and thus a slight change in their diversity and abundance may reflect ecological imbalance. Moreover, among a number of taxa, birds of river and riparian habitats are particularly important for monitoring, as they are relatively easily surveyable and identifiable, with little need of specific equipment or comprehensive knowledge.

The order Charadriiformes, also known as shorebirds are evolutionary important and phylogenetically enigmatic among the avian orders. Forming a morphologically diverse group, they are globally distributed, and largely migratory, representing a major part of aquatic bird diversity (Dehorter & Guillemain 2007). Charadriiformes belongs to a monophyletic group comprising of three monophyletic suborders; Lari, Scolopaci and Charadrii. These three suborders of the non-passerine birds are estimated to have been diverged from one another in the Cretaceous period (Paton *et al.* 2003).

Ibisbill *Ibidorhyncha struthersii* (Vigors, 1832), belonging to the order Charadriiformes, family Ibidorhynchidae, is considered a unique taxa for its perplexing apomorphies and plesiomorphies, resulting in challenges to the correct assessment of the systematic position of the species (Baker *et al.* 2007). Being morphologically unmistakable, the adult bird is characterised by a black face, crimson red bill and a black breast band (Pierce 1986, Knystautas 1996). Moreover, it shows little intraspecific variation with no subspecies described, is closely related to the oystercatchers, stilts, and avocets, and its name refers to a phenotypic resemblance to ibises.

The effective conservation measures that need to be taken for a species requires thorough understanding about its threat status. To this end, the current study was undertaken to evaluate the conservation status of Ibisbill by addressing the following questions; i) Which IUCN threat category does Ibisbill belong to? ii) What are the major threats that the species is facing in the study area?

Methods

Study area

The present study was carried out in the Kashmir division (NW Himalayas) of Union Territory of Jammu and Kashmir, India from August 2017 to July 2019. Surveys were also conducted to gather data on species presence in the Ladakh region as well. Sindh valley in central wildlife division (District Ganderbal) was selected as the intensive study site for identification of threats. River Sindh is located at 34.23 °N and 74.78 °E, with an average elevation of 1619 m (Dada *et al.* 2013). The river originates from Panjtarni snow fields (4250 m asl) and receives ample flow from the Amarnath (5003 m asl), Kolahoi (5425 m asl) and Thajwas Glacier (3000 m asl). Sindh is a fast-flowing torrential river in its upper and middle reaches while in the lower reaches it becomes calm and flows slowly (Siraj 2018, Haq *et al.* 2021a). The river comprises of small islands all along its course dominated by presence of huge rocks in the upper stretches. The beds of River Sindh comprise mainly of cobbles, boulders, with some underlying pebbles and sand at few places and thus provide an ideal habitat for riverine birds, especially for Ibisbill.

Criteria used for threat assessment

IUCN has defined five criteria (A–E) for the assessment of the threat status of any taxon in the world, however, only one of the five criteria needs to be fulfilled for assigning a threatened category. The five criteria (A–E) of the IUCN extinction risk assessment are based on the following three parameters: population reduction (criteria A and E), geographic range (criterion

Table 1. Summary of criterion B of the IUCN Red List Categories and Criteria (IUCN, 2019) for evaluating whether a taxon belongs to any of the threat category (Critically Endangered, Endangered and Vulnerable)

1. táblázat A Természetvédelmi Világszövetség (IUCN) Vörös Lista Kategóriák és Kritériumok (IUCN 2019) B kritériumának összegzése annak eldöntésére, hogy a vizsgált taxon megfelel-e bármelyik fenyegetettségi kategóriának (Súlyosan Veszélyeztetett, Veszélyeztetett, Sebezhető)

<table>
<tr><th colspan="5">Geographic range in the form of either B1 (extent of occurrence) and/ or B2 (area of occupancy)</th></tr>
<tr><th></th><th>Critically Endangered (CR)</th><th>Endangered (EN)</th><th colspan="2">Vulnerable (VU)</th></tr>
<tr><td>B1 Extent of Occurrence</td><td>< 100 km²</td><td>< 5000 km²</td><td colspan="2">< 20 000 km²</td></tr>
<tr><td>B2 Area of Occupancy</td><td>< 10 km²</td><td>< 500 km²</td><td colspan="2">< 2000 km²</td></tr>
<tr><td colspan="5">and at least two of the following three (a, b, c) conditions</td></tr>
<tr><td colspan="2">(a) severely fragmented or number of locations</td><td>=1</td><td>≤5</td><td>≤10</td></tr>
<tr><td colspan="5">(b) Continuing decline observed, estimated, inferred, or projected in any of (i) EOO; (ii) AOO area; (iii) extent and or quality of habitat; (iv) number of locations or subpopulations (v) number of mature individuals.</td></tr>
<tr><td colspan="5">(c) Extreme fluctuations in any of (i) EOO; (ii) AOO; (iii) number of locations or subpopulations; (iv) number of mature individuals.</td></tr>
</table>

EOO: Extent of Occurrence, AOO: Area of Occupancy

B) and abundance (criteria C and D) (Le Breton *et al.* 2019). We assessed extinction risk for Ibisbill based on criterion B, i.e. geographic range. According to criterion B, a geographic range should be in the form of either B1: Extent of Occurrence (EOO) or B2: Area of Occupancy (AOO) and a taxon should also fulfil at least two of three conditions (a, b and c) *(Table 1)*.

Calculation of geographic range parameters

The R statistical software package ConR (Dauby *et al.* 2017) was used for calculating various parameters of criterion B (EOO, AOO) in order to assess the IUCN assessment of threat category for Ibisbill. EOO is a polygon that is drawn to include all the known locations and/or occurrences of a species and AOO is the total occupied area index, computed by summarising areas of standard size (2 × 2 km) grid cells (Bland *et al.* 2017, IUCN 2019).

Identification of threats

Based on previous studies (Haq *et al.* 2022), the threats faced by the Ibisbill were recorded at six sites along River Sindh *(Figure 1)*. We refer to threats here as "proximate human activities or processes that have impacted or may impact species" (Salafsky *et al.* 2008). The major threats that represented disturbance to the bird included (i) mining (including manual extraction of

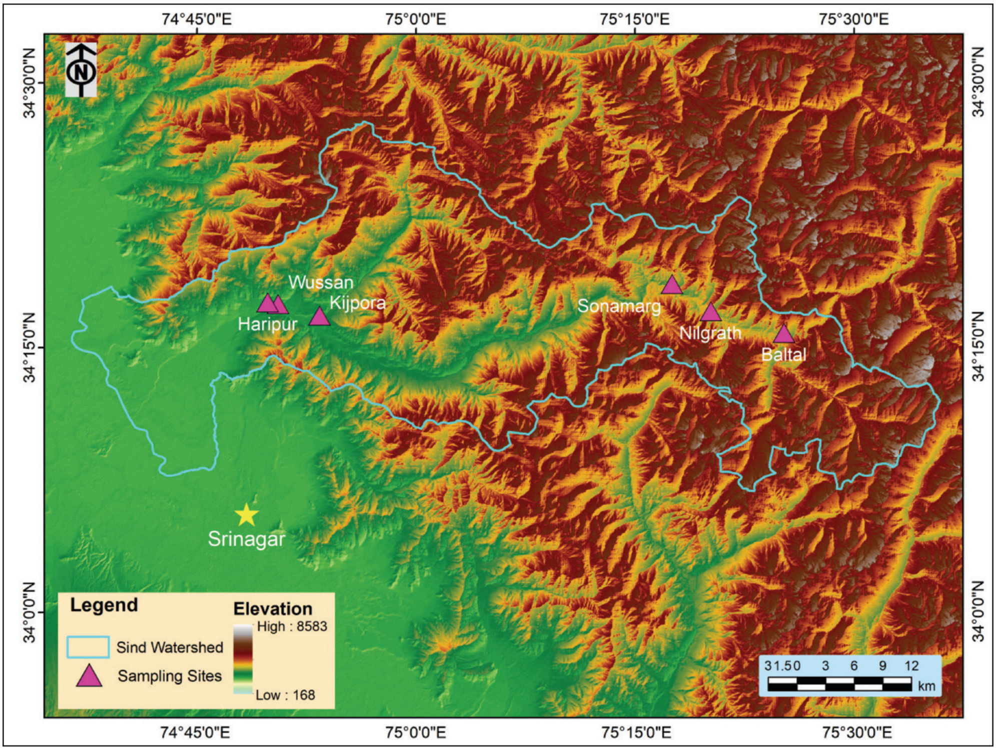

Figure 1. Map of the study area
1. ábra A vizsgálati terület térképe

sand, boulders and cobbles), (ii) human presence (movement of local inhabitants and tourists), (iii) grazing livestock (cattle and sheep), (iv) predation (presence of predators) and (v) vehicle movements near the stretches inhabited by Ibisbill (Haq *et al.* 2022). Conservation actions that need to be taken were compiled for implementation in order to propose areas for proper protection and to reduce the ongoing anthropogenic activities.

Results

Assessment of threat status

Calculations in the ConR package showed that the number of unique occurrences, sub-populations and locations for the Ibisbill turned out to be 27, 14 and 16, respectively. The calculated EOO and AOO were 35,927 km^2 and 100 km^2, respectively *(Figure 2)*. Based on

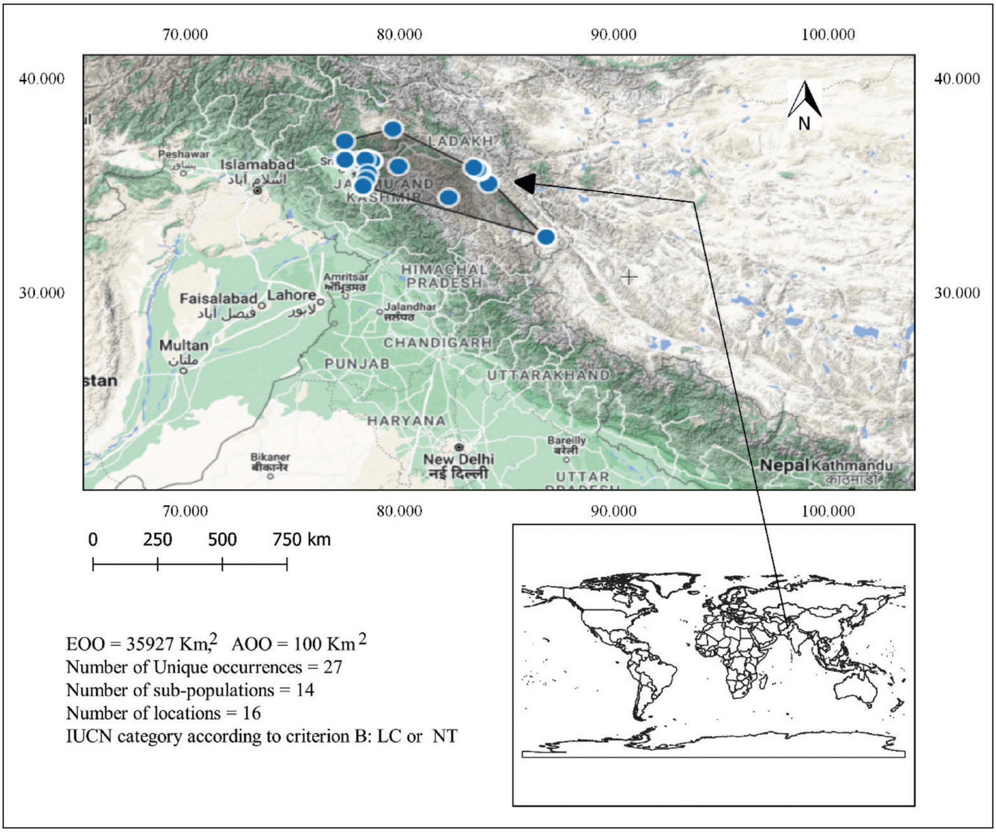

Figure 2. Occurrence sites of the Ibisbill in Kashmir and Ladakh Himalayan region are shown in blue dots. The convex hull used for calculating the EOO is shown as grey polygon

2. ábra A kardcsőrű töcs elfordulási helyei fekete pontokkal jelölve a Himalája Kasmír és Ladakh régiójában. A földrajzi elterjedés kiterjedésének (EOO) számítására használt konvex burok szürke poligonnal van jelölve

Table 2. EOO, AOO, occurrence sites, number of populations and locations, and threat status of Ibisbill in the Kashmir and Ladakh region of Indian Himalayas
2. táblázat A földrajzi elterjedés kiterjedése (EOO), az előfordulások összterülete (AOO), az egyedi előfordulások száma, a populációk és lokalitások száma, valamint a fenyegetettségi helyzet a kardcsőrű töcs esetén az indiai Himalája Kasmír és Ladakh régióiban

Taxa	EOO	AOO	No. of unique occurrences	No. of sub-populations	No. of locations	Category Criteria B
Ibisbill	35927 km^2	100 km^2	27	14	16	Endangered (by B2)

EOO: Extent of Occurrence, AOO: Area of Occupancy

the latter result, the Ibisbill qualifies as Endangered by criterion B2 (AOO < 500 km^2) in the study region *(Table 2)*. When the species is assessed globally (EOO = 7,440,000 km^2), the AOO value is likely to be much higher than the threshold of Endangered (500 km^2) or Vulnerable (2000 km^2), thus, the global status of the species is Least Concern (https://www.iucnredlist.org/species/22693672/93417952).

Threats identified

We recorded Ibisbill at six major sites along the 106-km-long River Sindh. These sites were discontinuous and more or less similar with respect to the presence of cobbles, boulders and having moderate flow of water. Ibisbill was not detected in the stretches with torrential flow and large stones. Human presence occurred most frequently (in five of six sites), followed by predation (four sites) and by mining and livestock grazing (three sites each) *(Table 3)*.

Table 3. Operative threats to Ibisbill in Sindh river of Kashmir Himalayan region
3. táblázat A kardcsőrű töcsöt fenyegető hatások a Sindh folyó mentén a himalájai Kasmír régióban

Taxa	Occurrence sites	Latitude	Longitude	Altitude (m) asl	Threats recorded
Ibisbill	Haripur	34.28729 N	75.82823 E	1695	Mining, Human presence, Livestock grazing
	Wussan	34.28127 N	74.85358 E	1723	Human presence, predation
	Kijpora	34.26884 N	74.88557 E	1757	Mining, Human presence, Livestock grazing, Predation, Vehicular movement
	Sonamarg	34.30405 N	75.29172 E	2678	Human presence, Vehicular movement
	Nilgrath	34.28923 N	75.3238 E	2718	Mining, Predation
	Baltal	34.25847 N	75.41225 E	2859	Human presence, Livestock grazing, Predation, Vehicular movement.

Discussion

Assessment of the threat status and identification of operative threats to a target species using standardised procedures plays a key role in formulating the appropriate conservation strategies and strengthens the planning of effective management (Collins-Kreiner *et al.*

2013). Factors to be considered when assessing impact of disturbance on bird species include frequency of the disturbance, whether rare or especially sensitive species are affected, and whether alternative habitats are available nearby (Hill *et al.* 1997).

The effect of anthropogenic activities on waders has been widely studied (Ruhlen *et al.* 2003, Lotze *et al.* 2011). However, no detailed or quantitative studies have been conducted on the ecology or the potential impacts of anthropogenic disturbances on Ibisbill from Indian Himalayan region. Disturbance has been observed to have direct impact on the behaviour of waterbirds including immediate response such as flushing, increased vigilance behaviour, calling and alteration in the ongoing acts like feeding, resting (Borgmann 2010). Besides other factors, tourism development may also influence the vulnerability of waders by predation (Clark & Nudds 1991). Ibisbill faces various anthropogenic threats emanating largely from mining, livestock grazing, human presence (fishing and tourism activities) and from other natural predators (Haq *et al.* 2022). While studying the response to a threat factor, the Ibisbill was found to take escape flights of more than 50 m during 35% (n=327) of the instances in Sindh river (Haq *et al.* 2022). An increase in the frequency of these defensive behaviours and in the associated physiological regulatory responses could potentially decrease nest attendance levels during the breeding season and result in an increased probability of breeding failure (Price 2008). The disturbances to Ibisbill in Sindh river were found to have a detrimental effect on the population size, as they were responsible for breeding failure and low recruitment rate of Ibisbill in the region (Haq *et al.* 2021b).

Extinction of a species is also forecasted by its small geographic range size (Harris & Pimm 2008), which is also relevant in the case of the highly specialized Ibisbill. Further, genetic, demographic and ecological events may strongly affect such species. Despite having a large geographic range in terms of EOO, the Ibisbill is known to inhabit specific habitats comprising of very small AOO. Thus, the current study contributes significantly to indices of global trends in threat status and identification of the areas for prioritisation of species for conservation.

Conclusions

The study assesses the threat status of the Ibisbill as Endangered based on IUCN criteria for threat assessment, in Kashmir and Ladakh region of Indian Himalaya. Furthermore, the threats faced by the bird was compiled and remedial measures and recommendations were formulated. Ibisbill, a habitat specialist bird, faces a threat from several ongoing anthropogenic activities in the river. If these continue unregulated, these activities may seriously affect the survival of the birds and the long-term persistence of the species.

Conservation recommendations

We have identified the areas which need immediate focus towards conserving Ibisbill and its habitat. The department of Wildlife protection, Jammu and Kashmir has to be informed and involved in protecting and releasing these sites from the disturbances. Since most of the area inhabited by the Ibisbill is non-protected, there is a dire need to highlight the plight of the bird for safeguarding its potential habitat. Nesting areas of the Ibisbill need to be marked properly and appropriate care should be ensured during the breeding season for successful hatching and recruitment. Mining (sand and boulder extraction) needs to be checked properly at the sites inhabited by the bird. The camping sites should be managed properly at a distance away from the river. In many range countries, the disturbances such as hydropower development projects, pollution by communal waste from populated areas to Ibisbill habitats are increasing. Therefore, regional cooperation for conservation of Ibisbill among range countries need to be initiated which should start with (i) an assessment of the suitable habitats for the species throughout its distribution range, (ii) regular monitoring at target sites to estimate local and global population sizes and to quantitatively assess threat factors (human settlements, hydropower projects, mining sites, recreation/tourism centre) and (iii) culminate in a spatial prioritisation of where to allocate conservation effort (area protection, monitoring, restoration/management efforts) for the long-term survival of the species.

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Genetic polymorphism in the mitochondrial D-loop of Oriental White-backed Vultures *(Gyps bengalensis)*

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Abstract Vultures are among nature's most successful scavengers, providing tractable models for ecological, economic, and cultural studies. Asian vultures have undergone dramatic declines of 90–99% in the subcontinent due to consequences of poisoning drugs, thereby being at a high risk of extinction. In Pakistan, surveys conducted previously focused mostly the cause of decline and breeding strategies only. Genetic profiling of vultures was still unmapped that could play a particular role in conservation endeavors and let researchers to genetically label individuals of threatened or endangered species. In this study, we examined genetic diversity and molecular phylogeny of Oriental White-backed Vultures by analyzing mitochondrial DNA (mtDNA) sequences. Genetic polymorphism was detected among individuals, and, on that basis, phylogenetic analysis was conducted through Bayesian analysis of DNA sequences using MCMC. Using multiple sequence alignment, two mutations, transversion T>G and transition G>A, were observed at nucleotide positions 1 and 2, respectively. Similarly, T/C heterozygosity at two positions, 53 and 110, and one heterozygous T/G locus at 130 position were also observed. The reference sequence, along with other samples of V1, V6, V7 and V9, was placed into a clade, while V2, V5, V11, V3, V4 and V10 samples were grouped into a two clade.

