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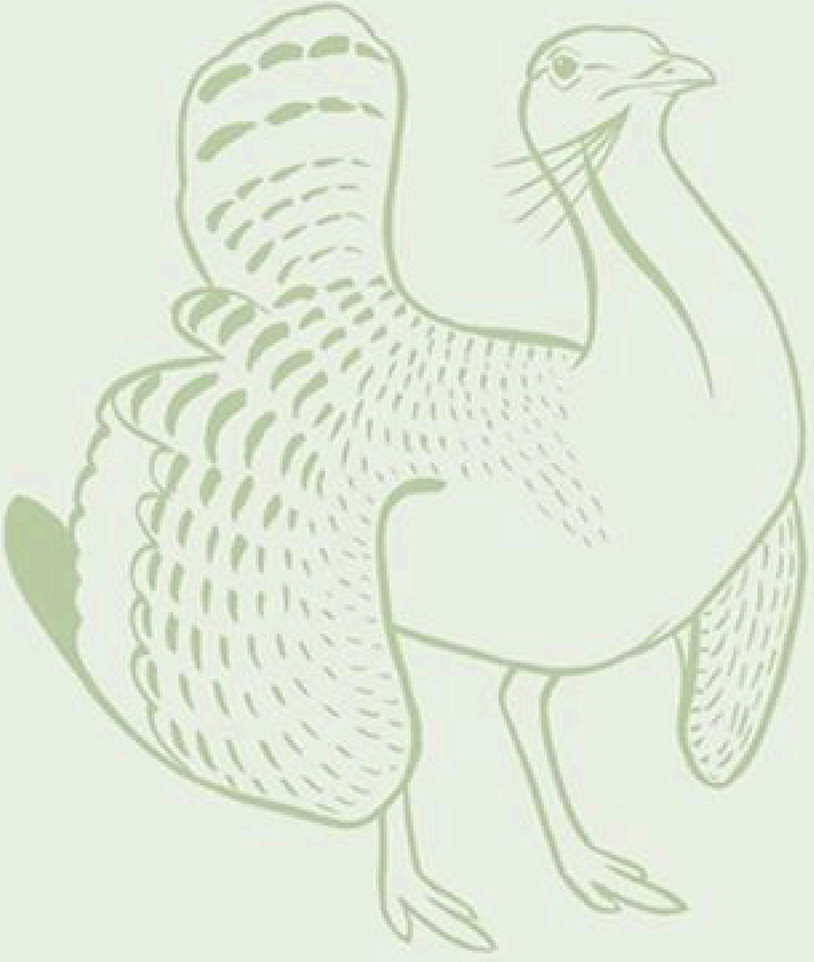

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Significance of the local anthropogenic effects in the dynamics of wild geese wintering on the Ramsar Site Lakes by Tata (Wetland City Tata)

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Abstract The Old Lake of Tata, an important wild goose roosting site in Central Europe, is unique in its location being in the middle of a city with a population of 24,000. Consequently, the site is subject to intensive human disturbances. Goose migration and wintering on the lake was studied for over 37 years (1984–2021). In addition to weather, hydrological, and feeding conditions, particular attention was focused on human influences (which were deemed generally disruptive, but also beneficial in some cases) that can induce significant changes in the population dynamics of the 30,000–50,000 wild geese from 13 species wintering here. Almost without exception, the largest observed changes in the abundance of wild geese roosting overnight on the Old Lake are due to anthropogenic effects. In the majority of cases, the most significant population changes are due to impacts associated with the operation and upkeep of the lake and the New Year's Eve fireworks. As a result, the winter operation of the Old Lake has been regulated by the Wild Goose Preference Mode since 2011, and the City of Tata has banned the use of fireworks during the winter period since 2018. Consequently, the conditions for wild goose migration have improved significantly. However, the situation of the Old Lake is still quite fragile and more conservation measures are needed to protect it further.

Keywords: wild goose migration, anthropogenic impacts, Tata, Ramsar

Összefoglalás Közép-Európa egyik legfontosabb vadlúd gyülekezőhelyének számító tatai Öreg-tó világviszonylatban is egyedülálló abban a tekintetben, hogy egy 24 000 lakosú város közepén helyezkedik el. Ebből adódóan rengeteg emberi zavaró hatás éri. A tavon zajló lúdvonulást és telelést 37 éven át (1984–2021 között) vizsgáltuk. Az időjárási, hidrológiai, táplálkozási körülmények mellett különös figyelmet fordítottunk azokra az emberi (általában zavaró, de néhány esetben kedvező) hatásokra, amelyek az itt telelő 13 fajba tartozó 30 000–50 000 vadlúd dinamikájában jelentős változásokat képesek indukálni. Az Öreg-tavon éjszakázó vadludak mennyiségében megfigyelt legnagyobb ütemű változások hátterében szinte kivétel nélkül valamilyen antropogén hatás áll. Az esetek többségében a tó üzemeltetésével összefüggő hatások, illetve a szilveszteri tűzijátékok okozzák a legjelentősebb állományváltozásokat. Ebből kifolyólag az Öreg-tó téli üzemeltetésében 2011-től bevezették a vadludakat preferáló üzemmódot, 2018-tól pedig Tata városa megtiltotta a tűzijátékok téli időszakban történő használatát. Mindezek eredményeként máris jelentősen javultak a vadlúdvonulás feltételei. Az Öreg-tó azonban még így is meglehetősen sérülékeny helyzetben van, aminek csökkentése érdekében további természetvédelmi intézkedések szükségesek.

Kulcsszavak: vadlúd vándorlás, antropogén hatás, Tata, Ramsar

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Introduction

The Old Lake of Tata is one of the most important wild goose s sites in the Pannonian region. Few Ramsar wetland areas in the world are subject to such a wide range of anthropogenic effects. This is due to its special location, which is practically in the middle of a city. The geese roosting here are subject to a number of human influences that often cause them to rouse en masse (Musicz 1992, 1998), and may even result in tens of thousands of geese leaving the wintering area in a matter of days (Musicz 2018a, 2018b). At the same time, there are also opposite effects that positively influence the arrival of geese and their stay in the area. From a nature conservation point of view, it is particularly important to explore both positive and negative anthropogenic factors affecting the population dynamics of wild geese.

Several studies have emphasized the importance of the Old Lake in Tata on wild goose migration in Europe (Sterbetz 1976, Fog 1982, Lebret 1982, Philippona 1983, Van den Bergh & Philippona 1986, Musicz 1990, 2014, Faragó *et al.* 1991, Faragó 1995, 1996). The Old Lake is among the top five wild goose roosting sites in Hungary in terms of annual maximum numbers of wild geese, but it is also where the highest numbers of wild geese in Hungary were observed in the winter of 2011/2012 and 2013/2014 (Faragó 2012, 2015).

Considering the effects influencing the migration dynamics of wild geese, publications have focused mainly on weather factors, climate change, and feeding conditions, while less attention has been paid to anthropogenic effects. Research on anthropogenic effects has largely been conducted on hunting aspects (Faragó 1982, 1995, Ebbinge 1985, 1991) and on correlations with agriculture (Fox *et al.* 2005, Rosin *et al.* 2012).

Few studies have addressed the relationship between water level fluctuations in wetlands and the migratory activity of different aquatic species. For the Hungarian-Slovakian section of the Danube, a study by Faragó and Hangya (2012) showed a clear correlation between water levels and bird numbers. In the case of artificially maintained and protected wetlands influenced by sluice systems, such as the Old Lake in Tata, optimal water level determination is of key importance from a nature conservation point of view, as noted by Musicz (1998, 2018a).

Relatively few publications address the conservation aspects of fireworks, despite the increasing human impact on bird populations and protected areas (Shamoun-Baranes *et al.* 2011, Karolewski *et al.* 2014). The use of fireworks, especially on New Year's Eve, is considered to be a dominant anthropogenic impact. The practice likely causes adverse changes in an increasing number of wetlands. The Old Lake of Tata is in a particularly vulnerable situation in this respect. Fireworks disturbances have occasionally displaced tens of thousands of wild geese from wetlands covered by the Ramsar Convention (Musicz 2018a, 2018b).

This study aims to answer the question of whether the two anthropogenic factors (lake management and fireworks) could be responsible for the sudden population fluctuations and regional dispersal of tens of thousands of wintering wild geese on the Old Lake of Tata.

Materials and Methods

Territory

The lakes by the Tata Ramsar site are located in the northwestern part of Hungary, in the long valley of the Által-ér river. A series of fish ponds, interspersed with karst springs, meadows, and ploughland comprises the site. The oldest wetland in the 1,897 hectare area (listed since 1989) is the Old Lake (N 47.64° E 18.33° N). With a water surface of 2.2 km^2, the lake is a priority aggregation site for northern wild goose species arriving to the Pannonian region *(Figure 1)*.

The Old Lake of Tata is the only Ramsar site in Europe that is almost completely encircled by a town. Although the lake is surrounded by monuments and houses, it provides a traditional resting place for tens of thousands of wild geese. It is one of the oldest fishponds in Hungary, with a dam that can be traced back to Roman times, a royal castle built in the 14th century, and a sluice system built in the 18th century *(Figure 2)*. The lake, a major tourist destination and the site of numerous festivals, is drained and fished every autumn.

Figure 1. Location of the Old Lake of Tata Ramsar site in Hungary
1. ábra A tatai Öreg-tó Ramsari terület földrajzi elhelyezkedése Magyarországon

Figure 2. Roosting wild geese on the Old Lake surrounded by the town of Tata (Photo: J. Milinte)
2. ábra Gyülekező vadludak a város által körülvett Öreg-tavon (Fotó: Milinte J.)

Studied wild goose species

Out of the 13 species of wild geese recorded to date, the Greater White-fronted Goose *(Anser albifrons)*, the Tundra Bean Goose *(Anser serrirostris)*, and the Greylag Goose *(Anser anser)* are considered to be the most common species at the Old Lake of Tata. The species usually occur or have occurred in groups of thousands or even tens of thousands. As our studies have focused on exploring the migratory dynamics of anthropogenic influences associated with large goose populations, our findings are primarily relevant to these species of geese. The combined share of the 10 other wild goose species occurring at the Old Lake of Tata comprises less than 1–2% of the population.

Greater White-fronted Goose

In the 37-year period under study, the Greater White-fronted Goose arrived to Hungary in the winter of 1989/1990 in the smallest numbers (maximum 15,100 individuals). In the winter of 2019/2020, the number of individuals wintering in Hungary exceeded 200,000 (Faragó 2020), while in the winter of 2020/2021 it was 265,000 (Faragó 2021c).

Until the mid-1990s, the species occurred in very small numbers (typically in the few hundreds) in the Tata area, constituting only 2–5% of the wild goose populations (Musicz 1990, 1998). It has since spectacularly replaced the role of the formerly dominant Tundra

Bean Goose at the Old Lake, and now accounts for at least 95% of the wintering wild goose masses in Tata (Musicz 2016). In recent years, the Old Lake has become a major aggregation site for the Greater White-fronted Goose in Hungary. The maximum wintering population on the lake exceeded 55,000 by 2020 (Faragó 2021c, Pellinger *et al.* 2021).

Greylag Goose

The Hungarian breeding population of Greylag Goose is estimated to be at least 2,000 pairs (Pellinger 2009, Faragó *et al.* 2016). In the 2010s, peak numbers of between 25,000 and 50,000 individuals were observed during migration periods in Hungary, usually in October or November (Faragó 2011, 2017, 2018, Faragó *et al.* 2016).

The Greylag Goose generally migrates through the Old Lake of Tata in small numbers. In the 1980s and 1990s, groups of fewer than 100 individuals were usually observed, followed by increasing numbers of 200–400 individuals since 1998 (Musicz 2014, 2016, Musicz *et al.* 2016). The Greylag Goose has been present on the lake in numbers that have occasionally reached several thousand specimens since the winter of 2007/2008, but typically only for shorter periods (maximum 2,500 specimens – Pellinger *et al.* 2021).

Tundra Bean Goose

The Tundra Bean Goose has been listed as a separate species in the IOC World Bird List since 2019 (Gill *et al.* 2021), it was previously part of the Bean Goose species as the tundra subspecies of the Bean Goose *(A. f. rossicus)*, together with the taiga subspecies *(A. f. fabalis)* and several other species. In this paper, we follow the IOC and Hungarian nomenclature. Our findings include the results for the former species *A. fabalis.*

The wintering population of the Tundra Bean Goose in Hungary peaked in November 1984 with 196,750 specimens (Faragó 1996, 2010, Faragó & Pellinger 2009), but in 1999 the maximum was less than 60,000 individuals and since 2012 it has not reached 10,000 individuals (Faragó 2014, 2015, 2016, 2017, Faragó 2021a, Faragó *et al.* 2017). In the winter of 2017/2018, only 362 individuals were recorded during January (Faragó 2021b). In parallel with the collapse of the wintering population in the Pannonian region, the numbers in southwestern Poland have exploded over the last decades (Ławicki *et al.* 2010a, 2010b, Wuczyński *et al.* 2012).

In line with the national trend, the Old Lake of Tata has also seen dramatic Tundra Bean Goose declines over the past two decades (Musicz 2016, 2018a, Musicz *et al.* 2016). In the winter of 1986/1987 the Old Lake saw a peak abundance 36,400 specimens; by the winter of 2019/2020, this number had dwindled to a mere 62.

A close relationship and area shift between the line of Lake Fertő – Kis-Balaton – western basin of Lake Balaton – Dráva river basin and the line of Old Lake – Lake Velence – Dinnyési-Fertő – Soponyai fish ponds – lower Danube for the wintering of the Bean Geese in Hungary is well known (Faragó & Pellinger 2009).

Frequency of monitoring

A complete survey of the wild goose population was conducted on more than 2,000 monitoring days between 1 October and 31 March 1984–2021 (during 37 wintering seasons) at the Old Lake of Tata. On average, counting was completed 2–3 days per week, but during periods of unusual winter water levels in the Old Lake and during fireworks around New Year's Eve, observations were conducted on a daily basis to take rapid population changes into account. Surveys were mainly performed during the mornings, but we also monitored flocks of geese arriving to the lake during the day and groups moving around the town, and occasionally observed the area around the Old Lake in the evening and at night. During the days around New Year's Eve, we recorded fireworks usage and, concurrently, the number of geese leaving the lake as well as the direction of their flight.

Examination of particularly rapid and significant stock fluctuations

We focused our analyses on particularly rapid and intense population changes where the number of wild geese roosting on the Old Lake increased or decreased by tens of thousands of individuals within a few days. Among the population declines, we examined 10 cases where the wild goose population declined by at least 80% within a few (<10) days and where the decline was as large as 10,000 individuals. Over the last 37 years, there have been several cases where the absolute decline exceeded 10,000. Cases where the absolute decline in numbers was very large, but the rate of decline was smaller in terms of rates, are not described. For example, in December 2020, the number of wild geese decreased from 50,500 to 17,800 in four days, but this decrease of 32,700 individuals did not reach 65%. In our study, we analysed the 10 fastest population declines. A more detailed case study is presented to describe the circumstances of the highest rate of population decline between 24 December 2008 and 3 January 2009.

Among the population increases, we analysed 10 cases where the number of geese overnighting at the site increased at least threefold within a few (<10) days, and where the increase exceeded 15,000 individuals. The circumstances of the fastest rate of population increase between 1 November 2017 and 7 November 2017 are presented in a more detailed case study.

In all cases, we analysed weather conditions in Hungary and Europe, the results of synchronous observation of wintering sites in Hungary in a given month (and in previous and subsequent months), and focused on the current state of the Old Lake of Tata (water level, possible ice cover), local anthropogenic influences, and the feeding sites. All factors have been considered in an attempt to identify the natural and anthropogenic causes of the large decline in wintering wild goose numbers.

Examination of meteorological factors

From the whole 37-year period, the meteorological data provided by the Hungarian Meteorological Service was analysed for the Tata region and categorized by months. The data included precipitation amount, the number of frosty days, the number of severe days (with minimum temperatures below –10 °C), coldest temperatures, the number of days with wind speeds above 15.0 m/s (stormy), the number of days with snow cover, and monthly maximum snow depth. We have considered European weather fronts, the unusually mild mid-winter periods, and in some cases, we even considered nearby earthquakes.

Investigating other wild goose roosting sites

Monthly standardized observations (Faragó 1995, 1996, 1998, 2008a, 2008b, 2010) covering 21 of the most important wild goose roosting sites in Hungary, conducted since 1984 within the framework of Hungarian Wild Goose Monitoring, provided the opportunity to monitor the distribution, spatio-temporal patterns, and redistribution of wild goose populations. These synchronous counts of waterfowl take place once in the middle of the month, and they naturally provide limited opportunities for comparison with the much more frequent monitoring surveys in Tata. However, in the case of a major population change in Tata, information from nearby wintering areas was exchanged orally as well (e.g. the Ipoly Valley, Slovakian areas, Dinnyési-Fertő, Csákvári-rét).

Increased attention has been paid to the accurate population assessment of wild geese roosting in nearby (<15 km) ponds, as in recent years it has become clear that the Old Lake of Tata has an intensive migratory dynamic link, mainly with the ponds of Ferencmajor, the Lake Asszony, and increasingly also with Lake Boldogasszonyi and the nearby Danube sections. The lakes, pond systems, and Danube stretches around Tata and in fact form a coherent wintering region for wild geese (Musicz 2021, Pellinger *et al.* 2021).

Use of unique tagging data

The ringing data supplied by the Hungarian Centre for Bird Ringing also provided important input for the interpretation of the extremely rapid population declines. Data with local (Tata and Komárom-Esztergom County) relevance were selected. In the case of the Greater White-fronted Goose, 486 observations (data records) of 54 ringed specimens were analysed up to 2018. For the Greylag Goose, we studied the records of 57 ringed individuals with 1,315 local references, while for the Bean Goose, we reviewed the records of 54 specimens ringed abroad with 280 occurrence records in Tata and its region. Ringing records that occurred within a few weeks time (max. 1 month) of the occurrence date in the Tata area in other parts of Hungary or in other European countries are of primary importance for the present study. Those cases are especially valuable where only a few days elapsed between two observations because these could indeed provide meaningful information concerning rapid population changes.

The number of wild geese with a *neck ring* has increased in Europe since 2010, so the most relevant data for our analysis are from the last 5–8 years.

Particularly valuable information for the spatio-temporal processes of major population fluctuations was provided by GPS-equipped Greater White-fronted Goose and Greylag Goose that were already in, arriving, or departing from the Tata area at the time of rapid population changes. In the case of Tundra Bean Goose, no tagged specimens have been found in Tata in recent years.

Anthropogenic impact assessment

The anthropogenic influences on the wintering conditions of wild geese are particularly diverse and intense in the Old Lake area due to its proximity to the city. Over the past 37 years, we have observed a variety of disturbance factors (Musicz 1992, 1998, 2018a). The impacts associated with the operation and upkeep of the lake (i.e. the artificial shaping of the water level) and the disturbance caused by fireworks are of particular significance. In addition, there are intermittent or occasional disturbances from kayakers, ice sailors, ice skaters, bird photographers, visitors walking into the mud of the drained lake, fishing boats, small aircraft flying low into the lake airspace, drones, and paragliders.

A separate group of disturbances are the various light effects that, unfortunately, are increasingly common in the Old Lake area. Street lights and decorative lights on the lakeshore, vehicle headlights, and the effects of light from people wearing headlamps are becoming increasingly significant disturbances. Larger scale disturbances include, in particular, the proliferation of greenfield industrial projects, road construction, the opening of gravel pits, and the large number of wind farms. Waterfowl hunting has been banned at the Old Lake since 1993 and is, therefore, not covered in this article.

Analysis of water level data and other lake characteristics

The prevailing water level and ice cover of the Old Lake is an important environmental factor for wild geese wintering in Tata. Winter water level data and other status characteristics for the last 37 years were provided by the North-Transdanubian Water Management Directorate, but were also supplemented by our own observations (occasional maps of the ice hummocks in different parts of the lake).

Fireworks activity testing

Among the environmental factors affecting wintering wild geese in Tata, the disturbance caused by fireworks at the end of the year is by far the most important. These disturbances were detected partly by field monitoring and partly by a webcam installed at the Old Lake from mid-November to mid-February 2017. This was supplemented in 2018 by a webcam on the observation tower on Kálvária Hill near the lake, which provides additional 24-hour surveillance throughout the year. The webcams, operated by a solar panel, provide real-time continuous monitoring, but also the possibility to retrospectively monitor the previous 6-hour period.

Statistical methods

For data processing and statistical interpretation Past v3.17 software was used (Hammer *et al*. 2001).

Results and Discussion

Over the past 37 years, the annual maximum number of wintering wild geese on the Old Lake of Tata has varied widely. There were years (1989/1990, 1993/1994) when the maximum number of wild geese on the Old Lake was less than 5,000, while in December 2013, the number exceeded 53,000 and reached 55,000 in November 2020. In the 1980s and 1990s, Bean Goose was clearly the dominant species at Old Lake until the autumn of 2000, when a national reversal occurred: the Greater White-fronted Goose became the dominant species. This natural reversal had also occurred nine years earlier. In the following five years, the Tundra Bean Goose population culminated in higher numbers, but from 2006 onwards, the Greater White-fronted Goose became the dominant species, a process that is still continuing today. In the meantime, the proportion of Greylag Geese has fluctuated between 1% and 5%. By 2020, the proportion of Tundra Bean Goose in the population had fallen to just a few thousandths *(Figure 3).*

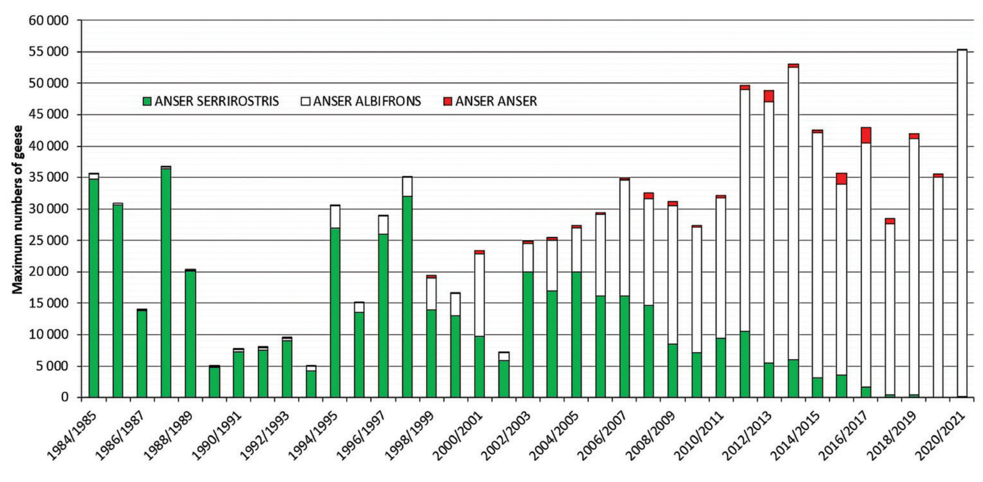


Figure 3. Maximum wintering number of wild geese on the Old Lake of Tata between 1984 and 2021
3. ábra A vadludak maximális telelő mennyisége a tatai Öreg-tavon 1984–2021 között

Several fluctuations in the dynamics of geese are usually observed each wintering season, both before and after the culmination. These changes, which are mostly due to natural migration ecology and European weather fronts, are spectacularly complemented by sudden population declines, and often the culmination phase is also marked by such sudden inflows. On several occasions in recent years, tens of thousands of wild geese have arrived to the Old Lake in this manner.

Effects of water management and pond management on the migration dynamics of geese

The Old Lake is usually in a drained state in winter, with a large, shallow water surface, providing an ideal resting place for the masses of wild geese that come here. According to many years of practical experience, this ideal condition can be observed in the 30–40 cm zone below the ’0’ graduation (127.55 m.a.s.l.) on the water level gauge near the Vecserei sluice, which is the basis for the official water level data of the Old Lake. At this water level, the total lake basin of 201.5 ha is covered by water for about 120 ha, while the remaining area is a dry land, rising from 10–30 cm deep shallow water. The much lower or much higher water levels, which are radically different from the ’natural’ state, are almost without exception caused by human influence and have already been interpreted as anthropogenic effects.

Over the past decades, the water levels in the Old Lake have fluctuated within very wide ranges during the wintering periods from October to March. Unusually low water levels were recorded in the winters of 1984/1985, 1994/1995, and 1995/1996, while in November 1993 or the winter of 2010/2011, water levels were much higher than the long-term average. The 37-year average of December water levels is –48 cm, but there was a month with water levels 1 m lower (1984) and 1 m higher (2010) than that. Similar extremes were observed in January, February, and March *(Figure 4)*.

The significantly differing water levels have proven to be a significant ecological factor over the past decades and can, to a significant extent, determine the temporary reduction or even increase of the role of the Old Lake in migration.