Keywords: vulture, *Gyps bengalensis*, phylogenetics, mitochondrial D-loop, Pakistan

Összefoglalás A keselyűk a természet legsikeresebb dögevői közé tartoznak, jól nyomon követhető modellt szolgáltatva ökológiai, gazdasági és kulturális kutatásokhoz. Az ázsiai keselyűpopulációk drámai, 90–99%-os csökkenésen mentek keresztül a szubkontinensen a gyógyszerek használatából fakadó mérgezések miatt, így magas a kihalásuk veszélye. Pakisztánban a korábban végzett vizsgálatok leginkább az állománycsökkenés okára és a költési stratégiák felderítésére összpontosítottak. A keselyűk genetikai profiljának feltérképezése még mindig nem történt meg, ami nagy szerepet játszhatna a konzervációs törekvésekben, valamint ezáltal a kutatóknak lehetőségük nyílna a veszélyeztetett fajok egyedeinek genetikai jelölésére. Ebben a tanulmányban a bengál keselyű genetikai diverzitását és molekuláris filogenetikáját vizsgáltuk mitokondriális DNS (mtDNS) szekvenciák elemzésével. Az egyedek között genetikai polimorfizmus volt megfigyelhető, így a DNS-szekvenciák alapján filogenetikai analízist végeztünk Bayesi módszerrel MCMC-t használva. Többszörös szekvencia-illesztést alkalmazva két mutációt, a T>G transzverziót és a G>A tranzíciót figyeltük meg az 1. és 2. nukleotid pozíciókban. Ugyanígy két pozíció T/C heterozigóta volt az 53. és a 110. helyen, valamint egy lókusz volt heterozigóta T/G a 130. pozícióban. A referenciaszekvencia a mi V1, V6, V7 és V9 mintáinkkal együtt egy kládba, míg a V2, V5, V11, V3, V4 és V10 mintáink egy másik kládba tartoztak.

Kulcsszavak: keselyű, bengál keselyű, *Gyps bengalensis*, filogenetika, mitokondriális D-hurok, Pakisztán

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Introduction

Vultures (Family Accipitridae) are scavengers with hunting strategies, thereby playing important ecological roles in maintaining ecosystem balance and aesthetic roles in environmental cleaning by feeding on animal's dead bodies, as well as controlling diseases spread by removing animal's carcasses. At a single time, vultures can consume more than 20% of their total body mass, therefore vulnerable to poisoning, particularly at carcasses that are laced with poison. Vultures are blessed with a unique digestive system that holds distinctive acids to dissolve bacterial strains of cholera anthrax, botulism and anthrax (Wink 1995). Mainly two taxonomic groups of vultures are known to date, and they fall into two broad categories: Old World and New World Vultures. They are the only vertebrate foragers but experienced the most rapid decline among birds over the past decades, which ranked them as one of the most threatened avian functional guilds in Asia and Africa (Buechley *et al.* 2016). Locally, eight species are present among these birds, and about 96% of decline in population of *Gyps bengalensis* was recorded in different areas of Pakistan, India, and Nepal during periods of 1991–1993 (Prakash 1999, Gilbert *et al.* 2002, 2004).

The main reason behind the decline of vulture's species is the unintentional poisoning of toxic NSAID (non-steroidal anti-inflammatory drug) Diclofenac by its veterinary use on cattle. Unfortunately, this catastrophic event continued unchecked for about 15 years due to usage of this anti-inflammatory drug (used to reduce the pain, fever and inflammation in livestock's) that emerged as the sole cause of rapid decline of Oriental White-backed Vulture population in Punjab, Pakistan (Oaks *et al.* 2001, Naidoo & Swan 2009). Vultures exposed to diclofenac during feeding on carcasses of livestock dozed with the drug and experienced kidney failure within a few days of their exposure (Swan *et al.* 2006). It has also been established that not only diclofenac but some other non-steroidal anti-inflammatory drugs (e.g. aceclofenac, carprofen, flunixin, ketoprofen, nimesulide and phenylbutazone) are also harmful to vultures and other scavenger birds (Cuthbert *et al.* 2006, Acharya *et al.* 2009). Since 2006, this drug has been proscribed by the administration of Pakistan, India and Nepal (Cuthbert *et al.* 2011). Although meloxicam is a single NSAID that was confirmed as a harmless drug designed for vultures (Harris 2013). Whereas further aspects that contribute to decline in vulture's populations include lack of food locally, habitat loss, severe climatic conditions, as well as poisoning via application of insecticides (Choisy 2013).

Information on the evolutionary history and genetic diversity of these species is considered critical for the success of both ex-situ and in-situ conservation. Mitochondrial DNA (mtDNA) of vultures was used due to some matchless properties, *i.e.* high copy number of mitochondria, inheritance from mother only, absence of recombination and increased mutation rate. Poulakakis *et al.* 2008 studied the phytogeography of Black Vultures *(Aegypius monachus)* in Europe based on microsatellite and mtDNA variation. Few avian conservation studies have used SNPs so far García and Arruga 2006 used SNPs to differentiate species of partridges for reintroduction, and Väli *et al.* 2010 used them to examine hybridization between two species of Spotted Eagles in Europe. Clearly, the future is upon us, given the power of high-throughput sequencing and the associated power utility of SNPs (Susan *et al.* 2011).

The White-tailed Eagle *(Haliaeetus albicilla)* is used as a major flagship and umbrella species for conservation work throughout large parts of Europe. Loss of genetic diversity can reduce both short-term viability and long-term adaptability (Frankham 2005). To assess the genetic impact of population declines during the twentieth century, we therefore studied the genetic variability at mitochondrial DNA (mtDNA) control region sequences and autosomal microsatellite markers in north and central European white-tailed eagle populations (Hailer *et al.* 2006).The Griffon vulture population in Serbia, similarly to many Balkan countries, experienced a rapid demographic decline starting from the mid-twentieth century, mainly due to the mass poisoning of the birds and the implementation of new veterinary measures that prohibited the deposition of dead animals in nature (Marinkovic & Karadzic 2008). Genetic diversity analysis is of great importance in modern-day conservation, and without knowing the genetic status of the population it is hard to implement proper conservation measures and secure long-term survival of the total population in the nowadays fast-changing environment and habitat fragmentation (Maudetr *et al.* 2002).

Population declines of *Gyps* vulture species across south Asia have been well-documented since they were first reported in 1999 (Prakash 1999, Prakash *et al.* 2003, Gilbert *et al.* 2006). As a result of these declines, the Oriental White-backed Vulture *Gyps bengalensis*, Long-billed Vulture *Gyps indicus* and the Slender-billed Vulture *Gyps tenuirostris* are all listed as 'Critically Endangered' (Murn *et al.* 2015).

Although *Gyps* vulture populations were probably declining slowly in many parts of the world during the 20th century, a very different situation existed in India, Nepal and Pakistan. Three species of vultures endemic to South Asia, oriental White-backed Vulture, Long-billed Vulture *(G. indicus)* and Slender-billed Vulture *(G. tenuirostris)*, are listed as being threatened with extinction after rapid population declines in the Indian subcontinent, which began in the 1990s (Prakash *et al.* 2007). The Oriental White-backed Vulture population in India in 2007 was estimated at one-thousandth of its level in the early 1990s (Green *et al.* 2007).

In the Punjab Province of Pakistan, annual rates of decline for breeding populations of Oriental White-backed Vultures have ranged between 11% and 61% per year since 2001. Conservation initiatives to address the vulture declines have included the establishment and development of conservation breeding centres (Murn *et al.* 2008).

Six, out of nine species of vultures found in India have been facing problem of existence and therefore declared as threatened. Of these, three species endemic to South Asia, the Indian White-backed Vulture, Long-billed Vulture *(Gyps indicus)* and Slender-billed Vulture *(Gyps tenuirostris)* are at high risk of global extinction and are listed as critically endangered because of rapid population declines within the last decade in the Indian subcontinent (Thaku *et al.* 2012).

In this study, we analyzed genetic polymorphism of mitochondrial D-loop in *Gyps bengalensis* for phylogenetic relationship of this breeding population. Our objectives were to check molecular diversity in these populations, providing a basis for a more effective conservation effort for the recovery of globally threatened vultures. In Pakistan, WWF conservation deeds are appreciable in this regard, according to the current reports of SAVE Consortium (Saving Asia's Vultures from Extinction) Changa Manga Conservation Centre had seven pairs of this species of which four pairs attempted to breed. Depending on the genetic structure, it could potentially be the appropriate source population for future reintroduction programs.

Materials & Methods

Sample collection

Biological samples were collected from 11 Oriental White-backed Vultures Collection sites from Ex-situ conservation, vulture Captive Breeding Facility under Punjab Vulture Restoration and Conservation project at Changa Manga Forest, Kasur, Pakistan *(Figure 1)* after formal permissions from the Punjab Wildlife & Parks Department, Pakistan.

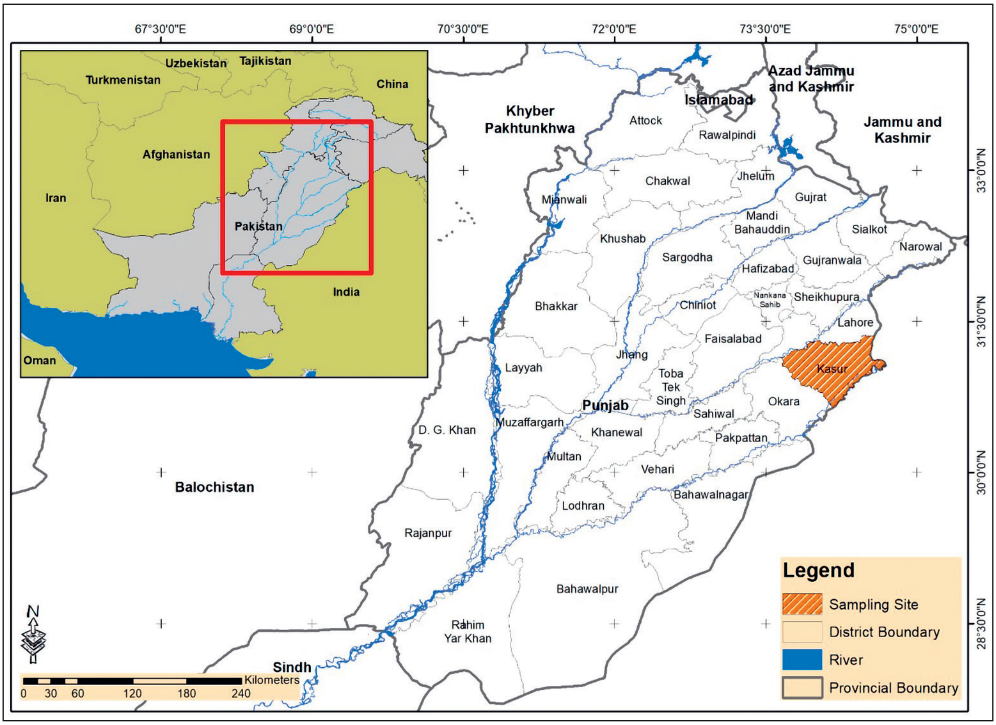

Figure 1. The samples collection sites from Punjab and Kasur
1. ábra A mintavételi helyek Punjab és Kasur tartományokban

DNA extraction

Genomic DNA was extracted from blood, by using protocol of Qiagen (QIAamp DNA Mini Kit, Cat No./ID: 51304). 1ml of blood samples was added in 350 µl Lysis Buffer (1 M Tris, 0.5 M EDTA, 5 M NaCl, 10% SDS), then centrifuged at 13.000 rpm for 5 min, and took pellet and repeated this step for 3 times at least, then added 40–50 µl of proteinase K and 70 µl of 10% SDS, centrifuged and took supernatant and added equal volume of phenol: chloroform: isoamyl alcohol (5:24:1), incubated for 10–15 min, centrifuged at 13,000 rpm for 3–5 min, and took supernatant.

Isopropanol (0.5 ml) was added to the samples, centrifuged again, then pellets were washed with 70% ethanol, and centrifuged again for 10 min. Samples were air dried for 12 hours. DNA samples were dissolved in 200 µl TE buffer or in DEPC-treated water. Then, a nanodrop method was applied at wavelength of 260/280 nm to check the purity and quantity of DNA samples (Sefc *et al.* 2003).

PCR amplification

Primers were designed for PCR amplification by Primer 3 program (Untergasser *et al.* 2012) We used the primers: GbCR4.L (5'-CGA TTC ATG GTA GCA GGT CA-3') and CSB1.H (5'-AAC ATG TCC AAC AAG CAT TCA-3') (Mullis *et al.* 1986). Next, extracted DNA was amplified by a subsequent PCR reaction (Mullis *et al.* 1986) in a total volume of 25 µl. PCR amplification consists of initial denaturation at 96 °C for 5 min, 30–35 cycles consisting of denaturation at 94 °C for 45 sec, annealing at 55 °C for 30 sec, and extension at 72 °C for 30 sec, then final extension at 72 °C for 10 min.

Sequencing and phylogenetic analysis

PCR samples were cleaned and washed with 70% ethanol and sent for sequencing by using a Beckman Coulter kit, and results were analyzed. More software's were used to analyze data including Codon Code Aligner Version 5.1.4 (CodonCode Corporation) for sequences alignment as well as for analysis, and BEAST v1.10.4 (Tamura *et al.* 2013) for phylogenetic analysis. Furthermore sequences were passed through BioEdit v7.0.9.1. (Hall 1999). Haplotype and nucleotide diversity were estimated using DNaSP v4.1 software (Librado & Rosaz 2009). Using the maximum parsimony (MP) method, we chose the tree having the highest likelihood value as our best from the NETWOK 4.5.1.2 (Bandelt 1999) and the species trees were reconstructed using the coalescent-based model implemented in BEAST v1.10.4 (Tamura *et al.* 2013). The two mtDNA loci were partitioned by codon for all analyses. The tree was constructed from 481 bp, using the first high variation section fragment (LVH). The reference sequence (previous sequence taken from GenBank) was used to derive a phylogenetic analysis to identity the haplogroup in the current isolate. Genetic variation data include demographic historical information. mtDNA of mismatch distribution (for comparison) is usually used to investigate demographic events (Slatkin & Hudson 1991, Rogers & Harpending 1992).

It has been developed two standards; not multimodal distribution rules must have a constant population size, and growing populations have a uniform single peak distribution.

Before the system phylogeny analysis, JModeltest 2.1.7 determined the appropriate DNA model analysis. In this analysis, the best model of HKY+1 was found for Neighbour connection parameters using the appropriate Bayesian information criteria. The original tree was analyzed by BioNJ using PhyML 3.1. Using a 100-start analysis to estimate the maximum probability node supports non-parametric pseudo-random. Bayesian analysis was implemented using MrBayes 3.2 (Geyer 1991). Finally, a random starting shirt was used for the analysis, yielding 10.000.000 generations for all species sequence data (100% of bootstrap support). It has been studied by every 100 generations frequently used to check the generation and transition by Markov Chain model. The pairwise difference matrix is performed using NETWOK 4.5.1.2 (Bandelt *et al.* 2001). The non-parametric bootstrap (henceforth referred to simply as the bootstrap) is a computer-based statistical technique that uses data resampling to estimate values of interest. The bootstrap sample is then analysed to infer a phylogenetic tree. The exact relationship of bootstrap values, Pboot, to posterior probability values, $P(\tau \mid D)$ is an open and important question in phylogenetic analysis. While the theory of each measure is largely independent, it has been posited that they should be equivalent. Therefore, a need exists for additional studies focusing on the behaviour and relationship of the bootstrap and posterior probability measures. Unfortunately, an analytical solution is not readily apparent for this study. The experiment used a paired design to compare the bootstrap values obtained using these two procedures for 1.000 simulated samples for a single point in the model space.

This tree was modified by manually adding six taxa (namely*: Circus approximans, Accipiter badius, Accipiter melanoleucus, Accipiter minullus, Elanus scriptus* and *Elanus axillaris*), based on a consensus tree of random phylogenies.

Results & Discussions

Although, the total number of vultures in captivity is slightly less than required to maintain current levels of heterozygosity, the difference in the number of breeding vultures necessary to maintain allelic diversity is much greater. The current captive breeding population constitutes only 27% of this number, and political and logistical barrier separating the captive breeding facilities potentially hamper exchange of individuals and the maintenance of genetic diversity even further.

In spite of these changes in food supply, the Cape Griffon *(Gyps coprotheres)* of southern Africa was the only member of the genus considered to be in danger of global extinction until the late 1990s. This species is believed to have been affected by multiple threats (BirdLife International 2007). It was then recognized that populations of vultures endemic to South Asia were declining rapidly across the Indian subcontinent for unknown reasons (Pain *et al.* 2008). To help mitigate this marked reduction in abundance, supplementary feeding stations (SFS; colloquially termed “vulture restaurants”) have been created worldwide, often without consideration of the scientific evidence supporting the suitability of the practice (Cortes *et al.*

2016). In order to ensure that wild vulture populations recover and that a safe environment exists for captive-bred vultures to be released, the veterinary use of diclofenac and other similarly vulture-toxic NSAIDs needs to be eliminated. This is a huge challenge, but one that must be undertaken to prevent the extinction of these iconic birds (Mukherjee *et al.* 2014).

Far less information is available on the population trends of vultures in Nepal, as until relatively recently political instability in the country has prevented repeat surveys from being undertaken in lowland areas where resident Oriental White-backed and Slender-billed Vultures were formerly most abundant (Chaudhary *et al.* 2012). Globally, 61% of vulture species are threatened with extinction and are declining mainly due to anthropogenic pressures (Ogada *et al.* 2012). It is vital to understand threats to vultures in terms of land use and local human livelihoods. Previous research in Africa focused on the human dimensions of vulture conservation in commercial farming and protected areas (Pfeiffer *et al.* 2015).

This vulture population in Pakistan was experiencing rapid decline from diclofenac poisoning during the monitoring period, and so nest densities could have been lower than a potential maximum. Apart from differences in habitat (arid Nagarparkar, wetland dominated Keoladeo, coastal mangroves Sundarbans and forest plantation Changa Manga), the way spatial extent of the breeding areas in each study area may provide another explanation. Possible solutions of conservation, reintroducing in to the nature. Long-term effects of the problem of current technical bottlenecks that prevent better use of genomics to resolve conservation issues of vultures. Possibilities of using NSAID drugs in breeding and remove the harmful drugs from the food chain.

Result of this study suggest that this age-class is potentially still at risk in vulture population. Even with the goal of complete removal of diclofenac and other harmful non-steroidal anti-inflammatory drugs from the environment, it is possible that residual quantities of diclofenac remain in livestock carcasses and threat vultures. The establishment of a Vulture Safe Zone (VSZ) in the study area in 2012 saw the beginning of a new phase of environmental monitoring and conservation to address this issue. Across the approximately 8,000 km^2 VSZ, a range of activities such as livestock health camps, awareness-raising sessions in villages and consultations with veterinary dispensaries are all aimed at highlighting the risks to vultures from diclofenac and emphasizing the need to maintain the ban on its use in livestock. We have sequenced eleven samples for the identification of single nucleotide polymorphisms (SNP). There were two SNPs, and three heterozygous conditions were identified *(Table 1)*. T>G and G>A conversions were identified at positions 1 and

Table 1. SNPs, Heterozygous conditions and mutation types
1. táblázat SNP-k (egypontos nukleotid-polimorfizmusok), heterozigóta állapotok és mutáció típusok

Position	RefSeq	Changed position	Total samples	Transition/Transversion
1	T	G	V5	Transversion
2	G	A	V1, V3, V4, V5, V7, V10, V11	Transition
53	T/C	(Heter.)	V1, V2, V3, V4, V5, V7, V9, V10, V11	
110	T/C	(Heter.)	V1, V2, V3, V4, V5, V7, V9, V10, V11	
130	T/G	(Heter.)	V1, V2, V3, V4, V5, V7, V9, V10, V11	

2, respectively, as compared with the reference sequence (A/C). Two T/C heterozygous positions (53 and 110), and one T/G heterozygous position (130) were identified in all samples *(Table 1)*. However mtDNA is haploid, heteroplasmy (more than one mtDNA type in an individual) may occur. It is now thought that all individuals are heteroplasmic at some level – many above the limits of detection in DNA sequence analysis – thus heteroplasmy can be explanation for the occurrence of heterozygous positions in our sequences. All the other genetic information is given in *Table 1*.

Furthermore, an important next step in the monitoring of this colony is to determine breeding success. Comparing breeding success with pre-decline populations (and those that were suffering acute mortality from diclofenac poisoning could offer an indication of what levels of additive mortality exist for this population. Similarly, comparison of breeding productivity with the nearby colony of Long-billed Vultures will be important to see if the colonies are both (or neither) affected by similar rates of mortality which changes the genetic diversity of the vultures.

Finally, dispersal behavior of birds from this population must also be assessed. Oriental White-backed Vultures can range over vast distances, so it is not unlikely that birds may be dispersing across a wide area in the same way that birds may have arrived to the Nagarparkar

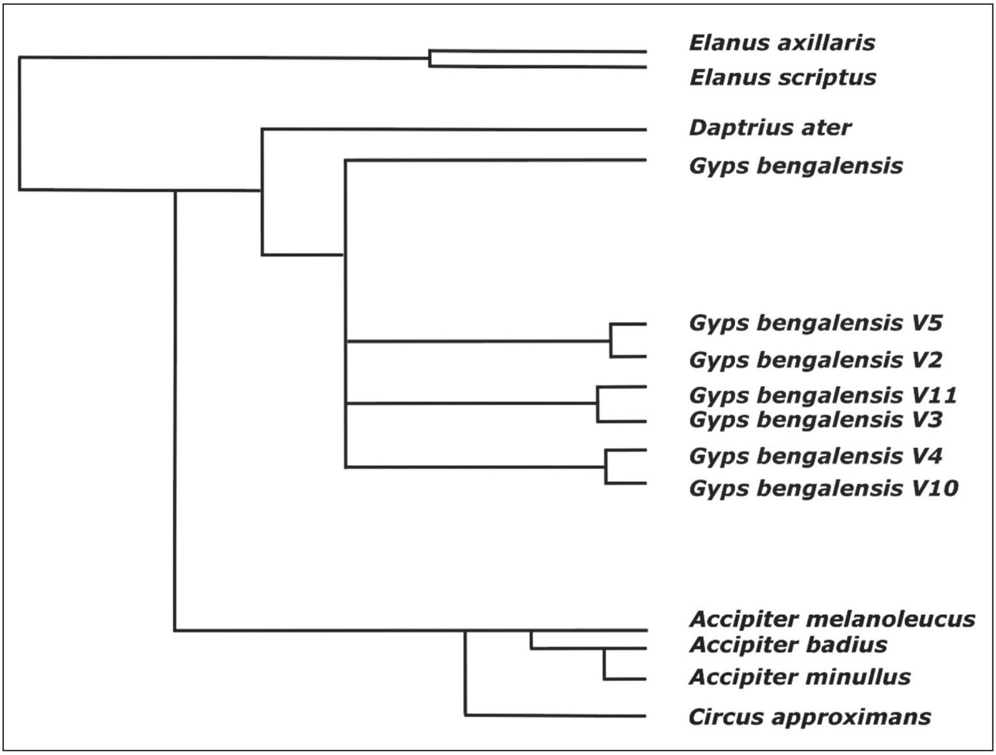

Figure 2. Phylogenetic analysis of samples of *Gyps bengalensis*. Other species on the tree represent raptorial birds

2. ábra A bengál keselyű minták filogenetikus analízise. A fán szereplő többi faj a ragadozómadarakat képviseli colony from adjacent areas such as Gujarat in India. Looking at the results of molecular diversity and a phylogenetic analysis of Oriental White-backed Vultures, using a mitochondrial D-loop marker, no new species was identified. In samples V5 and V2, V11 and V3, as well as V4 and V10, share a common clade and have a common ancestor. Reference sequences, V9, V7 and V1, appeared to closely be related to each other. Sequence samples V5 and V6 of *Gyps bengalensis* are more closely related to *Gyps bengalensis* reference material *(Figure 2)*.

However, the long-term conservation value of a Vulture Safe Zone will be reduced if there are limited opportunities for vultures to nest in the spatial patterns that optimize the dynamics of their breeding colonies. Based on the results presented here in genetic analysis of mitochondrial D loop marker, and in addition to the removal of unsafe veterinary drugs, a key component of Vulture Safe Zone work should be the preservation of nest tree distributions that can support large colonies of clustered nests of Oriental White-backed Vultures.

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Observations on parental care of the Eurasian Spoonbill *(Platalea leucorodia)* during the post-fledging dispersal

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Abstract The number of available publications on the post-fledging parental care of wading birds (herons, ibises, spoonbills, and storks) and many other bird species is limited. In this study, I summarised the available knowledge collated from the observations of the Eurasian Spoonbills *(Platalea leucorodia)* in Hungary. A part of the available data is based on observations of colour-ringed individuals. The latest feeding event of a young by its parent was observed at Lake Csaj on 5 October 2020. The youngsters were being fed by their parents for 43 days (observed maximum) during the post-fledging dispersal. However, I estimated that this behaviour could even last for as long as 53 days. The parents lead (care for) their yearlings for 51 days (observed maximum), again I estimated that it could potentially last for a longer period of 63 days. The estimated length of parental care and feeding period could be longer or a little bit shorter during the post-fledging dispersal because it was not possible to follow the life of the families exactly. During parental care (feeding and leading of chicks), the majority of the colour-ringed Spoonbills were observed 2–26 km to the natal colonies of yearlings and the breeding colonies of the adults. However, on some occasions, they were 111–145 km far from those colonies. During the post-fledging dispersal, Spoonbills care for their chicks for a longer time than the European breeding heron species. A possible reason could be that the bills of young Spoonbills are not appropriate for fishing effectively at the beginning of fledging because of their shorter length and their less efficient hydrodynamic effect during lateral sweeping. Another reason could be that Spoonbills are tactile foragers and need more time to learn fishing. Based on data of a juvenile followed by a GPS device, learning the migration route and stop-over sites from parents or experienced adults could be important for Spoonbills, otherwise, young migrating alone with no accomplished individuals may not find the optimal routes and the proper stop-over areas. In the case of Spoonbills, we still do not know exactly the features of parental care during the post-fledging dispersal and have even less data on it during the migration. Thus, I request potential observers along the Adriatic Flyway to record the observations of parent-offspring interactions (feeding by parents, begging) particularly the Hungarian colour-ringed adults and/or young individuals and send data to the author's e-mail address.

Keywords: parental feeding, begging, parental leading, post-fledging dispersal and migration, Threskiornithidae, colour-ringing

Összefoglalás Általánosan elmondható, hogy a gázlómadarak (gémek, íbiszek, kanalasgémek és gólyák) és sok más madárfaj kirepülést követő utódgondozásáról kevés a hozzáférhető irodalmi adat. Ebben a dolgozatban a kanalasgémek hazai megfigyeléseit foglaltam össze. Az adatok egy része színes gyűrűs megfigyeléseken alapul. A legkésőbbi etetést 2020. október 5-én a tömörkényi Csaj-tavon észleltem. A színes gyűrűs madarak megfigyelései alapján az etetések a kirepülést követően 43 napig (észlelt maximum), míg a becslések alapján 53 napig tartanak. Ugyancsak a színes gyűrűs kanalasgémek megfigyelései szerint a kirepülés után 51 napig (észlelt maximum), míg a becslések szerint 63 napig vezetik az öregek a fiókákat. A megadott becsült etetési és fiókavezetési időszakok akár hosszabbak, vagy kissé rövidebbek is lehetnek, mivel a családok életét nem lehetett pontosan követni. A szülői gondoskodás alatt a legtöbb esetben a gyűrűs kanalasgémeket az adott évben az általuk használt gémteleptől 2–26 km-re figyelték meg, de az elmozdulások a dokumentált esetek kisebb részében 111–145 km-esek is lehettek. A kanalasgémek hosszabb ideig gondozzák a fiókákat a kirepülés után, mint a hazai gémtelepeken költő gémfélék. Ennek lehetséges oka, hogy a fiatal kanalasgémek csőre a kirepülést követően még nem alkalmas a hatékony halászatra a hossza és a hidrodinamikai tulajdonságai miatt és/vagy, mivel a kanalasgémek tapintással kutatják fel a zsákmányukat, lehetséges, hogy több időre van szükségük megtanulni halászni. Egy GPS-jeladós fiatal kanalasgém útvonala alapján valószínű, hogy fontos a szülőktől, vagy más tapasztalt öregektől megtanulni a vonulási útvonalat és a megállóhelyek (stop-over) elhelyezkedését, mert a tapasztalt példányok nélkül vonuló fiatalok nem biztos, hogy megtalálják azokat. A kanalasgémeknél – más gázlómadarakhoz hasonlóan – a kirepülés utáni diszperzió során az utódgondozást még nem ismerjük részleteiben és a vonulás alatti gondoskodásról még ennél is kevesebbet tudunk, ezért kérjük a megfigyelőket, hogy az Adriai Madárvonulási Útvonalon jegyezzék fel a magyar színes gyűrűs öreg és/vagy fiatal kanalasgémek megfigyelése esetén is a szülő-fióka interakciókat (etetés, koldulás) és azt juttassák el a szerző e-mail címére.

Kulcsszavak: etetés, koldulás, fiókavezetés, kirepülés utáni diszperzió és vonulás, íbiszfélék, színes gyűrűzés

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Introduction

Parental care during the post-fledging dispersal and migration is essential for offspring to learn important behaviour elements from their parents and develop their own skills: fledglings 1) improve their flight skills, 2) learn how to socialise with other conspecific individuals, 3) acquire foraging behaviour and food preferences, 4) learn how to avoid predators, and 5) acquire migration routes (McIntyre & Collopy 2006, Hayes & Barzen 2016). The learning process during the post-fledging period is vital. However, despite its importance it has not been investigated in detail and is one of the least-studied life history stages of many bird species because it is often difficult to follow fledglings after they leave their nests or colonies (McIntyre & Collopy 2006).

The Eurasian Spoonbill *(Platalea leucorodia)* (hereafter Spoonbill) is a regular colonial breeder in Hungary and its population was estimated at 850–1,300 pairs between 2002 and 2013 and then 550–900 pairs between 2014 and 2020 (Pigniczki 2021). In Hungary, the first Spoonbills appear in the breeding colonies from late February and early March (Haraszthy 2019). The egg-laying and the start of the incubation are estimated at approximately from the middle of March onwards (Pigniczki 2015, 2021). Both parents take part in the incubation, and then, both of them care for and feed the chicks (Cramp 1977, Bloomfield 2001). Adults feed their chicks by regurgitation (Cramp 1977). In Hungary, Spoonbills start to fledge from the end of May and fledging could last until the middle of August, depending on the date of egg-laying (Pigniczki Cs. unpublished data). The fledging estimated by Cramp (1977) happens when the chicks are 45–50 days old, however, they leave their nests and move to a trampled down place close by when they are approximately 30 days old. Hancock (1992) judged the fledging when the chicks are 45–54 days old. In England, Bloomfield (2021) found that nestlings leave their nests when they are 31–42 days old (mean: 37.2 days). After fledging and departure from the colonies, parents move together with their young up to at least ten weeks. During this period, parental feeding continues as long as young increasingly find their food alone (Cramp 1977). At the same time, Cramp (1977) reported that the age of independence of the young is not yet known.

There is no further literature information available on parental care of Spoonbills when the families started their post-fledging dispersal. The main goal of this publication is to provide an overview of the observations related to parental care of Spoonbills during the post-fledging dispersal in Hungary and attract the attention of the ornithologists, ring-readers, birdwatchers, and bird photographers working in the field to collect and report behavioural data.

Material and Methods

I used the data of the colour-marked Spoonbills and one of the GPS-tagged individuals for this study. The method of colour-ringing in Hungary was described in detail in several previous publications (Pigniczki & Végvári 2015, Pigniczki *et al.* 2016). The chicks were captured for colour-marking when their ages were estimated at 3–5(–6) weeks old. During the ringing, it was not possible to measure the chicks thus no estimation is available on their accurate age. I used the records of colour-ringed Spoonbills 1) marked in Hungary and observed in Hungary or abroad and 2) the data of individuals with Serbian or Croatian origin and observed in Hungary.

An adult female Spoonbill was trapped in her foraging area at a fishpond (Lake Csaj) on 25 March 2018 and then was equipped with a GPS device. I used some continuously inspected leg loops to catch her. The GPS device mounted on her back is an OrniTrack E-25 2G GPS-GSM device (produced by Ornitela) the weight of which is approximately 25 g. This type of device basically forwards the collected data via a 2G network. If the 2G network is not available then the data transmission happens via a GSM network. It was not possible to weigh the captured individual, but the bird was in a good condition (no sharply outstanding sternum was detected by hand). According to Triplet *et al.* (2008), a Spoonbill weighs 1800–2400 g, thus the mass of the device and harnesses did not exceed the 3% of the bodyweight of the captured specimen.

Not only parental feeding of the young was treated as an indicator of parental care, but also the begging behaviour of the chicks if it was tolerated by the adult. Strange adults do not tolerate begging fledglings that belong to other families and these young were often attacked, however, parents never attacked their own offspring (Pigniczki Cs. unpublished observations). Thus, if an adult tolerated begging but young was not being fed I treated that as an indicator of parental leading. During begging, young are following adults and bobbing their heads rhythmically up and down and giving a continuous, high-pitched shrilling, and rasping "cheeerr" call (Cramp 1977, Bloomfield 2021, Pigniczki Cs. unpublished observations). When a parent feeds an offspring, the young put its bill into the pharynx or the throat of the parent to have the regurgitated food *(Figure 1)* (Cramp 1977, Pigniczki Cs. unpublished observations).

During the field observations, two methods were used to collect data on parental care. One of the methods was a scan sampling: when I observed a Spoonbill flock, I regularly noted if I observed that a yearling was being fed by an adult. If the birds were not marked it was not possible to decide whether that family was observed previously or not. During 2020 (between 14 July and October 5) and on 27 June 2021, I noted if pieces of food were fallen

Figure 1. An adult Spoonbill is feeding its fledged offspring. Note that the young insert its bill into the throat of the adult to get the regurgitated food (Photo: Cs. Pigniczki)

1. ábra Öreg kanalasgém eteti a kirepült fiókáját. A fiatal a csőrét a szülő torkába dugja, hogy megszerezze a visszaöklendezett táplálékot (Fotó: Pigniczki Cs.)

into the water while parents fed their young. The other method was noting the behaviour of the colour-ringed juveniles and adults: in this case, I recorded 1) the begging behaviour of juveniles toward adults, and 2) when yearlings were being fed by adults.

When I estimated the length of parental leading and feeding of youngsters during post-fledging dispersal, I regarded the age of the Spoonbills uniformly as three weeks (21 days) old during marking because no biometric data of nestlings is available. I considered the age of the fledging to be 50 days. Hence, it was generally estimated that Spoonbills fledged on the 29th day after their marking. In the case of a fledgling, where the difference between the day of its marking and the day of its first observation out of the colony was lesser than 29, I treated the day of the first observation out of its colony as the estimated fledging day. The difference in days between the last day when parental care (leading or feeding by parents) was observed in the case of a given young and the estimated fledging day and plus one day added is the estimated length of parental care. In the case of observed length of parental care, I applied the difference between the first and the last day when parental care was detected and plus one day added.

The natal colony of a youngster was identified if that has a colour-mark. It was possible to determine the breeding colony of the adults, 1) if a GPS device indicated that, 2) if a colour-ringed adult was observed in a particular colony, 3) if a colour-ringed adult was regularly observed during the breeding period in the vicinity of a colony, and 4) if a colour-ringed adult led or fed at least one colour-ringed fledgling. To calculate the distance between two locations, I used the 'fossil' package in R (Vavrek 2011).

Results

During post-fledging dispersal, adult Spoonbills led their fledglings to wetlands near their natal colonies. While yearlings stay in one location, adults may forage in other areas and I often observed that adults arrive to feed their fledged young in the 'kindergartens'. Juveniles were also observed when they learnt fishing: they often caught pieces of reed from the water and they held them in their bills.

Adults care for their chicks for quite a long. They fed their chicks even in early October (1 October 2019 and 5 October 2020) as was observed in the case of unmarked families at Lake Csaj *(Table 1)*. The youngsters obtain their food from the throat of their parents. It was observed seven times on 48 occasions (14.6%) that smaller or bigger pieces of

Table 1. The date of the latest observed parental feeding of young between 2017 and 2021 in Hungary

1. táblázat Az évente legkésőbb megfigyelt etetések dátumai 2017–2021 között Magyarországon

Date of observations of the last parental feed in each year
12 September 2017
20 September 2018
1 October 2019
5 October 2020
20 August 2021

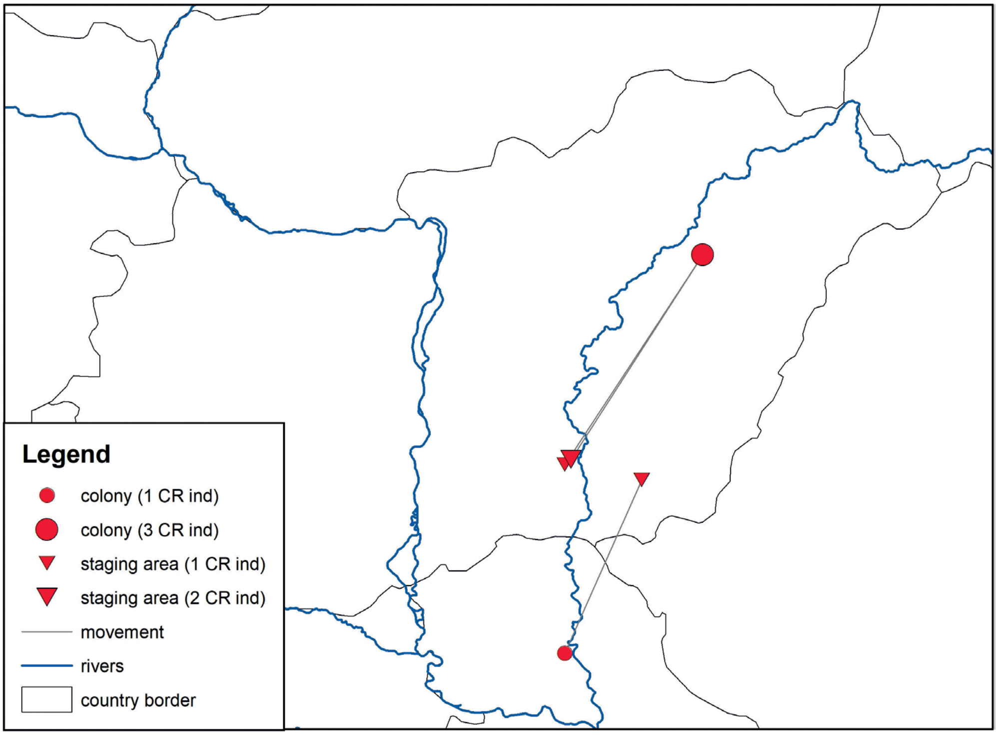

Figure 2. Long-distance movements (over 100 km) of Spoonbill families during parental care between the natal/breeding colonies and the staging areas during dispersal movements

2. ábra A kanalasgém-családok hosszú távú (100 km feletti) diszperziós elmozdulása az utódgondozás alatt a kelési/fészkelőtelep és a későbbi tartózkodási helyek között

Table 2. The latest dates, the distances from the natal/breeding colonies, the estimated and the observed lengths of periods when young Spoonbills are fed by their parents or the adult Spoonbills feed their chicks. In some cases, a row contains the data of young and their parents as well. In these cases, where the data of offspring and parents differ from each other are separated with interrupted lines. The table contains those data when the parents and/or yearlings had colour rings

2. táblázat Kanalasgém fiatalok legkésőbbi etetésének, vagy az öregek legkésőbbi fiókaetetésének dátuma, távolsága a kikelési-/fészkelőhelytől, a becsült és a megfigyelt etetéses gondoskodás hossza. Bizonyos esetekben egy-egy sor tartalmazza a fiatal és az öreg adatait is. Ezekben az esetekben, ahol az utódok és a szülők adatai között eltérés van, szaggatott vonallal választottam el. A táblázatban az azonosítható, színes gyűrűs egyedek adatai szerepelnek