By analysing the monthly water level values and the corresponding monthly geese maxima at the Old Lake, we found that the abundance of geese during October and November is inversely proportional to the water levels. We did not find a strong correlation between the two, but in general, the lower the water level, the easier it is for larger flocks of wild geese

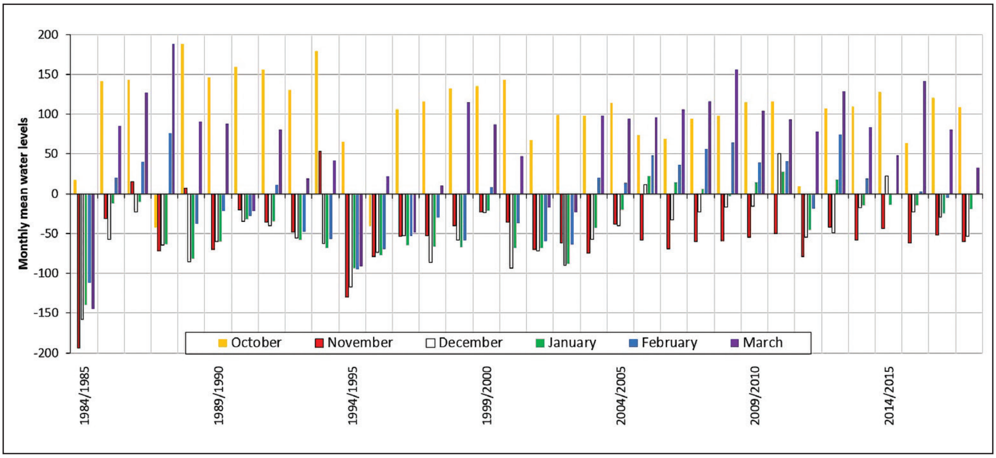
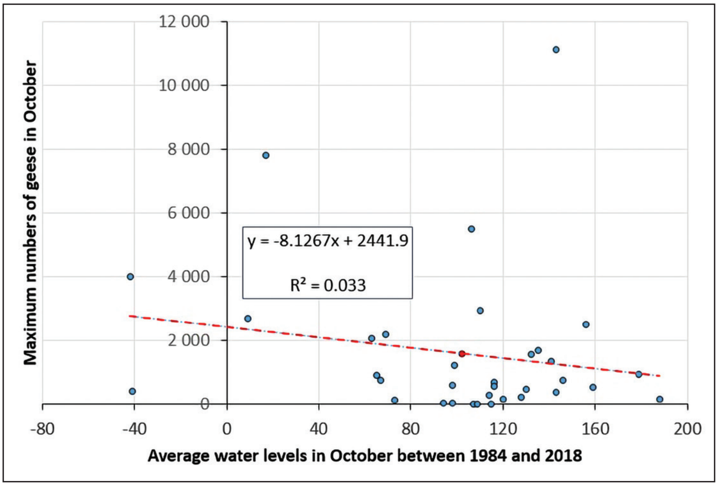
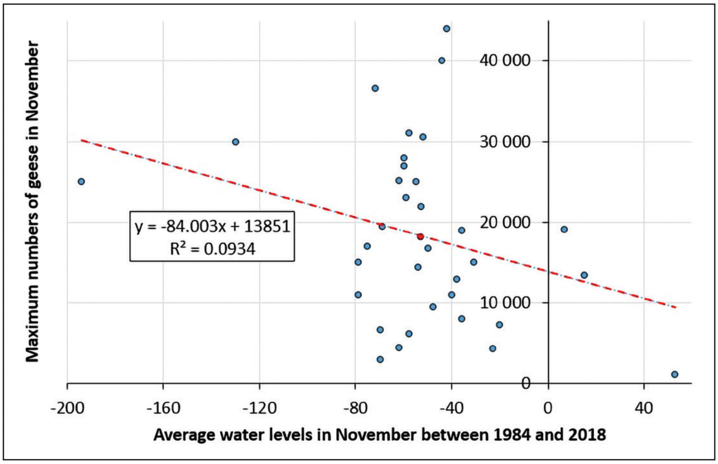


Figure 4. Monthly mean water levels in the winter seasons on the Old Lake Tata between 1984 and 2017
4. ábra A téli időszak havi közepes vízállás értékei a tatai Öreg-tavon 1984–2017 között





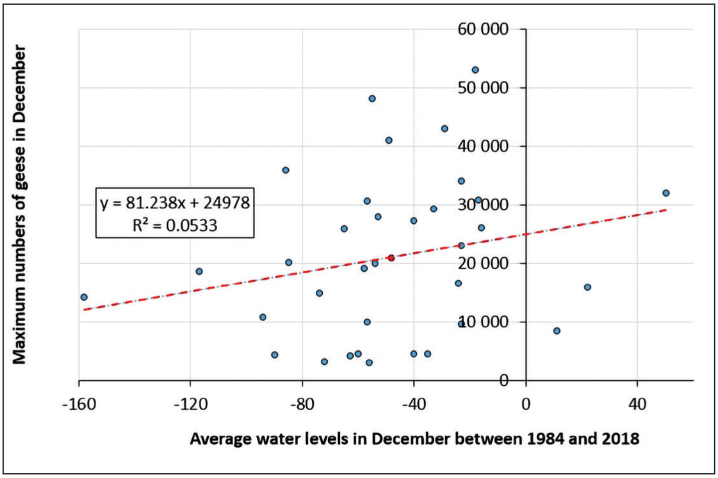


Figure 5. Relationship between the average monthly water depth of the Old Lake and the maximum monthly number of geese between 1984 and 2018 (October-November-December)

5. ábra Az Öreg-tó havi átlagos vízmélységének összefüggése a havi maximális lúdmennyiséggel 1984–2018 időközében (október-november-december hónapok)

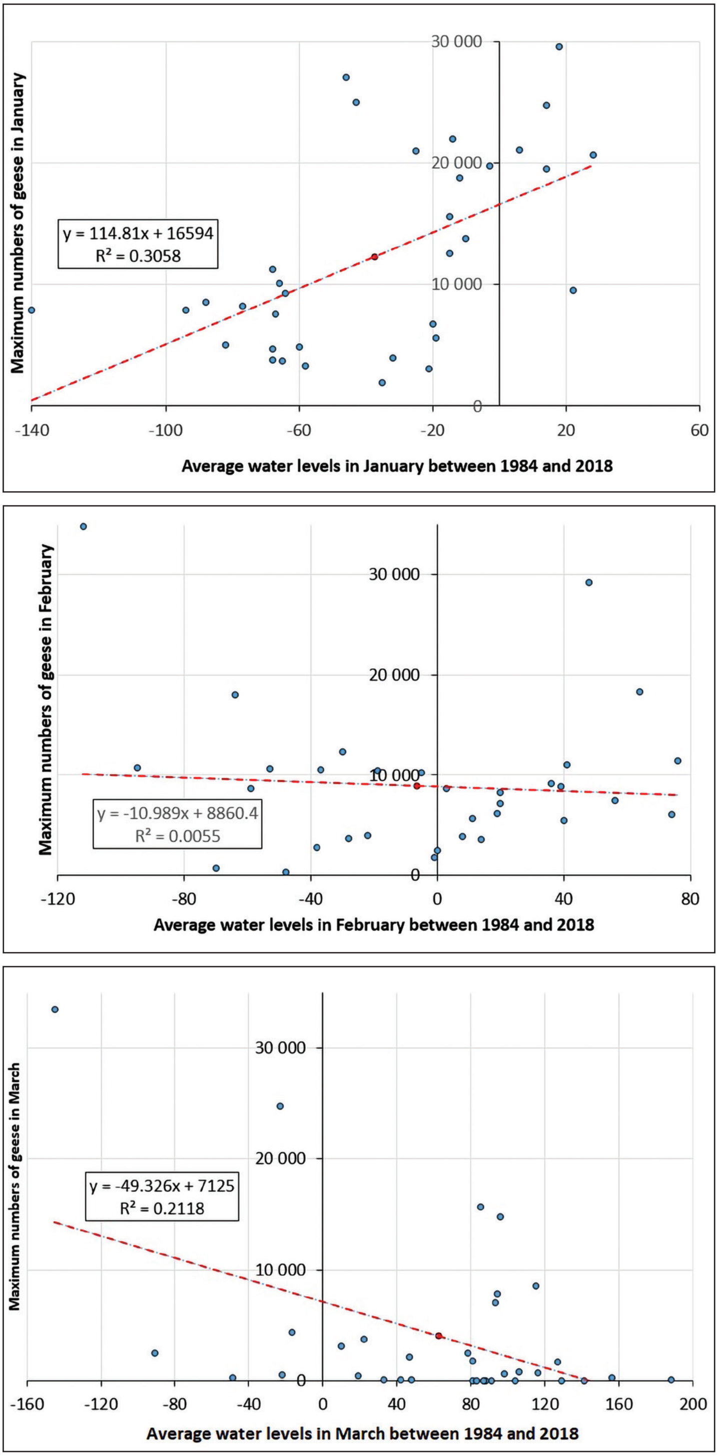


Figure 6. Relationship between the average monthly water depth of the Old Lake and the maximum monthly number of geese between 1984 and 2018 (January-February-March)

6. ábra Az Öreg-tó havi átlagos vízmélységének összefüggése a havi maximális lúdmennyiséggel 1984–2018 időközében (január-február-március hónapok) to develop on the lake during the fall. It is interesting to note that the largest October goose peak (11,100 individuals) occurred in October 1986, when the monthly mean water level was 143 cm. It is also noteworthy that 30,000 geese were recorded during a November (1994) water level of –130 cm (i.e. extremely low) *(Figure 5)*.

In the months of December and January, the correlation between water levels and wild goose abundance is reversed and a linear relationship can be observed, although again with very low R^2 values. For optimal lake management, it can be argued that higher water levels should be aimed for allowing more wild geese to gather on the lake. To facilitate fishing, the lake bed is often almost completely dry in December *(Figures 5, 6)*.

In February and March, there is also an inverse correlation between monthly water level values and goose maxima *(Figure 6)*. Nevertheless, for the benefit of the wild geese, significantly lower water levels than the long-term averages would be desirable both in February and March.

Given the extremely wide range of water levels, there is a range (around 60 cm) that can be considered optimal for wild geese and other waterfowl. Out of the annual maximum abundances recorded over the past 37 years, 29 (i.e. almost 80% of the cases) fell within the –43 cm ± 30 cm water level range *(Figure 7)*.

At this water level range, about 60–130 hectares of the 201 hectare lake basin are covered by water of varying depths, but predominantly shallow (10–60 cm). Large reefs appear in the middle of the lake at these levels.

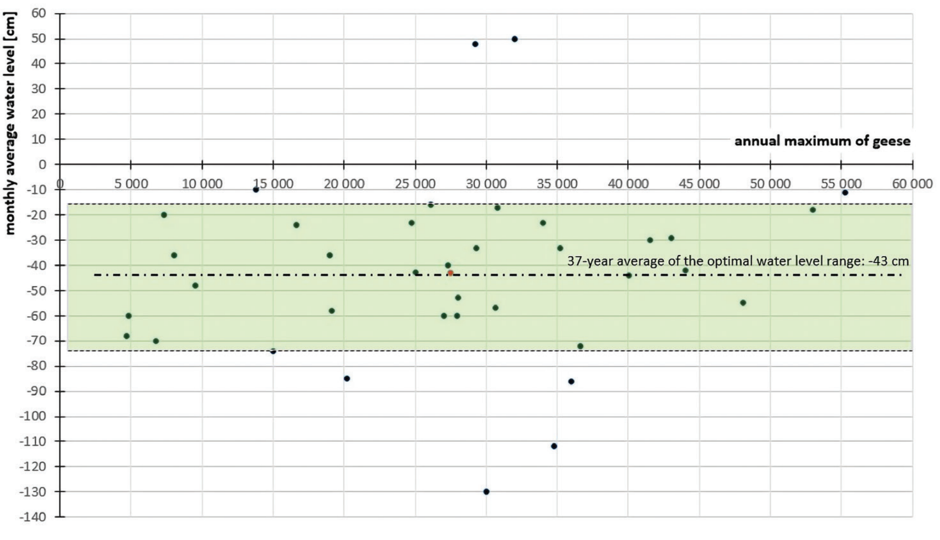


Figure 7. Relationship between the annual maximum of wild geese wintering on the Old Lake and the average monthly water levels between 1984 and 2021 (the optimal water level range is marked with a green field)

7. ábra A tatai Öreg-tavon telelő vadludak éves maximumainak összefüggése az adott havi közepes vízállás értékekkel 1984–2021 között (az optimális vízszint tartomány zöld mezővel jelölve)

Effects of fireworks on wild goose population dynamics

Given the fact that the Old Lake is located in the middle of a town with a population of around 24,000, fireworks in the town during the migration and wintering season (especially on New Year's Eve) are a major shock to the masses of wild geese resting on the lake. Tens of thousands of geese can flee from the lake overnight and recovery usually takes days, if not weeks. On New Year's Eves between 1984 and 1995, the average rate of abrupt decline was 42%; between 1995 and 2007 it was 44%; while between 2007 and 2017 there was an increasing frequency of events in which up to 90–100% of the geese assembled on the lake fled (Musicz 2018a, 2018b).

In some years, significant population shifts have occurred within the Pannonian region as a result of New Year's Eve fireworks. In particular, a close population dynamical link was observed between the Old Lake of Tata and the Dinnyési-Fertő (60 km) in 2005–2010. Between 26 December 2006 and 1 January 2007, the number of geese at the Old Lake decreased by about 24,000, while on the Dinnyési-Fertő the number increased by 29,000 on 1 January. In December 2017, a Greater White-fronted Goose ("Lilly17"), equipped with a satellite transmitter, stayed in the Tata area for 18 days. After the New Year's Eve fireworks on December 31, it fled to the Kiskunság saline lakes area, about 105 km to southeast (Musicz 2018b).

Fireworks do not always cause geese to flee long distances. Since the 2010s, wild geese have increasingly found temporary shelter in nearby fishponds (within 15 km) or sections of the Danube.

To reduce the disturbing effects of fireworks, the Municipality of Tata – on the initiative of citizens – created a municipal decree in October 2018 to ban the use of fireworks between 1 November and 28 February. The ban covers most of the inner area of Tata and some areas outside the city. This local ordinance is also a national example, as Tata is the first city in Hungary to restrict the use of fireworks for nature conservation purposes (protection of wintering wild geese on the lake) (Musicz 2018b).

To date, we have experienced three New Year's Eve events since the regulation was introduced, and these have been very positive, with an average of 85% of the overnight geese masses remaining on the lake during these three events, compared to an average of 31% in the previous ten years *(Figure 8)*.

The data for the 10 years before and three years after the ban are normally distributed according to the Shapiro-Wilk test. The F-test indicates no significant difference between the variances (F=2.094; p=0.358). According to the two-sample t-test applied on this basis, the means are significantly different (t=3.0506, P=0.011).

The vast majority of Tata's population adheres to the restrictions; however, some fireworks explosions on New Year's Eve continue to occur, but these are orders of magnitude less disruptive than before.

Cases of particularly rapid decline of wild goose populations

Over the past 37 years, the largest decline in the population of wild geese on the Old Lake occurred between 24 December 2008 and 3 January 2009, when all the wild geese (more

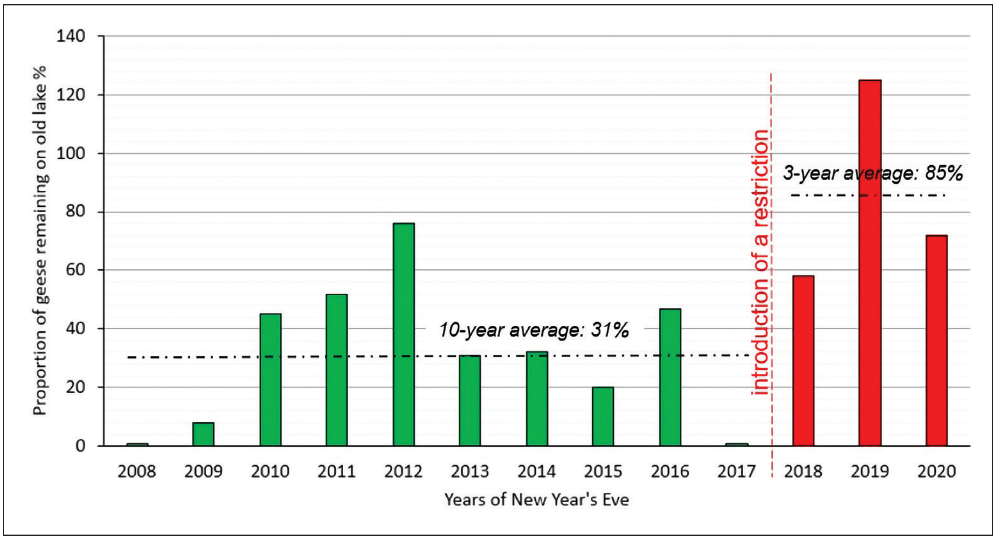


Figure 8. The proportion of wild geese remaining on the Old Lake of Tata during the New Year's fireworks

8. ábra A szilveszteri tűzijátékok ellenére a tatai Öreg-tavon maradó vadludak aránya

than 23,000 specimens) left the area within 10 days. This was surpassed in volume by the second half of December 2019, when nearly 32,000 wild geese left the lake in nine days (this was a 92% decline). In addition to natural factors, fireworks and excessively high water levels in the lake were responsible for the significant population declines in the vast majority of cases, but light effects and disturbance by skaters on the lake were also notable *(Table 1)*.

Case study of the fastest rate of decline of the wintering wild geese on the Old Lake – between 24 December 2008 to 3 January 2009

The situation of wild goose migration in Tata and major wintering areas in Hungary

2008/2009 brought the greatest number of wild geese (nearly 160,000) to Hungary at a somewhat unusual time (February). In both December and January, a group of around 120,000 wild geese was observed in the country. In these two months, the national maximum was recorded in Tata with 31,000 and 27,000 specimens respectively. Between these two dates, however, there was a very sharp drop in numbers on the Old Lake. The steep decline was caused by the fireworks that accompanied New Year's Eve. As a result, approximately 23,000 geese left the Old Lake within 10 days.

At the same time, the number of geese on Lake Fertő and in the ponds of Rétszilas increased spectacularly (by 16,000), but the most striking increase occurred in 2008 at Dinnyési-Fertő, which is the traditional wintering site closest to Tata. On 30 December 2008, only 3,300 geese spent the night there, but two days later the number suddenly rose to 18,000 (verbal report by L. Fenyvesi), while the number of geese on the Old Lake dropped to less than 100.

Table 1. The most significant and fastest population decreases of wintering geese as well as their natural and anthropogenic causes on the Old Lake of Tata between 1984 and 2019

1. táblázat A telelő vadludak legnagyobb és leggyorsabb ütemű állománycsökkenései, valamint azok természetes és antropogén okai a tatai Öreg-tavon 1984–2019 között

Date	Goose population of the Old Lake of Tata at the period's beginning/end	Rate of decrease [%, individuals]	Natural factors	Anthropogenic effects
24.12.2008. 03.01.2009.	23,200 0	100.0% 23,200	Lake freezing over	Fireworks, skaters
24.12.2017. 01.01.2018.	11,000 0	100.0% 11,000	Warm front, full moon	Ideal water level but fireworks
20.02.2006. 25.02.2006.	29,300 140	99.5% 29,160	Overly warm weather	Too high water level
28.12.2020. 06.01.2021.	17,700 1,050	94.1% 16,650	Mild, cloudy and cold, clear days alternate	Ideal water level but fireworks
17.01.2013. 25.01.2013.	21,000 1,450	93.1% 19,550	Snow cover, earthquake	Too high water level
18.12.2019. 27.12.2019.	34,300 2,700	92.1% 31,600	Lots of rain	Disturbing lighting, fireworks, too high water level
31.12.2009. 01.01.2010.	18,100 1,500	91.7% 16,600	Warm front, full moon	Fireworks
29.12.2013. 01.01.2014.	22,000 2,300	89.5% 19,700	Ideal circumstances	Fireworks
16.12.2007. 23.12.2007.	17,000 2,200	87.1% 14,800	Lake freezing over	Ideal water level but skaters
03.02.2009. 07.02.2009.	18,300 2,400	86.9% 15,900	Strong snow	Overly high water level

Some of the geese flushed from the Old Lake by the fireworks were forced to the nearby stretches of the Danube, and the number of geese in the ponds of Ferencmajor a few kilometres away increased from 5,000 to 9,000 during these days. This direct link is also confirmed by the large lynx tagged in late November 2008 in the Old Lake, which was found on 17 January 2009 in the ponds of Ferencmajor.

Local, national and European weather patterns

During late December and early January, the weather in western and southern Europe was quiet and uneventful due to anticyclonic effects. High temperatures ranged from 6–12 °C in the west and from 12–17 °C along the Mediterranean. However, it was extremely cold in the Ural Mountains, with daytime temperatures reaching only –5 to –10 °C. Along a frontal system stretching from Scandinavia to the Black Sea, snow fell in many places, with rain in the south. The Carpathian Basin experienced cold, wintry weather. In the Tata region, precipitation (mainly rain) was well above the long-term average from December to March in the winter of 2008/2009.

Table 2. The most significant and fastest population increases of wintering geese as well as their natural and anthropogenic causes on the Old Lake of Tata between 1984 and 2019
2. táblázat A telelő vadludak legnagyobb és leggyorsabb ütemű állománynövekedései, valamint azok természetes és antropogén okai a tatai Öreg-tavon 1984–2019 között

Date	**Goose population of the Old Lake of Tata at the period's beginning/end**	**Rate of increase [x, individuals]**	**Natural factors**	**Anthropogenic effects**
01.11.2017. 07.11.2017.	20 16,950	847.5x 16,930	Stormy cold front from western Europe, full moon	Water level tending to ideal
10.11.2018. 14.11.2018.	110 41,500	370.0x 41,390	Very mild weather, S wind turns to NW	Water level tending to ideal
31.10.2016. 06.11.2016.	114 16,700	146.5x 16,586	Unusually many geese in Hungary	Water level tending to ideal
11.01.2004. 16.01.2004.	2,700 25,000	9.3x 22,300	Strong warm front, freezing of other lakes in the region	Ideal water level
17.11.2020. 26.11.2020.	5,800 46,900	8.1x 41,150	Humid, foggy weather, first frosty days	Water level tending to ideal
03.01.2010. 13.01.2010.	3,000 19,500	6.5x 16,500	Suddenly mild weather	Rapid coming back of geese disappeared during fireworks
14.02.2006. 20.02.2006.	6,000 29,300	4.9x 23,300	Warm front, Danube flooding	Overly high water level but partly frozen
06.11.2014. 13.11.2014.	7,500 36,000	4.8x 28,500	Cold front from western Europe, goose groups from south	Water level tending to ideal; outstanding feeding conditions
20.12.1987. 27.12.1987.	5,600 26,000	4.6x 20,400	Mild weather, melting ice	Water level tending to ideal
07.11.1987. 14.11.1987.	10,200 36,600	3.6x 26,400	Mild, rainy weather	Strong hunting

The status of the Old Lake of Tata and local disturbances

The shallow waters of the drained Old Lake were largely covered by ice on 27 December 2008. New Year's Eve fireworks were also a major disturbance. The cold weather, which dipped below –10 °C at the beginning of January, caused the ice thickening that a festival of thousands of skaters and ice sailors was held on 10 January, preventing the arrival of the evening geese flocks from taking their overnight accommodation.

Cases of particularly rapid growth of the wild goose population

The most spectacular population increase of wild geese at Old Lake in the past 35 years took place in November 2017, when the number of wild geese at the lake jumped almost 850-fold (to nearly 17,000 individuals) in just one week. Another very spectacular increase of around 370 times occurred in November 2018, when the number of geese jumped from 110 to 41,500 in four days. Each of the 10 fastest growing populations exceeded 16,000 in absolute numbers *(Table 2)*.

Case study of the fastest rate of population growth of wintering wild geese at the Old Lake (between 1 November 2017 to 7 November 2017)

Wild goose migration in Tata and other important wintering areas in the Pannonian region

Regarding the arrival of wild geese in autumn, a new phenomenon has been observed in Tata since 2015: the larger flocks of geese arriving in the area in autumn do not start to gather in the Old Lake, but in the surrounding ponds with less human impact (Lake Asszony, ponds of Ferencmajor, Lake Boldogasszony). By the end of October 2017, the combined numbers of Greater White-fronted Goose, Greylag Goose, and Bean Goose had risen to nearly 5,000 in Lake Asszony, while no overnighting of any groups had occurred on the Old Lake. In the Slovak-Hungarian section of the Danube, water levels were relatively high during this period, so there were no reefs to attract the geese.

After 1 November, however, there was an extremely strong inflow into the Old Lake, partly due to the volume migrating from the surrounding lakes (this was when the water level of the Old Lake became ideal and the reefs began to appear), and partly due to successive new groups coming to the area. By 7 November, nearly 17,000 geese were already congregating on the lake, and by 11 November they had reached 28,000. This number was one of the highest of the 170,000 wild geese present in Hungary in November, with only the 35,000 geese observed in the nearby Lake Velence/Dinnyési-Fertő area being higher. There were also significant numbers of wild geese in the Hortobágy (28,000), the ponds of Biharugra (22,000), Lake Fertő (14,000), and Soponyai (10,000) fish ponds (Faragó 2021b).