Colour-ring code	Age	Natal/breeding colony	Location and date of last observation of parental fed	Distance from colony (km)	Estimated length of periods when young are being fed by their parents (days)	Observed length of periods when young are being fed by their parents (days)
JH53	1	Hortobágy Fishpond	Lake Csaj, 27 July 2017	140	37	NA
JL50	1	Lake Péteri	Lake Csaj, 8 September 2017	13	53	40
JH72	1	Lake Fehér (Szeged)	Nagy-Széksós-tó, 14 August 2017	18	11	9
JH79	1	Lake Fehér (Szeged)	Lake Csaj, 12 September 2017	25	33	30
JP06	1	Ürbő Fishpond	Apaj-puszta, 2 July 2018	3	7	NA
RaBf/GYG	10				NA	NA
JP31	1	Lake Fehér (Szeged)	Fertő (Szeged), 13 July 2018	4	8	NA
J997	4				NA	NA
JP58	1	Lake Fehér (Szeged)	Fertő (Szeged), 1 August 2018	4	19	16
J806	6				NA	NA
JP59	1	Lake Fehér (Szeged)	Fertő (Szeged), 19 July 2018	4	6	NA
J806	6				NA	NA
JT49	1	Hortobágy Fishpond	Hortobágy Fishpond, 17 June 2019	2	6	6
aGBf/GYL	12				NA	NA
JP95	1	Lake Fehér (Szeged)	Nagy-Széksós-tó, 11 August 2019	19	43	43
J02F	1	Lake Fehér (Szeged)	Nagy-Széksós-tó, 1 August 2019	18	31	18
J03F	1	Lake Fehér (Szeged)	Nagy-Széksós-tó, 11 August 2019	18	40	40
E|MT	1	Becej Fishpond (Serbia)	Lake Fehér (Kardoskút), 2 August 2011	111	37	NA
B/W [TK]	6				NA	NA
B/R [GK]	15	Lake Fehér (Szeged)	Nagy-Széksós-tó, 9 August 2017	18*	NA	NA

Table 3. The latest observations, the distances from the natal/breeding colonies, the estimated and the observed lengths of periods when young Spoonbills are led by their parents or the adult Spoonbills lead their chicks. In some cases, a row contains the data of young and their parents as well. In these cases, where the data of offspring and parents differ from each other are separated with interrupted lines. The table contains those data when the parents and/or yearlings had colour rings

3. táblázat Kanalasgém fiatalok szüleik általi vezetésének a legkésőbbi megfigyelési dátuma, távolsága a kikelési-/fészkelőhelytől, a becsült és a megfigyelt gondoskodás hossza. Bizonyos esetekben egy-egy sor tartalmazza a fiatal és az öreg adatait is. Azokban az esetekben, ahol az utódok és a szülők adatai között eltérés van, szaggatott vonallal választottam el őket. A táblázatban az azonosítható, színes gyűrűs egyedek adatai szerepelnek

Code	Age	Natal/breeding colony	Location and date of last observation of parental leading	Distance from colony (km)	Estimated length of parental leading (days)	Observed length of parental leading (days)
J662	1	Lake Kolon	Böddi-szék, 5 July 2013	17	14	2
J663	1	Lake Kolon	Böddi-szék, 5 July 2013	17	14	3
J364	1	Lake Csaj	Lake Csaj, 28 July 2015	4	56	33
L/L [HE]	12				NA	NA
J936	1	Hortobágy Fishpond	Büdös-szék (Pusztaszer), 8 August 2015	145	63	51
J816	1	Lake Fehér (Szeged)	Lake Csaj, 28 July 2015	26	44	44
GYa/RBfG	7				NA	NA
JL38	1	Lake Csaj	Lake Csaj, 27 July 2017	4	49	49
JL51	1	Lake Péteri	Lake Csaj, 8 September 2017	13	53	42
JL70	1	Lake Péteri	Lake Csaj, 7 September 2017	13	42	34
JH75	1	Lake Fehér (Szeged)	Nagy-Széksós-tó, 21 August 2017	18	18	13
J18H	1	Hortobágy Fishpond	Nagy-szik, 16 July 2020	21	47	47
J173	7	Hortobágy Fishpond	Lake Csaj, 3 September 2017	141	NA	NA
J278	5+	Vidre-ér Fishpond	Lake Csaj, 27 August 2019	7	NA	NA

the regurgitated food fell into the water while adults fed their young. After 13 days of her estimated fledging, a juvenile (ring code: J707) foraged independently at Lake Fehér (near Gátér) on 16 June 2015. J707 foraged on small food (probably aquatic animals) and later, held a twig in her bill.

Based on observations of colour-ringed juveniles, adult Spoonbills care for and feed their chicks not only in the vicinity of their natal colonies (range: 2–26 km), but also in larger distances (111 km, 140 km, 141 km, and 145 km) *(Figure 2, Table 2, Table 3)*. Observations on long-distance dispersal (100+ km) were detected from late July. Adult Spoonbills lead their young from one place to another. One of the adults [ring code: LYa/(Bf)RY] led and fed its offspring to two different locations: the first place was at Fertő (near Szeged) on 28 June 2015 and five days later, the other place was at Lake Csaj on 3 July 2015, with 28 km distance between them.

During the post-fledging dispersal, 6–43 days was the length of the observed parental feeding period of young *(Table 2)*. It was estimated that youngsters were being fed by parents for up to 53 days (7.6 weeks) (range: 6–53 days, median: 31 days) after their fledging. 2–51 days was the length of observed parental leading of yearlings *(Table 3)*. It was estimated that the parental leading could last even for 63 days (nine weeks) (range: 14–63 days, median: 45.5 days) after fledging.

Discussion

Based on observations during the post-fledging dispersal, I estimated the maximum length of parental feeding and leading periods in Central European Spoonbills and they could last for 7.6 and 9.0 weeks, respectively. These periods could be even longer 1) if the chicks were older during their markings and fledged earlier than it was estimated and 2) if it was not possible to follow the parent-young relationships until the end. In contrast, these periods could also be shorter if some of the young fledged later than they were estimated. Cramp (1977) stated that parental care may last up to at least ten weeks. The result of this paper gives a roughly similar estimation. However, parental care could be even longer and needs further observations to have a better understanding of this.

Spoonbills have quite a long parental care period during post-fledging dispersal compared to other species that live in the same environment and are closely related to them. The adults in the Ardeidae family feed their young with regurgitated food. Young chicks get their food directly into their bills, whilst older offspring are being fed with food regurgitated on to the floor of the nest. It was described in the case of Eurasian Bitterns *(Botaurus stellaris)*, Black-crowned Night Herons *(Nycticorax nycticorax)*, Squacco Herons *(Ardeola ralloides)*, Cattle Egrets *(Bubulcus ibis)*, Little Egrets *(Egretta garzetta)*, Grey Herons *(Ardea cinerea)* and Purple Herons *(Ardea purpurea)* (Cramp 1977). In the case of the Little Egret, young are independent after one month (four weeks) of fledging (Voisin 1991). The yearlings of Grey Herons are independent after two weeks of their fledging or a bit later (Voisin 1991). According to Cramp (1977), young of other species of the Ardeidae family that breed in Europe become independent shortly after fledging. Young Spoonbills get their food from their parents when they put their bills into their parents' throats. The length of the parental feeding period during the post-fledging dispersal was estimated at minimum ten weeks by Cramp (1977). Based on observations of colour-ringed young Spoonbills, they were being fed by their parents for roughly six weeks (43 days) and moved together with their parents for more than seven weeks (51 days). However, estimations based on recent results suggest that yearlings could be fed by adults for eight weeks and with parents caring for them for up to nine weeks. Probably, the different lengths of the parental care in the case of Spoonbills and herons could be caused by the different foraging techniques and the time needed to learn them: herons are visual foragers while Spoonbills are tactile foragers (Martin & Katzir 1994, Swennen & Yu 2005). Young Great Cormorants *(Phalacrocorax carbo)* are being fed for 40–50 days after fledging and they also insert their heads into their parents' bills (Cramp 1977). In the case of the European Shag *(Gulosus aristotelis)* young are being fed by their parents for 40 days after fledging. When young Shags become independent they spend more time for foraging than adults, because yearlings are inexperienced in that activity (Daunt *et al.* 2007). Young Spoonbills are led by their parents to sites with shallow water where they can start the learning process of fishing. Only a very limited number of observations and notes are available on the exact process of when and how the juveniles start to learn to obtain their prey. A juvenile foraged independently on 16 June, 13 days after of her estimated fledging, however, it was supposed to be being fed by their parents, too. Compared to other species of Ardeidae, young herons start to learn how to catch their prey in the water at the foot of the heronry as soon as they are able to fly. During this period, young herons also visit their nests several times a day and are still taking nourishment from their parents (Voisin 1991).

The longer period of parental care could also be the result of the shape and the length of the bill of young Spoonbills. Spoonbills use their spoon-shaped, flattened, and broad bills and lateral sweeping to shed a vortex off their bill tips that result in hydrodynamic suction on the bottom, which disturbs and makes their prey move. To achieve this, the bill should be kept close to the bottom during foraging (Weihs & Katzir 1994, 2008). According to Cramp (1977), the full bill length is attained when Spoonbills are 3–6 months old. Based on field observations, the bill length of young could be obviously short after fledging (Pigniczki Cs. unpublished observation). It is very likely that a short and growing bill is not optimal for effective fishing at the beginning, besides, the fledglings are inexperienced foragers.

Parents and young Spoonbills move together at the beginning of the post-fledging dispersal. Yearlings are led by their parents to 'kindergartens' in wetlands within the vicinity of the colonies. Parents and their young were often observed between 2–26 km distances from the natal colonies of the chicks in the 'kindergartens'. From late July, there is evidence that parents and juveniles moved together over a hundred kilometres from their breeding or natal colonies. Previously, it was found that juvenile Spoonbills' maximal distance to their natal colonies increased with months during the post-fledging dispersal (Pigniczki & Végvári 2015).

Interestingly, a young Spoonbill was photographed in NE Italy, 408 km far from its natal colony, quite early, on 30 June 2009, after 39 days of its marking (Pigniczki 2015). That individual seemed to start its migration very early because it was observed out of the Carpathian Basin (Pigniczki & Végvári 2015). Based on the recent results, parents, in general, care for their young in June, thus I assume that the adults led that young to Italy. Unfortunately, no data is available whether or not that the youngster was being fed by their parents in Italy. In Hungary, young Spoonbills are being fed by their parents during summer and some of them could potentially be fed up until September, and even as late as early October when most of the juveniles are on their migration or have arrived at their wintering areas (Pigniczki *et al.* 2016). During the migration and the arrival to the wintering areas, parental care was never reported, probably, because the observers did not pay attention to this phenomenon. In the case of waterbirds, I did not find available data where parents fed their offspring during migration. However, there are several migratory waterbirds where the parents and their offspring migrate together such as described in the case of swans (*Cygnus* sp.), geese (*Anser* sp., *Branta* sp.), and cranes (Gruidae) (Cramp 1977, Hayes & Barzen 2006, Jonker *et al.* 2011, Kölzsch *et al.* 2020). During parental care, the young geese and cranes learn migratory routes from their parents which is called cultural transmission (Hayes & Barzen 2006, Jonker *et al.* 2011). A juvenile Spoonbill with a GPS device left the Balkans and moved to Italy in January when adult Spoonbills stay in their wintering areas. Thus, this young Spoonbill probably moved alone or together with inexperienced conspecifics and although reached Italy flying along a suboptimal route, it seems that he did not find optimal stop-over areas to forage and died within four days (Pigniczki Cs. unpublished data). Based on the GPS track of this young Spoonbill, I suppose that the cultural transmission from parents or strange experienced adults could be essential for Spoonbills as well to learn the migratory routes and the optimal and adequate stop-over sites.

As a summary, the long parental care in the case of Spoonbills could be the result of many factors such as 1) the learning of tactile fishing could be difficult and young need more time to become experienced foragers, 2) their bills are short after the fledging and are not proper tools for fishing. During their parental care, they might have long-distance dispersal movements.

Call for observation of parental care of Eurasian Spoonbills along the Adriatic Flyway

My data analysis indicates that Spoonbill could be an ideal bird species to study the parental care and the parental feeding of young because the Spoonbill is 1) a highly sociable species, 2) easy to detect when yearlings are begging or being fed, and 3) due to colour-rings, easy to identify and follow the individuals. I request ornithologists, birdwatchers, and bird photographers to pay attention and read the code of colour-rings of Hungarian Spoonbills and help collect data on the parental care along the Adriatic Flyway (Hungary, Austria, Slovakia, Romania, Serbia, Croatia, Slovenia, Bosnia & Herzegovina, Montenegro, Greece, Italy, Algeria, Tunisia, and Libya) during the post-fledging dispersal and migration. The requested data are the followings:

1. date and length of the observation,
2. location with exact coordinates,
3. the number of individuals in the Spoonbill flock,
4. the number of yearlings and adults,
5. the code of colour-rings,
6. relationship between colour-ringed adult and colour-ringed young (if both of them have colour-rings),
7. behaviour of colour-ringed birds:

    a. colour-ringed adult is followed by a begging juvenile (no observation of being fed)
    b. colour-ringed adult feed juveniles
    c. colour-ringed begging young
    d. colour-ringed young is being fed
    e. colour-ringed juvenile forage independently
    f. no sign of parent-offspring interaction
8. photo or video if available.

I kindly request that the observers send any collected data to myself for accurate recording.

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A comparative study on the nesting materials used by House Sparrow *(Passer domesticus)* for Open and Inbox nests

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Abstract House Sparrow *(Passer domesticus)* is a bird species connected to humans, widely distributed in most of the human settlements. They build nests in the crevices of our homes. The recent changes in house design minimized the crevices to build the nests and lead to habitat loss. The nest boxes are the alternative ways to re-establish the decreased House Sparrow population. We investigated the usage of nesting materials by House Sparrow with reference to construction of Open nests and Inbox nests in our study area. Because of its flexible attitude, it utilizes all the available materials for nest construction. In our study area at Jangareddigudem, we have examined about 100 Inbox nests and 40 Open nests. There are around 29 varieties of nesting materials used by Sparrows in the examined Inbox nests and around 27 varieties in the Open nests. Dūrvā grass found to be the major component of all the analyzed nesting materials in both Open and Inbox nests that constituted 43% and 36.5% respectively. The other structural materials such as coconut fibre (3.5%) and broom fibre (6%) were found to be more in Inbox nests. Synthetic fibre was more in Open nests (3.7%). We found significant differences between the Open and Inbox nests with respect to quantum of each nesting material type used, weight of the nests and time taken for nest construction by House Sparrow.

Keywords: House Sparrow, nesting materials, nest boxes, Inbox nests, Open nests

Összefoglalás A házi veréb *(Passer domesticus)* kultúrakövető madárfaj, amely a legtöbb emberi településen előfordul. Fészkeiket az épületek réseibe építik. A házak tervezésének – építésnek a közelmúltban bekövetkezett változásai minimálisra csökkentették a fészkelési lehetőségeket, ami az élőhelyük csökkenéséhez vezetett. A fészekodúk kihelyezésével a lecsökkent házi veréb populáció helyreállítható. A faj fészkelőanyag-használatát vizsgáltuk két fészektípusban (nyitott fészek, odú). A madarak rugalmas hozzáállása miatt minden rendelkezésre álló anyagot felhasználtak a fészeképítéshez. A Jangareddigudemben található vizsgálati területünkön körülbelül 100 fészekodút és 40 nyitott fészket vizsgáltunk meg. A vizsgált odúkban 29 féle fészkelőanyagot találtunk, a nyitott fészkekben pedig 27 félét. Az összes elemzett fészkelőanyag fő összetevője a csillagpázsit volt mind a nyitott fészkekben, mind az odúkban, amely aránya 43%, illetve 36,5% volt. A többi fészekanyag, például a kókuszrost (3,5%) és a seprűszál (6%) nagyobb arányban volt az odúkban. A szintetikus szál több volt a nyitott fészkekben (3,7%). Az odúkban levő és a nyitott fészkek között szignifikáns különbségeket találtunk az egyes használt fészkelőanyag-típusok mennyisége, a fészkek tömege és a fészeképítéshez szükséges idő tekintetében.

Kulcsszavak: házi veréb, fészekanyag, odú, nyitott fészek

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Introduction

House Sparrow *(Passer domesticus)* is a widely distributed bird species in human settlements (Summer-Smith 1988). The population of this tiny bird is under decline in several parts of the world (Royal Society for Protection of Birds 2002, Hussain *et al*. 2016). Though these birds were protected initially by the Red List (Paul 2015), they have been shifted to least concerned species (Bird life international, 2018), and their population is still declining at some parts of the globe (Berigan *et al*. 2020). One of the major reasons for their decline is the lack of nesting sites (Vincent 2005). As the House Sparrow is an opportunistic secondary nesting bird, it can utilize the nest boxes for its breeding activities (Bhattacharya *et al*. 2011, Chetan 2012, Balaji 2014, Mahesh & Suseela 2021). It can cope-up in all kinds of ecological conditions, and can use available and suitable nesting materials for nest construction including grasses, feathers, stalks, threads, fibres, paper, wool, etc. (Indykiewiez 1990).

The modified house architecture is unable to provide crevices as in the tiled-roofs or thatched houses. In search of breeding space, House Sparrow invaginated the present RCC buildings and started using available spaces such as ventilators, gaps of false roofs made up of POP ceilings, sanitary pipes, behind the compressor of air-conditioners, on electrical meter boxes etc. Construction of a secured nest in open areas requires more effort and also more material.

In this study, we concentrated on the usage of nesting materials by House Sparrow for nest construction in open areas (henceforth referred as 'Open nests') and a comparison was made with the nests constructed in provided nest boxes (henceforth termed as 'Inbox nests'). Statistical analysis was carried out by using T-test (paired two samples for mean) to identify the significant differences between Open and Inbox nests. The hypotheses made to carry out the present study are as follows.

Hypotheses of the study

H_{01}: The quantum of each material type used by House Sparrow for construction of Open nests and Inbox nests is the same.

H_{a1}: The quantum of each material type used by House Sparrow for construction of Open nests and Inbox nests significantly differs.

H_{02}: The overall weight of Open and Inbox nests constructed by House Sparrow remains the same.

H_{a2}: The overall weight of Open and Inbox nests constructed by House Sparrow significantly differs.

H_{03}: There is no significant difference in nest construction time by House Sparrow for the construction of Open nests and Inbox nests.

H_{a3}: There is significant difference in nest construction time by House Sparrow for the construction of Open nests and Inbox nests.

Methods

Jangareddigudem (17.1223° N, 81.2923° E) is an upland area of West Godavari District of Andhra Pradesh state of India. Its altitude is 74 meters above the median sea level, with 15.8 km^2 area. Jangareddigudem is a semi-urban town with a tropical climate. The town consists more of open areas *(Figure 1)* and is selected as the study area for conservation of House Sparrow by providing nest boxes since 2014. The occupancy rate of the nest boxes was reported as 97% in this town (Mahesh & Suseela 2021).

The total number of Inbox nests in each nest box was determined by carefully separating the nests starting from the uppermost nests to the bottom nest.

The abandoned Open nests and Inbox nests were collected, and analyzed for the nesting materials. In case of Open nests, they were collected carefully and the entire nesting material was made free from the dried excreta. The entire nest was then weighed and the nesting materials were separated systematically from core (inner cup) of the nest to outer structural materials. The quantity of all the nesting materials was determined. For Inbox nests, the nests were removed from the rear window, and then made them free from dried pellets and dust. Then each of the nests was weighed separately to find out the quantity of nesting materials in each nest. The nesting materials were separated carefully and calculated for their proportion.