With the influx of wild geese in November 2017, four Greater White-fronted Geese and eight Greylag Geese with rings were recovered from Old Lake and the nearby fish ponds. None of the three Hungarian-ringed and one Dutch-marked Greater White-fronted Geese provided any clues as to the direction of their arrival. Most of the six Czech-marked and two Austrian-marked Greylag Geese indicated a likely northwest-southeast migration route.

Local, national, and European weather patterns

October 2017 was much stormier than the multi-year average, with four days reaching wind speeds of 15 m/s. In the final days of October, the Carpathian Basin was also hit by a few days of rain with gale-force winds from northwest. At the beginning of November, sunny but frosty weather set in, with anticyclonic effects dominating Central Europe. The migratory activity of the assembling geese was probably enhanced by the full moon on 4 November. From 5 November, winds shifted south and much milder weather arrived in the Tata area. Precipitation during November was above the long-term average (48.7 mm).

The status of the Old Lake of Tata and local disturbances

On 3 November, the water level of the Old Lake began to reach ideal levels and the first reefs appeared. There were no major disturbances on the Old Lake or on other lakes around Tata during these days.

In both cases, the influence of weather fronts was detectable, but there was also the anthropogenic factor: the draining of the lake made water levels ideal for the wild geese masses. This has become particularly clear in recent years. Geese are usually already present in large numbers in Hungary and the wider Tata area in early November, but it is only on days when the reefs of the lake emerge that they start to migrate to the Old Lake by the thousands or tens of thousands. Geese are particularly fond of this shallow, reef-strewn stretch of water. This is the reason for six of the 10 fastest population growth rates.

In addition to water management, the most important anthropogenic impacts include the rapid return of wild geese scared away by New Year's Eve fireworks and by the heavy waterfowl hunting in the wider Tata area, which repeatedly drives wild geese to the Old Lake. Since 1994, the beneficial effects of the hunting ban at the Old Lake have been clearly visible. Also noteworthy among the anthropogenic effects are the exceptional feeding conditions in some years (e.g. 2014) (much larger area sown with winter wheat).

Conclusions

Wild goose monitoring over the last 37 years has revealed a complex system of local disturbance in the vicinity of the Old Lake in Tata, which has significantly influenced the migration dynamics and spatial dispersal of wild geese in several cases.

The 37-year time series of wild goose maxima clearly identifies the main threats and, at the same time, the effects of positive actions that have opened an 'era', a new chapter in the decades-long history of wild goose migration on the Old Lake. The negative effects of the 1986–1988 dredging and the unusually high water levels of 1988–1994 can be clearly identified. The beneficial effects of the 'wild goose friendly' operation introduced by the lake's manager in 2011 and the ban on fireworks from 2018 are also evident *(Figure 9)*.

Due to disturbance factors in the confined environment of the Old Lake, the dispersal of the wintering goose population in the Tata area is increasing. Within a 15 km radius of Tata, there are now four wetlands where the number of geese occasionally exceed 10,000. At times, more wild geese roost on these than on the Old Lake (Pellinger *et al.* 2021). This trend is likely to increase in the future, which may result in the Old Lake, in combination with the surrounding wetlands, increasingly being able to play a significant role in goose migration.

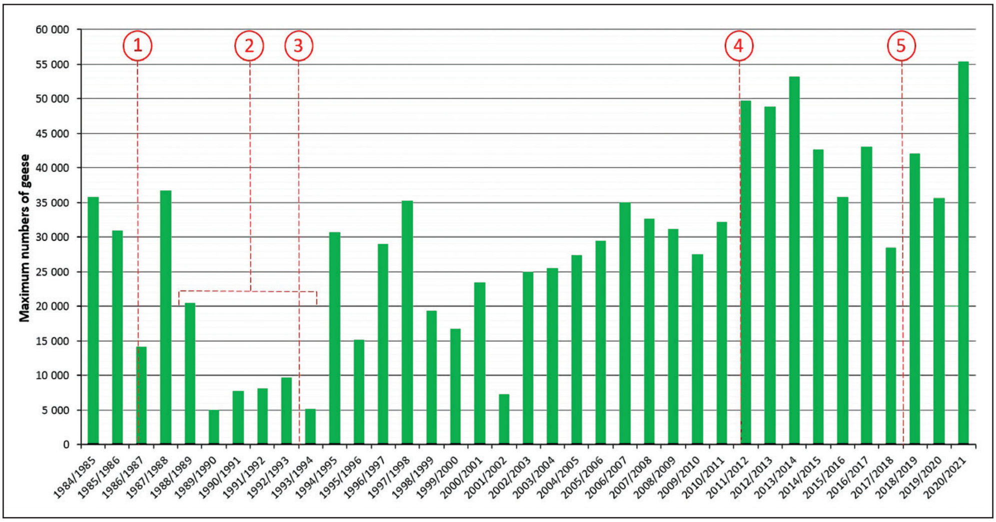


Figure 9. Changes in the annual maximum of geese wintering on the Old Lake of Tata affected by the main anthropogenic effects between 1984–2021 (Legend: 1: disturbing effects of dredging; 2: autumn water level much higher than the multi-year average; 3: start of a ban on waterfowl hunting; 4: start of 'wild goose mode' of water management; 5: start of fireworks ban)

9. ábra A tatai Öreg-tavon telelő vadludak éves maximumainak változása a fontosabb antropogén hatásokkal összefüggésben 1984–2021 között (Jelmagyarázat: 1-tókotrás zavaró hatásai; 2-túl magas őszi vízállás; 3-vízivad vadászati tilalom kezdete; 4-a vízgazdálkodás „vadludas üzemmódjának" kezdete; 5-tűzijáték tilalom kezdete)

The disturbance effects caused by fireworks were so significant that geese sometimes left the whole area of Tata. Sometimes even other wild goose roosting sites (e.g. Lake Fertő, Dinnyési-Fertő, Kiskunság saline lakes) located 50–100 km away experienced noticeable population changes. The most significant disturbance at the Old Lake is undoubtedly resulted by the New Year's Eve fireworks. The restriction of fireworks usage in the winter, which is unique in Hungary, was introduced in 2018, and it has fulfilled the hopes of effectively contributing to the containment of the wild geese population that gather on the Old Lake. It is particularly important that the ban (accepted by the vast majority of the residents of Tata) remains in effect in terms of space and time.

Anomalies (excessively high or low water levels) in the operation of the Old Lake (mainly due to fishing) can also cause notable disturbance. No significant correlation was found between geese leaving and excessive draining of the lake, suggesting that fishing on the Old Lake – subject to compliance with the main nature conservation rules – is compatible with the interests of goose migration. Major draining of the lake will not cause the wintering goose population to leave if the duration of the draining does not exceed 1-2 weeks. This has been the main objective of the lake's manager over the last 10 years, which is probably why the R^2 values for this factor are very low. In the future, other factors (e.g. natural predators in the case of a drying lake bed or even the presence of humans) should be taken into account. Since 2017, this has been increasingly possible to observe with the infra-red webcam installed in the lake bed from mid-November to the end of January.

The population of wintering wild geese on the Old Lake is increasingly a subject to extremely rapid and large-scale changes. All of the 10 most significant population declines of the last 37 years have occurred in the last 15 years, and three of the last four wintering seasons have seen such extreme cases. It is also intriguing that regular overwintering periods are experiencing spectacular growth (sometimes within a month) followed by equally rapid declines. This was the case in the winters of 2005/2006, 2008/2009, 2009/2010, 2017/2018 and 2020/2021. The main causes of these rapid changes can clearly be identified as human impacts (in particular the usage and upkeep of the lake and the fireworks). We concluded that, among the different scales of disturbance, the most significant cause of the sudden decline (6 out of 10 cases studied) was the light and noise of New Year's Eve fireworks. In addition, excessive water levels in the night roosts were also the cause of significant declines in several cases (4 out of 10). The very rapid dynamics of the wintering wild geese population on the Old Lake indicate that these are becoming increasingly vulnerable due to the anthropogenic causes. As a consequence, the lake is expected to become an important place in wild goose migration for increasingly shorter periods of time in the future.

These anthropogenic conditions often amplify the effects of favorable weather and snow conditions. This is particularly true in the case of Tata, where systematic efforts are being made to improve the conditions for the migration of wild geese, a major tourist attraction (the city was awarded the 'Wetland City' prize by the Secretariat of the Ramsar Convention in 2018 in recognition of its efforts in this regard).

Today, the increased vulnerability of the Old Lake of Tata is still somewhat compensated by local conservation measures and the social awareness-raising effect of the wild goose festivals (the Wild Goose Festival of Tata), but to preserve the international (Ramsar) importance of the Old Lake, it is essential to introduce a more comprehensive approach to urban development and management.

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References

Bergh, L. M. J. van den & Philippona, J. 1986. The occurrence of geese (mainly Bean Geese) at Tata in the West of Hungary. – Aquila 92: 65–80.

Ebbinge, B. S. 1985. Factors determining the population size of arctic-breeding geese, wintering in Western Europe. – Ardea 73: 121–128.

Ebbinge, B. S. 1991. The impact of hunting on mortality rates and spatial distribution of geese, wintering the western Palearctic. – Ardea 79: 197–209.

Faragó, S. 1982. A vízivadvadászat Magyarországon 1970–1980. [Waterfowl hunting in Hungary between 1970 and 1980]. – Nimród Fórum 1982. november: 14–26. (in Hungarian)

Faragó, S., Kovács, G. & Sterbetz, I. 1991. Goose populations staging and wintering in Hungary 1984–1988. – Ardea 79(2): 161–163.

Faragó, S. 1995. Geese in Hungary 1986–1991. Numbers, Migration and Hunting Bags. – IWRB Publ. 36. Slimbridge UK.

Faragó, S. 1996. A Magyar Vadlúd Adatbázis 1984–1995. Egy tartamos monitoring [Data Base of Geese in Hungary 1984–1995: A long-term monitoring]. – Magyar Vízivad Közlemények 2: 3–168. (in Hungarian)

Faragó, S. 1998. A Magyar Vízivad Információs Rendszer [The Hungarian Waterfowl Information System]. – Magyar Vízivad Közlemények 4: 3–16. (in Hungarian)

Faragó, S. 2008a A Magyar Vízivad Monitoring standardizált megfigyelési területei [The standard observations units of Hungarian Waterfowl Monitoring (HWM)]. – Magyar Vízivad Közlemények 16: 21–48. (in Hungarian)

Faragó, S. 2008b A vonuló vízivadfajok állományainak tér-idő mintázata Magyarországon. Az 1996–2004 közötti időszak elemzése [Time-space pattern of migratory waterfowl species in Hungary. An analysis of the period 1996–2004.]. – Magyar Vízivad Közlemények 16: 49–200. DOI: 10.17242/MVvK_26.01 (in Hungarian)

Faragó, S. 2010. Numbers and distributions of geese in Hungary 1984–2009. – Ornis Svecica 20: 144–154.

Faragó, S. 2011. A vadlúd monitoring eredményei a 2010/2011-es idényben Magyarországon [Results of Geese Monitoring in Hungary in the season 2010/2011]. – Magyar Vízivad Közlemények 20–21: 201–250. (in Hungarian)

Faragó, S. 2012. A vadlúd monitoring eredményei a 2011/2012-es idényben Magyarországon [Results of Geese Monitoring in Hungary in the season 2011/2012]. – Magyar Vízivad Közlemények 22: 1–50. (in Hungarian)

Faragó, S. 2014. A vadlúd monitoring eredményei a 2012/2013-as idényben Magyarországon [Results of Geese Monitoring in Hungary in the season 2012/2013]. – Magyar Vízivad Közlemények 24: 1–50. (in Hungarian)

Faragó, S. 2015. A vadlúd monitoring eredményei a 2013/2014-es idényben Magyarországon [Results of Geese Monitoring in Hungary in the season 2013/2014]. – Magyar Vízivad Közlemények 25: 1–54. DOI: 10.17242/MVvK_Monitoring/25-2 (in Hungarian)

Faragó, S. 2016. A vadlúd monitoring eredményei a 2014/2015-ös idényben Magyarországon [Results of Geese Monitoring in Hungary in the season 2014/2015]. – Magyar Vízivad Közlemények 27: 3–54. DOI: 10.17242/MVvK_27.01 (in Hungarian)

Faragó, S. 2017. A vadlúd monitoring eredményei a 2015/2016-os idényben Magyarországon [Results of Geese Monitoring in Hungary in the season 2015/2016]. – Magyar Vízivad Közlemények 29: 3–52. DOI: 10.17242/MVvK_29.01 (in Hungarian)

Faragó, S. 2020. A Magyar Vízivad Monitoring Adatbázisa 2019/2020. Kutatási jelentés [Data base of Hungarian Waterfowl Monitoring in 2019/2020, Research Report]

Faragó, S. 2021a A vadlúd monitoring eredményei a 2016/2017-es idényben Magyarországon [Results of Geese Monitoring in Hungary in the season 2016/2017]. – Magyar Vízivad Közlemények 31–32: 1–50. (in Hungarian)

Faragó, S. 2021b A vadlúd monitoring eredményei a 2017/2018-as idényben Magyarországon [Results of Geese Monitoring in Hungary in the season 2017/2018]. – Magyar Vízivad Közlemények 31–32: 301–352. (in Hungarian)

Faragó, S. 2021c. A Magyar Vízivad Monitoring Adatbázisa 2020/2021. Kutatási jelentés [Data base of Hungarian Waterfowl Monitoring in 2020/2021, Research Report]

Faragó, S. & Pellinger A. 2009. Vetési lúd [Bean Goose]. – In: Csörgő, T., Karcza, Zs., Halmos, G., Magyar, G., Gyurácz, J., Szép, T., Bankovics, A., Schmidt, A. & Schmidt E. (eds.) Magyar madárvonulási atlasz [Hungarian Bird Migration Atlas]. – Kossuth Kiadó, pp. 112–114. (in Hungarian with English Summary)

Faragó, S. & Hangya, K. 2012. Effects of water level on waterbird abundance and diversity along the middle section of the Danube River. – Hydrobiologia 697: 15–21. DOI: 10.1007/s10750-012-1166-1

Faragó, S., Kovács, Gy. & Hajas, P. P. 2016. Nyári lúd *(Anser anser)* fajkezelési terv Magyarországon [Management plan for Greylag Goose *(Anser anser)* in Hungary]. – Magyar Vízivad Közlemények 28: 81–114. DOI: 10.17242/MVvK_28.03 (in Hungarian)

Faragó, S., Musicz, L. & Hajas, P. P. 2017. Vetési lúd *(Anser fabalis)* fajkezelési terv Magyarországon [Management plan for Bean goose *(Anser fabalis)* in Hungary]. – Magyar Vízivad Közlemények 30: 117–153. DOI: 10.17242/MVvK_30.03 (in Hungarian)

Fog, M. 1982. Number of Bean Goose (*Anser fabalis* sp.) in the wintering areas. – Aquila 89: 123–125.

Fox, A. D., Madsen, J., Boyd, H., Kuijken, E., Norriss, D. W., Tombre, I. M. & Stroud, D. A. 2005. Effects of agricultural change on abundance, fitness components and distribution of two arctic-nesting goose populations. – Global Change Biology 11: 881–893. DOI: 10.1111/j.1365-2486.2005.00941.

Gill, F., Donsker, D. & Rasmussen, P. (eds.) 2021. IOC World Bird List (v11.1). – DOI: 10.14344/IOC.ML.11.1.

Hammer, Ø., Harper, D. A. T. & Ryan, P. D. 2001. PAST: Paleontological Statistics Software Package for Education and Data Analysis. – Palaeontologia Electronica 4: 1–9.

Karolewski, K., Bochenski, M., Ciebiera, O., Markulak, D. & Jerzak, L. 2014. New Year's eve fireworks impact on the number of Magpies on the roosting place. – International Studies of Sparrows 38: 27–29. DOI: 10.1515/isspar-2015-0029

Ławicki, Ł., Staszewski, A. & Czeraszkiewicz, R. 2010a Migration and wintering of the Bean Goose *Anser serrirostris* and White-fronted Goose *A. albifrons* in Western Pomerania in 1991–2008 [Wędrówka i zimowanie gęsi zbożowej *Anser serrirostris* i gęsi białoczelnej Anser albifrons na Pomorzu Zachodnim w latach 1991–2008]. – Ornis Polonica 51: 93–106. (in Polish)

Ławicki, Ł., Wylegała, P., Polakowski, M., Wuczyński, A. & Smyk, B. 2010b New data of Bean Goose *Anser fabalis* and White-fronted Goose *Anser albifrons* migration and wintering in Poland. – Goose Bulletin 11: 10–14.

Lebret, T. 1982. Goose observations in the Pannonic region in October–December 1980 and in March 1981. – Aquila 89: 187–191.

Musicz, L. 1990. Vadlúdmozgalmak vizsgálata a tatai Öreg-tavon az 1984–1989 közötti időszakban [Wild goose movements on the Tata Öreg Lake 1984–1989]. – Aquila 96–97: 19–35. (in Hungarian)

Musicz, L. 1992. A tatai Öreg-tó vadlúdforgalmának antropogén hatásvizsgálata [Anthropogenic impact assessment of wild goose presence at the Old Lake of Tata]. – LIMES – Komárom-Esztergom Megyei Tudományos Szemle 2: 29–40. (in Hungarian)

Musicz L. 1998. A vadlúdtelelés ökológiai-természetvédelmi vonatkozásai a tatai Öreg-tavon [Ecology and nature conservation relation of wild goose wintering at the Öreg-tó ("Old Lake") Tata, Hungary]. – Partimadár 6–7: 25–43. (in Hungarian)

Musicz, L. 2014. Vadlúd monitoring a Tatai-tavakon [Monitoring of wild geese on the Lakes of Tata]. – Szélkiáltó 16: 17–19. (in Hungarian)

Musicz, L., Bátky, G. & Csonka, P. 2016. Vadlúdfajok arányainak változása Tatán 2000–2015 között [Changes in the proportion of wild goose species in Tata between 2000–2015]. – Szélkiáltó 17: 14–21. (in Hungarian)

Musicz L. 2016. A tatai Öreg-tavon telelő vadlúdcsapatok vonulási mintázatának változása 1984–2016 között [Migration pattern changes of wild Goose groups wintering on the Old Lake Tata between 1984 and 2015]. – Magyar Vízivad Közlemények 28: 67–80. DOI: 10.17242/MVvK_28.02 (in Hungarian)

Musicz, L. 2018a The significance of local anthropogenic effects on the movements of wintering geese in the Pannonian region. – 18th Conference of Goose Specialist Group, Klaipéda, Lithuania, 27–30 March, 2018. Conference abstracts, p.73.

Musicz, L. 2018b A tűzijátékok zavaró hatása a tatai Öreg-tavon telelő vadludakra [The impact of fireworks on wild geese wintering on Öreg-tó, Tata]. – Aquila 125: 7–10. (in Hungarian)

Musicz, L. 2021. A Tatai Öreg-tó Ramsari-terület vadlúd állományának komplex vizsgálata. [A complex analysis of the goose population of „Lakes by Tata" Ramsar site]. – PhD Thesis, University of Sopron Disszertáció. pdf (uni-sopron.hu) (in Hungarian)

Pellinger, A. 2009. Nyári lúd [Greylag Goose]. – In: Csörgő, T., Karcza, Zs., Halmos, G., Magyar, G., Gyurácz, J., Szép, T., Bankovics, A., Schmidt, A. & Schmidt E. (eds.) Magyar madárvonulási atlasz [Hungarian Bird Migration Atlas]. – Kossuth Kiadó, pp. 119–122. (in Hungarian with English Summary)

Pellinger, A., Musicz, L., Tatai, S., Laber, J., Bátky, G. & Bodor, Á. 2021. Átvonuló és telelő vadludak eloszlásának változásai a Dunántúl északi részén [Changes in the distribution of migrating and wintering wild geese in the northern part of Transdanubia]. – Rence – A Fertő-Hanság Nemzeti Park Igazgatóság Tanulmánykötetei 6: 191–227. (in Hungarian with English Summary)

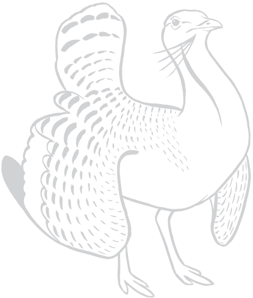
Philippona, J. 1983. Beobachtungen von Wildgänsen bei Tata/VR Ungarn [Observations of wild geese at Tata/Hungary]. – Der Falke 30(7): 225–227. (in German)

Rosin, Z. M., Skórka, P., Wylegała, P., Krąkowski, B., Tobolka, M. & Myczko, Ł. 2012. Landscape structure, human disturbance and crop management affect foraging ground selection by migrating geese. – Journal of Ornithology 153: 747–759. DOI: 10.1007/s10336-011-0791-1

Shamoun-Baranes, J., Dokter, A. M., van Gasteren, H., van Loon, E. E., Leijnse, H. & Bouten, W. 2011. Birds flee en mass from New Year's fireworks. – Behavioural Ecology 22(6): 1173–1177. DOI: 10.1093/beheco/arr102

Sterbetz, I. 1976. A vadlúdvonulás alakulása a magyarországi gyülekezőhelyeken [Development of wild-geese migration on the Hungarian gathering-places]. – Aquila 82: 181–194. (in Hungarian)

Wuczyński, A., Smyk, B., Kołodziejczyk, P., Lenkiewicz, W., Orłowski, G. & Pola, A. 2012. Long-term changes in numbers of geese stopping over and wintering in south-western Poland. – Central European Journal of Biology 7(3): 495–506. DOI: 10.2478/s11535-012-0031-6



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Assemblage of wetland bird Species in Purbasthali Oxbow Lake, West Bengal, India: Implications for Management

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Abstract The present study attempts to assess the composition, abundance and diversity of avifauna with respect to their habitat in and around the Purbasthali wetland, based on both primary data collected through the point count method during 2017–2019 and literature data. Among the total 77 species (encompassing 10 orders and 19 families), 39 species are migrants, 18 are rare and 24 species show declining global trend. According to their habitat, they are sub-divided into three categories i.e. *waterfowls* (live in open water, 20 species), *waders* (live in bank areas/water edge area, 45 species) and *wetland associated* (live in nearby trees, 12 species). The Shannon-Wiener Diversity Index (H') and the Evenness Index (E') are used to examine the diversity within and between the habitats. The result reveals higher diversity and evenness of the waders in comparison to others. The maximum diversity (H′=3.02) and evenness (E′=0.79) has been recorded for the waders in 2019, whereas the least values (H′=1.02, E′=0.34) have been found in 2016 for the waterfowls. Relative Diversity Index affirms the dominance of the *Anatidae* family. The birds of the area have now been seriously threatened by human intervention.

Keywords: Shannon-Wiener Diversity Index, Purbasthali wetland, habitat, Relative Diversity Index, migratory birds

Összefoglalás Jelen tanulmány a Purbasthali nevű vizes élőhely és környezete madárfaunájának összetételét, abundanciáját és diverzitását mutatja be a 2017–2019 között pontszámlálással gyűjtött és irodalmi adatok alapján. Az összesen 77 – 10 rendbe és 19 családba sorolható – faj közül 39 vonuló, 18 ritka, 24 faj esetén az egyedszám csökkenő globális trendet mutatott. Élőhelyük szerint e fajok három kategóriába sorolhatók: vízimadarak (nyílt vizet kedvelők, 20 faj), gázlómadarak (partmenti területeket kedvelők, 45 faj) és a vizes élőhelyekhez kötődők (közeli fákat kedvelők, 12 faj). Az élőhelyeken belüli és azok közötti sokféleség vizsgálatára Shannon-Wiener diverzitás indexet (H') és egyenletesség indexet (E') használtunk. Az eredmények alapján a gázlómadarak diverzitása a legmagasabb, és ennek a csoportnak az eloszlása a legegyenletesebb. A maximális diverzitást (H′=3,02) és egyenletességet (E′=0,79) a gázlómadarak esetén regisztráltuk 2019-ben, a legkisebb értékeket (H′=1,02, E′=0,34) pedig a vízimadaraknál 2016-ban. A relatív diverzitás index megerősíti az Anatidae család dominanciáját. A térség madarait jelentősen veszélyezteti a területen végzett emberi beavatkozás.

Kulcsszavak: Shannon-Wiener diverzitás index, Purbasthali vizes élőhely, habitat, relatív diverzitás index, vonuló madarak

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Introduction

Wetlands are considered as home of unique and diverse species of plants and animals especially the water birds (Garg 2015). Freshwater wetlands harbour more than 40% of bird species worldwide (Rajpar & Zakaria 2010). Birds live in wetlands and its surroundings are known as 'wetland birds' which include waterfowl, waders/shorebirds and birds reliant or associated to wetlands (Kumar *et al.* 2005). All species in the first two groups depend on the wetland, live in wetlands or its banks/ water edge for nesting, breeding, feeding and roosting, whereas the last category live in the nearby trees and scrubs.