Statistical analysis was carried out by using T-test (paired two samples for mean) to identify the significant differences between Open and Inbox nests with respect to quantum

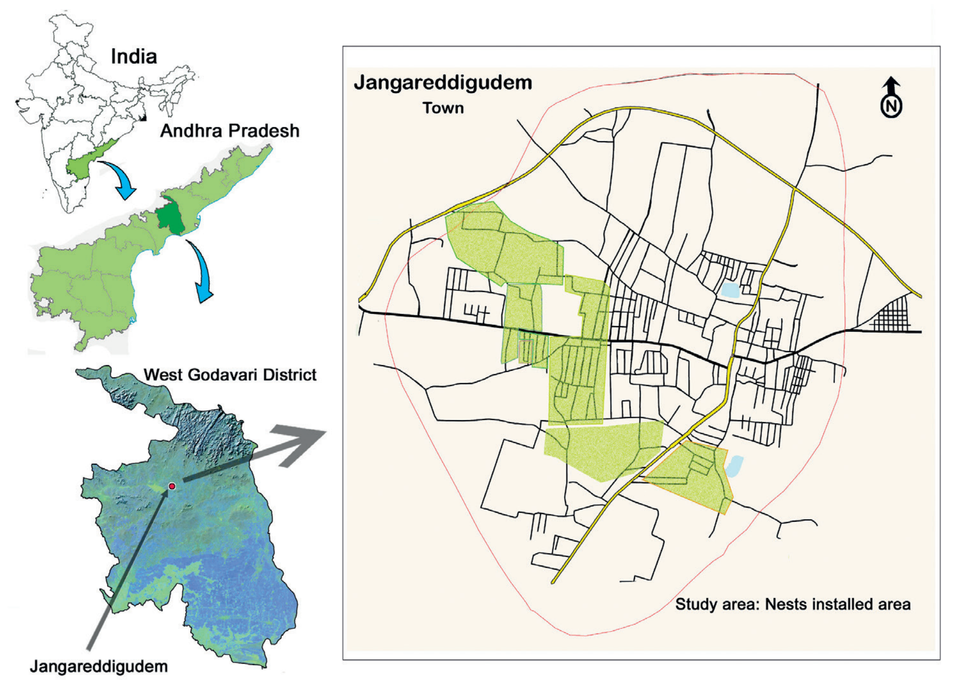


Figure 1. Map showing the study area (Nest boxes installation area)
1. ábra A vizsgálati terület térképe

of each nesting material type used, weight of the nests and time taken for nest construction by the House Sparrow. For statistical analysis, the nesting materials collected from 40 Inbox and 40 Open Nests was compared to find out the significant differences between the two.

Construction time for the nests was also observed. For Open nests, paired sparrows use to search for crevices. If any secured place they find, will start the collection of materials. For Inbox nests from the day of installation we observed for occupancy, then onwards regular monitoring was done for collection of nesting materials from structural and up to the thermal insulators.

Results

For the analysis of nesting materials, we surveyed around 40 Open nests, those were made on sanitary pipes, ventilators, and available crevices. The average weight of Open nests was found to be 94 g (range: 52–142). Among the observed Open nests, 37.5% were constructed with more than 100 g of material, 27.5% of the nests build with 91 to 100 g of material, and 35% of the nests with 50 to 90 g of nesting material.

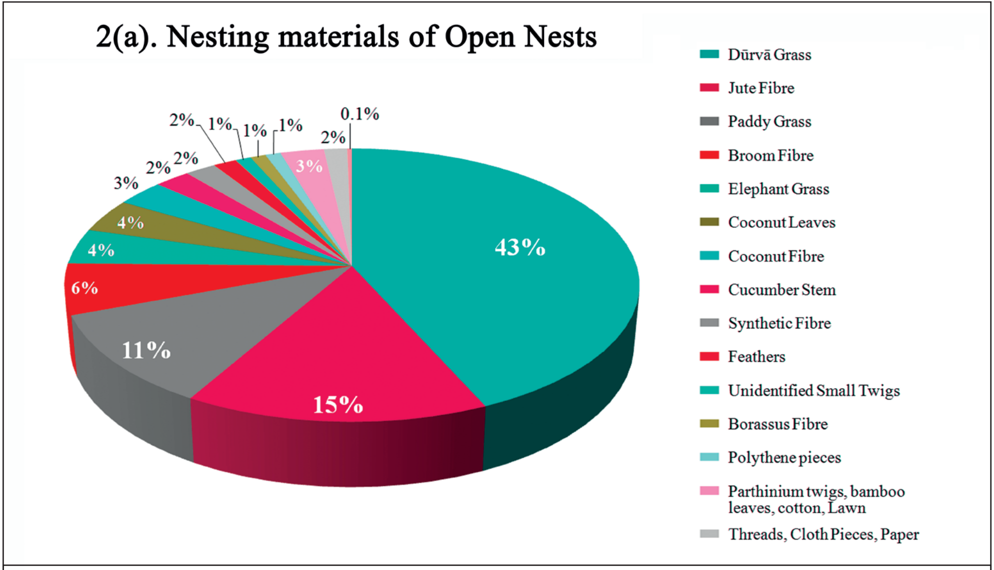
Among the nesting materials of Open nests, highest quantity was observed with dūrvā grass *(Cynodon dactylon)* with 43%, and the second-largest quantity was found with jute fibre *(Corchorus olitorius)* with 15.41%. In the Open nests, the paddy grass *(Oryza sativa)* was used as the outermost structural material *(Figure 2).*

With regard to the frequency of nesting materials, dūrvā grass was found in all the nests. The lining material that is jute fibre was found in 83% of the nests. Paddy grass was found only in 62.5% of the nests that were made on sanitary pipes. Broom fibre *(Thysanolanena maxima)* was used in 95% of the nests *(Table 1).*

In all the nest boxes, we have noticed more quantity of the nesting material in the first nest (Bottom most Inbox nest). The Sparrows built the first Inbox nest by collecting structural materials up to a height of 3 inches. To make the nest to such a height, they collect the available structural materials such as dūrvā grass, coconut fibre, broom fibre, etc. Of the examined nests, the average weight of the bottom nest was found to be around 42.2 g. On observation, it was noticed that Sparrows fill the first nest (bottom most) up to the closure of the rear window (an opening at the rear bottom, used to remove old Inbox Nests, without any disturbance to the nest box). From the second nest onwards, the weight of the nests was found to decrease by 20 g. The average weight of the second nests was about 21.23 g. The average weight of all the nests, from bottom to superficial layers was shown in *Figure 3*. For the construction of the 11th nest, Sparrows collected only 10.7 g of nesting material *(Figure 3)*, which comes to around one-fourth of the first nest. The weight of the nests is inversely proportional to the height of the nest box.

The data shown in *Table 2* represent the number of nests in examined 100 nest boxes. In 50% of the nest boxes, Sparrows made seven Inbox nests. In rare case, as in case of nest box no. 23, consisted of 14 Inbox nests *(Table 2).*

Among all the surveyed Inbox nests, on an average more than one third of the nesting material was dūrvā grass that constituted around 36.4%. Lining material, jute fibre (extracted from gunny bags, jute rope and jute thread (used for packing of grocery) was



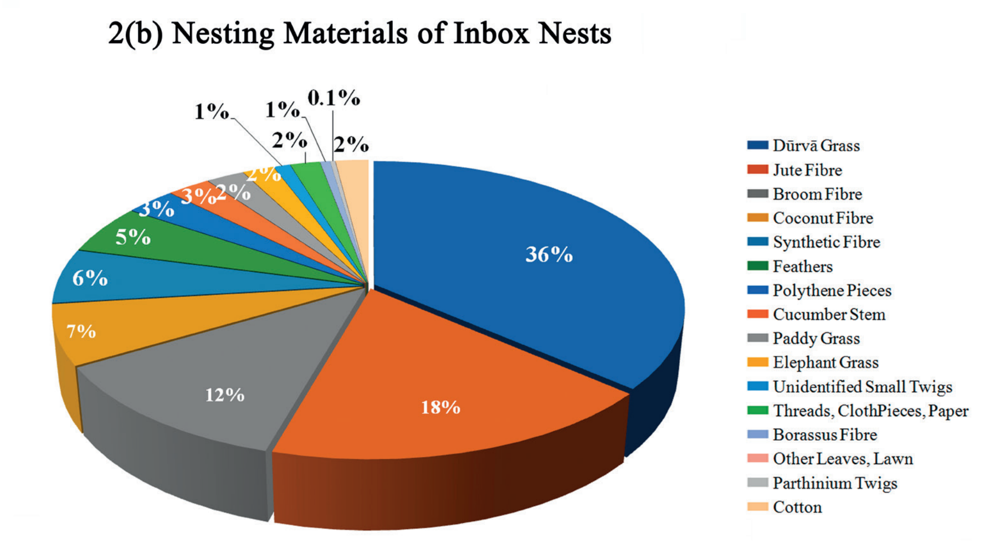


Figure 2. Composition of nesting materials in Open nests and Inbox nests
2. ábra A fészekanyag összetétel a két fészektípusban

the second-largest material, and constituted around 18.08%. Along with dūrvā grass, broom fibre and coconut fibre were found in all the nests, and constituted around 5.03% and 2.78% of the total weight, respectively *(Table 1)*.

Out of 15 structural materials that were tested, null hypothesis (H_0) was rejected for 11 (dūrvā grass, goose grass *(Eleusine indica)*, paddy grass, carrot grass *(Parthenium hysterophorus)* twigs, unidentified leaves, coconut leaves, lawn, cucumber stem (*Cucumis* sp.), bamboo leaves, palmyra *(Borassus flabillier)*, small twigs), indicating that there is significant difference in the quantum of these structural materials between the two types of

Table 1. Composition of nesting materials in Open and Inbox nests
1. táblázat A fészekanyag összetétele a két fészektípusban

Structural Materials					
				Open nests	Inbox nests
No.	**Species / material**	**Origin**	**Parts used**	**Weight (g)**	**Weight (g)**
1	Dūrvā grass (*Cynodondactylon*)	P	Stem & Leaves	40.4	15.38
2	Broom fibre (*Thysanolanena maxima*)	P	Inflorescence	5.5	5.03
3	Coconut (*Cocas nucifera*)	P	Fibre of fruit	3.12	2.78
4	Cucumbers (*Cucumis species*)	P	Stem	2.15	1.11
5	Paddy grass (*Oryza sativa*)	P	Grass	10.55	0.97
6	Goose grass (*Eleusine indica*)	P	Leaves	4.025	0.82
7	Small twigs (unidentified)	P	Twigs	1.2	0.42
8	Palmyra (*Borassus flabillifer*)	P	Fibre of petiole	0.875	0.27
9	Neem (*Azadirecta indica*)	P	Leaves	0.15	0.025
10	Nylon threads	M	Fibre	0.67	–
11	Carrot grass (*Parthenium hysterophorus*)	P	Twigs	0.75	0.15
12	Drumstick (*Moringa oleifera*)	P	Bark	–	0.1
13	Leaves (Unidentified)	P	Leaves	0.65	0.1
14	Tamarind (*Tamarindus indica*)	P	Leaves	0.17	0.087
15	Lawn grass (*Zoysia japonica*)	P	Leaves	0.65	0.075
16	Coconut (*Cocas nucifera*)	P	Leaf fibre	0.45	0.075
17	Bamboo(*Bambusa vulgaris*)	P	Leaves	0.65	0.02
18	Jujube (*Zizipusjujube*)	P	Thorns	–	0.01

Lining and core of nest					
19	Jute (*Corchorus olitorius*)	P	Fibre	14.475	7.63
20	Synthetic fibre	M	Fibre	2.05	2.5
21	Polythene	M	Pieces	0.85	1.31
22	Cotton (*Gossypium herbaceum*)	P	Cotton fibres	0.775	0.825
24	Cloth pieces	M	Pieces	0.625	0.375
25	Domestic fowl (*Gallus gallus*)	A	Feathers	1.67	1.87
26	Silk cotton (*Bombax cieba*)	P	Fibre	0.23	0.051
27	Threads	M	Small pieces	0.64	0.025
28	Paper	M	Small pieces	0.57	0.175
29	House Sparrow (*Passer domesticus*)	A	Feathers	–	0.01
30	Cigarette buds	M	Fibre	0.101	0.01

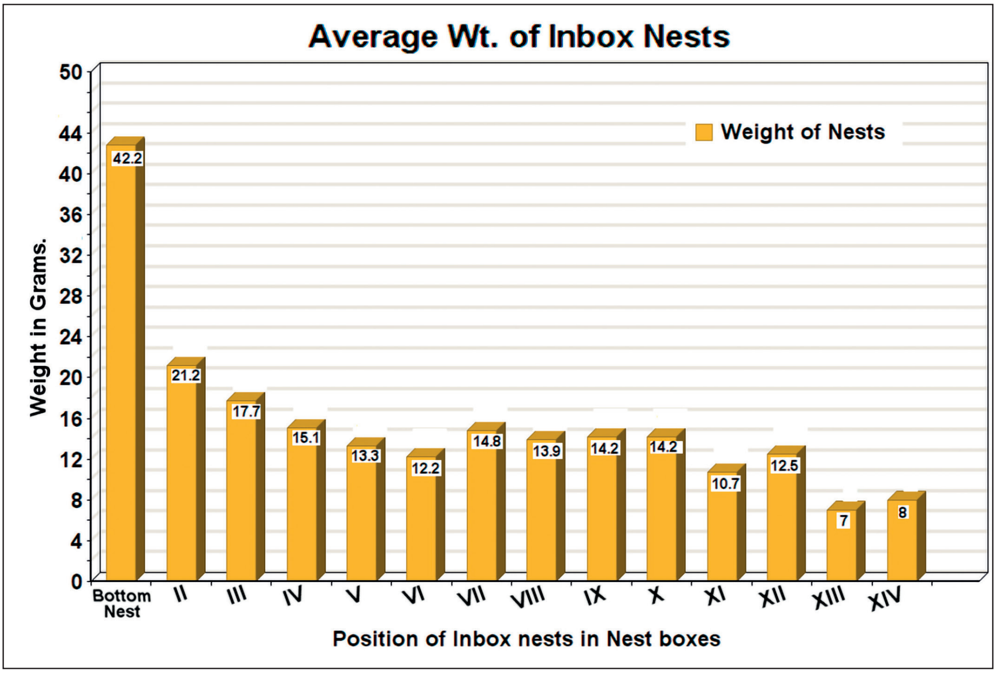


Figure 3. The average weight of Inbox nests in nest boxes
3. ábra A fészekoduba rakott fészkek átlagos tömege

nests. These 11 structural materials were mainly used for construction of outermost part of the nest i.e. at base level to build a basic platform. In case of Open nests, Sparrows need more of structural materials to make this outermost platform to fill surrounding gaps where they make the nests. For the other four structural materials (coconut fibre, broom fibre, tamarind leaves *(Tamarindus indica)* and neem leaves *(Azadirecta indica)* leaves), H_0 was accepted indicating that there is no significant difference between the two types of nests with regard to the quantum of these four structural materials. Among these four structural materials, Sparrows use coconut fibre and broom fibre to form the inner rim of the basic platform and hence there is no much difference in their quantum between the two types of nests. Whereas neem leaves and tamarind leaves were used as outer structural materials and were rarely found in few of the nests *(Table 3a)*.

Table 2. Number of nests in nest boxes (layer wise)
2. táblázat A fészkek száma a fészekodúkban

Nest location in the Nest box from bottom to top	First Nest / Bottom Nest	Second	Third	Fourth	Fifth	Sixth	Seventh	Eighth	Ninth	Tenth	Eleventh	Twelfth	Thirteenth	Fourteenth
No. of examined nest boxes that contained the specified nests	100	98	95	90	75	63	50	40	17	13	8	4	3	1

Table 3. T-test results of comparative analysis of quantum of each nesting material type (a), weight of nests (b) and construction time (c) between Open and Inbox nests

3. táblázat Az egyes fészkelőanyag-típusok mennyiségére (a), a fészkek tömegére (b) és az építési időre (c) vonatkoztatott összehasonlító elemzés (T-teszt) eredményei a két fészektípus között

(a) Analysis of Structural Materials

S. No.	**Material**	**Mean**		**Variance**		**Obs.**	**H.M.**	**df**	**t-Stat**	**H_0**
		Open Nests	**Inbox Nests**	**Open Nests**	**Inbox Nests**					
1	Dūrvā grass	40.4	15.388	197.990	79.122	40	0	39	9.3576	Rejected
2	Goose grass	4.025	0.825	14.794	3.840	40	0	39	4.7554	Rejected
3	Paddy grass	10.55	0.975	251.638	1.102	40	0	39	3.8933	Rejected
4	Carrot grass	0.75	0.150	1.423	0.490	40	0	39	2.8006	Rejected
5	Leaves (unidentified)	0.65	0.100	1.874	-1.231	40	0	39	2.5619	Rejected
6	Coconut leaves	0.45	0.075	0.450	0.075	40	0	39	2.7328	Rejected
7	Lawn	0.65	0.075	3.926	0.225	40	0	39	1.9936	Rejected
8	Cucumber stem	2.15	1.113	5.874	2.019	40	0	39	2.2483	Rejected
9	Bamboo leaves	0.65	0.025	5.413	2.459	40	0	39	1.6899	Rejected
10	Palmyra fibre	0.875	0.275	3.599	0.615	40	0	39	1.7883	Rejected
11	Neem leaves	0.15	0.025	0.285	0.025	40	0	39	1.4035	Accepted
12	Small twigs	1.2	0.425	1.497	5.635	40	0	39	1.7247	Rejected
13	Tamarind leaves	0.175	0.088	0.302	0.306	40	0	39	1.0447	Accepted
14	Coconut fibre	3.2	2.788	8.677	7.217	40	0	39	0.6536	Accepted
15	Broom fibre	5.5	5.038	13.897	12.441	40	0	39	0.5261	Accepted

*tested at 5% level of significance

(b) Analysis of Lining Materials

S. No.	**Material**	**Mean**		**Variance**		**Obs.**	**H.M.**	**df**	**t-Stat**	**H_0**
		Open Nests	**Inbox Nests**	**Open Nests**	**Inbox Nests**					
1	Jute fibre	14.475	7.638	37.846	33.538	40	0	39	5.2678	Rejected
2	Threads	0.641	0.026	0.657	0.026	40	0	39	4.9152	Rejected
3	Paper pieces	0.575	0.025	2.199	0.025	40	0	39	1.1380	Accepted
4	Cloth pieces	1.025	0.225	4.128	0.281	40	0	39	1.4034	Accepted
5	Polythene pieces	0.850	1.640	0.797	7.494	40	0	39	-1.7311	Accepted
6	Silk cotton	0.231	0.051	0.445	0.103	40	0	39	1.4818	Accepted
7	Cigarette buds	0.102	0.076	0.147	0.072	40	0	39	0.3295	Accepted
8	Synthetic fibre	2.050	2.563	2.459	4.490	40	0	39	-1.2313	Accepted
9	Cotton	0.825	1.188	1.379	2.317	40	0	39	-0.2169	Accepted
10	Feathers	1.675	1.875	3.097	2.894	40	0	39	-0.4554	Accepted

*tested at 5% level of significance

(c) Analysis of Weight of the Nest and Construction Time

Parameter considered	Mean		Variance		Obs.	H.M.	df	t-Stat	H_0
	Open Nests	Inbox Nests	Open Nests	Inbox Nests					
Weight of the nest	94.07	42.203	497.97	160.99	40	0	39	11.03	Rejected
Construction Time	22.57	14.75	8.096	1.423	40	0	39	15.943	Rejected

*tested at 5% level of significance

It is also evident from the results that the usage of dūrvā grass in Open nests was significantly more compared to their usage in Inbox nests ($P < 0.001$). The quantum of paddy grass was also more in Open nests with 11.21% of the total weight, which is significantly more in Open nests than Inbox nests ($P \leq 0.001$). The cucumber stem was widely used in both Open and Inbox nests, almost in equal number of nests (occurred in around 70% of the nets). But the quantum of cucumber stems was significantly more in Open nests ($P = 0.015$).