Congregation of waterfowls in any wetland denotes the rich health of the waterscape and *vice versa* (Gregory 2006). Subsequently, wetland birds play crucial roles in sustaining the natural balance of the aquatic ecosystem (Clout & Hay 1989), possess different trophic levels of a food chain and thus, help supply of energy and maintain species diversity in an ecosystem (Hadley *et al.* 2012), carry out the role of predators, pollinators, herbivores, pest controller, agents of seed dispersal, and vectors of invertebrates and nutrients (Bibi & Ali 2013). Moreover, they are also considered as very sensitive to any sort of alterations in the environment (Koli 2014) and thus, they are often rationally called as effective bio-indicators of the wetland ecosystem (Li *et al.* 2009).

Various authors like Kumar *et al.* (2016), Chatterjee *et al.* (2017), Chen *et al.* (2019), Hamilton *et al.* (2019) and Luo *et al.* (2019) have analysed the distribution, density, diversity, composition and abundance of wetland birds around the world with diverse perspectives and interests. Nowadays, analysing the spatio-temporal dynamism of structure and diversity of avian communities has become essential to assess the impact of anthropogenic activities on the natural systems of wetlands as well as to ascertain the responses of the water birds to such environmental challenges (Cahill *et al.* 2013). Moreover, such studies have been proved to be helpful in explaining the importance of the protection of diverse wetland habitats on the globe for conservation of water birds (Rittiboon & Karntanut 2011).

Endowed with a rich variety of wetland habitats, India provides favourable breeding and wintering grounds for various migratory and vagrant water birds (Hardy *et al.* 1987). About 15% (n=1340) of the bird species of the world are found across the Indian subcontinent (Grimmett *et al.* 2011). Among them, 310 species can be brought under the categories of wetland birds (Grimmett & Inskipp 2007). However, the natural wetlands over the last two centuries have faced tremendous anthropogenic pressure worldwide due to the escalation of human interventions and resultant environmental changes (Turner *et al.* 2000, Kahara *et al.* 2012). More than half of the global wetlands have either been lost or transformed during the past century, and the remaining are experiencing degradation due to reckless human intercession (Fraser & Keddy 2005). Any adverse changes in the function of ecosystem affects the birdlife associated with the wetland (Bhattacharjee & Bargali 2012). Hence, the loss and deterioration of wetlands has not only affected the water birds but also threatened the birds reliant on or associated to wetlands (Ma *et al.* 2010). Studies like Prasad *et al.* (2002) and Reginald *et al.* (2007) have depicted how significant loss of the Indian wetlands has adversely affected the composition of bird community to a significant proportion.

The floodplain wetlands, located over the riparian tract of lower Gangetic plain of West Bengal, are biologically prolific and rich repository of water birds. The Purbasthali wetland is an important repository of various resident and migratory water bird species as it provides favourable space for breeding, foraging, roosting and watering (Mandal & Siddique 2018). Avifaunal diversity of Purbasthali wetland has been studied by Chowdhury (2017), Mandal and Siddique (2018), Mandal *et al.* (2018), Debnath *et al.* (2018) and Chakraborty *et al*. (2021). These works presented a checklist based on both water birds and terrestrial birds living around the wetland. Subsequently, 74 water bird species have been identified by Mandal and Siddique (2018) however; Debnath *et al.* (2018) found 86 species, Chowdhury (2017) reported 89 species while Chakraborty *et al*. (2021) has identified 27 species. Mandal *et al.* (2018) have emphasized how human intervention has reduced the diversity among waders. Thus, it can be stated that its wetland bird assemblages are not properly recorded, evaluated or even documented by the researchers or any proper authority. Therefore, the following objectives have been set up for the present research to address the gaps:

a) to prepare a complete inventory of wetland birds found in Purbasthali Lake based on available secondary data and acquired field-generated primary information;
b) to assess the temporal diversity of avifauna in different habitat niches i.e. open water, water-edge or bank areas and nearby trees/shrubs;
c) to ascertain the existing threats to the avian species of the concerned wetland

Materials and Methods

The Study Site

The Purbasthali oxbow lake (locally known as *Chupi Beel*) is an abandoned channel of the River Bhagirathi on its right bank. The lake extends from 23°25′54″ N to 23° 27′54″ N and 88°19′45″ E to 88°21′54″ E, covering a total water area of 2.19 km^2 (Mandal *et al.* 2018).

The wetland is located along the margin between the districts of Nadia and Purba Barddhaman in the lower Gangetic deltaic region of West Bengal *(Figure 1)*. This water body bears a unique blend of both lacustrine (lentic) and riverine (lotic) ecosystems due to its connectivity with the river at its southern part through a narrow strait (Ganesan & Khan 2008). The crescent-shaped lake was formed during 1989–1991 by the lateral shifting of the river course with simultaneous erosion-deposition processes. Sequential erosion at the left bank with the sediment accretion in its opposite bank has dissociated the meander loop from the prime course of the River (Bandyopadhyay *et al.* 2014).

Data Sources and details of surveying

Both primary and secondary data have been used to fulfil the stated objectives. Due to lack of authentic governmental data, detailed chronological data on avifaunal community encompassing their order, family, species, and population for the years of 2014–2016

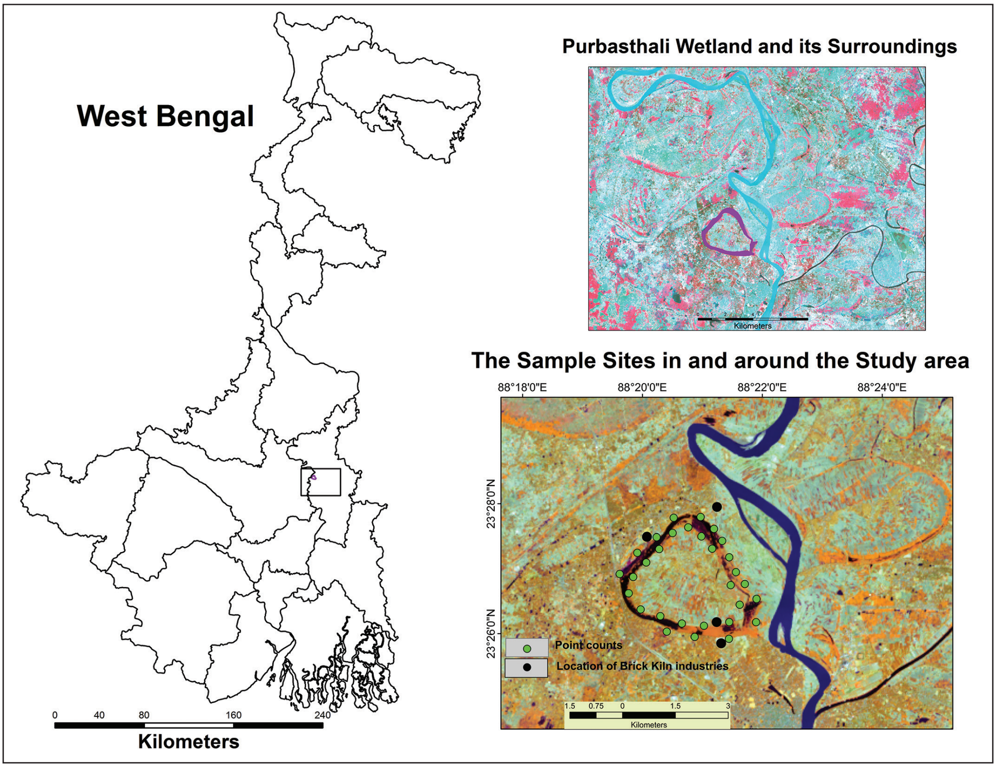

Figure 1. Location of the study area
1. ábra A vizsgálati terület elhelyezkedése

have been accessed from the *Junglees*, a Non-Government Organization (NGO), which professionally collects temporal (annual) data on birds for a long time. The researchers have participated actively in the enumeration processes during 2017–2019 to observe and record the migratory and resident birds found in the wetland. The primary survey has been conducted by following the field guidelines of Ali and Ripley (1987). The bird species in different habitats in and around the wetland have been observed and documented through the point count method. Three types of habitats has been identified from the wetland area i.e. open water, water edge area/transitional zone between land and water, and adjacent trees / shrubs, and bird's data have been collected according to these categories. Point count method, one of the most suitable methods of enumeration of the highly visible bird species across different habitats, has been operated from a fixed location for a definite period at any season of the year (Issa 2019). For the present study, the sighted bird species have been identified, counted and enlisted during the most active day period (i.e. 6 am – 10 am and 4 pm – 6 pm) (Kumar & Gupta 2013) from 30 fixed points *(Figure 1)* with a circle of 50 m radius for 10 minutes interval at every point. The lake has been monitored twice in every month during the aforesaid period to examine the monthly variation in species richness. The actual population under each species has been counted with eye-estimation in January in each year due to greatest abundance of the birds during winter.

A comprehensive inventory including their common (local) name, scientific name, taxonomic position (orders, families, and species), dispersal status, habitat, IUCN status (2017), food habit, and global trends of the population has been prepared after identification, enumeration, and documentation of the avifaunal community of the Purbasthali lake. Works of Mazumdar (2017), Chowdhury (2017), Mandal and Siddique (2018), Debnath *et al.* (2018) and Kumar and Sharma (2018) have intensively studied to frame the checklist. The identified birds have been classified into three categories according to their habitat preference/ location i.e. 'wetland-dependent' or 'waterfowls, 'water edge' or 'waders' and 'wetland associated' or 'terrestrial'. Further, they are also sub-divided into four groups based on the frequency of observation (Khan & Nahar 2009), which are 'Very common' (Vc) (sighted nearly 80–100% during field visit), 'Common' (Co) (50–79.9%), 'Fairly common' (Fc) (20–49.9%) and 'Rare' (Ra) (<19.9%).

Methods of analysis

Species richness refers to the total number of species in a particular habitat. **Relative Diversity Index** (***RDI***) has been used to assess the relative abundance of bird species by using the succeeding formula (Mandal & Siddique 2018):

$$\boldsymbol{RDI = \frac{n}{N} X\ 100}$$

Where, n = Total number of birds in a species, and N = Total number of birds across all species

To determine the habitat wise diversity of birds, **Shannon-Wiener's Diversity Index** (***H'***) has been employed, by following the equation below (Kumar & Sharma 2018):

$$\boldsymbol{H' = -\sum_{i=1}^{s} pi\, ln(pi)}$$

Where, S=Total number of species in the community; pi = Proportion of the total sample fit in to the i^{th} species, and $ln(pi)$ = Natural logarithm of this proportion.

In this study, the diversity of bird communities was assessed within habitat (**α** diversity) and compared between habitats (**β** diversity). Subsequently, the Evenness index (***E'***) was computed to reveal the diversity within and between the selected habitats of the birds by using the following equations (Kumar & Sharma 2018):

$$\boldsymbol{E' = H'/\, ln\, S}$$

Where, ***H'***=Shannon-Wiener's Diversity Index and ***S***=number of species in selected wetland, and natural algorithm.The value of the index ranges from zero and one, where higher values (closer to one) represents higher evenness (Smith & Wilson 1996).

Result and Discussion

Composition and assemblage of avifauna

The ecological suitability of the lake attracts various native as well as migratory bird species (of different orders and families) to settle here or to choose it as a winter destination for feeding, breeding, grazing and swimming. A total of 77 bird species that belong to 10 orders and 19 families directly depended upon or associated with the concerned wetland, have been recorded during the study period. *Table 1* represents the detailed inventory of the identified bird species of the site studied.

The order Charadriiformes has hadthe highest number of families (6) and species (26), whereas the order Anseriformes has represented by only one family i.e. Anatidae, with 14 species. Naturally, the Anatidae family has shown the highest RDI value (18.18) followed by the Charadriidae (14.29), Ardeidae (11.69) and Scolopacidae family (11.69). On contrary, the lowest RDI value (1.30) has been represented by nearly six avian families like Podicipedidae, Rostratulidae, Burhinidae, Hirundinidae, Pandionidae and Apodidae *(Table 2)*.

It should be noted that spatial variations within a natural habitat effectively determine the abundance and occurrence of bird species (Pennington & Blair 2011). In order to understand the impact of habitat structure on the communities, the observed birds species of the area are classified into three groups based on their habitat preferences. The three distinctive habitats, chosen for further analysis, are A. **open water** (**waterfowls**), B. **water edge** or **bank areas** (**waders or shorebirds**), and C. **adjacent trees** and **shrubs** (**wetland associated**). Preference of habitat varies with families according to their feeding habit, availability of nesting materials, and behaviour (Malik & Joshi 2013). Several birds (like herons, egrets) are generally found in the bank areas or in the surrounding agriculture fields, whereas some others (like jacanas) prefer to take shelter in the dense, floating water hyacinth or other hydrophytes. The waterfowls used to dive in open water. Nearby trees and shrubs also provide shelters to some other arboreal birds like kingfishers, Shikra *(Accipiter badius)*, Osprey *(Pandion haliaetus)*. A large part of the wetland is still deep enough to hold a significant volume of freshwater. Moreover, this part of the water body is devoid of any type of hydrophytes, especially water hyacinths. The presence of such a vast open water surface provides a suitable arena for the members of the Anatidae family as they mostly prefer such type of environment (Benoit & Askins 1999). Naturally, the relative diversity of the Anatidae family has reported the maximum. On contrary, a notable proportion of the bank areas or water edge areas are used for cropping purposes. Even a larger segment of the shallow submerged water edge areas has been reclaimed and utilized as seedbeds. The abundance of waders (shorebirds) is getting influentially controlled by the availability of food (mainly small invertebrates) in the agricultural fields as invertebrates are less available in deep water (Murkin & Kadlec 1986). Increasing areas of cropland offer the ecological niche of several bird families like Charadriidae, Ardeidae, and Scolopacidae. Though, they are confined to the bank areas and adjacent agricultural fields, their relative diversity are significant.

Among the recorded species, 50.56% are migratory birds, who visit the wetland during the winter days, whereas 49.44% are resident birds *(Figure 2)*. The most common resident birds

Table 1. Checklist of wetland birds found in and around of the Purbasthali Wetland
Dispersal Status: R resident, M migratory;
Habitat Location: OW open water, WE water edge, T trees and Shrubs;
Abundance: ++++ very common, +++ common, ++ fairly common, + rare;
IUCN Status: LC least concern, NT near threatened, VU vulnerable;
Trend: ? unknown, ↑ increasing, ↓ decreasing, → stable
1. táblázat A Purbasthali vizes élőhelyen előforduló madárfajok

Order	**Family**	**Common name**	**Scientific name**	**Dispersal Status**	**Habitat location**	**Abundance**	**IUCN status**	**Trend**	**Feeding Habits**
Podicipediformes	**Podicipedidae**	Little Grebe	*Tachybaptus ruficollis*	R	OW	++++	LC	↓	Carnivorous
Pelecaniformes	**Phalacrocoracidae**	Great Cormorant	*Phalacrocorax carbo*	R	OW	++	LC	↑	Carnivorous
		Indian Shag	*Phalacrocorax fuscicollis*	R	OW	+++	LC	?	Carnivorous
		Little Cormorant	*Microcabro niger*	R	OW	+++	LC	?	Carnivorous
	Ardeidae	Grey Heron	*Ardea cinerea*	R	WE	+++	LC	?	Carnivorous
		Purple Heron	*Ardea purpurea*	R	WE	+++	LC	↓	Carnivorous
		Indian Pond Heron	*Ardeola grayii*	R	WE	++++	LC	?	Carnivorous
		Night Heron	*Nycticorax nycticorax*	R	WE	+++	LC	↓	Carnivorous
		Cattle Egret	*Bubulcus ibis*	R	WE	++++	LC	↑	Carnivorous
		Intermediate Egret	*Ardea intermedia*	R	WE	++++	LC	↓	Carnivorous
		Great Egret	*Ardea alba*	R	WE	+++	LC	?	Carnivorous
		Little Egret	*Egretta garzetta*	R	WE	++++	LC	↑	Carnivorous
		Yellow Bittern	*Ixobrychus sinensis*	R	WE	++	LC	?	Carnivorous
	Threskiornithidae	Glossy Ibis	*Plegadis falcinellus*	R	WE	+	LC	↓	Carnivorous
		Black Ibis	*Pseudibis papillosa*	R	WE	+	LC	↓	Carnivorous
		Spoon Bill	*Platalea leucorodia*	R	WE	+++	LC	?	Carnivorous
		Black-headed Ibis	*Threskiornis melanocephalus*	R	WE	++	NT	↓	Carnivorous

Order	Family	Common name	Scientific name	Dispersal Status	Habitat location	Abundance	IUCN status	Trend	Feeding Habits
Ciconiiformes	Ciconiidae	Asian Openbill Stork	*Anastomus oscitans*	M	WE	++++	LC	?	Carnivorous
		Lesser Adjutant Stork	*Leptoptilos javanicus*	R	WE	+	VU	↓	Carnivorous
Anseriformes	Anatidae	Greylag Goose	*Anser anser*	M	OW	+	LC	?	Omnivorous
		Ruddy Shelduck	*Tadorna ferruginea*	M	OW	+++	LC	?	Omnivorous
		Northern Pintail	*Anas acuta*	M	OW	+++	LC	↓	Omnivorous
		Common Teal	*Anas crecca*	M	OW	+++	LC	?	Omnivorous
		Cotton Pygmy-goose	*Nettapus coromandelianus*	M	OW	++++	LC	→	Omnivorous
		Gadwall	*Marecas trepera*	M	OW	+++	LC	↑	Omnivorous
		Eurasian Pigeon	*Mareca penelope*	M	OW	++	LC	↓	Omnivorous
		Garganey	*Spatula querquedula*	M	OW	+++	LC	↓	Omnivorous
		Northern Shoveller	*Spatula clypeata*	M	OW	+++	LC	↓	Omnivorous
		Common Pochard	*Aythya ferina*	M	OW	++	VU	↓	Omnivorous
		Tufted Duck	*Aythya fuligula*	M	OW	++	LC	→	Omnivorous
		Red-crested Pochard	*Netta rufina*	M	OW	++++	LC	?	Omnivorous
		Lesser Whistling Duck	*Dendrocygna javanica*	R	OW	++++	LC	↓	Omnivorous
		Ferruginous Pochard	*Aythya nyroca*	M	OW	+	NT	↓	Omnivorous
Gruiformes	Rallidae	Bailon'sCrake	*Zapornia pusilla*	M	WE	+	LC	?	Omnivorous
		Water Cock	*Gallicrex cinerea*	R	WE	+++	LC	?	Omnivorous
		Purple Swamp Hen	*Porphyrio porphyrio*	R	WE	+++	LC	?	Omnivorous
		White-breasted Waterhen	*Amaurornis phoenicurus*	R	WE	+++	LC	?	Omnivorous
		Indian Moorhen	*Gallinula chloropus*	R	WE	+++	LC	→	Omnivorous
		Common Coot	*Fulica atra*	R	WE	+++	LC	↑	Omnivorous

Order	**Family**	**Common name**	**Scientific name**	**Dispersal Status**	**Habitat location**	**Abundance**	**IUCN status**	**Trend**	**Feeding Habits**
Charadriifofmes	**Jacanidae**	Bronze-winged Jacana	*Metopidius indicus*	R	WE	+++	LC	?	Omnivorous
		Pheasant-tailed Jacana	*Hydrophasianus chirurgus*	R	WE	+	LC	↓	Omnivorous
	Charadriidae	River Lapwing	*Vanellus duvaucelii*	M	WE	+	NT	↓	Omnivorous
		Grey-headed Lapwing	*Vanellus cinereus*	M	WE	++	LC	?	Omnivorous
		Red-wattled Lapwing	*Vanellus indicus*	R	WE	++++	LC	?	Omnivorous
		Yellow-wattled Lapwing	*Vanellus malabaricus*	R	WE	++	LC	?	Omnivorous
		Pacific Golden Plover	*Pluvialis fulva*	M	WE	++	LC	?	Carnivorous
		Little Ringed Plover	*Charadrius dubius*	M	WE	++	LC	?	Carnivorous
		Little Stint	*Calidris minuta*	M	WE	++	LC	?	Insectivorous
		Marsh Sandpiper	*Tringa stagnatilis*	M	WE	++	LC	?	Insectivorous
		Ruff	*Philomachus pugnax*	M	WE	+	LC	↓	Insectivorous
		Lesser Sand Plover	*Charadrius mongolus*	M	WE	++	LC	?	Carnivorous
		Black-Winged Stilt	*Himantopus himantopus*	M	WE	+++	LC	↑	Insectivorous
	Scolopacidae	Common Snipe	*Gallinago gallinago*	M	WE	+	LC	↓	Insectivorous
		Common Red Shank	*Tringa tetanus*	M	WE	++	LC	?	Insectivorous
		Green Shank	*Tringa nebularia*	M	WE	+	LC	↑	Insectivorous
		Spotted Redshank	*Tringa erythropus*	M	WE	++	LC	?	Insectivorous
		Common Sandpiper	*Actitis hypoleucos*	M	WE	++++	LC	↓	Insectivorous
		Wood Sandpiper	*Tringa glareola*	M	WE	+	LC	→	Insectivorous
		Green Sandpiper	*Tringa ochropus*	M	WE	+	LC	?	Insectivorous
		Pin-Tailed Snipe	*Gallinago stenura*	M	WE	++++	LC	?	Insectivorous
		Temminck's Stint	*Calidris temmincki*	M	WE	+	LC	?	Insectivorous
	Rostratulidae	Painted Snipe	*Rostratula benghalensis*	R	WE	++	LC	?	Omnivorous
	Burhinidae	Great Stone Plover	*Esacusre curvirostris*	R	WE	+	NT	?	Carnivorous
	Laridae	Whiskered Tern	*Chlidonias hybrid*	M	OW	+++	LC	→	Omnivorous
		River Tern	*Sterna aurantia*	R	OW	++++	NT	↓	Omnivorous

Order	Family	Common name	Scientific name	Dispersal Status	Habitat location	Abundance	IUCN status	Trend	Feeding Habits
Passeriformes	Hirundinidae	Barn Swallow	*Hirundo rustica*	R	T	++++	LC	↓	Insectivorous
	Motacillidae	White Wagtail	*Motacilla alba*	M	T	+	LC	→	Insectivorous
		Yellow Wagtail	*Motacilla flava*	M	T	++	LC	↓	Insectivorous
		CitrineWagtail	*Motacilla citreola*	M	T	+	LC	↑	Insectivorous
Coraciiformes	Alcedinidae	Pied Kingfisher	*Ceryle rudis*	R	T	+++	LC	?	Carnivorous
		Common Kingfisher	*Alcedo atthis*	R	T	+++	LC	?	Carnivorous
		Stork-billed Kingfisher	*Pelargopsis capensis*	R	T	+++	LC	↓	Carnivorous
		White-breasted Kingfisher	*Halcyon smyrnensis*	R	T	+++	LC	↑	Carnivorous
Accipitriformes	Accipitridae	Marsh Harrier	*Circus spilonotus*	M	T	+	LC	↑	Carnivorous
		Indian Shikra	*Accipiter badius*	R	T	+++	LC	→	Carnivorous
	Pandionidae	Osprey	*Pandion haliaetus*	R	T	++	LC	?	Carnivorous
Apodiformes	Apodidae	Asian Palm Swift	*Cypsiurus balasiensis*	R	T	++	LC	?	Insectivorous

Table 2. Relative diversity of avian families
2. táblázat A madár családok relatív diverzitása

Name of the Families	No of Species	RDI	Name of the Families	No of Species	RDI
Podicipedidae	1	1.30	Rostratulidae	1	1.30
Phalacrocoracidae	3	3.90	Burhinidae	1	1.30
Ardeidae	9	11.69	Laridae	2	2.60
Threskiornithidae	4	5.19	Hirundinidae	1	1.30
Ciconidae	2	2.60	Motacillidae	3	3.90
Anatidae	14	18.18	Alcedinidae	4	5.19
Rallidae	5	6.49	Accipitridae	3	3.90
Jacanidae	2	2.60	Pandionidae	1	1.30
Charadriidae	11	14.29	Apodidae	1	1.30
Scolopacidae	9	11.69	Total (N)	77	100

found in the area are Lesser Whistling Duck *(Dendrocygna javanica)*, Pond Heron *(Ardeola grayii)*, Cattle Egret *(Bubulcus ibis)*, while, Red-crested Pochard *(Netta rufina)*, Ruddy Shelduck *(Tadorna ferruginea)*, Gadwall *(Mareca strepera)* are the few most abundant migrants. Nearly 18.18% of the species have been observed very frequently, hence are termed as 'very common'. Another 33.77% of birds are common and have been sighted frequently. Nearly 24.68% of birds are fairly common and the rest 23.38% of birds are rare *(Figure 2)*. Two vulnerable species i.e. Common Pochard *(Aythya ferina)* (Order: Anseriformes, Family: Anatidae) and Lesser Adjutant Stork *(Leptoptilos javanicus)* (Order: Ciconiiformes, Family:

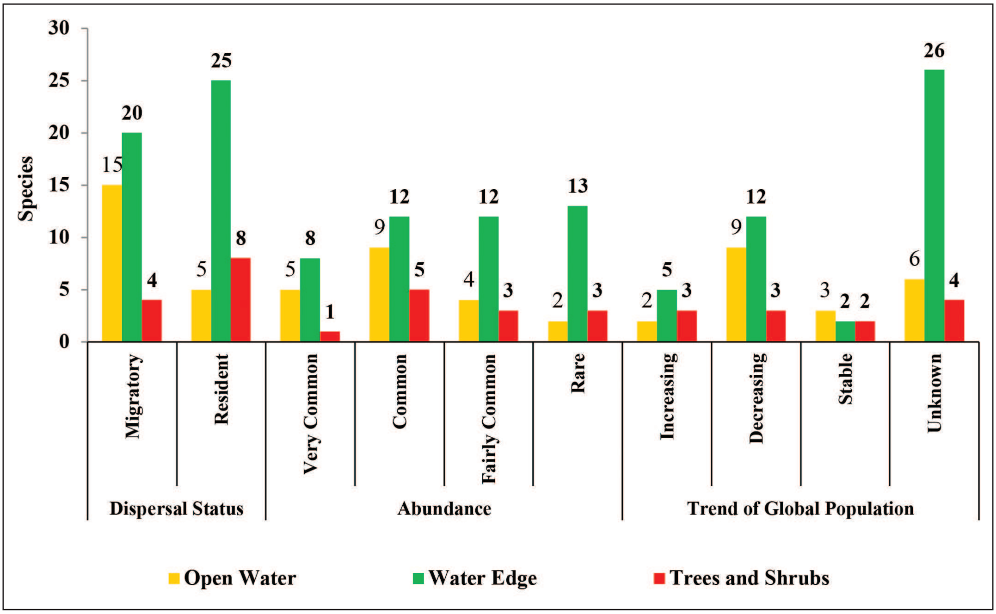

Figure 2. Status of the wetland birds, Purbasthali oxbow lake
2. ábra A vízimadarak jellemzői a Purbasthali holtág területén

Ciconidae) as per IUCN's Red Data Book has been found during the survey. Furthermore, five near-threatened species namely Black-headed Ibis *(Threskiornis melanocephalus)* (Order: Pelecaniformes, Family: Threskiornithidae), Ferruginous Pochard *(Aythya nyroca)* (Order: Anseriformes, Family: Anatidae), and three members of Charadriiformes, i.e. River Lapwing *(Vanellus duvaucelii)* (Family: Charadriidae), Great Stone Plover (curlew) *(Esacus recurvirostris)* (Family: Burhinidae) and River Tern *(Sterna aurantia)* (Family: Laridae) have been observed in the area during enumeration. *Figure 2* shows the dispersal status, abundance and global trend of bird species classified according to habitat.