Among the tested ten lining materials, H_0 was accepted for eight (feathers, cloth pieces, synthetic fibre, cotton, silk cotton, paper pieces, cigarette buds and polythene pieces), indicating that there is no significant difference in the quantum of these lining materials between the two types of nests. This is because of the uniform size of the cup of this bird. Lining materials such as jute fibre and threads rejected null hypothesis, indicating that quantum of these materials between the two types of nests is significantly different. The usage of jute fibre was significantly more in Open nests, which cover 15.41% of the total weight of the nests. It could be due to the uneven space that is left after construction of basic platform (outermost and inner rim) using structural materials *(Table 3b)*.

To comparison, the jute fibre was occurred in all Open nets and its usage was found in 87% of the Inbox nests, with a quantity of 15.41% of the total weight in Open nests and 18% in Inbox nests. Feathers were also found to be predominant lining materials found in 77% of the both types of nests. But the quantum of feathers was significantly more in Inbox nests with 4.43% of the total weight. The usage of synthetic fibre and polythene pieces was more in Inbox nests than Open nests.

With regard to the type of materials (both structural and lining) that were used for construction of nests by House Sparrows, there is no much difference between the two types of nests.

From the results of statistical analysis, it is clear that the weight of the nests is more for Open nests and the analysis rejected the null hypothesis with a significant P value ($P < 0.001$). Hence from the test results, it is concluded that the weight of nesting materials used by House Sparrow for construction of Open nests and Inbox nests significantly differ *(Table 3c)*.

From the results, it is clear that Open nests takes more time for their construction compared to Inbox nests, and the analysis rejected the null hypothesis with a significant P value ($P < 0.001$). Hence, from the test results, it is concluded that the construction time taken by the House Sparrow for construction of Open nests and Inbox nests significantly differ, and it is more for Open nests *(Table 3c)*.

Discussion

As per the observations by Fitzgerald and Martin (2005), House Sparrow exhibit behavioral flexibility. They utilize any kind of nesting materials for their nest building, which are readily available in their surroundings. Our current study also revealed the same flexible behavior. For construction of nests, Sparrows collected mostly green material. The observational studies of Dubiec *et al*. (2013) revealed that green materials in the avian nests decrease the abundance of parasites and pathogens. In addition to this, the volatile compounds of green materials also stimulate the immune system of nestlings.

The developed new generations by our nest box installations, utilized 29 different varieties of nesting materials. The uncommon materials like the peels of moringa stem *(Moringa oleifera)* were also found in the nests. Usage of paddy grass was found only in few nests as there is no paddy cultivation in the study area. Whatever the paddy grass that is present in the nests may be obtained from the cattle fodder.

The comparative analysis of the quantum of each of the nesting material type (both structural and lining materials) used by House Sparrow for constructing Open and Inbox nests revealed a significant difference with regard to quantum of each of the structural material type that is used by the Sparrows to construct both the nests. This could be due to the fact that Sparrows need more of the structural material to build a safer outer base in case of Open nests, and comparatively lower quantity of each of the structural material is sufficient to build Inbox nests due to availability of basic structure.

When construction comes to the core of nest with lining materials, our observations state that the quantum of each of the lining material that is used remains almost same. This is because each of the species make cup according to its need and its size. Hence, there is no much variation in the quantity of each of the lining material that is used between Open and Inbox nests. In Inbox nests, synthetic fibre was found to be more compared to Open nests and this could be attributed to the availability of materials during construction time. But coming to the type of materials, almost similar kind of structural and lining materials were observed in both the nests.

On comparison, the Open nests needed two times more structural materials than the Inbox nests. Utilization of dūrvā grass was 7% more in Open nests. The paddy grass also found 9% more in Open nests than Inbox nests. This could be due to the fact that during initial phase of construction of Open nests, structural materials use to fell down and several attempts are needed to form the base which necessitates the use of more dūrvā grass and paddy grass. This comparison also indicates that they need double time period to build the Open nests with more effort. Utilization of goose grass was found in 10% of Inbox nests, where in Open nests it was used around 60% of nests. Bamboo leaves were found in only two Open nests where the bamboo plants are present. The structural materials like palmyra fibre *(Borrassus flabillifer)*, lawn, coconut leaves and lining materials like cotton (*Gossipium* sp.), cotton threads, jute threads constituted only 2% of the volume of Open nests.

Usage and composition of nesting materials depends on the availability of nesting materials in that location (Wimberger 1984). The same was reflected in our observations, at nest box no.339, most of the constructive material was peels of moringa stem. In two of the nest boxes, fowl feathers were the major nesting materials, where the rearing of domestic fowl was noticed. At the nest box no.90, sparrow feathers (wing feathers primaries and secondaries) were found in the core of the nest. Usage of anthropogenic materials was also noticed and they may also benefit the nestlings. As per Suárez-Rodtguez *et al*. (2013), the presence of cigarette butts in the sparrow nests prevented the pests, it could be short term but increased proportion of cigarette butt in nests along with the cellulose also increases genotoxicity in the blood cells of the nestlings (Suárez-Rodtguez & Garcia 2014). In our studies, we found butts of cigarettes only in two nests, but no mortality was recorded. The presence of anthropogenic materials like nylon fibre caused the death of two fledglings in one Open nest during our observational studies. The fibres coiled irregularly around the legs of the fledglings and that detained them in the nest.

According to Dhanya *et al*. (2016), the presence of animal matter like fowl feathers and human hair (in the core of the nest) may increase the presence of parasites and this could be a potential menace for the tolerance of nestlings and the procreation capacity of the parents. Lombardo *et al*. (1995) experimentally proved the importance of feather lining in the nests and its benefits to birds. Feather lining benefits the birds in two ways, as insulators and also act as a barrier between nest parasites and nestlings. Nest insulation affects the brooding performance. Lombardo *et al*. (1995) in their experimental studies on Tree Swallows *(Tachycineta bicolor)* proved that the nests without feather lining delayed the incubation and hence resulted in production of fewer fledglings than the feather lining nests.

In our studies, feathers were found in 84% of the Inbox nests and 80% of the Open nests. The fledgling success was recorded 83% in the Inbox nests. Since remote time, most of the birds used feathers to construct the core of the nests because of their thermos resistant property. The dry protein substances like feathers and hairs after shedding from the body never become a habitat for parasites, unless they expose to water continuously. In our studies, as most of the nest boxes were installed under the roof, there is no chance of wetting of the nests due to rains and other reasons. As per the observations of Blue Tits *(Cyanistes caeruleus)* nests by Hanmer *et al.* (2017), increase in the anthropogenic material in the nests decreases the arthropods and also affects the breeding success. As per our observational studies, there was no impact of usage of anthropogenic materials in the abundance of arthropods and the breeding success.

Conclusion

In conclusion, Open nests need more of structural materials than the Inbox nests. This also indicates that sparrows need more time and effort to build Open nests compared to Inbox nests. The total amount of materials used for the construction of first nest in Inbox nests is less than half of the Open nest material. In other words, Sparrows need to collect more than double the quantity of nesting material for the construction of Open nests. This also requires more effort and time. There is 10 to 12 days delay in egg laying from the day of start of nest construction in Open nests, than the Inbox nests and it may impact successive breeding attempts of the pair of the Sparrows.

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Substrate influences foraging selection by Eurasian Green Woodpeckers *Picus viridis* in autumn and winter: observations in Hungary over a 20-year period

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Abstract The foraging behaviour and habitat use of the Eurasian Green Woodpecker *Picus viridis* at various sites in Hungary over a 20-year period was documented. Detailed observations were recorded on foraging behaviour at hard substrates; in quarries, cliffs and human made structures of brick and stone construction. Using Chi-square tests on the frequency of observations of birds at hard substrates foraging sites, we compared usage during periods of snow cover and those without. Birds were found to be more frequently observed at hard substrates during periods of snow cover because these remained largely free of snow. We supposed that this response was due to invertebrate prey becoming increasingly scarce generally across typical foraging sites, i.e. grasslands and meadows during harsh winter conditions. Accessibility to the alternative sites became important as a source of food because availability of prey was more reliable. Vertical surfaces of hard substrates such as those associated with quarries, cliffs and buildings may be important to sustain Eurasian Green Woodpeckers in periods of snow cover where these provide a valuable foraging resource.

Keywords: *Picus viridis*, woodpecker, foraging, terrestrial-feeding, snow-cover, substrate, ants

Összefoglalás A zöld küllő *(Picus viridis)* táplálékkeresési szokásait és élőhelyhasználatát vizsgáltuk Magyarország különböző területein 20 éven keresztül, 2002–2021 között. Részletes megfigyeléseket végeztünk a madarak kemény felületeken, kőbányákban, sziklákon, illetve ember által épített tégla- és kőépítményeken történő táplálékkeresése során. Ezeken a táplálkozóhelyeken a madarak megfigyelésének gyakoriságát Khi-négyzet tesztek segítségével hasonlítottuk össze hótakaráskor és a hótakarás nélküli időszakokban. Hóborításkor a madarakat gyakrabban figyeltük meg ezeken a kemény felszíneken, mivel ezek nagyrészt hómentesek maradtak. Feltételezésünk szerint ez annak tudható be, hogy zord időjárási körülmények között a gerinctelen zsákmányállatok egyre ritkábban fordulnak elő az egyébként jellemző táplálkozási helyeken, mint a füves területek vagy rétek. Az alternatív helyszíneken biztosabbá vált a zsákmány elérhetősége, ezért fontos táplálékbázist jelentettek a madarak számára. Mindezekből arra következtetünk, hogy hóborítottság esetén a kemény aljzatok – például kőbányák, sziklák és épületek – függőleges felületei fontosak lehetnek a zöld küllők túléléséhez, mert értékes táplálékforrást szolgáltatnak.

Kulcsszavak: zöld küllő, harkályok, táplálkozás, hóborítottság, aljzat/felület, hangyák

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Introduction

The global range of Eurasian Green Woodpecker *Picus viridis* falls almost entirely within the Western Palearctic region. The species occurs from Britain in the west to Russia and Turkmenistan in the east, and from Norway and Sweden in the north to Italy and the Balkans in the south, and in the southeast to Iran and Iraq. It is absent from Finland and islands such as Corsica, Sardinia, Malta, Crete, Cyprus and Ireland. The Eurasian Green Woodpecker is polytypic, with three subspecies recognised: *viridis* in Britain, Scandinavia, continental Europe (including Hungary) and western Russia, *karelini* in Italy, the southern Balkans, the Caucasus and east to Turkmenistan, and *innominatus* in the Zagros Mountains of Iran and Iraq (Gorman 2014, Winkler & Christie 2014). It is replaced in Spain and Portugal by Iberian Green Woodpecker *Picus sharpei,* a species which was previously regarded as a subspecies of *P. viridis* (Perktas *et al.* 2011, Pons *et al.* 2011).

Around 95% of its total population is considered to occur in Europe (BirdLife International 2021), where the species is rather uniformly distributed, occurring in both lowlands and uplands, to around 2,300 m (Wilk 2020). Birds are typically sedentary, although post-breeding dispersal by juveniles is common (Glutz von Blotzheim & Bauer 1980, Gorman 2004, Török 2009).

The species ranges across three eco-climatic regions: temperate, Mediterranean and boreal. Wooded pastures, parks, groves, woodlands with clear-cut areas, open plantations with glades or grassy patches, orchards, gardens, sports-fields, paddocks, farmland and floodplain woods with grassy dykes are all inhabited. Open broadleaved and broadleaf-conifer woodlands are favoured over coniferous stands. Drier ground with a sunny aspect is preferred for foraging over shady and damp areas because it generally harbours abundant ants (Alder & Marsden 2010).

In many countries, Eurasian Green Woodpecker has become increasingly synanthropic, even expanding into urban areas. The overall trends are positive with the European population found to have moderately increased in recent decades (Wilk 2020). In the IUCN Red List of Threatened Species, it is classified as Least Concern (BirdLife International 2021). In Hungary, where all of the observations documented here took place, the species occurs nationwide and is locally fairly common, with 15,000–17,000 breeding pairs currently estimated (Gorman *et al.* 2021).

As is characteristic for species in the *Picus* genus, the staple diet of Eurasian Green Woodpecker is soil-, ground- and mound-living ants (Glutz von Blotzheim & Bauer 1980, Cramp 1985). Ants from the genera *Lasius*, *Formica* and *Myrmica* are mainly consumed and are taken in all stages (egg, larva, pupa, adult) and collected directly from the ground surface or dug from their colonies (Blume 1996, Raqué & Ruge 1999). Most food is sought on the ground with short-grazed and mown grasslands preferred (Alder & Marsden 2010). Indeed, though other sympatric woodpeckers will, to varying degrees, forage terrestrially, Eurasian Green Woodpecker has become a specialist of grassland habitats. The species, however, also forages in trees and bushes and searches for invertebrates in cowpats, fungi, spider-webs and leaf-litter. Soil-dwelling invertebrates such as earthworms are taken as well as the larvae of beetles and caterpillars, spiders and wasp and bee grubs (Glutz von Blotzheim & Bauer 1980). Nevertheless, like many woodpeckers, this species is opportunistic and resourceful when circumstances dictate and vegetable matter, such as windfall fruit, nuts and berries, are also occasionally consumed (Glutz von Blotzheim & Bauer 1980, Gorman 2004).

In the autumn and winter, Eurasian Green Woodpecker is reported to be sensitive to prolonged periods of snow cover affecting accessibility to food (De Bruyn *et al*. 1972, Glutz von Blotzheim & Bauer 1980). The importance of access to a reliable source of food is critical for a species which is vulnerable to harsh winter weather (Rolstad *et al*. 2000). With 20 years of observational records of Eurasian Green Woodpeckers utilising hard surfaces in natural and human sites (quarries, cliffs and buildings), the aim of this study was to elucidate any patterns in foraging behaviour during the autumn-winter period. We wanted to identify whether Eurasian Green Woodpeckers showed any strong selection for foraging substrate by accounting for differences in snow cover.

Methods

Study area: The observations documented here by one of the authors (GG) took place across Hungary over a twenty-year period (2002–2021). A total of 58 observations were made of individuals visiting quarry walls, cliffs and buildings: 30 in stone quarries (both abandoned and active), 17 on buildings (walls and rooftops of houses, apartment blocks, hotels, abandoned military base) and 11 on limestone and other cliffs. All involved single adult birds: 31 males and 27 females. Some of the locations were situated within or by woodlands, but others were in quite open areas in villages and suburban areas.

Study period: All observations were made in Hungary, in autumn and winter, from September to March, as follows: September (1), October (1), November (10), December (12), January (15), February (13), March (6). On 39 of the 58 occasions snow of various depths covered the ground in the surrounding area. Although locations were visited by the author at all times of day, the vast majority of observations of foraging woodpeckers were in the afternoon hours (54 from 58).

Analysis: Using the frequency of observations Chi-square tests were used to 1) explore whether there were differences between male and female Eurasian Green Woodpeckers, using cross tabulation to perform a test for association. Following which 2) a Goodness of fit test for homogeneity was undertaken to compare the frequency of observations of birds at hard surfaces when snow was present and absent compared to an expected theoretical frequency. Because the cross tabulation is for two categories in each of these tests, there is a risk of rejecting the null hypothesis when it is in fact true, therefore Yates's correction was applied to adjust for this (Fowler & Cohen 1996).

Table 1. Observations of *Picus viridis* foraging on quarry walls, cliffs and buildings over a 20-year period (2002–2021) in Hungary (A – male, B – female)

1. táblázat A zöld küllők kőfejtő-falakon, sziklákon és épületeken való táplálkozásának megfigyelései egy 20 éves periódus alatt (2002–2021) Magyarországon (A – hím, B – tojó)

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Location	Foraging habitat	Month/Year	Time (approx.)	Sex	Ground snow-cover
Bükk Hills	quarry	Jan 2002	12.30	Male	Yes
Tatabánya	quarry	Dec 2002	17.00	Male	Yes
Zemplén Hills	quarry	Nov 2003	14.00	Male	No
Gerecse Hills	cliff	Dec 2003	13.00	Male	Yes
Vértes Hills	cliff	Mar 2004	15.00	Male	No
Mátra Hills	quarry	Jan 2005	12.30	Male	Yes
Gerecse Hills	cliff	Dec 2005	14.00	Male	Yes
Visegrád Hills	building	Oct 2006	10.30	Male	No
Bükk Hills	quarry	Nov 2006	13.00	Male	No
Vértes Hills	quarry	Feb 2008	11.00	Male	Yes
Bükk Hills	quarry	Nov 2008	14.30	Male	No
Budaörs	cliff	Dec 2008	15.00	Male	Yes
Pilis Hills	quarry	Jan 2009	14.00	Male	Yes
Tata	building	Jan 2010	11.00	Male	Yes
Tata	building	Feb 2011	14.00	Male	Yes
Bükk Hills	quarry	Mar 2011	15.00	Male	No
Pilis Hills	quarry	Nov 2011	15.00	Male	No
Visegrád Hills	quarry	Jan 2012	14.30	Male	Yes
Zemplén Hills	quarry	Nov 2012	16.00	Male	No
Aggtelek	cliff	Jan 2013	15.00	Male	Yes
Budapest	building	Dec 2013	12.30	Male	Yes
Tokaj	quarry	Mar 2014	16.00	Male	No
Visegrád Hills	building	Nov 2015	16.30	Male	No
Tatabánya	quarry	Jan 2016	14.00	Male	Yes
Aggtelek	quarry	Dec 2016	15.00	Male	Yes
Zemplén Hills	building	Feb 2017	12.30	Male	Yes
Pilis Hills	quarry	Dec 2017	15.00	Male	Yes
Tatabánya	quarry	Feb 2018	12.30	Male	Yes
Buda Hills	quarry	Feb 2018	15.00	Male	Yes
Bükk Hills	quarry	Jan 2019	11.30	Male	Yes
Vértes Hills	cliff	Mar 2021	16.00	Male	No

Ⓑ

Location	Foraging habitat	Month/Year	Time (approx.)	Sex	Ground snow-cover
Aggtelek	building	Feb 2002	15.00	Female	Yes
Pilis Hills	building	Feb 2003	13.30	Female	No
Budapest	building	Jan 2004	14.30	Female	Yes
Börzsöny Hills	quarry	Dec 2004	13.00	Female	Yes
Tokaj	quarry	Jan 2006	16.00	Female	Yes
Fertőd	building	Feb 2007	12.30	Female	Yes
Visegrád Hills	building	Nov 2007	15.00	Female	No
Tatabánya	quarry	Mar 2009	13.00	Female	No
Pilis Hills	cliff	Nov 2009	13.30	Female	No
Gödöllő	building	Feb 2010	12.30	Female	Yes
Börzsöny Hills	quarry	Sept 2010	15.00	Female	No
Budapest	building	Dec 2011	13.00	Female	Yes
Gerecse Hills	cliff	Feb 2012	17.00	Female	Yes
Mátra Hills	quarry	Feb 2013	13.00	Female	Yes
Gerecse Hills	cliff	Jan 2014	14.00	Female	Yes
Bükk Hills	building	Nov 2014	15.30	Female	No
Börzsöny Hills	quarry	Feb 2015	16.00	Female	Yes
Balf	quarry	Dec 2015	14.00	Female	Yes
Tata	building	Jan 2016	12.30	Female	Yes
Börzsöny Hills	quarry	Jan 2017	13.00	Female	Yes
Zemplén Hills	quarry	Mar 2017	13.00	Female	No
Visegrád Hills	cliff	Jan 2018	14.30	Female	Yes
Tapolca	building	Dec 2018	13.00	Female	Yes
Pilis Hills	cliff	Jan 2019	14.00	Female	Yes
Pilis Hills	quarry	Dec 2019	13.00	Female	Yes
Visegrád Hills	quarry	Jan 2021	14.00	Female	Yes
Pilis Hills	building	Feb 2021	13.30	Female	No

Results

Differences between male and female observations were examined by comparing their respective frequencies at hard surfaces in the winter months *(Figure 1)*. There was a positive association regardless of snow-cover and therefore no significant difference between the frequencies of observations for male and female Eurasian Green Woodpeckers ($X^2 = 0.04$,

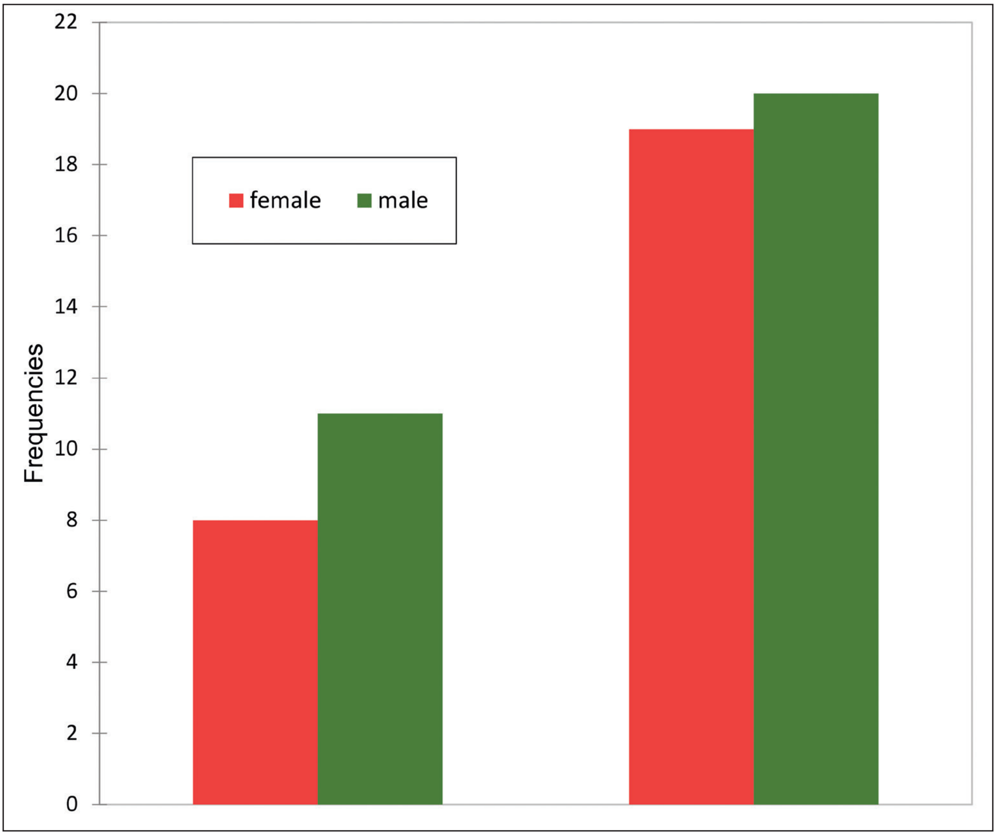

Figure 1. Frequencies of observations of male and female Eurasian Green Woodpeckers at hard surface substrates during periods without (left) and with (right) snow-cover

1. ábra A hím és tojó zöld küllő egyedek megfigyelési gyakorisága kemény felületeken hótakaró nélküli (bal) és hótakarás alatti (jobb) időszakokban

df = 1 P = 0.85, n = 58). Therefore, the data for male and female woodpeckers was pooled. We found a significantly higher frequency of observations at hard surfaces when snow was present ($X^2 = 6.22$, df = 1, $P < 0.01$, n = 58). Using a Goodness of Fit test (Fowler & Cohen 1996) it was concluded that there were significantly more birds foraging at hard surfaces during snow-cover.

Discussion

On quarry and cliff walls birds usually worked on certain spots, often at cracks and crevices, and did not move around actively as, for example, Grey-headed Woodpeckers have been observed to do when visiting such sites (Gorman 2020). When not disturbed, the birds tended to forage for between 30–50 minutes which is not dissimilar to foraging duration at other times of the year in other typical habitat as witnessed in a radio-tracking study in southern

UK (D. Alder own observation). On buildings, birds explored vertical walls, but also in rain gutters, beneath eaves and on one occasion the thatched roof of a cottage. Hosking (2011) reported a similar observation of bird in Suffolk, England, that regularly visited his newly re-thatched house in the autumn.Ant-based diet is influenced by season (Rolstad *et al.* 2000). Severe winters, with low temperatures and deep snow, affect the terrestrial ants that Eurasian Green Woodpeckers predominantly feed upon. For example, in Britain the hard winter of 1962/63 is thought to have severely impacted this woodpecker, with declines reported from many localities (Dobinson & Richards 1964). After another severe winter in Britain in 1981–82, local declines were again noted, some lasting several years (Glue 1993). Although terrestrial ants form the staple diet of this species, when these are not available other more accessible prey is sought and a shift in foraging areas and diet occurs. Such seasonal shifts are not unusual in the *Picus* genus. It has been observed that Grey-headed Woodpecker *Picus canus* also changes foraging locations, and hence prey sought, when deep snow or frozen ground impedes feeding (Rolstad & Rolstad 1995, Gorman 2020). In Sweden, Edenius *et al.* (1999) found that in periods with heavy snow, the same species moved from open-ground foraging to feeding on carpenter ant *(Camponotus)* colonies in tree trunks and stumps above ground level. In Japan, Matsuoka and Kojima (1979) found that in winters with deep snow the proportion of the Black Ant *Lasius niger*, which is a major food resource in spring and summer, was almost absent in the diet of Grey-headed Woodpeckers.

Rolstad *et al*. (2000) found that Eurasian Green Woodpecker was able to use Formica nests beneath tree cover in winter which was accessible because the trees afforded some protection from heavy snow. Being large nest mounds, these were more accessible than other sources in exposed open habitats e.g. meadows. Similarly, such nests are perennial and can be used for many years by the ants and thus, are a predictable source of food for the birds. It may be that hard surfaces as we have found which remain clear of snow can help to sustain at least some of the winter foraging requirements for Eurasian Green Woodpeckers. Clearly, more work is required to fully understand this behaviour particularly by identifying the invertebrates being taken and also looking at the microclimate conditions in such areas compared to other known foraging sites and controls.

It is likely that Eurasian Green Woodpeckers forage on rock walls and buildings in winter when their favoured ant prey become dormant and difficult to retrieve from hard or snow-covered ground (De Bruyn *et al.* 1972). As probing, gleaning and licking are the primary feeding techniques of Eurasian Green Woodpeckers, deep snow and frozen ground inhibits them from accessing terrestrial ant colonies that have moved far below ground. Consequently, the shift to sites above ground level, such as vertical walls and places where the temperature is above freezing such as buildings, to feed on the invertebrates that are found there occurs. The shift to foraging on rocky walls and buildings appears to be an adaptive response to seasonal variation in prey availability.

Foraging of this kind by this species has been occasionally discussed, mainly in the German literature, for example by Baier (1973), Löhrl (1977), Glutz von Blotzheim and Bauer (1980) and Blume (1996). This behaviour has, however, sometimes been interpreted and presented as rather uncommon. Anecdotal reports often refer to it as being unusual.

Foraging on buildings may be more frequent than reported, as the skittishness of this species probably means that birds immediately flee from such sites when people approach.

Eurasian Green Woodpeckers foraging on different non-terrestrial substrates when weather conditions prompt them to do so, is probably a regular shift, at least for populations in areas where winters are harsh. The fact that most foraging visits documented here occurred in afternoon hours also seems to indicate that warmth by the sun is important as invertebrates are more active and accessible at such times.

We suggest that the shifts in the foraging behaviour of Eurasian Green Woodpeckers described here, take place frequently and are not at all unusual.

These woodpeckers can and will dig through snow using their large, stout bill. Funnel-shaped, tunnel-like holes, 5–10 cm in diameter and of varying depth (up to 60 cm but sometimes more) are often bored into ant mounds and through snow to reach prey (Gorman 2015). But when low temperatures and hard frost result in ants retreating deep below ground or into their mounds, they become inaccessible and other food resources are then sought. At such times the availability of food at alternative sites away from the ground can be vital to this woodpecker when foraging for the terrestrial ants that form its staple diet becomes difficult.

Acknowledgements

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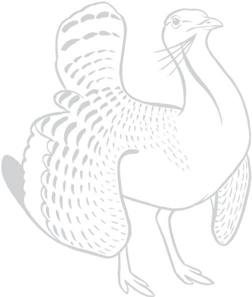

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Multi-species settlement by secondary hollow-nesting passerine birds in a European Bee-eater *(Merops apiaster)* colony

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Abstract Simultaneous nesting of six species of secondary hollow-nesting passerine birds in abandoned European Bee-eater *Merops apiaster* nest-holes has been detected and described. The holes were occupied by Great Tit *Parus major*, Spotted Flycatcher *Muscicapa striata*, European Pied Flycatcher *Ficedula hypoleuca*, Black Redstart *Phoenicurus ochruros*, Eurasian Tree Sparrow *Passer montanus* and White Wagtail *Motacilla alba*, which formed a multi-species settlement in a European Bee-eater colony.

Keywords: *Merops apiaster, Parus major, Phoenicurus ochruros, Ficedula hypoleuca, Muscicapa striata, Passer montanus,* nest holes

Összefoglalás A gyurgyalag *(Merops apiaster)* régi odúiban hat másodlagosan üregben fészkelő madárfaj egyidejű fészkelését írták le: széncinege *(Parus major)*, szürke légykapó *(Muscicapa striata)*, kormos légykapó *(Ficedula hypoleuca)*, házi rozsdafarkú *(Phoenicurus ochruros)*, mezei veréb *(Passer montanus)* és barázdabillegető *(Motacilla alba)*, amelyek egy többfajos közösséget alkottak egy gyurgyalag telepen.

Kulcsszavak: széncinege, házi rozsdafarkú, kormos légykapó, szürke légykapó, mezei veréb, odúk

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Introduction

The European Bee-eater *Merops apiaster* is recognized as an effective ecosystem engineer (Casas-Criville & Valera 2005, Sekercioglu 2006, Gyurácz *et al*. 2013, Smalley *et al*. 2013), i.e. a species that creates, modifies, or maintains a habitat, creates changes in the state of biotic and abiotic conditions that directly or indirectly affect the availability of resources for other species (Jones *et al*. 1997, Wilby 2002). Namely, European Bee-eater nest holes are utilized by other bird species for nesting that are unable to dig nest holes in banks, andvertical walls of outcrops.

Birds build nests to keep and protect eggs and chicks, to provide optimal humidity and temperature conditions necessary for the development of embryos and chicks (Heneberg

& Simecek 2004, Deeming 2016, Kristofik *et al*. 2017). Nests of different species vary significantly, and the structure of the nest is species-specific (Biddle *et al*. 2018). However, several patterns can be detected (Nagy *et al*. 2019). Intraspecific variation of nests in terms of location, size, and composition of building material has been described (Britt & Deeming 2011, Briggs & Deeming 2016, Biddle *et al*. 2018).

Many species of birds use closed nests made by other animals such as the European Bee-eater. Therefore a mixed-species breeding association of birds is formed in European Bee-eater colonies. The use European Bee-eater nest-holes by Common Kestrel *Falco tinnunculus*, Little Owl *Athene noctua*, Common Swift *Apus apus*, Eurasian Wryneck *Jynx torquilla*, European Roller *Coracius garrulus*, Eurasian Hoopoe *Upupa epops*, Northern Wheatear *Oenanthe oenanthe*, Common Starling *Sturnus vulgaris*, Eurasian Tree Sparrow *Passer montanus*, and House Sparrow *Passer domesticus* has been previously recorded (Shupova 1999, Malovichko & Konstantinov 2000, Gyurácz *et al*. 2013, Urban *et al*. 2013). Birds find protection from predators and bad weather conditions in European Bee-eater burrows, (Gyurácz *et al*. 2013). Recently, the list of birds inhabiting Bee-eater holes has been expanded. It is described that Coal Tit *Periparus ater* (Klitin 1972), Eurasian Blue Tit *Cyanistes caeruleus* (Buchko & Skilskiy 1995), Great Tit *Parus major* (Belik 1999, Shupova & Chaplygina 2017), Collared Flycatcher *Ficedula albicollis* (Knysh & Savinskaya 2013) also nests in European Bee-eater burrows. These bird species might try to find secure nesting sites or reduce rivalry with other hole-nesters (von Haartman 1957).

The purpose of ouris study is to analyze the species composition of hollow-nesting passerine birds that inhabited burrows, forming a poly-species settlement in one of the colonies of European Bee-eater.

Materials and Methods

We found a multi-species nesting settlement of birds in the vertical walls of the main parent outcrops on the Khotiv village Kyiv region. It is located directly on the outskirts of the residential zone of a settlement, and was established in a European Bee-eater colony existing for several years. The territory is part of the Kyiv plateau, belongs to the north of the forest-steppe zone and is located at the northern border of the European Bee-eater nesting area in Ukraine (50°19’18.6”N 30°29’40.3”E). The climate is moderately continental with warm summers and mild winters. Recent studies have noted a warming trend in the region (Netsvetov *et al*. 2018). The average annual temperature is 7.5 °C, the average temperature in July is 19.8 °C. Average annual precipitation is 606 mm with a maximum in July (75 mm).

We found the nesting settlement on 05th on July 2018. Observations were carried out in 2019 (06 May; 12 June; 09 July), 2020 (04 May; 03 June; 02 July; 16 July) and 2021 (04 May; 15 June; 06 July; 22 July). The discovered burrows were mapped with a brief description of the biotope, the location of the nest; and photographic recording of observations were taken. The reproductive cycle stage (incubation, feeding of chicks) was established via visual observation and was based on observation, and were categorized the number of birds leaving the nest, arriving to the nest with food, and other characteristics of bird behavior.

Results

The multi-species colony is separated from the human settlement by a dirt road, bordered by a meadow and a forest. The nests are located in a steep wall of a natural ravine. The cliff is composed of soft rocks, there are layers of loess, loams, clays, sandstones. The surface of the ravine is a dry meadow presence dominance of bushgrass *Calamagrostis epigeios*, common agrimony *Agrimonia eupatoria*, narrow-leaved meadow-grass *Poa angustifolia* and with young European aspen *Populus tremula*, silver birch *Betula pendula*, Scots pine *Pinus sylvestris*, boxelder *Acer negundo*, European dewberry *Rubus caesius*, common dogwood *Cornus sanguinea* was recorded. The total length of the ravine is 143 m, the length of the wall free from vegetation is about 35 m, and the height is 7 m, in the left part it is vertically divided into two ledges. The European Bee-eater is the dominant species here, but thethe number of occupied nests was quitebreeding pairs is low during the study period. The number of occupied burrows were: nine burrows in 2018, five in 2019, six in 2020, four in 2021.

European Bee-eaters dig their nests choose those parts of the cliff for nesting that are not covered by vegetation. The destruction of the cliff wall after rains is often observed. Often European Bee-eaters dug burrows in the central and upper part of freshly collapsed wall. The height of burrows on all colonys in the vicinity of Kyiv is 90–580 cm (on average,

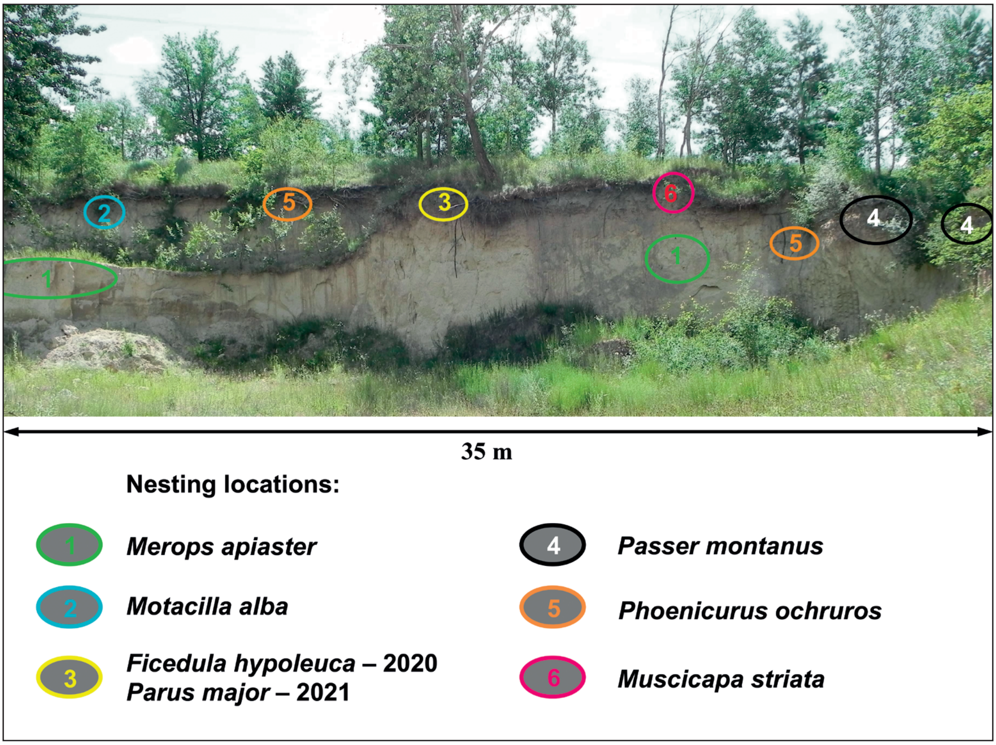

Figure 1. Distribution of species on nesting in burrows in a multi-species settlement
1. ábra A különböző fajok fészkeinek elhelyezkedése a fészektelepen

213.86±15.88; CV=48%; n=42). In this described colony, European Bee-eater burrows (n=27; 2018–2021) were located at an altitude of 90–400 cm (on average 200.1±8.81; CV=23%). The diameter of the European Bee-eater burrows is 5.5–6.5 cm (on average 6.08±0.12; CV=7%).

Seven species of secondary hollow-nesting birds settled in the burrows of the European Bee-eater, in different years: in 2018 – one pair of Common Starling and five pairs of Eurasian Tree Sparrow; in 2019 – one pair of Great Tit and four pairs of Eurasian Tree Sparrow.

The richest years regarding species composition of the colony were 2020 and 2021 *(Figure 1)*. On 16 July 2020, we observed here, simultaneously with nesting European Bee-eater, two pairs of Black Redstart *Phoenicurus ochruros*, one pair of European Pied Flycatcher *Ficedula hypoleuca*, one pair of Spotted Flycatcher *Muscicapa striata*, and four pairs of Eurasian Tree Sparrow. European Bee-eater, Eurasian Tree Sparrow, European Pied Flycatcher, Spotted Flycatcher, one pair of Black Redstart fed chicks in burrows. Adults of the second pair of Black Redstart and four juveniles were feeding together near the cliff. Black Redstart females periodically clashed with each other and with the flying male Barn Swallow *Hirundo rustica*.

Secondary hollow-nesters occupied old burrows at a height of 2 m or higher *(Table 1)*. Spotted Flycatcher, European Pied Flycatchers and Great Tit settled at the highest. All species chose burrows, next to which, not far from the entrance, there were roots or branches of woody plants. Birds used them to outlook posts, stopped on them before entering a burrow, to look out for insects, or for long duration observation. The diameter of the burrow entrance inhabited by passerine birds was also significantly larger than the diameter of the European Bee-eater burrows. This indicates that the secondary hollow-nesting birds (with the exception of Tree Sparrow) occupy very old burrows, partially destroyed and do not create competition for the European Bee-eater. The Tree Sparrow is less capricious, settles in different burrows.