The observed bird community of the wetland has been sub-divided into three groups according to their habitat associations. Nearly ¼ (25.97%) of the identified bird species have been found in the deep and clean open water area. All the 14 species of Anatidae family (Order: Anseriformes), one species of Podicipedidae family (Order: Podicipediformes), three species of Phalacrocoracidae (Order: Pelecaniformes) and two species of Laridae family (Order: Charadriiformes) have been recorded in the area fall in this category. On contrary, the maximum share of bird species (58.44%) has been observed at the water edge areas of the wetland. Families like Ardeidae, Charadriidae, Scolopacidae, Jacanidae prefer to live in those transitional zones and nearby crop fields due to the easy availability of food and shelters. Besides, several other bird species like kingfishers, Asian Palm Swift *(Cypsiurus balasiensis)*, Osprey, Shikra, and wagtails have been recorded at nearby trees and shrubs of the wetland, which constitutes 15.58% of the avian community.

It should be further mentioned that nearly 32.43% of the recorded species observed in the area show a declining trend in their global population, whereas 13.51% have reported growth at the global level. Only 9.46% bird species reveal a stable condition, whereas the global population status of nearly half of the observed species is unidentified to date. Moreover 37.66% of birds observed in the site are omnivores, 38.96% are carnivores and the rest 23.38% are insectivores. The concerned water body provides all kind of food and prey for the birds found here.

Temporal Variation of Species Richness and abundance

The time series analysis exhibits that all of these three categories of wetland birds found in the area show a positive trend of population growth over the observation period of 2014–2019 *(Figure 3)*, which is conclusively a promising fact. The birds live in open water and bank areas/ water edge show quite fluctuations in their abundance. On contrary, the birds sheltered in the nearby trees exhibit a slow but steady growth, though their share in the total population is very low (3.1% in 2019) *(Table 3)*.

Species richness of the waders is much higher than the other two categories, while the birds dependent on wetlands exhibit the lowest species richness. Maximum 20 species of waterfowls were observed during 2017–2019, while their lowest richness (17) was found in 2014. In case of the waders, the lowest richness (31) were recorded in 2014 and the maximum (44) in 2019 *(Figure 4)*.The temporal fluctuation in the richness of bird species preferring trees/shrubs is insignificant. It should be specified that, despite less species diversity, the population size of the waterfowls are notably higher than the birds of other habitats. They comprise the lion's share of the total bird population of the area over the years. For example,

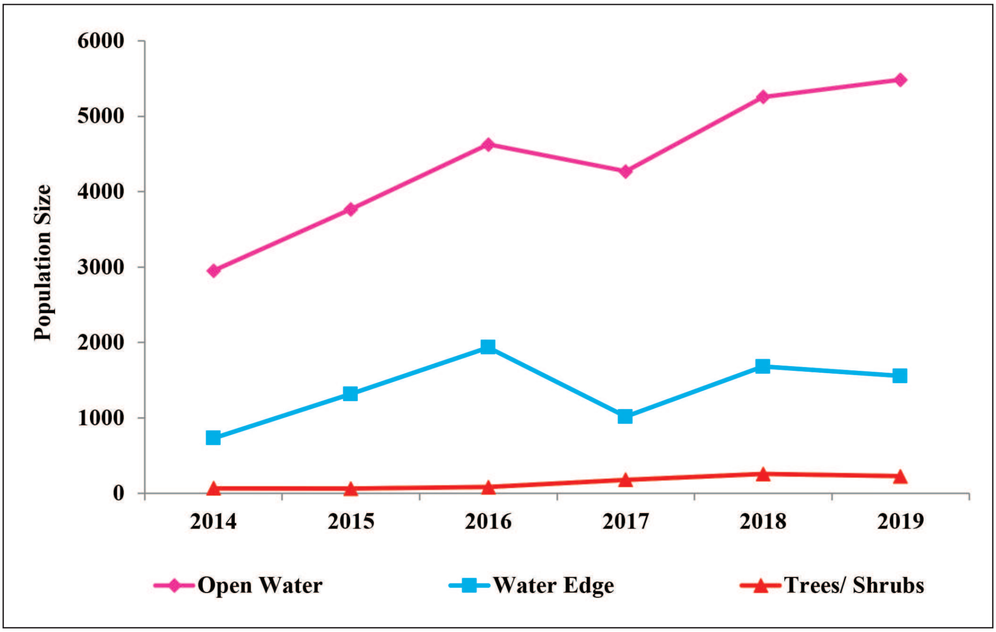

Figure 3. Temporal variation in bird population, 2014–2019
3. ábra A madárpopulációk nagyságának változása 2014–2019 között

in 2019 the recorded number of the waterfowls was 5,485, which accounts for 75.50% of the bird population *(Table 3)*. On the contrary the number of waders and wetland associates was only 1,555 (21.40%) and 225 (3.10%) respectively. More specifically, the greater number of the Lesser Whistling Duck is solely responsible for the dominance of waterfowls in the birds' flock. For example, the number of Lesser Whistling Ducks was 4,008 in 2019, which comprises73.07% of the waterfowls and 55.17% of the total bird population of the area. Since 2014 the Lesser Whistling Duck has constantly occupied the dominant position in the birds' colony and shown an increasing trend over time. From 2014 to 2018, this specific bird species accounts for nearly 38.72%, 48.60%, 53.46%, 53.47% and 51.99% of the total bird population of the area respectively. Hence, they have been intentionally excluded

Table 3. Year-wise birds population, 2014–2019
3. táblázat A különböző madárpopulációk egyedszáma 2014–2019 között

Year	Waterfowls		Waders		Wetland Associated		Total
	Number	%	Number	%	Number	%	
2014	2,952	78.72	730	19.47	68	1.81	**3,750**
2015	3,766	73.18	1,317	25.59	63	1.22	**5,146**
2016	4,626	69.66	1,933	29.11	82	1.23	**6,641**
2017	4,268	78.13	1,016	18.60	179	3.28	**5,463**
2018	5,255	73.06	1,681	23.37	257	3.57	**7,193**
2019	5,485	75.50	1,555	21.40	225	3.10	**7,265**

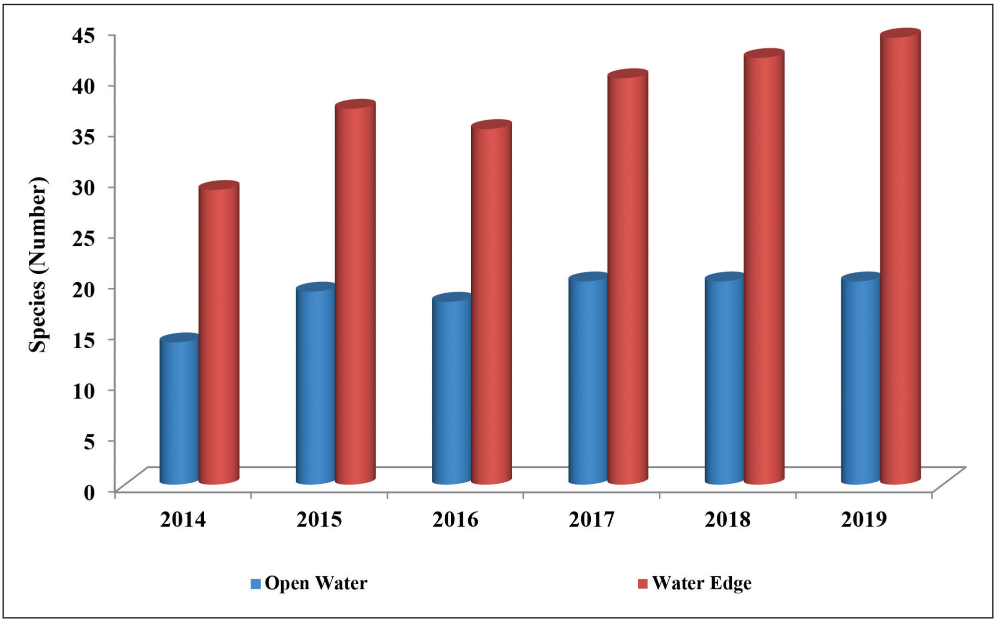

Figure 4. Temporal variation in species richness, 2014–2019
4. ábra A fajgazdagság változása 2014–2015-ben

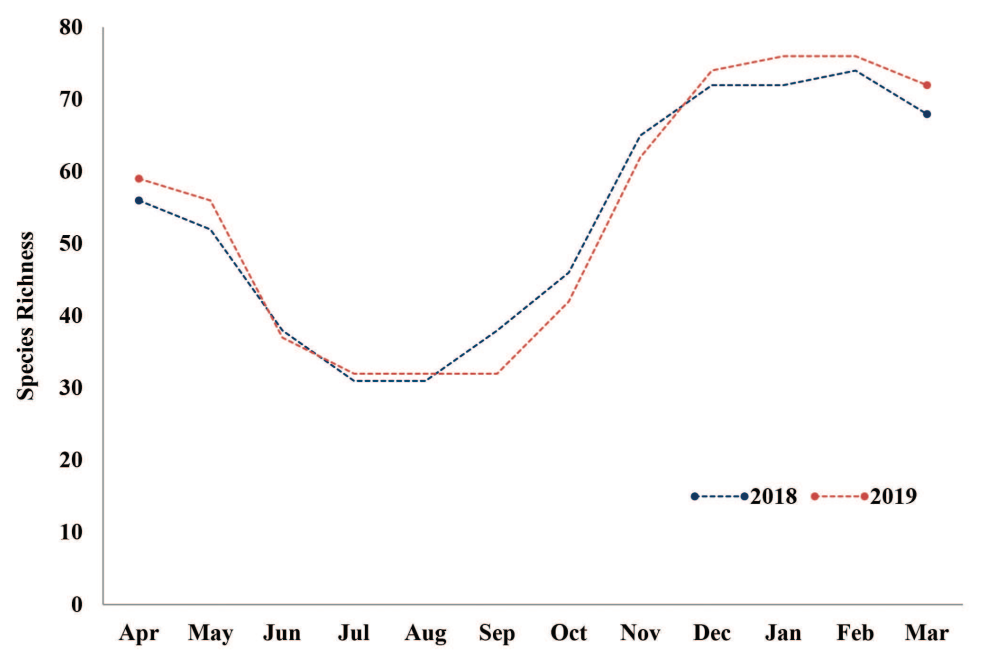

Figure 5. Monthly variationin species richness, 2018–2019
5. ábra A fajgazdagság havonkénti változása 2018–2019-ben

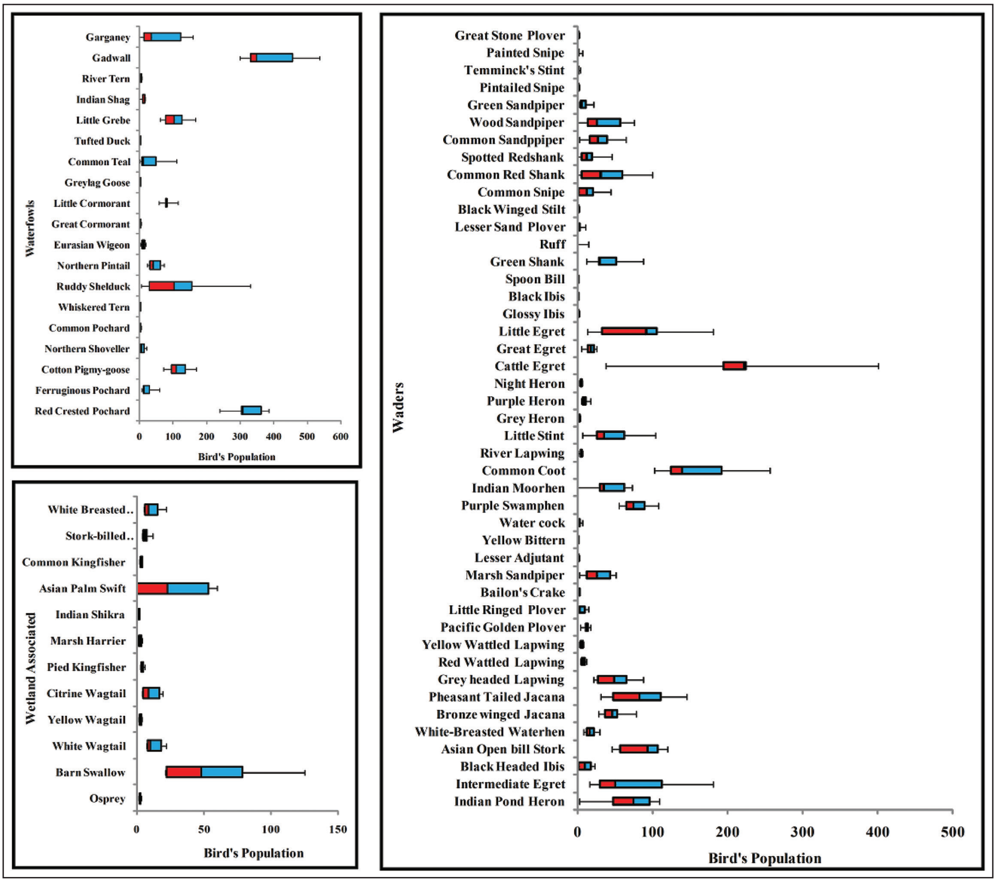

Figure 6. Temporal variation of population of bird species
6. ábra A madárpopulációk időbeli változása

while creating box plots, as their huge fleet distorts the diagrammatic representation of the temporal variation of species *(Figure 6)*.

The species richness has achieved its maximum level during the winter season due to the inflow of a considerable number of migratory birds, while it was reduced during the monsoon *(Figure 5)*. The primary survey has identified 39 winter migrants in the study site, which normally arrive at the beginning of November and stay till the mid of March. Thus, the wetland turns into bliss of migratory birds and a popular winter destination for the bird-watchers and nature-lovers. The monthly variation of species richness in 2018 and 2019 showed an almost similar trend *(Figure 5)*. During the rainy days, adjacent lowlands, floodplains, ponds or any other surface depressions, even the nearby paddy fields get inundated. As a consequence, the extent of the natural habitat as well as the ecological niche of the resident birds gets expanded. This phenomenon helps the birds to spread over a vast area, which reduces species richness in the specific wetland. On the contrary, in summer the wetland becomes the only source of food and shelter for the water birds as other water storages of the area become dried up. Naturally, the species richness lies at a higher level. Additionally, the arrival of a few summer migrants (like Lesser Whistling Duck) and residents (like Pheasant-tailed Jacana or *Hydrophasianus chirurgus*, Pond Heron, and egrets) has enhanced the species richness during those days.

The temporal data also reveals that most of the waterfowl and shorebird species show remarkable variability in their numbers within the stipulated period *(Figure 6)*. In contrast, the wetland associated birds exhibit striking consistency in number, except one species, of Barn Swallow *(Hirundo rustica)* (Order: Passeriformes, Family: Hirundinidae). Primarily, the members of the Ardeidae family alike various types of egret (Cattle Egret, Little Egret, Intermediate Egret) and heron (Indian Pond Heron) have shown wide fluctuations in their population. Similarly, the flocks of a few open water bird species like Ruddy Shelduck, Garganey *(Spatula querquedula)*, Gadwall *(Mareca strepera)* and Common (Eurasian) Teal *(Anas crecca)* have shown oscillating nature over the stipulated period.

Habitat wise Species Diversity, Richness, and Distribution

The Shannon-Wiener Diversity Index (H′) and Evenness index (E′) is computed for every year separately based on the habitat-wise gathered data *(Table 4)*. Both the H′ and E′ values are found to be less for the open water bird species compared to the other species lived in the water edge area and nearby trees/shrubs during the stipulated period of study. On an average, the birds in water edge areas record the highest H′ value (2.90), whereas, species that live in the trees show the maximum evenness value (0.78). The highest H′ value (3.02) is possessed by the water edge birds during 2019, and maximum E′ value (0.94) is reported by the birds of trees/shrubs in 2014. Similarly, the lowest H′ (1.02) and E′ value (0.34) were recorded for the open water birds in 2016. It has also been observed that the diversity values for the bird species of water edge areas show insignificant fluctuation, whereas birds of the other two categories confirm distinctive variation over the observation periods.

The birds settled in nearby trees / scrubs have shown the lowest species richness, though their H′ and E′ values are higher than open water birds. Maximum species richness but low population figure has been recorded for the water edge birds, where population under various species is not much different from each other and no species can claim its dominance. Hence,

Table 4. Habitat-wise Species Diversity (H′) and Evenness Index (E′)
4. táblázat Fajdiverzitás és kiegyenlítettség a különböző élőhelyeken

Year	Open Water		Water Edge		Trees and Shrubs	
	H′	E	H′	E	H′	E
2014	1.70	0.59	2.83	0.74	2.33	0.94
2015	1.34	0.45	2.95	0.77	2.14	0.86
2016	1.02	0.34	2.83	0.74	2.01	0.81
2017	1.19	0.40	2.82	0.74	1.76	0.71
2018	1.19	0.40	2.97	0.78	1.57	0.63
2019	1.14	0.38	3.02	0.79	1.83	0.73
Average	1.26	0.43	2.90	0.76	1.94	0.78

the diversity of waders and wetland associated birds has become high alike the evenness. On the contrary, low H′ and E′ values of open water birds reveal the remarkable inter-species difference in abundance of the birds and relative dominance of one or two species. The massive dominance of Lesser Whistling Duck in the open water certainly leads to the lowest species diversity and evenness in the studied wetland.

Present threats to the wetland birds

Purbasthali Wetland has been the source of different services to the native rural folks and thus they have utilized the water body in nearly twelve ways such as the collection of material goods (like food, fodder, fuel, clay, humus, etc.), fishing, cattle bathing, bird watching, boating, wetland agriculture (Mandal *et al.* 2020). Indiscriminate extraction of resources and higher degree of utilization nowadays has imposed serious threats upon the physico-chemical as well as biological health of the concerned wetland (Mandal *et al.* 2018). Worldwide anthropogenic interference has extensively damaged the natural habitat of birds through isolation and fragmentation, which have crucially impacted upon the existence and variety of birds (Westphal *et al.* 2006). Human-induced alterations in land use/cover mosaic in different parts of the globe have driven out a significant number of bird species from their original habitats (Burgess *et al.* 2002).

Discharge of effluent, infilling, conversion of bank areas or water edge areas for agriculture, aquaculture, over extraction and utilization of wetland resources are few most common forms of threats, in which reclamation has been pointed out as the most important cause for endangering the birds' species in the Asian region (Kumar *et al.* 2005). The present study site is not an exception. The authors have identified four such distinct human operations, which have become bane to the biodiversity of the lake. These are agricultural expansion and shrinkage of the wetlands area, pollution, establishment of brick kilns *(Figure 1)* and hunting/ trapping of birds. In addition, the use of pesticides and insecticides in the crop fields reduces the food availability for the birds. The use of the sub-marshal pumps for irrigation purposes creates noises which affect the bird abundance in the bank areas. Moreover, farmers often use firecrackers to repel the waders from their fields along the banks of the wetland. The agricultural activities at the bank areas date back to the origin of the water loop, which has expanded gradually and at present a sizable portion of the banks has been transformed into permanent or semi-permanent crop fields through the reclamation of the wetland area. Such human activities have turned the ecotone (transitional areas) into the zone of human-bird conflict, which has directly destroyed the habitat as well as the ecological niche of the waders and affecting their abundance. As the response of birds to changes in habitat differs according to their strategies, few species have successfully adopted the change, while few others get tremendously affected due to their incapability to do so.

The Land Use and Land Cover (LULC) analysis *(Table 5)* of the studied region has been prepared by Maximum Likelihood technique in Arc-GIS. Landsat Satellite Images of 30 meter resolution of 1990, 2000, 2010 and 2020 have been used for the analysis. The LULC is extracted on the basis of 3 km radius from centre of the wetland that exhibits the transformation of land use/cover mosaic over the time. With the help of LULC exact transformation of certain

Table 5. Conversion of water area into other land use categories, 1990–2020
5. táblázat A vízterület átalakítása más földhasználati kategóriákba 1990–2020 között

Land use land cover class/Year	**1990**	**2000**	**2010**	**2020**	**Total**
Water bodies	4.0797	3.492	2.2419	1.4598	11.2734
Agriculture land	11.5128	13.6836	14.7096	15.192	55.098
Vegetation	6.6438	5.346	4.7448	1.9557	18.6903
Built up area	2.007	4.5441	6.2307	9.405	22.1868
Other	4.0419	1.2195	0.3582	0.2727	5.8923

land categories are found. *Table 5* disclose rapid shrinkage of the water body from 4.08 km^2 in 1990 to 1.45 km^2 in 2020 with significant increment of cropland and built-up space. Expansion of agricultural land and incessant urban growth are the two prime responsible factors of such conversion. The agricultural area has been increased from 11.51 km^2 in 1990 to 15.19 km^2 in 2020, while the urban area has also been increased from 2 km^2 to 9.41 km^2 in 2020 within the stipulated period. Such swift depletion of open water space of the wetland will become the most potential threat to wetland birds of the area in near future. The works of Mandal and Siddique (2018) and Mandal *et al.* (2018) have already stated that agricultural expansion in the Purbasthali wetland has lessen the abundance of the waders.

In spite of higher degree of human intervention, the water edge areas provide the richest habitat for the birds in terms of species assemblage. *Figure 2* shows 45 species are found in those areas, out of which 20 are migratory and 25 are resident. Nearly 20 species are common to very common, whereas another 13 species are rarely seen. Besides, 12 birds species, observed in the banks, have shown a declining trend in numbers at the global level. One vulnerable species i.e. Lesser Adjutant Stork and two near threatened species namely Great Stone Plover (Order: Charadriiformes, Family: Burhinidae) and Black-headed Ibis (Order: Pelecaniformes, Family: Threskiornithidae) are also found in this zone. Thus, the water edge areas need attention for conservation because further degradation and interruption may cause reduction in the population size or extinction of species. In comparison to the other two types, the waders have shown very less increment in their fleet over the past six years. Apart from them, the waterfowls and the wetland associated birds also need protection. For example, the problem of eutrophication, algal bloom and other invasive species have thwarted the diving and grazing habits of open water birds in many parts of the lake. Over extraction of resources and exhaustive uses of the water body by human groups, trim down the appositeness of the wetland as the habitat of birds. Preventive and curative measures are immediate needs to resolve those problems.

Some precautionary measures were taken through community participation to restrain the negative impact of human interference as well as to make Purbasthali attractive to migratory birds. The local people have also started to get benefits from the promotion of seasonal tourism based on the migratory birds in winter. As a result, the stakeholders have become interested to protect the migratory birds, whereas clear negligence for the resident species has been witnessed. Based on its avifaunal diversity, the lake possesses an immense possibility to be developed as an ecotourism site.

To make the wetland a healthy habitat for birds, the Purbasthali Lake, particularly its bank areas should be protected from human interventions. Use of chemical pesticides/fertilizers in crop fields should be restricted. Some stringent prohibition should be imposed on unrestrained picnic activities, especially the high decibel sound boxes. The ecotourism guidelines should be strictly imposed to control tourist activities. Plastic material and food packets should be banned while boating and bird-watching. The more scientific method should be adopted for fishing, which will not disturb the assembly of the birds. Above all, mass awareness is imperative to sustain the health of the wetland and its ecological pulse.

Conclusion

Although the value of ecosystem services extended by the Purbasthali wetland is immense in nearby rural life and livelihood, its role as a habitat of birds is equally great, rather more imperative. The *beel* hosts an excellent avifaunal diversity. Thus, its appraisal as mere human common pool resource indeed seems prejudiced and unjust. Now, it has become crucial to assess the ecological value of the concerned wetland in regulating the nature and conserving the biodiversity. Thus, it is essential to gather detailed knowledge and database about its floral and faunal communities. The study represents a comprehensive and informative account of the wetland birds of Purbasthali Lake, which may help the researchers in further studies or the administration in future planning. It is promising that despite conspicuous human intervention, the lake still provides food and shelter to 77 bird species, resident and migratory. This phenomenon signifies that the state of the health of the lake is good enough, though at present the avifauna of the area faces a high degree of human disturbances, which may prove disadvantageous to the abundance of few bird species. Specifically, the waders are facing serious threats due to human encroachment in the bank areas. The entire ecological set up of the Purbasthali Lake has been vulnerable by anthropogenic interference. In such a context, community participation in management of the wetland may become a solution for sustaining the biological resources, especially the birds. Proper management of the *beel* will not only improve the environmental condition for its resident species, but will also attract more migratory and vagrant species in near the future. Awareness among the stakeholders along with the efforts of the local authorities in a participatory manner regarding the conservation of the wetland will help to retain the ecological propriety of this habitat (wetland) of avifauna. Furthermore, the preservation efforts will be equally auspicious to the local human groups too.

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References

Ali, S. & Ripley, S. D. 1987. Compact Handbook of the Birds of India and Pakistan together with those of Bangladesh, Nepal, Bhutan and Sri Lanka. – Oxford University Press, Delhi

Bandyopadhyay, S., Kar, N. S., Das, S. & Sen, J. 2014. River systems and water resources of West Bengal: a review. – Geological Review of India Special Publication 3: 63–84. DOI: 10.17491/cgsi/2014/62893

Benoit, L. K. & Askins, R. A. 1999. Impact of the spread of *Phragmites* on the distribution of birds in Connecticut tidal marshes. – Wetlands 19(1): 194–208.

Bhattacharjee, A. & Bargali, H. S. 2012. Status and trend of water birds in wetlands around Corbett. – The Corbett Foundation, Uttarakhand, India

Bibi, F. & Ali, Z. 2013. Measurement of diversity indices of avian communities at taunsa barrage wildlife sanctuary, Pakistan. – The Journal of Animal & Plant Science 23(2): 469–474.

Burgess, N. D., Doggart, N. & Lovett, J. 2002. The Uluguru Mountains of Eastern Tanzania: the effect of forest loss on biodiversity. – Oryx 36(2): 140–152. DOI: 10.1017/S0030605302000212

Chakraborty, A., Barman, H., Saha, G. K. & Aditya, G. 2021. Waterbird assemblage in an emerging wetland of West Bengal, India: characterization for conservation management. – Ornis Hungarica 29(1): 1–19. DOI: 10.2478/orhu-2021-0001

Chatterjee, A., Adhikari, S. & Mukhopadhyay, S. K. 2017. Effects of waterbird colonization on limnochemical features of a natural wetland on Buxa Tiger Reserve, India during wintering period. – Wetlands 37: 177–190. DOI: 10.1007/s13157-016-0851-7

Choudhury, S. 2015. A study on avifaunal species diversity of Purbasthali Oxbow Lake, West Bengal, India. – International Journal of Science and Research 6(3): 866–871.

Chen, Q., Xu, G., Wu, Z., Kang, P., Zhao, Q., Chen, Y., Lin, G. & Jian, S. 2019.The effects of winter temperature and land use on mangrove avian species richness and abundance on Leizhou Peninsula, China. – Wetlands 40(1): 153–166. DOI: 10.1007/s13157-019-01159-6

Clout, M. N. & Hay, J. R. 1989. The importance of birds as browsers, pollinators and seed dispersers in New Zealand forests, New Zealand. – New Zealand Journal of Ecology 12(Suppl.): 27–33.

Debnath, S., Biswas, S. & Panigrahi, A. K.2018. Present status and diversity of avian fauna in Purbasthali bird sanctuary, West Bengal, India. – Agricultural Science Digest 38(2): 95–102. DOI: 10.18805/ag.d-4711

Fraser, L. H. & Keddy, P. A. 2005. The World's Largest Wetlands: Ecology and Conservation. – Cambridge University Press, Cambridge, DOI: 10.1017/CBO9780511542091

Ganesan, L. & Khan, R. A. 2008. Studies on the ecology of zooplankton in a floodplain wetland of West Bengal, India. – In: Sengupta, M. & Dalwani, R. (eds.) Proceedings of Taal 2007, The 12th World Lake Conference, pp. 67–73.

Garg, J. K. 2015. Wetland assessment, monitoring and management in India using geospatial techniques. – Journal of Environmental Management 148: 112–123. DOI: 10.1016/j.jenvman.2013.12.018.

Gregory, R. D. 2006. Birds as biodiversity indicators for Europe. – Significance 3(3): 106–110. DOI: 10.1111/j.1740-9713.2006.00178.x

Grimmett, R., Inskipp, C. & Inskipp, T. 2011. Birds of the Indian Subcontinent. – Oxford University Press, India

Grimmett, R. & Inskipp, T. 2007. Birds of Southern India. – Om Books International, New Delhi

Hadley, S. J. K., Hadley, A. S. & Betts, M. 2012. Acoustic classification of multiple simultaneous bird species: A multi-instance multi-label approach. – Journal of Acoustical Society of America 131(6): 4640–4650. DOI: 10.1121/1.4707424

Hamilton, S. G., King, S. L., Russo, G. D. & Kaller, M. D. 2019. Effect of hydrologic, geomorphic, and vegetative conditions on avian communities in the Middle Rio Grande of New Mexico. – Wetlands 39(5): 1029–1042. DOI: 10.1007/s13157-019-01156-9

Hardy, A. R., Stanley, P. I. & Greeting, S. P. W. 1987. Birds as indicator of the intensity of use of agriculture pesticide in UK. – In: Diamond, A. W. & Falion, F. N. (eds.) The value of Birds. – International Council for Birds Preservation, ICBP Tech. Publ., Cambridge, pp. 119–132.

Issa, M. A. A. 2019. Diversity and abundance of wild birds species' in two different habitats at Sharkia Governorate, Egypt. – Journal of Basic and Applied Zoology 80(34): 1–7. DOI: 10.1186/s41936-019-0103-5

Kahara, S. N., Duffy, W. G., DiGaudio, R. & Records, R. 2012. Climate, management and habitat associations of avian fauna in Restored Wetlands of California's Central Valley, USA. – Diversity 4(4): 396–418. DOI: 10.3390/d4040396

Khan, S. I. & Naher, H. 2009. Birds in Kurigram district of Bangladesh. – Journal of Threatened Taxa 1(4): 245–250. DOI: 10.11609/JoTT.o1698.245-50

Koli, V. K. 2014. Diversity and status of avifauna in Todgarh-Raoli Wildlife Sanctuary, Rajasthan, India. – Journal of Asia-Pacific Biodiversity 7(4): 401–407. DOI: 10.1016/j.japb.2014.10.005

Kumar, A., Sati, J. P., Tak, P. C. & Alfred, J. R. B. 2005. Handbook on Indian Wetland Birds and their Conservation. – Zoological Survey of India, MoEF, GoI.

Kumar, P. & Gupta, S. K. 2013. Status of wetland birds of Chhilchhila wildlife sanctuary, Haryana, India. – Journal of Threatened Taxa 5(5): 3969–3976. DOI: 10.11609/JOTT.O3158.3969-76

Kumar, P. & Sharma, A. 2018. Wetland birds assemblages in man-made sacred ponds of Kurukshetra, India. – Proceedings of the Zoological Society 72(1): 61–73. DOI: 10.1007/s12595-018-0259-x

Kumar, P., Rai, D. & Gupta, S. K. 2016. Wetland bird assemblage in rural ponds of Kurukshetra, India. – Waterbirds 39(1): 86–98. DOI: 10.1675/063.039.0111

Li, Z. W. D., Bloem, A., Delany, S., Martakis, G. & Quintero, J. O. 2009. Status of Water Birds in Asia – Results of the Asian Water Bird Census: 1987–2007. – Wetlands International, Kuala Lumpur, Malaysia

Luo, K., Wu, Z., Bai, H. & Wang, Z. 2019. Bird diversity and water bird habitat preferences in relation to wetland restoration at Dianchi Lake, south-west China. – Avian Reseach 10: 21. DOI: 10.1186/s40657-019-0162-9

Ma, Z., Cai, Y., Li, B. & Chen, J. 2010. Managing wetland habitats for water birds: an international perspective. – Wetlands 30: 15–27. DOI: 10.1007/s13157-009-0001-6

Mandal, M. H. & Siddique, G. 2018. Water Birds at Purbasthali Oxbow Lake: A Geographical Study. – Researchers World IX. (Special Issue): 7–19.

Mandal, M. H., Siddique, G. & Roy, A. 2018.Threats and opportunities of ecosystem services: A geographical study of Purbasthali Oxbow Lake. – Journal of Geography, Environment and Earth Science International 16(4): 1–24. DOI: 10.9734/JGEESI/2018/43229

Mandal, M. H., Roy, A. & Siddique, G. 2020. Spatial dynamics in people-wetland association: an assessment of rural dependency on ecosystem services extended by Purbasthali Wetland, West Bengal. – Environment Development Sustainability 23(1): 10831–10852. DOI: 10.1007/s10668-020-01089-y

Malik, D. S. & Joshi, N. 2013. Habitat selection pattern of migratory avifauna in relation to nutrients in Asan wetland at Doon Valley (Garhwal Himalaya), India. – International Journal of Recent Scientific Research 4(10): 1470–1475.

Mazumdar, S. 2017. Composition of avian communities in a human-modified wetland Okhla bird sanctuary, India: with notes on conservation initiatives. – Proceedings of the Zoological Society 72: 319–333. DOI: 10.1007/s12595-017-0239-6

Millennium Ecosystem Assessment (MEA) 2005. In Ecosystems and Human Well-being: Wetlands and Water Synthesis. – World Resources Institute: Washington DC, USA

Murkin, H. R. & Kadlec, J. A. 1986. Relationships between waterfowl and macro invertebrate densities in a northern prairie marsh. – The Journal of Wildlife Management 50(2): 212–217.

Pennington, D. N. & Blair, R. B. 2011. Habitat selection of breeding riparian birds in an urban environment: Untangling the relative importance of biophysical element and spatial scale. – Diversity and Distribution 17(3): 506–518. DOI: 10.1111/j.1472-4642.2011.00750.x

Prasad, S. N., Ramachandra, T. V., Ahalya, N., Sengupta, T., Kumar, A., Tiwari, A. K., Vijayan, V. S. & Vijayan, L. 2002. Conservation of wetlands of India - a review. – Tropical Ecology 43(1): 173–186.

Rajpar, M. N. & Zakaria, M. 2010. Density and diversity of water birds and terrestrial birds at Paya Indah Wetland Reserve, Selangor Peninsular Malaysia. – Journal of Biological Sciences 10(7): 658–666. DOI: 10.3923/jbs.2010.658.666

Reginald, L. J., Mahendran, C., Kumar, S. S. & Pramod, P. 2007. Birds of Singanallur Lake, Coimbatore, Tamil Nadu. – Zoos' Print Journal 22(12): 2944–2948. DOI: 10.11609/JoTT.ZPJ.1657.2944-8

Rittiboon, K. & Karntanut, W. 2011. Distribution of resident birds in a protected tropical habitat in south Thailand. – Journal of the International Society for Southeast Asian Agricultural Sciences 17(2): 95–103.

Smith, B. & Wilson, J. B. 1996. A consumer's guide to evenness indices. – Oikos 76: 70–82.

Turner, R. K., van den Berg, J. C. J. M., Soderqvist, T., Barendregt, A., van den Straaten, J., Maltby, E. & van Ireland, E. C. 2000. Ecological-economic analysis of wetlands: scientific integration for management and policy. – Ecological Economics 35: 7–23. DOI: 10.1016/S0921-8009(00)00164-6

Westphal, C., Steffan-Dewenter, I. & Tscharntke, T. 2006. Bumblebees experience landscapes at different spatial scales: possible implications for coexistence. – Oecologia 149: 289–300. DOI: 10.1007/s00442-006-0448-6

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Seasonal dynamics of bird assemblages in commercial plantations of *Pinus radiata* in southern-central Chile

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Abstract Forest plantations are a relatively new environment in Chile, with impacts of different magnitude on different taxa. Birds are one of the taxa mostly used as ecological indicators to evaluate the effect of different types of disturbances, due to their level of specialization in the use of different microhabitats. We assessed the effect of the age of *Pinus radiata* plantations (2 sites of 8 and 12 years, respectively) and seasonal (three seasons) on the diversity of bird assemblages. The abundance of species from both sites was assessed through acoustic and visual survey records, which were also compared at trophic guild level. Differences were observed at both sites, with higher diversity calculated for Cerro Ñielol (8 years) compared to Rucamanque (12 years). In seasonal terms, diversity was not affected in Cerro Ñielol, while in Rucamanque significant differences were observed during fall. Guild composition of both localities also showed differences, resulting in higher diversity for Cerro Ñielol in most cases. Despite the fact that the younger plantation presented the highest diversity, elements associated with the landscape are discussed, as well as particularities in the use of space by the species.

Keywords: diversity, ecological matrix, guilds, seasonality, vegetation cover

Összefoglalás Az ültetvény-erdők viszonylag új környezetet jelentenek Chilében, és eltérő hatást gyakorolnak a különböző taxonokra. A madarak az egyik olyan csoport, melyeket gyakran használnak ökológiai indikátorként a különböző típusú zavarások hatásának értékelésére, mivel eltérő mikrohabitatok használatára specializálódtak. Felmérték a Monterey-fenyő *(Pinus radiata)* ültetvények kora (két terület, 8, illetve 12 éves állományok) és a szezonalitás (3 évszak) hatását a madárközösségek diverzitására. A fajok abundanciáját mindkét területen akusztikus és vizuális felméréssel vizsgálták, melyeket a trofikus guildek szintjén is összehasonlították. A két terület között eltéréseket figyeltek meg: Cerro Ñielol (8 év) esetén magasabb diverzitást számoltak, mint Rucamanque (12 év) esetén. A szezonalitást figyelembe véve Cerro Ñielolban nem tapasztaltak változást, míg Rucamanqueban jelentős különbségeket figyeltek meg az őszi időszakban. Mindkét terület guild összetétele is eltéréseket mutatott, ami a legtöbb esetben Cerro Ñielolban eredményezett magasabb diverzitást. Annak ellenére, hogy a fiatalabb ültetvény mutatta a legmagasabb diverzitást, vizsgálták a tájképi elemeket és a fajok térhasználatának sajátosságait is.

Kulcsszavak: diverzitás, ökológiai mátrix, guildek, szezonalitás, növénytakaró

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Introduction

The loss and transformation of habitats are among the main threats to biodiversity (Caughley 1994). The replacement of native forest cover by monoculture plantations is a relatively recent phenomenon in Chile (Donoso & Lara 1996), where the Monterrey pine *(Pinus radiata)* is one of the most widely used for this conversion (Lavery & Mead 1998). This fact is presented through significant effects at the landscape level (Echeverría *et al.* 2008), where the original habitat has been reduced to scattered fragments with different degrees of connectivity. In this context, the local fauna must adapt to this new ecological scenario in order to persist. Despite the recognition of plantations as "green deserts" due to the low diversity, there is evidence that they could support various taxa (Bremen & Farley 2010, Estades *et al.* 2012), depending on the presence of specific environmental conditions suitable for certain species. However, the high degree of specialism in some species can significantly limit their occupation (Newbold *et al.* 2008) affecting the community composition and the functionality performed locally (Lindenmayer *et al.* 2005). In this way, the resulting community structure would allow to improve the understanding of the interactions between species with different resource requirements, under the spatial context that the plantations can provide (Devictor *et al.* 2008).

Bird communities in South America comprise a mosaic of groups with different places of origin and times of speciation (Jaksic & Feinsinger 1991). This heterogeneous origin is reflected in its community organization throughout the Chilean territory (Gantz *et al.* 2009, Zúñiga 2014), differentiated through space and food use (Rozzi *et al.* 1996, Díaz *et al.* 2005). The belonging of these species to specific groups delimited by specific uses of resources, called guilds, allows them to limit actions associated with competition, which enables their coexistence (Root 1967, Jaksic 2002). However, the conversion of native forest into plantations can result in the decrease of species richness whose habitat preferences are associated with the resources lost with changes derived from human disturbances (Arriaga-Weiss *et al.* 2008). Additionally, seasonal variations represent changes in the availability of resources, as a consequence of phenological processes in plants (Ratchke & Placey 1985), and invertebrate development (Borror *et al. 1*992), which affect the spatial dynamics of bird species. This situation implies additional changes in the composition of bird communities in the plantations, as the resources vary in response to seasonality.

Although the problem of the effect of *Pinus radiata* plantations and bird assemblages has been addressed in central and south-central Chile (Estades 1994, Estades & Temple 1999, Pérez 2004), there are information gaps regarding their composition in seasonal terms, which is of interest regarding species interaction and their impact on the environment. Another issue of particular importance is associated with the age of the plantation, since the development of the vegetation cover could result in niches available for these species (Macarthur & Macarthur 1962). This is supported by the fact that *Pinus radiata* plantations present variations in the accompanying vegetation as their age increases (Brockerhorff *et al.* 2003), by which a change in the associated animal communities is expected (Hernández *et al.* 2013, Rodríguez-Pérez *et al.* 2018).

The objective of this study is to examine the avifauna present in two sites with different age of *Pinus radiata* plantations in south-central Chile in a seasonal context. The hypothesis to be tested is about the dissimilarity of bird assemblages at both sites according to the age of plantations, which could be reflected both at the level of diversity of the whole diversity and guild composition. Likewise, variations of assemblages through seasons would also be expected.

Material and Methods

The two study sites were located around two protected areas. The first is near Cerro Ñielol (hereinafter Cerro Ñielol; 38º43'S, 72º35'W), a protected area that is close to the city of Temuco, in south-central Chile *(Figure 1)*. It is characterized by deciduous forests, represented mainly by the roble-laurel-lingue formation (*Nothofago-Perseetum*; Oberdorfer 1960). The present *Pinus radiata* plantation is eight years old. Its contiguous location with the city of Temuco gives it the presence of an urban matrix. The other plantation is 12 years old, and is close to the Predio Rucamanque (hereinafter Rucamanque; 38º39'S, 72º36'W), which is a private protected area characterized by evergreen forest formations, with olivillo *(Lapagerio-Aextoxiconetum punctatii)* and roble-laurel-lingue *(Nothofago-Perseetum)* associations (Ramírez 1989). In its environment there is a matrix associated with grasslands and scrub, as well as the occurrence of small-scale crops (Zúñiga *et al.* 2020). A preliminary comparison of vegetation of both sites allowed to establish that Rucamanque has a greater canopy cover (Mann-Whitney test, U=65, P=0.001), shrub

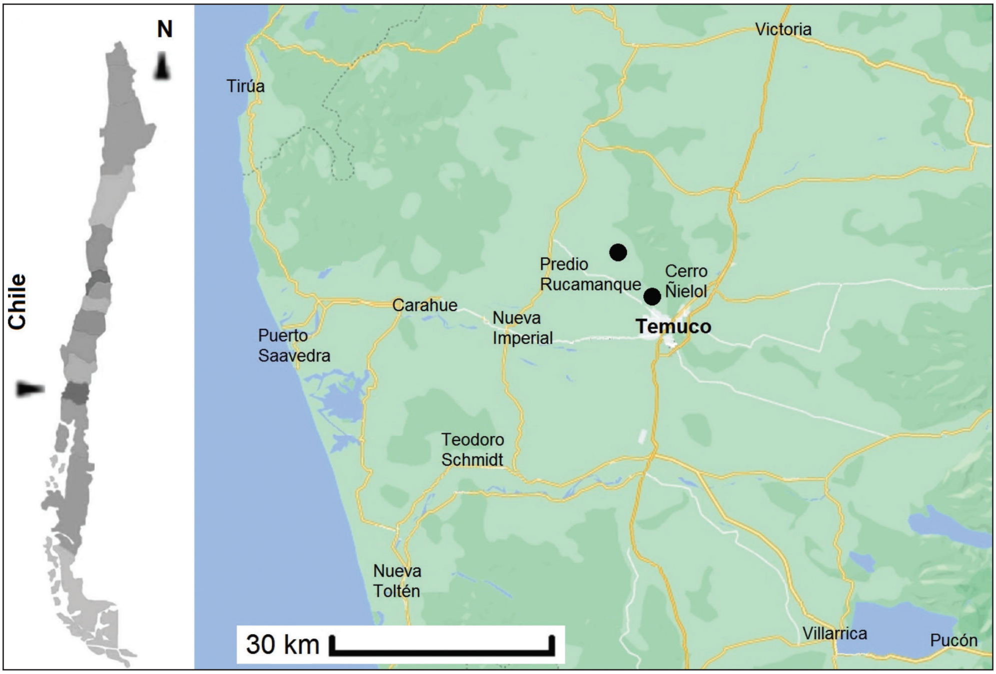

Figure 1. Study area. Sampling sites are highlighted with black dots
1. ábra A vizsgálati terület. A mintavételi helyeket fekete pontok jelölik

cover (U=255,48, P=0.028), litter (U=0, P<0.0001) and dead wood (U=107, P=0.026). However, the two sites did not show significant differences with respect to herbaceous vegetation cover (U=226, P=0.170).

During the period from April to December 2018 (fall, winter and spring in the southern hemisphere), point count method with a fixed radius of 50 m were carried out at both sites (Bibby *et al.* 1992), consisting of counting individuals of the different species through listening and observation stations in 50 m wide. These counts were performed fortnightly during the mornings (07:00 to 13:00 hrs.) and carried out within 4 minutes, after a waiting period of 3 minutes. Four recording stations were considered for each site. The acoustic identification of the birds was based on Egli (2006), and the nomenclature established by Jaramillo (2005).

The data for each site were expressed as the percentage representation of each species by site and season (Estades 1994, Zúñiga 2014), which allows to establish dominance for each case. The diversity at each site was calculated with the Shannon index (Shannon 1948), maximum diversity (Morin 2011) and evenness (Pielou 1966) for each season. To compare the diversity between stations in each locality, the procedure of Hutchenson (1970) was used, which was also applied to compare the effect of each station between both localities. To avoid the occurrence of type I error as a consequence of multiple comparisons, the Bonferroni procedure (Holm 1979) was performed. At each site, to evaluate the effect of seasonality on species abundance, the Kruskal-Wallis test was used (Quinn & Keough 2002). The effect of site at each season was examined using Mann-Whitney tests (Quinn & Keough 2002). Additionally, the community similarity was assessed in each season using the Bray-Curtis index (Odum 1950), which where performed with Biodiversity Profesional beta software (Mc Aleece *et al.* 1997).

To determine the effect of seasonality on trophic guilds, the composition of each group was compared across these periods. In this way, each registered species was represented within a particular guild, which is comprised of insectivores, granivores, omnivores and carnivores (Estades 1994, Rozzi *et al.* 1996). The importance of each guild in the different seasons was estimated using the Levins index (Levins 1968, Zúñiga *et al.* 2020), and its standardization was obtained through the index by Colwell and Futuyma (1971), due to the eventual inequality in the representation of species through the seasons. This index varies between 0 and 1, according to homogeneity in species composition. Their comparison was made using the Hutchenson procedure (Hutchenson 1970). All analyses (except Bray-Curtis index) were performed with XLSTAT trial version.