In this ravine, the burrows occupied by Black Redstart are located at a distance of about 20 m, but they are not in the line of sight. This is prevented by the bend of the ravine and

Table 1. Parameters of burrows occupied by Passeriformes Note: H – distance from the nest to the top edge of the cliff; h – distance from the nest to the bottom edge of the cliff; D – diameter of the tunnel entrance

1. táblázat Az énekesmadár fajok által elfoglalt költőüregek paraméterei. H – a fészek és a fal felső szélének távolsága, h – a fészek és a fal alsó szélének távolsága, D – az üreg bejáratának átmérője

Species	**h (m)**	**H (m)**	**D (cm)**
Motacilla alba	2.0	0.1	7.5
Ficedula hypoleuca	5.0	0.4	11.0
Muscicapa striata	6.0	0.1	7.0
Phoenicurus ochruros	3.0–3.50	0.40–1.0	8.0–10.0
Parus major	5.0	0.40	11.0
Passer montanus	2.0–2.10–2.20–3.50	0.25–0.25–0.50–1.0	5.5–5.5–6.0–7.0

the growth of young European aspen on the ledge. In 2021, Black Redstart settled in the same burrows as in 2020; on 06 June 2021, both pairs fed chicks in the nests, the males periodically vocalised. On 22 July 2021, after heavy rainfall, the ravine wall collapsed in several places. The burrows in which the Black Redstart nested partially survived, but we did not observe any birds. The height of the burrows occupied by Black Redstart is 3–3.5 m from the base of the cliff, and the distance to the upper edge of the cliff was 0.4 to 1.0 m. One burrow was located on the upper ledge of the ravine and was hidden by European aspen, the second was located in the high part of the ravine, but was also covered with separate tree branches. Before visiting the burrow, birds landed on the branches of fallen or living tree.

Eurasian Tree Sparrows settled in an aggregated manner in the left part of the ravine, hidden by tree branches. Burrows here are almost destroyed; the height of the wall is about 1–1.5 m, in some places the wall of the cliff is overgrown with herbaceous vegetation. The number of breeding pairs decreased from five to one pair from 2018 to 2021. A possible reason for this may be that after the collapse of the ravine wall as a result of heavy rainfall, several European Bee-eater nests were destroyed, and they reconstructed two of the old burrows in the part of the ravine where the Eurasian Tree Sparrow nested. Eurasian Tree Sparrow were deprived of nest sites or nesting opportunities.

Previously suspected nesting of White Wagtail *Motacilla alba* was confirmed in 2021, when a Eurasian Sparrowhawk *Accipiter nisus* showed up at the colony. The first alarm signal was given by the European Bee-eater, reacting to which, all the birds united to expel the hawk. White Wagtail was found in the left part of the ravine above the ledge at a distance of less than 10 m from the Black Redstart nest.

Spotted Flycatcher and European Pied Flycatcher nested in cliff burrows only in 2020, in the upper part on the border with the soil layer under the hanging plant roots. In 2021, the nesting site of European Pied Flycatcher was taken by a Great Tit pair. These burrows are very old, washed out and deformed by soil erosion.

Discussion

European Bee-eaters used the same burrows for several years (Shupova 1999, Bastian *et al.* 2013). By re-using existing nesting burrows they might potentially save time and energy, and can allocate it layingto other activities during the laying period (Brust *et al.* 2015). 22–54% of European Bee-eaters have used the same cliff for nesting several times, and in 26% of the breedings the same burrows were chosen by the same pair (Brust *et al.* 2015). The use of hollows several times was also noted for hollow-nesters. It is also known that the same holes might be used consecutively by different bird species (Wesołowski 1989) As the cliff becomes overgrown and it becomes difficult to enter the burrow, European Bee-eaters leave old burrows and dig new ones. Urbán *et al.* (2013) believe that thickets of woody vegetation in front of burrows prevent birds from flying in and out of the burrow. The height of the European Bee-eater burrows is very variable and depends on the height of the cliffs in the region. For example, in the steppe zone of Ukraine, the height of the European Bee-eater burrows, on average 315.70 cm, and the distance to the upper edge of the cliff is 81.80 cm (Shupova 1999).

The Great Tit nests in hollows, artificial nests, crevices of various buildings, cracks in rocks, cavities under tree roots. It is suggested that the noted feature of the ecology of the genus *Parus* is associated with their evolution in mountainous landscapes. Then, settling on the plains, tits switched to a forest lifestyle, but retained a high plasticity of their nest-building instincts (Belik 2009). Nesting of Great Tit in European Bee-eater burrows has already become a common occurrence in the steppe (Belik 1999, Shupova & Chaplygina 2017).

In flat landscapes of Ukraine Black Redstart subspecies *Ph. o. gibraltariensis* behaves like a synanthrope and nests, as a rule, in human structures (Knysh 2013, Shupova 2014), old nests of Barn Swallows (Monteagudo 1999, Knysh 2013). There is a known case of joint nesting of Common House Martin *Delichon urbicum* and Black Redstart in the Common House Martin nest (Krynski 2017). In Tibet, birds of the subspecies *Ph. o. phoenicuroides* tend to nest in burrows created by other animals (Chen *et al.* 2011, Lu *et al.* 2011). Perhaps we are witnessing the next stage of adaptation of the *Ph. o. gibraltariensis* to the conditions of habitation of the secondary area, they start nesting outside the urbanized biotopes, where the resource of nesting sites is exhausted, and the naturalisation of the species in the forest-steppe zone of Ukraine in the absence of rocks. At the same time, the birds follow the skills of a related subspecies, nesting in the clay walls of cliffs.

It should be noted that earlier, in the biotopes of the residential zone, we already observed a conflict between Black Redstart females, and, most likely, this is normal for the species. In the residential area, we determined the closest location of the nests of this species – 100 m (Shupova 2018).

For 10 years, we have been recording the nesting of White Wagtail and Black Redstart under the roof of the same building in the territory of the Feofania Park and have never observed a conflict between them. White Wagtail is a species that for a long period has been characterized by nesting in cavities among bare tree roots in cliffs of river and lake banks, in grooves of sandy cliffs, in forest ravines recess, and in burrows of birds and rodents (Chaplygina 2014).

Muscicapidae species are characterised by high flexibility in the selection of nesting sites and use a wide range of nesting locations. The previously described Spotted Flycatcher positions are shallow niches: hollows, rotted tree and branch breaks, old woodpecker hollows, Common Blackbird and Barn Swallow nests, divarication of branches, niches in human structures, artificial nests (Stoate & Szczur 2006). European Pied Flycatcher nests mainly in tree holes and artificial nests. European Pied Flycatcher nests in clear tree hollows free of parasites (Wesołowski 2000). The lack of clean hollows may have forced the pair to settle in the old European Bee-eater burrow, which has not been used for a long time.

Eurasian Tree Sparrow is known to use the European Bee-eater burrows for nesting (Shupova 1999, Gyurácz *et al.* 2013, Stubbe *et al.* 2016). It is described that Eurasian Tree Sparrow often visit European Bee-eater burrows during their absence, clog them with stems and leaves of herbaceous vegetation, fluff and other bulky material, and make a spherical nest inside the burrows (Malovichko & Konstantinov 2000). European Bee-eaters are aggressive towards sparrows trying to settle in the colony, therefore Eurasian Tree Sparrows nest on the periphery of the colony in the old part left by Bee-eaters.

When building nests, birds select materials and conditions according to genetically established criteria (Briggs *et al.* 2019). At the same time, nest building is a plastic behavior. Individual experiences allow birds to build different nests, resulting in geographic differences in the used building material and where they are located (Briggs *et al*. 2019, Mainwaring *et al.* 2021). The lack of free nesting sites on the territory of tree plantations can also stimulate nesting in burrows of hollow nests. Wesołowski (1989) argued that competition for holes in natural forests is less intense than in plantations. It was also shown that the combined density of secondary hole-nesting birds did not decrease when hollows were blocked in an experiment (Waters *et al*. 1990). Birds can adapt to new conditions in a changing environment (Mänd *et al*. 2005). In landscapes with a shortage of woody plants, and, accordingly, hollows, nesting in burrows gives birds the opportunity to colonize new territories and may result in population number increase or even enlargement of the species range.

The number of breeding European Bee-eater pairs in the region is highly variable. From 2018 to 2020 it steadily decreased not only in the described colony, but also in the neighbouring one: three pairs in 2018 and in 2019, two pairs in 2020. In 2021, the number of nesting burrows of European Bee-eater has increased to four here. The weather conditions of the region are not optimal for the breeding of this species, often at the beginning of summer there are prolonged rains. It was shown that the existence of European Bee-eater populations depends on changes in weather and climatic conditions in the area of the colony, especially at the northern borders of the species range (Arbeiter *et al*. 2016). Long periods of rains with low temperatures might force the birds to start nesting several weeks later (Bastian *et al*. 2011). In the vicinity of Kyiv, European Bee-eaters begin nesting after the rains stop and a sufficient number of insects appear to provide an opportunity to feed their offspring. The destruction of the cliff wall as a result of landslides after rains is often observed, accompanied by the loss of clutches. For example in 2021, part of the ravine on the outskirts of the village collapsed along with nesting burrows. We associate the increase in the number of nesting birds in the neighbouring colony with the fact that couples migrated here, whose nests were destroyed in the first colony. The surface of the soil above this cliff has a slope towards the opposite wall of the cliff, so it is not washed out by rains.

Observations of the European Bee-eater nesting in the vicinity of Kyiv show that the noticed increase in the species range in the last 25 years (Essel *et al*. 2016, Stubbe *et al*. 2016, Cattaneo 2018, Yankevich *et al*. 2018, Londei 2020) does not lead to an increase in the number of birds in the forest-steppe zone, the conditions of which remain extreme for the European Bee-eater.

Thus, the European Bee-eater behavior acts as a pioneer species, reclaiming fresh cliffs in the outcrops of the main parent rocks; and as ecosystem engineer species: its old nests are the nucleus of the multi-species settlements formationstations for birds setting up closed nests. We think that the identified cases of Spotted Flycatchers, European Pied Flycatchers and

Great Tits breeding in burrows are indicate a tendency for dendrophils nesting in hollows, and they explore new nesting biotopes because of the lack of trees. This process is caused by a decrease in the area of tree plantations, an increase in urbanised areas, the use of new construction, communication technologies and the changing species composition of plants. Species with plasticity in nest-building behavior inhabit new biotopes and landscapes. The adaptation of secondary hollow-nesting birds to nesting in burrows facilitates their settling in arid regions, where there has always been a shortage of large trees with hollows. As a consequence, the development of new territories will lead to the expansion of the nesting areas of more adaptable species. For Black Redstart, Common Starling, House Sparrow that evolved in mountainous landscapes, the way of nesting in burrows is close to species-specific (in cracks of rocks). In terms of nesting and feeding style, the biotopes of cliffs are also more similar to rocks than to forest biotopes. Thus, the colonisation of burrows with these birds should be regarded as a logical choice by birds of not only species-specific nesting sites, but also habitats that are close to species-specific.

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Insect ectoparasites of the Red-backed Shrike *Lanius collurio* in the Iberian Peninsula

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Abstract Of the 26 Red-backed Shrike *(Lanius collurio)* sampled with the aim of characterizing insect ectoparasites, five birds were parasitized by louse flies. Two species were identified, *Ornithophila metallica* and *Ornithomya fringillina*. The first species is reported for the first time in the Red-backed Shrike, and the second represents a new host association in the Iberian Peninsula.

Keywords: bird, louse flies, *Ornithophila*, *Ornithomya*, passerine, Spain

Összefoglalás A megvizsgált 26 tövisszúró gébics *(Lanius collurio)* közül öt volt kullancsléggyel parazitálva. Ezek két fajhoz *(Ornithophila metallica, Ornithomya fringillina)* tartoztak. Az első fajt először mutatták ki tövisszúró gébicsen, a második fajjal alkotott parazita-gazda kapcsolat pedig az Ibériai-félszigeten először bukkant fel.

Kulcsszavak: madár, kullancslégy, *Ornithophila*, *Ornithomya*, énekesmadarak, Spanyolország

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Introduction

The Red-backed Shrike *Lanius collurio* is a long-distance migratory songbird widely distributed in the Palearctic and the Afrotropical regions. During the breeding season, it is distributed on extent areas of Europe and Central Asia. Often linked to the semi-open habitats of the Eurosiberian region during the breeding season, it occupies mountain fresh areas with lower summer dryness in the Mediterranean region. It winters on the semiarid savanna areas of the southeast of Africa (Bruderer & Bruderer 1993, Lefranc & Workfolk 1997, Shirihai & Svensson 2018).

In the Iberian Peninsula, the Red-backed Shrike is distributed from the Pyrenees to the Cantabrian Mountains, Galicia, North Portugal, and the northern Iberian system. In recent decades, it has colonized the freshest areas of the Meridional Iberian System and Central System Mountains (Hidalgo *et al.* 2020, Tellería *et al.* 2020). This distribution range expansion into new Mediterranean areas cannot hide its global population trends. In the north and west Europa has been documented a population regression of 38% over the last 30-40 years. The breeding population in Iberia decreased by 54% between 1998 and 2016, reaching to 95% reduction in the Eurosiberian region at the northern Iberian Peninsula (Escandell 2017, Tellería 2018).

Due to declining populations, the Red-backed Shrike is protected in European countries and also by international organizations (BirdLife International 2017). From a conservational point of view, it is important to know the community of their ectoparasites, since ectoparasites can negatively impact the health status of bird populations and transmit microbial infections (Owen *et al.* 2010). In the Red-backed Shrike, ectoparasites representing three orders of insects have been detected in the Palearctic regions: flies, lice, and fleas (Diptera, Phthiraptera and Siphonaptera).

Brueelia cruciata and *Philopterus coarctatus* (suborder Ischnocera) have been cited in Central and Eastern Europe (Hellenthal *et al*. 2004, Costicã 2007, Ilieva 2009, Vas *et al*. 2012a, 2012b, Dik *et al*. 2017). From Amblycera, the genus *Myrsidea* in Bulgaria (Ilieva 2009), *Menacanthus eurysternus* and *M. camelinus* in Central Eastern Europe region (Hellenthal *et al*. 2004, Costicã 2007, Ilieva 2009, Vas *et al*. 2012a, 2012b).

Three species of fleas (Siphonaptera), *Ceratophyllus garei*, *C. borealis* and *C. gallinae* have been cited of the Red-backed Shrike in Central Europe (Kristofík *et al*. 2002). Three species of louse flies (Diptera: Hippoboscidae) have been recorder until now, *Ornithoica turdi* in Croatia (Trilar & Krčmar 2005), *Ornithomya avicularia* and *O. fringillina* in Central Europe (Sychra *et al*. 2008, Oboňa *et al.* 2019a).

Currently, there are no studies of insect ectoparasites of the Red-backed Shrike in the Iberian Peninsula (Tellería *et al*. 2020). The lack of information encourages us to explore the insect ectoparasite fauna of the Red-backed Shrike in Spain.

Materials and Methods

The study has been carried out at two ringing stations (Collado Cerrado [40º 52' N, 03º 45' O] and Arroyo de la Laguna [41º 09' N, 03º 36' O], Madrid, Spain). These ringing sites are located on mountain passes between 1.480 to 1.530 m a.s.l., both covered by cattle pasturelands *(Festuca indigesta, Nardus stricta)* with scattered thorny bushes (*Rosa* sp.) and creeping juniper *(Juniperus communis nana)*, surrounded by brooms *(Cytisus oromediterraneus)* and Scots Pine *(Pinus sylvestris)*. As in other mountains in the Mediterranean basin, this habitat heterogeneity results in a high biological richness during the breeding season on relatively small areas (Lewis 1970, Myers *et al.* 2000).

Shrikes were captured from May to August 2021 using mist nets. All shrikes were placed individually into a single-use cotton bags to avoid mixing ectoparasites among birds. Louse flies were collected directly from the shrikes and stored in capped tubes containing 70% ethanol until their identification in laboratory. For each identified specimen, we recorded the taxon, collection date and site, collector and person who identified it. The louse fly specimens were identified to species under a stereo microscope. To avoid biases in these determinations, all specimens were identified and classified by the same researcher (I. Bernal). Nomenclature of louse flies follows Hutson (1984).

Shrikes were aged and sexed according to Svensson (2009). Descriptive statistics were computed using Quantitative Parasitology on the Web, while confidence intervals (CI) were calculated according to Sterne (Reiczigel *et al*. 2019).

Results

Out of 26 individual Red-backed Shrike captured, five birds were infested with louse flies (prevalence = 19.2%; CI (Stearns) = 6.6–39.4%). We did not detect other orders of insect ectoparasites mentioned above (Phthiraptera and Siphonaptera). In one of the shrikes, it was not possible to collect sample. All the infested shrikes carried a single fly specimen. Two species of louse flies were identified, *Ornithophila metallica* and *Ornithomya fringillina (Figure 1)*.

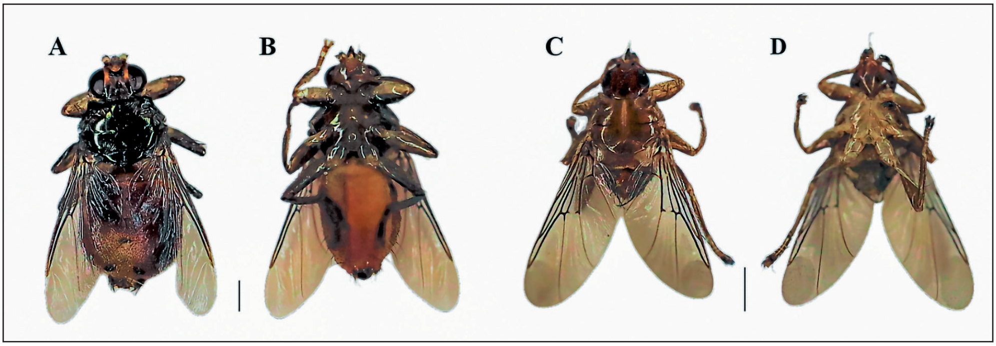

Figure 1. A and B – *Ornithophila metallica*, habitus, C and D – *Ornithomya fringillina*, habitus. A and C – dorsal view, B and D – ventral view. Unit scale: 1 mm

1. ábra A és B – *Ornithophila metallica*, C és D *Ornithomya fringillina*, A és C – háti nézet, B és D – hasi nézet, méretegység: 1 mm

***Ornithophila metallica* Schiner, 1864**

Material examined: 1 ♂ ex *Lanius collurio* (first calendar year), 16.VII. 2021.

***Ornithomya fringillina* Curtis, 1836**

Material examined: 1 ♀ ex *Lanius collurio* (first calendar year), 24.VII. 2021; 1 ♂ ex *Lanius collurio* (first calendar year), 24.VII. 2021; 1 ♂ ex *Lanius collurio* (adult female), 24. VII. 2021.

The samples were collected of the same family of shrike, from two first calendar year bird and one adult female.

Discussion

Two genera of louse flies (*Ornithophila* and *Ornithomya*) have been recorded from the Red-backed Shrike.

Ornithophila metallica is a polyxenous species distributed across the southern parts of the Palearctic, Afrotropical, Oriental and Australasian regions. In Europe, their records represent introductions by migratory birds (Nartshuk & Matyukhin 2019, Oboňa *et al.* 2019b, Gaponov & Tewelde 2020). Previously, *O. metallica* had been recorded in another species of the *Lanius* genus in Tajikistan (Red-tailed Shrike *Lanius phoenicuroides*, in Nartshuk & Matyukhin 2019). In the Iberian Peninsula, *O. metallica* has been recorded in raptors, cuckoos and passerines

(Cordero del Campillo *et al.* 1994, Carles-Tolrá 1998, 2001, Talabante *et al.* 2019). Therefore, our results represent the first host record of Red-backed Shrike.

Furthermore, we add a new host association for the Iberian Peninsula with the first record of *Ornithomya fringillina* on the Red-backed Shrike. This louse fly is a polyxenous species distributed around the Palearctic and Nearctic regions (Oboňa *et al.* 2019b, Gaponov & Tewelde 2020). *O. fringillina* is restricted more by habitat than by host specificity, preferring relatively lower areas with closed vegetation (Hutson 1984). This association had previously been documented in 25 Red-backed Shrikes (prevalence 100%) in Slovakia (Oboňa *et al.* 2019a). However, in the Iberian Peninsula this fly only has been cited in Barn Swallow *Hirundo rustica* and Common House Martin *Delichon urbicum* (Cordero del Campillo *et al.* 1994).

The confirmed presence of *O. metallica* in specimens of Red-backed Shrike is the first record of this species from this host. The long-distance migratory movements of this host across Europe and Africa makes this species an interesting host to detect new species of ectoparasites previously not recognized.

Knowledge of the ectoparasites of migratory birds, such as the Red-backed Shrike, can increase the documented species richness of a region, as well as improve our understanding of these species.

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