Results

During the total sampling effort of 672 minutes, a total of 28 species were observed, of which 27 were registered in Cerro Ñielol and 20 in Rucamanque (57.14% of shared species), with different representation through seasons (Kruskal-Wallis test) (*Table 1*). It is highlighted that in Cerro Ñielol 11 species presented significant differences in their abundances among the sampled seasons, compared to the 9 in Rucamanque *(Table 1)*. However, when seasonality was compared between sites, 9, 6 and 10 species were obtained with significant

Table 1. Percentage representation of bird species in different stations in the two sampled locations (F: fall; W: winter; S: Spring). Statistic values of Kruskal-Wallis tests (H) is indicated, the statistical significance (p), and the guild to which each species belongs. In bold, the cases where there was statistical significance are indicated. C: carnivorous; G: granivorous; I: insectivorous; O: omnivorous

1. táblázat A madárfajok százalékos aránya a különböző területeken a két mintavételi helyen (F: ősz; Ny: tél; S: tavasz). A Kruskal-Wallis tesztek statisztikai értékeit (H), a statisztikai szignifikanciát (p) és a guildet is jeleztük, amelyhez az egyes fajok tartoznak. Félkövéren szedve azokat az eseteket tüntettük fel, ahol statisztikailag szignifikáns eredményt kaptunk. C: ragadozó; G: magevő; I: rovarevő; O: mindenevő

Site	**Cerro Ñielol**					**Rucamanque**				
Sspecies	**F**	**W**	**S**	**H**	**p**	**F**	**W**	**S**	**H**	**p**
	%	**%**	**%**			**%**	**%**	**%**		
Anairetes parulus (I)	–	–	1.38	4.50	0.105	–	–	–	–	–
Aphrastura spinicauda (I)	2.89	2.66	6.94	0.42	0.807	3.05	5	3.44	6.03	**0.048**
Caracaras chimango (C)	0.72	2.66	–	6.33	**0.042**	–	–	–	–	–
Cinclodes patagonicus (I)	0.72	1.33	0.69	7.62	**0.027**	3.05	2	–	4.75	0.093
Colaptes pitius (I)	0.72	4	–	6.72	**0.035**	–	1	–	7.71	**0.021**
Columba araucana (G)	2.17	6	2.77	6.06	**0.048**	1.52	–	6.03	7.71	**0.021**
Curaeus curaeus (O)	–	0.66	–	4.50	0.105	–	–	2.58	–	–
Diuca diuca (G)	5.07	5.33	1.38	5.74	0.057	1.52	–	10.34	7.714	**0.021**
Elaenia albiceps (O)	–	–	2.08	4.50	0.105	–	–	4.31	7.714	**0.021**
Eugralla paradoxa (O)	6.52	4.66	–	6.72	**0.035**	1.52	10.99	6.03	5.992	**0.050**[*]
Glaucidium nana (C)	–	2.66	2.08	2.93	0.230	0.76	6	1.72	1.556	0.459
Lepthastenura aegithaloides (I)	–	–	–	3.20	–	–	0.99	–	3	0.223
Mimus thenca (O)	–	–	1.38	4.50	0.105	–	–	–	–	–
Nothoprocta perdicaria (O)	–	2	3.47	4.13	0.127	–	–	–	–	–
Pteroptochos tarnii (O)	6.52	10.66	0.69	4.90	0.086	15.28	6.99	12.06	3.342	0.188
Scelorchilus rubecula (O)	7.97	3.33	6.94	5.83	0.054	22.92	13.01	7.75	0.957	0.620
Sicalis luteola (G)	5.07	–	1.38	6.33	**0.042**	–	–	–	–	–
Sporagra barbatus (G)	12.31	6	9.02	7.26	**0.027**	–	–	6.03	6.82	**0.033**
Syctalopus magellanicus (O)	7.97	7.33	–	6.47	**0.039**	26.65	14	7.75	4.118	0.128
Sylviorthorhynchus desmursii (I)	–	–	–	–	–	1.52	1	–	4.935	0.085
Sturnella loyca (O)	1.44	1.33	1.38	5.54	0.063	2.29	–	–	4.571	0.102
Tachycineta leucopyga (I)	3.62	4	6.94	4.14	0.126	–	3	3.44	5.534	0.063
Troglodytes aedon (I)	5.79	7.33	13.19	5.88	0.053	1.52	11	14.65	5.907	0.052
Theristicus caudatus (C)	5.79	1.33	8.33	6.77	**0.034**	–	–	–	–	–
Turdus falcklandii (Om)	2.89	6	11.80	5.69	0.058	9.16	10	12.06	6.167	**0.046**
Vanellus chilensis (O)	5.79	6.66	–	4.18	0.123	–	–	–	–	–
Xolmis pyrope (I)	6.52	6	0.69	6.00	**0.050**	7.64	11	1.72	4.068	0.131
Zonotrichia capensis (G)	9.42	8	17.36	7.20	**0.027**	1.52	3.99	6.03	6.085	**0.048**

Table 2. Effect of seasonality over the abundance of species at each site. Statistical values of Mann-Whitney tests (U) and statistical significance (p) are indicated. Significant differences are marked in bold

2. táblázat A szezonalitás hatása a fajok egyedszámára az egyes helyszíneken. A Mann-Whitney tesztek statisztikai értékeit (U) és a statisztikai szignifikanciát (p) tüntettük fel. A jelentős eltéréseket félkövérrel jelöltük

Species	**Fall**		**Winter**		**Spring**	
	U	**p**	**U**	**p**	**U**	**p**
Anairetes parulus	–	–	–	–	1.50	0.121
Aphrastura spinicauda	3	0.487	7	0.275	4	0.827
Caracaras chimango	0	0.053	9	**0.037**	–	–
Cinclodes patagonicus	0	**0.034**	3.50	0.653	7.50	0.121
Colaptes pitius	3	0.317	1	0.105	–	–
Columba araucana	5.50	0.637	1.50	0.121	0	**0.046**
Curaeus curaeus	–	–	7.50	0.121	–	–
Diuca diuca	0	**0.046**	9	**0.037**	9	**0.043**
Elaenia albiceps	–	–	–	–	5.50	0.658
Eugralla paradoxa	5	0.796	5.50	0.658	9	**0.037**
Glaucidium nana	6	0.317	7	0.246	7.50	0.114
Lepthastenura aegithaloides	–	–	7.50	0.121	–	–
Mimus thenca	–	–	–	–	7.50	0.121
Nothoprocta perdicaria	–	–	7.50	0.121	3	0.317
Pteroptochos tarnii	8.50	0.077	8	0.105	9	**0.046**
Scelorchilus rubecula	9	**0.046**	6.50	0.369	9	**0.046**
Sicalis luteola	9	**0.037**	–	–	1.50	0.121
Sporagra barbatus	9	**0.037**	0	**0.046**	9	**0.046**
Syctalopus magellanicus	9	**0.050**	9	**0.046**	9	**0.043**
Sylviorthorhynchus desmursii	6	0.317	9	**0.034**	–	–
Sturnella loyca	1	0.105	6	0.317	0	**0.034**
Tachycineta leucopyga	0	**0.037**	2	0.197	1	0.105
Troglodytes aedon	0	**0.043**	4.50	1	8.50	0.072
Theristicus caudatus	9	**0.037**	9	**0.034**	0	**0.034**
Turdus falcklandii	6.50	0.369	7	0.246	6	0.487
Vanellus chilensis	7.50	0.121	9	0.034	–	–
Xolmis pyrope	1	0.105	6	0.507	5.50	0.637
Zonotrichia capensis	0.50	0.072	0.50	0.072	0	**0.046**

differences for fall, winter and spring, respectively *(Table 2)*. Regarding the similarity in the conformation of species through seasons, the highest values were obtained between fall and winter at both sites *(Table 3)*. In contrast, spring communities showed low similarities with those of the other seasons.

In general terms, a greater diversity, as well as a higher value of maximum diversity were observed in Cerro Ñielol than in Rucamanque *(Table 4)*. However, in the case of community evenness, its value was higher in Rucamanque than Cerro Ñielol in only one season (spring). In this way, significant differences in diversity were observed when comparing the sites in all seasons (Hutchenson test, t=3.14, P=0.0019; t=4.71, P<0.0001; t=3.63, P=0.0003 for fall, winter, and spring, respectively). When the effect of the seasons on the diversity of each site was independently compared, a variety of responses were obtained. In Cerro Ñielol, no significant differences in diversity were observed among these seasons, while in Rucamanque, differences were found in two cases, when fall vs. winter and fall vs. spring were compared (Hutchenson t-test, t=8.43, t=9.11, respectively; P<0.0001 in both cases).

Table 3. Bray-Curtis similarity at both sites through the seasons sampled
3. táblázat Bray-Curtis hasonlóság mindkét területen a mintavételi évszakok során

Season	**Winter**		**Spring**	
	Cerro Ñielol	**Rucamanque**	**Cerro Ñielol**	**Rucamanque**
Fall	72.11	61.93	57.74	53.41
Winter	–	–	54.56	56.11

Table 4. Diversity (H´), maximum diversity (Hmax) and evenness (J´) of the bird assemblages in the examined seasons
4. táblázat A madárközösségek diverzitása (H´), maximális diverzitása (Hmax) és egyenletessége (J´) a vizsgált évszakokban

	H´		**Hmax**		**J´**	
	Cerro Ñielol	**Rucamanque**	**Cerro Ñielol**	**Rucamanque**	**Cerro Ñielol**	**Rucamanque**
Fall	2.70	2.18	2.94	2.70	0.91	0.80
Winter	2.82	2.43	3.04	2.77	0.93	0.87
Spring	2.58	2.55	2.99	2.70	0.86	0.94

Table 5. Diversity of the different guilds (plus standardized index) in the two study sites, through the sampled seasons.
5. táblázat A különböző guildek sokfélesége (plusz standardizált index) a két vizsgálati helyszínen, a mintavételi évszakokban

Guilds	**Fall**		**Winter**		**Spring**	
	Cerro Ñielol	**Rucamanque**	**Cerro Ñielol**	**Rucamanque**	**Cerro Ñielol**	**Rucamanque**
Insectivores	4.88 (0.76)	3.55 (0.61)	4.90 (0.78)	4.32 (0.47)	3.26 (0.45)	2.24 (0.41)
Granivores	3.73 (0.68)	2.66 (0.83)	3.90 (0.96)	1.47 (0.47)	2.58 (0.39)	3.79 (0.93)
Omnivores	5.03 (0.60)	3.87 (0.57)	6.33 (0.66)	4.76 (0.94)	3.69 (0.44)	4.86 (0.77)
Carnivores	1.24 (0.24)	1 (0)	2.77 (0.88)	1 (0)	1.47 (0.47)	1 (0)

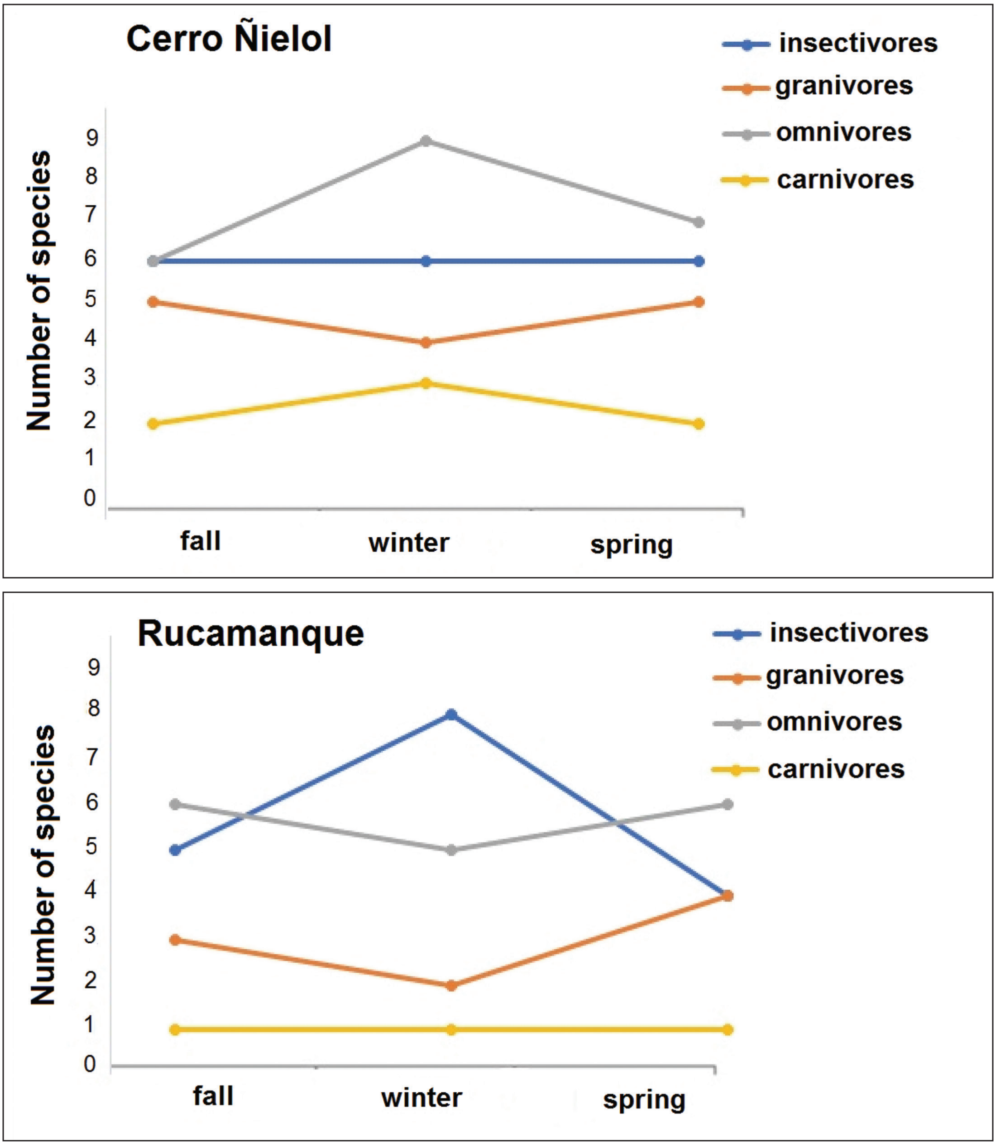

Figure 2. Variation of guilds across sampled seasons in the studied sites
2. ábra A guildek változása a mintavételi évszakok során a vizsgált területeken

Regarding the guild composition, four groups (insectivores, granivores, omnivores and carnivores) were observed, with a predominance of omnivores, followed by insectivores and granivores, respectively *(Figure 2)*. A variation in their diversity was also observed at both sites *(Table 5)*, which was reflected both locally and seasonally. Thus, a decrease in the number of granivorous species was observed in winter in both localities, with a simultaneous increase in omnivores, subsequently increasing the general species richness in spring. In Cerro Ñielol, Hutchenson's t-test yielded significant differences in the case of insectivores (fall vs. spring, t=2.55, P=0.0124), granivores (fall vs. spring, t=4.13, P<0.0001), and omnivores (fall vs. spring, winter vs. spring, t=37.00, t=4.20, respectively; P<0.0001 in both cases). In Rucamanque, significant differences were obtained for insectivores (fall vs. winter, fall vs. spring, t=44.34, t=6.56, respectively; P<0.0001 in both cases), granivores (fall vs. spring and winter vs. spring; t=10.99, t=10.30, respectively; P<0.0001 in both cases), and omnivores (fall vs. winter and winter vs. spring; t=3.73, p=0.0009; t=3.70, P=0.0003, respectively). When both localities were compared, differences were observed for insectivores (fall vs. winter, t=4.48, P<0.0001), granivores (fall vs. winter, fall vs. spring, winter vs. spring; t=10.33, t=8.58, t =5.13, respectively; P<0.0001 in both cases), omnivores (fall vs. winter and winter vs. spring; t=6.07, t=6.69, respectively; P<0.0001 in both cases) and carnivores (t=5.42, P=0.0016 for spring).

Discussion

The species richness of the assemblages at both sites was lower than that observed in forested habitats at the same latitude (Zúñiga *et al.* 2020), which reflects the restriction of plantations for their use by birds. This finding is consistent with that reported in other ecosystems (Volpato *et al.* 2010), where the community species richness and diversity in plantations are only fractions of what is experienced in the natural habitats. A remarkable fact is the predominance of generalist habitat species at both sites, whose ability to use different strata of vegetation would allow them to occupy the plantations for activities related to foraging and resting (Díaz *et al.* 2005). In this way, species with higher habitat requirements were found in low abundance, as well as in the limits of the points where the sampling was carried out, which shows that they would use the plantations in a marginal way. Despite the above, at both sites, the species richness was higher than reported in *Pinus radiata* plantations of similar age (Estades 1994), whose management involved significant removal of the shrub cover. This coverage would facilitate the expansion of niches for birds, reported in other locations (Friend 1982, Luck & Korodaj 2008). Moreover, nesting peculiarities of some species demonstrate the existence of requirements associated with both the structural complexity of the vegetation and the floristic diversity (Laiolo 2002), a situation that in the case of plantations is restricted. Despite the nesting records of some species in this type of environment (Estades 1999), it is necessary to assess this situation at both sites to determine if these plantations are suitable to carry out this process.

An aspect of special consideration at both sites is the matrix effect, which could explain some part of the variation in species richness (Dunford & Freemark 2005). This fact has been observed in *Pinus ponderosa* plantations in Patagonia Argentina, where the composition of bird communities would respond to the interaction of this environment with native vegetation through the landscape (Lantschner *et al.* 2008). In this sense, the greater proximity of Cerro Ñielol with the urban environment allowed the interaction of species with a greater degree of spatial and trophic flexibility (Díaz *et al.* 2005), which would increase the diversity of the assemblage. In contrast, Rucamanque is a site essentially surrounded by native forest, more linked to forest specialist birds (Díaz *et al.* 2005). Scattered shrub vegetation would be a facilitating element for this occurrence (Tomasevic & Estades 2008), which is should be considered in management plans.

The variation in the guild composition at both sites presents similarities with reports in other plantations, where winter would be the most critical season (Zúñiga *et al.* 2020), with omnivores being the least affected group. This fact would be explained by the ability of this group to use a wide spectrum of resources (Chubaty *et al.* 2013), which will add the spatial flexibility of these species by using nearby patches of habitat. The variation in the composition of insectivores at the seasonal level contrasts with that observed in other habitats of the same latitude, including plantations of *Eucalyptus* sp. (Zúñiga *et al.* 2020), where decreases in the diversity of this guild have been observed in winter. An explanation for this phenomenon would be the structural complexity resulting from its surroundings, given the age of both sites. This condition would favor the occurrence of insects through the vertical profile, thus enabling their use by birds of this guild (Poch & Simonetti 2013). One aspect to highlight is that although species with a high degree of specialization in habitat use were detected, such as *Aphrastura spinicauda* and *Lepthastenura aegithaloides* (Díaz *et al.* 2005), they only presented occasional records, which suggests a note of caution regarding their use about their preference as habitat.

The low presence of granivores at both sites is consistent with what was observed in *Eucalyptus* plantations (Zúñiga *et al.* 2020), which would be explained by the low richness of plant species that can provide seeds (Becerra & Simonetti 2020). Likewise, the low representation of carnivores in the study area could be the consequence of a sampling artifact, which minimizes their detection (Márquez *et al.* 2004). However, the high degree of vegetation closure resulted in a restriction for their occurrence due to the need for open environments for hunting activities (Pavez 2004). An aspect of special consideration in Cerro Ñielol is associated with its condition as a peri-urban site, which favors the presence of dogs. There are reports that dogs decrease the diversity of birds through predation (Banks & Bryant 2007), which would generate evasive behaviors in species, thus limiting their detection. In contrast, in Rucamanque there are no dogs around the plantations, which would partially restrict this process at this site. Although dogs have been detected around Rucamanque, they have a very low frequency of occurrence, for which they would not significantly affect the population dynamics of local bird species.

In conclusion, although differences were found in the diversity of birds at both sites, the older plantation did not present the highest species richness, limiting the applicability of the hypothesis of environmental heterogeneity as a promoter of diversity. Effects associated with the spatial context, mainly around the ecological matrix, would explain part of the variation of this pattern. In this way, it is necessary to extend the study on bird assemblages on a larger spatial scale, which would allow determining the exchange of species between both environments. Furthermore, seasonality had an important effect on the configuration of the assemblage composition on a temporal scale. The quantification of the plantation resources during their productive cycle is of special relevance, as to establish effective management measures for the management of microhabitats thus facilitating the occurrence of local wildlife.

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References

Arriaga-Weiss, S., Calmé, S. & Kampichler, C. 2008. Bird communities in rainforest fragments: guild responses to habitat variables in Tabasco, Mexico. – Biodiversity and Conservation 17:173–190. DOI: 10.1007/s10531-007-9238-7

Banks, P. B. & Bryant, J. V. 2007. Four-legged friend or foe? Dog walking displaces native birds from natural areas. – Biology Letters 3(6): 611–613. DOI: 10.1098/rsbl.2007.0374

Becerra, P. I. & Simonetti, J. A. 2020. Native and exotic plant diversity plant species diversity in forest fragments of a coastal landscape of central Chile. – Bosque 41(2): 125–136. DOI: 10.4067/S0717-92002020000200125.

Bibby, C. J., Burguess, N. D. & Hill, D. A. 1992. Bird census techniques. – Academic Press, London

Borror, D. J., Triplehorn, C. A. & Johnson, N. F. 1992. Study of insects. 6th ed. – Saunders College Publishing, Orlando, E. U. A.

Bremen, L. L. & Farley, K. A. 2010. Does plantation forestry restore biodiversity or create green deserts? A synthesis of the effects of land-use transitions on plant species richness. – Biodiversity and Conservation 19: 3893–3915. DOI: 10.1007/s10531-010-9936-4

Brockendorff, E. G., Ecroyd, C. E., Leckie, A. L. & Kimberley, M. O. 2003. Diversity and succession of adventive and indigenous vascular understory plants in *Pinus radiata* plantation forests in New Zealand. – Forest Ecology and Management 185(3): 307–326. DOI: 10.1016/S0378-1127(03)00227-5

Caughley, G. 1994. Directions in conservation biology. – Journal of Animal Ecology 63(2): 215–244. DOI: 10.2307/5542

Chubaty, A. M., Ma, B. O., Stein, W., Gillespie, D. R., Henry, L. M., Phelan, C., Palsson, E., Simon, F. W. & Roitberg, B. D. 2013. On the evolution of omnivory in a community context. – Ecology and Evolution 4(3): 251–265. DOI: 10.1002/ece3.923

Colwell, R. & Futuyma, J. 1971. On the measurement of the niche breadth and overlap. – Ecology 52(4): 567–572. DOI: 10.2307/1934144

Devictor, V., Julliard, R. & Jiguet, F. 2008. Distribution of specialist and generalist species along spatial gradients of habitat disturbance and fragmentation. – Oikos 117(4): 507–514. DOI: 10.1111/j.0030-1299.2008.16215.x

Díaz, I. A., Armesto, J. J., Reid, S., Sievking, K. E. & Wilson, M. F. 2005. Linking forest structure and composition: Avian diversity in successional forests in Chiloé Island, Chile. – Biological Conservation 123(1): 91–101. DOI: 10.1016/j.biocon.2004.10.011

Donoso, C. & Lara, A. 1996. Utilización de los bosques nativos en Chile: pasado, presente y futuro [Utilization of native forests in Chile: past, present and future]. – In: Armesto, J. J., Villagrán, C. & Arroyo, M. K. (eds.) Ecología de bosques nativos de Chile [Ecology of Native Forests of Chile]. – Editorial Universitaria, Santiago, Chile, pp. 367–387. (in Spanish)

Dunford, W. & Freemark, K. 2005. Matrix matters: effect of surrounding land uses on forests birds near Ottawa, Canada. – Landscape Ecology 20: 497–511. DOI: 10.1007/s10980-004-5650-5

Echeverría, C., Coomes, D., Hall, M. & Newton, A. E. 2008. Spatially explicit models to analyze forest loss and fragmentation between 1976 and 2020 in southern Chile. – Ecological Modelling 212(3–4): 439–449. DOI: 10.1016/j.ecolmodel.2007.10.045

Egli, G. 2006. Voces de aves chilenas (Compact disc) [Chilean bird voices]. – Unión de Ornitólogos de Chile, Santiago de Chile (in Spanish)

Estades, C. 1994. Impacto de la sustitución del bosque natural por plantaciones de *Pinus radiata* sobre una comunidad de aves en la octava región de Chile [Impact of the replacement of the natural forest by *Pinus radiata* plantations on a community of birds in the eighth region of Chile]. – Boletín Chileno de Ornitología 1: 8–14 (in Spanish)

Estades, C. F. 1999. Nidificación de aves en un rodal maduro de *Pinus radiata* [Nesting of birds in a mature *Pinus radiata* stand]. – Boletín Chileno de Ornitología 6: 35–38 (in Spanish)

Estades, C. F. & Temple, S. 1999. Deciduous-forest bird communities in a fragmented landscape dominated by exotic pine plantations. – Ecological Applications 9(2): 719–727. DOI: 10.1890/1051-0761(1999)009[0573:DFBCIA]2.0.CO;2

Estades, C. F., Grez, A. A. & Simonetti, J. A. 2012. Biodiversity in Monterrey pine plantations. – In: Simonetti, J. A., Grez, A. A. & Estades, C. F. (eds.) Biodiversity Conservation in Agroforestry Landscapes: Challenges and Opportunities. – Editorial Universitaria, Santiago, Chile, pp. 77–98.

Friend, G. R. 1982. Bird populations in exotic pine plantations and indigenous eucalypt forests in Gippsland, Australia. – Emu 82(2): 80–89. DOI: 10.1071/MU9820080

Hernández, J., Acuña, M. P., Corvalán, P. & Simonetti, J. A. 2013. Assessing understory in forest plantations using laser imaging detection and gradient (LIDER). – Revista Chilena de Historia Natural 86(4): 433–442. DOI: 10.4067/S0716-078X2013000400005

Holm, S. 1979. A simple sequentially rejective multiple test procedure. – Scandinavian Journal of Statistics 6(2): 65–70.

Hutchenson, K. 1970. A test for comparing diversities based in Shannon formula. – Journal of Theoretical Biology 29(1): 151–154. DOI: 10.1016/0022-5193(70)90124-4

Jaksic, F. 2002. Ecología de comunidades [Ecology of Communities]. – Ediciones Universidad Católica de Chile, Santiago, Chile (in Spanish)

Jaksic, F. & Feinsinger, P. 1991. Bird assemblages in temperate forests of North and South America: a comparison of diversity, dynamics, guild structure and resource use. – Revista Chilena de Historia Natural 64: 491–510.

Jaramillo, A. 2005. Aves de Chile [Birds of Chile]. – Lynx Ediciones, Barcelona, España (in Spanish)

Laiolo, P. 2002. Effect of habitat structure, habitat composition and diversity on a forest bird community in a north-western Italy. – Folia Zoologica 51(2): 121–128.

Lantschner, M. V., Rusch, V. & Peyrou, C. 2008. Bird assemblages in pine plantations replacing native ecosystems in NW Patagonia. – Biodiversity and Conservation 17(5): 969–989. DOI: 10.1007/s10531-007-9243-x

Lavery, P. B. & Mead, D. J. 1998. Pinus radiata: a narrow endemic from North America takes on the world. – In: Richardson, D. (ed.) Ecology and Biogeography of *Pinus*. – Cambridge University Press, Cambridge, pp. 432–449.

Levins, R. 1968. Evolution in Changing Environments. – Princeton University Press

Lindenmayer, D., Blanchard, W., Tennant, P., Barton, P., Ikin, K., Mortelliti, A., Okada, S., Crane, M. & Michael, D. 2015. Richness is not all: how changes in avian functional diversity reflect major landscape modification caused by pine plantations. – Diversity and Distributions 21(7): 836–847. DOI: 10.1111/ddi.12328

Luck, G. W. & Korodaj, T. N. 2008. Stand and landscape-level factor related to bird assemblages in exotic pine plantations: implications for forest management. – Forest Ecology and Management 255(7): 2688–2697. DOI: 10.1016/j.foreco.2008.01.029

Macarthur, R. H. & Macarthur, J. W. 1962. On bird species diversity. – Ecology 42(3): 594–598. DOI: 10.2307/1932254

Mc Aleece, N., Gage, J. D. G., Lambshead, P. J. D. & Paterson, G. K. J. 1997. Biodiversity Professional Statistics Analysis Software. – Jointly developed by the Scottish Association for Marine Science and the Natural History Museum London https://www.sams.ac.uk/science/outputs/

Márquez, C., Sánchez, I. M. & Rau, J. 2004. Técnicas de observación y estimación de abundancia de aves rapaces [Observation techniques and estimation of abundance of birds of prey]. – In: Muñoz-Pedreros, A. & Yáñez, J. (eds.) Aves rapaces de Chile [Birds of Prey in Chile]. – CEA Ediciones, Valdivia, pp. 253–263. (in Spanish)

Morin, P. 2011. Community Ecology. – Wiley-Blackwell, Sussex

Newbold, T., Scharlemann, J. P. W., Butchart, S. H. M., Şekercioğlu, Ç. H., Alkermande, R., Booth, H. & Purves, D. W. 2008. Ecological traits affect the response of tropical forest bird species to land-use intensity. – Proceedings of the Royal Society B 280: 20122131. DOI: 10.1098/rspb.2012.2131

Oberdorfer, F. 1960. Pflanzensoziologische Studien in Chile-Ein Vergleich mit Europa [Plant sociological studies in Chile-A comparison with Europe]. – Flora et Vegetatio Mundi 2: 1–208. (in German)

Odum, E. 1950. Bird populations of the Highlands (North America) Plateau in relation to plant succession and avian invasion. – Ecology 31: 587–605. DOI: 10.2307/1931577

Pavez, E. 2004. Descripción de las aves rapaces chilenas [Description of Chilean birds of prey]. – In: Muñoz-Pedreros, A. & Yáñez, J. (eds.) Aves rapaces de Chile [Birds of Prey in Chile]. – CEA Ediciones, Valdivia, pp. 29–103 (in Spanish)

Pérez, M. F. 2004. Efecto de la estructura de plantaciones de *Pinus radiata* D. Don sobre su calidad como hábitat para aves en Constitución [Effect of the structure of *Pinus radiata* D. Don plantations on its quality as a habitat for birds in Constitución]. – Memoria para la obtención del título de Ingeniero Forestal, Universidad de Chile (in Spanish)

Pielou, E. C. 1966. The measurement of diversity in different types of biological collection. – Journal of Theoretical Biology 13: 131–144. DOI: 10.1016/0022-5193(66)90013-0

Poch, T. J. & Simonetti, J. A. 2013. Insectivory in *Pinus radiata* plantations with different degree of structural complexity. – Forest Ecology and Management 304: 132–136. DOI: 10.1016/j.foreco.2013.04.044

Quinn, G. P. & Keough, M. J. 2002. Experimental Design and Data Analysis for Biologists. – Cambridge United Press

Ramírez, C., Hauenstein, E., San Martín, J. & Contreras, D. 1989. Study of the flora of Rucamanque, Cautin Province, Chile. – Annals of the Missouri Botanical Garden 76(2): 444–453. DOI: 10.2307/2399493

Ratchke, B. & Placey, E. 1985. Phenological patterns of terrestrial plants. – Annual Review of Ecology and Systematics 16: 179–214. DOI: 10.1146/annurev.es.16.110185.001143

Rodríguez-Pérez, J., Herrera, J. M. & Arizaga, J. 2018. Mature non-native plantations complement native forests in bird communities: canopy and understory effects on avian habitat preferences. – Forestry 91(2): 177–184. DOI: 10.1093/forestry/cpx053

Root, R. B. 1967. The niche exploitation pattern of the Blue-gray Gnatcatcher. – Ecological Monographs 37(3): 317–350. DOI: 10.2307/1942327

Rozzi, R., Martínez, D., Willson, M. F. & Sabag, C. 1996. Avifauna de los bosques templados de Sudamérica [Avifauna of South American temperate forests]. – In: Armesto, J., Villagrán, C. & Arroyo, M. K. (eds.) Ecología de los bosques nativos de Chile [Ecology of Native Forests of Chile]. – Editorial Universitaria, Santiago, pp. 135–152 (in Spanish)

Shannon, C. 1948. A mathematical theory of communication. – Bell System Technical Journal 227: 379–423.

Tomasevic, J. A. & Estades, C. E. 2008. Effect of the structure of pine plantations on their "softness" as barriers for ground-dwelling birds in south-central Chile. – Forest Ecology and Management 255(3–4): 810–816. DOI: 10.1016/j.foreco.2007.09.073

Volpato, G. H., Prado, V. M. & Dos Anjos, L. 2010. What can tree plantations do for forest birds in fragmented forest landscapes? A case study in southern Brazil. – Forest Ecology and Environment 260(7): 1156–1163. DOI: 10.1016/j.foreco.2010.07.006

Zúñiga, A. H. 2014. Composición de un ensamble de aves en un fragmento boscoso del sur de Chile [Composition of an assemblage of birds in a forest fragment of southern Chile]. – Gestión Ambiental 27: 35–45 (in Spanish)

Zúñiga, A. H., Fuenzalida, V. & Sandoval, R. 2020. Dinámica estacional de ensambles de aves en tres hábitats del centro-sur de Chile [Seasonal dynamics of bird assemblages on three habitats of Southern-central Chile]. – Ecología en Bolivia 55(2): 73–84 (in Spanish)

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Population density, habitat preference, and breeding biology of Chukar Partridge *(Alectoris chukar)* in Malakand division, Khyber Pakhtunkhwa, Pakistan

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Abstract The field biology of Chukar Partridge *(Alectoris chukar)* was studied in Malakand division, which is an important range of its distribution in Pakistan. The abundance of the species at different altitudes was studied using both transect trails of various lengths and point counts at certain spots. The average monthly population density was estimated to be 1.448±0.466 birds/ha. The population density in August, September, and October 2020 was significantly ($P<0.05$) higher, as compared to the rest of the months. Chukar Partridge sightings were the lowest in December, with mean±SD = 0.996±0.147 birds/ha and the highest in October 2.333±0.202 birds/hae in all the study sites. At each study site, maximum activity habitats were marked and selected for breeding ecology study. Data was collected on breeding time, nest site selection, nest size, incubation period, and hatching success. The breeding season in this species starts in February, with the peak months being March and April, when calls are frequently heard. The frequency of calls varied from 0.15 to 0.3 per minute. Throughout the breeding season, twelve nests were observed using binoculars and camera traps from a distance to avoid disturbance, wherever possible and without disrupting the species. Nesting sites were mostly on slopes under the eaves of *Dodonea viscosa, Calotropis procera* and *Zizyphus oxyphylla*. Dry leaves of *Poa annua, Cynodon dactylon, Dichanthium annulatum,* small twigs of bushes, and downy feathers were used as nesting materials. The average diameter of nest (n=12) was 25.43±3.4 (mean±SD) cm. Overall, the mean clutch size was 94.5±30.187 egg/nest with an incubation period of 22–24 days. The hatching success rate was 82% with 315 successfully hatched chicks, while the fledging rate was 83% (265 fledged out of 315 hatched in, n=42 nests).

Keywords: *Alectoris chukar*, population dynamics, nesting, ecological factors, Pakistan

Összefoglalás A csukár *(Alectoris chukar)* fészkelésbiológiáját vizsgáltuk Malakand tartományban, mely a faj pakisztáni elterjedésének fontos területe. A faj abundanciája tengerszintfeletti magasság szerinti mintázatát transzekt- és pontszámlálásos módszert alkalmazva mértük fel. Az átlagos havi populációsűrűség értéke 1,448±0,466 madár/ha volt. A populációsűrűség augusztusban, szeptemberben és októberben szignifikánsan ($P<0,05$) nagyobb volt, mint a többi hónapban. A megfigyelések száma decemberben volt a legalacsonyabb (0,996±0,147 madár/ha), és októberben a legmagasabb (2,333±0,202 madár/ha). Fészkelésökológiai vizsgálatok elvégzésére az összes vizsgált területen kiválasztottuk és megjelöltük a maximális aktivitással jellemzett élőhelyeket. Ennek folyamán adatokat gyűjtöttünk a költés időzítéséről, a fészkelőhely-választásról, a fészek méretéről, a költés időtartamáról és a költési sikerről. A faj költése februárban kezdődik, márciusban és áprilisban éri el csúcsát, mikor a nászhangokat a leggyakrabban lehet hallani. A nászhangok gyakorisága 0,15–0,3/perc értékek között váltakozott. A költési időszakban 12 fészket figyeltünk kézitávcsövek és kameracsapdák segítségével olyan távolságból, hogy ne okozzunk zavarást. A fészkek leggyakrabban *Dodonea viscosa, Calotropis procera Zizyphus oxyphylla* növényfajok takarásában kerültek elő. Fészekanyagként főleg *Poa annua*, *Cynodon dactylon, Dichanthium annulatum* száraz levelei, illetve kisebb cserjeágak és pehelytollak szerepeltek. A fészkek átlagos átmérője 25,43±3,4 cm (n=12) volt. Az átlagos fészekaljméret 94,5±30,187 tojás/fészek volt. A kotlás 22–24 napig tartott. A 42 fészekből összesen 315 csibe kelt ki, a költési siker 82%-os volt, 265 fióka repült ki, a kirepülési siker 83% volt.

Kulcsszavak: *Alectoris chukar*, populációdinamika, fészkelés, ökológiai tényezők, Pakisztán

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Introduction

Chukar Partridge (*Alectoris chukar* Gray, 1830) is a game bird belonging to the order Galliformes, family Phasianidae, and subfamily Phasianinae. (Ahmad *et al*. 2017). Chukar Partridge can be found in Mongolia, China, South Russia, Pakistan, India, Afghanistan, Southwest Asia, and North east Africa (Roberts 1991). In mountainous regions of Pakistan, the Chukar Partridge is widespread. Sindh, Baluchistan, Punjab, Khyber Pakhtunkhwa, Malakand division, Margalla hills, Kohistan, and Gilgit Baltistan are among the provinces in Pakistan where the species is found. Chukars normally breed once a year, depending on environmental factors, but two broods may occur if sedentary conditions are favorable (WA 2011). Males perform a display in the spring, involving a head tilt and a pop-out wing, to attract females. Male and female began calling and participating in "tidbitting gigs," (del Hoyo *et al*. 1994). Chukars can be found in flocks of 2–4 or 5–7 birds in the region. Depending on the elevation, they start copulating in March and raise their brood mostly in mid or early April to July. The breeding starts right on time at low height when compared to higher elevation (Roberts 1991) that in the Himalayas it moves to snow-topped fields and does not start breeding until late June (Roberts 1991). The Chukar is mostly found at an elevation of 1000 to 4500 m but in Pakistan, they may move down to 600 m elevation above mean sea level (asl) (Rasmussen & Anderton 2005). Chukars are upland birds that prefer dry rocky hills and ravines with little vegetation or scrub. In cold season they inhabit low altitudes ranging from 1,000 to 1,500 m. During the summer they inhabit high elevation areas up to 4500 m. They prefer areas mostly close to crop fields on steep slopes and forage in the mountains and gorges cultivated in the hills. They live in groups in the spring and late summer, but this is not the case in the fall. (Roberts 1991). The Chukar Partridge population has dropped sharply in many countries in recent years (Cetin *et al*. 1997). Habitat depletion, predators, agricultural technology, growth of bird-breeding farms, widespread use of herbicides and insecticides, and excessive hunting are all factors contributing to the population decline (Gaudioso *et al*. 2002, Grimmett *et al*. 2008). Increased human population has a negative impact on the Partridge's population and habitat (Bhattacharya *et al*. 2009). The Chukar population is also impacted by predators and harsh environmental factors. (Christensen 1996). Chukar Partridge is the national bird of Pakistan. However, this bird lacks sufficient scientific data on the population ecology and breeding of its native areas. Although the Malakand division is part of its native home range, no data from this region has previously been recorded. The aim of this study is to analyses the population density, habitat preferences and some aspects of Chukar Partridge breeding biology in the Malakand division. Highly favored environments have been established, and breeding populations have been identified for long-term tracking studies to determine nesting, incubating, and reproductive success.

Material and Methods

Study area

The proportion of suitable habitat and the distribution patterns of species are greatly affected by variations in elevation gradient, topography, and vegetation characteristics in the Malakand division. On the basis of altitudinal range and vegetation structure differences, four zones were selected for the current study. Three line transects were formed in each zone *(Figure 1)*. The dominant tree species of the selected zones include *Ficus racemose, Melia azedarachta, Eucalyptus lanceolate, Pyrus pashia, Ficus palmate, Cedrus deodara, Quercus baloot, Pinus wallichiana* and *Pinus roxburghii* Sarg. Shrubs include *Desmodium elegans, Berberis lyceum, Datura stramonium, Dodonea viscosa, Calotropis procera, Zizyphus oxyphylla, sarcococca saligna, Hedera nepalensis, Jasminium officinalis* and *Rubus fruticosus.* Herbs include *Sorghum helepense, Poa annua, Stachys parviflora, Heliotropium strigosum, Stellaria media, Origanum vulgare, Solanum xanthocarpum* and *Hypricum perforatum*. Wild sheep, bears, jackals, wolves, rodents, monkeys, foxes, Indian Crested Porcupine *(Hystrix indica)*, Chukar Partridge *(Alectoris chukar)*, Black Francolin *(Francolinus francolinus)*, Grey Francolin *(Ortygornis pondicerianus)* and other avian species are among the area's major wildlife.

Throughout the study period, the climatic factors studied (air temperature, relative humidity, wind speed, wind direction, cloud cover, rainfall, and relative humidity) are given in *(Table 1)*. The climate of this area is semi-arid with moderate hot, long summer and short, cool, windy, rainy winter.

Table 1. Climatic parametres recorded in Malakand division from January to December 2020
1. táblázat Makaland tartományban 2020 január és december között mért klimatikus paraméterek

Months	Temperature (°C)			Relative Humidity (%)	Wind speed	Wind direction	Cloud cover (%)	Rainfall (mm.)
	Max	Min	Mean					
January	12.5	8.7	11	64%	8	S	8	81
February	14.0	10.5	12.2	60%	3.8	N	6	98
March	18.3	15.0	17	76%	4	SE	7	125
April	21.8	17.8	20	78%	5.5	SE	9	105
May	27.3	23.0	25	43%	7	N	9	47
June	29.0	26.0	28	44%	9	S	7	30
July	35.6	31.3	34	70%	4	SW	2	135
August	38.8	33.6	36.2	78%	8.3	SE	3	132
September	32.3	27.0	30	63%	9.5	SW	6	64
October	28.2	23.8	26	30%	7	NE	6	28
November	23.0	20.0	22	31%	11	SE	5	22
December	19.2	16.0	18	33%	6.8	S	9	48

Sampling strategies

The area is divided into equal-sized grids in the distribution range, and the grids for the tracks were selected randomly. Since the distribution of Chukar Partridge in the study region is not random, but is more determined by variations in topography and landscape characteristics, tracks were walked as the behavior of the species and landscape characteristics. Monitoring was carried out by the line transect method (Buckland 2001), while dogs were used during the non-breeding season to increase the chances of observation (Besnard *et al*. 2010). The sites were chosen because they were on an elevation gradient and were at least 20–30 km apart. Due to the vast distance among sites, the first trail was completed within the first two weeks of a month at each site, followed by the second trail during the remaining two weeks. During the study, 288 trails with a total length of around 1 km were walked. For population density, both direct and indirect data were used. The number of sighted birds was divided by the transect area to get the density (number of birds per ha) (100 ha = 50 m × 1000 m, length × width of transect). Feathers, droppings, nests, and voice calls were used to look for bird nests and determine where they roost and feed. Vegetation associated with nesting was also reported. The number of calls of Chukar and time spent at each site were recorded. The following formula was used to determine the frequency of calls:

$$\text{Calling frequency} = \frac{\text{Total No of calls}}{\text{Total observation time}}$$

All experimental results were examined by statistical package for social sciences IBM (SPSS) version 20 software database. Observations were considered statistically significant at (P<0.05).

During the months of February, March, April, May and June, the study site for the nest was assessed, considering the equal potential of all habitats for Chukar Partridge hosting and breeding activities. They were frequently visited and watched for signs of nesting around their habitat prior to the start of the breeding season. i.e., the last week of February. The nests were spotted by walking across the study area on an existing route, while randomly encountered female birds were followed to their nests. Once an active nest was confirmed, it was labelled and given a unique number. The nest's location, general appearance, and composition are all noted. Signed nests were visited twice a week with the utmost caution to ensure that breeding birds were not disturbed. After egg laying, the frequency of visits was increased on alternating days to monitor incubation and hatching times. The date of the first and last egg laid, the number of eggs, the date of hatching, and the number of eggs hatched were all recorded.

Habitat preference

Along its distribution range, the Chukar Partridge uses a variety of habitat types and favors different breeding grounds. We also recorded the vegetation cover and habitat components associated with Chukar breeding habitat preferences and breeding ecology through various

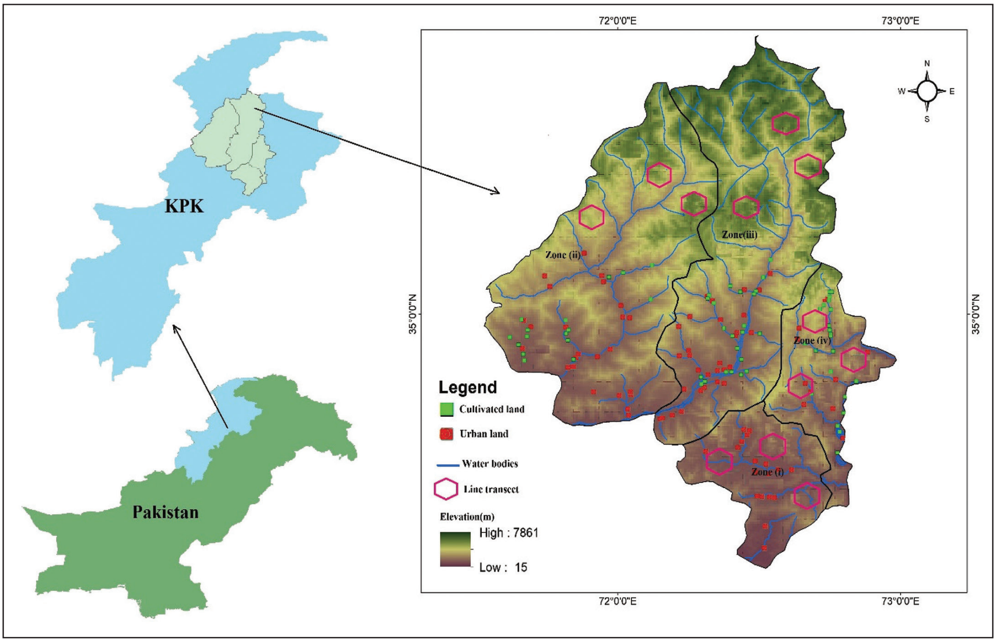

Figure 1. Map of Malakand division, Pakistan showing administrative details and site of the study area

1. ábra Makaland tartomány térképe (Pakisztán), amely mutatja a közigazgatási határokat és a vizsgált területet

vegetation zones. The study sites were located between 200 and 7,861 m asl. Based on differences in altitudinal range and vegetation structure, the study sites were divided into four main habitat types *(Figure 1)*. Zone – I at lower elevation between 400 to 2300 m asl were dominated by *Zizyphus oxyphylla, Sarcococca saligna, Dodonea viscosa, Hedera nepalensis*, Zone – II located between 680 to 3,800 m asl dominated by *Berberis lyceum, Datura stramonium, Dodonea viscosa, Zizyphus oxyphylla.* Zone – III at higher elevation with cultivated land located between 500 to 7,861 m asl dominated by *Desmodium elegans, Dodonea viscosa, Calotropis procera* and Zone – IV located between 875 to 4,400 m asl near to agriculture land dominated by *Dodonea viscosa, Jasminium officinalis, Rubus fruticosus*.

Results

The distribution of Chukar Partridge population was found to vary greatly in the different study areas. Male calls were abundant throughout the breeding season, and individuals remained active during the day. Female calls were also heard at certain locations, likely as a result of predator or other nearby danger. Males, however, were extremely vocal during the breeding season, and were seen consistently calling throughout the habitat.

We first assessed their abundance at all sites, based on direct encounter or male breeding calls, in order to investigate some aspects of Chukar Partridge breeding ecology *(Table 2)*.

Table 2. Mean Chukar Partridge density (ind/ha) in the Malakand division based on direct sightings and call counts at all study sites. The values provided are (Mean±SE)

2. táblázat A csukár átlagos állománysűrűsége (egyed/ha) Makaland tartományban, közvetlen megfigyelések és nászhang-felmérések alapján (átlag±SD)

S. No.	Months	Zone-I	Zone-II	Zone-III	Zone-IV	Sum of all Zones
1	Aug	0.357±0.021	0.550±0.050	0.403±0.025	0.467±0.015	1.777±0.186
2	Sep	0.470±0.020	0.666±0.015	0.540±0.020	0.593±0.025	2.270±0.185
3	Oct	0.476±0.015	0.666±0.015	0.563±0.015	0.626±0.015	2.333±0.202
4	Nov	0.280±0.026	0.463±0.015	0.373±0.015	0.443±0.025	1.560±0.227
5	Dec	0.173±0.011	0.323±0.015	0.236±0.020	0.263±0.015	0.996±0.147
6	Jan	0.270±0.020	0.336±0.015	0.276±0.015	0.343±0.025	1.226±0.070
7	Feb	0.230±0.010	0.380±0.010	0.350±0.020	0.37±0.020	1.330±0.217
8	Mar	0.203±0.025	0.320±0.020	0.253±0.015	0.280±0.020	1.056±0.128
9	Apr	0.186±0.020	0.303±0.020	0.250±0.020	0.276±0.015	1.016±0.153
10	May	0.240±0.010	0.320±0.020	0.263±0.015	0.296±0.020	1.120±0.086
11	Jun	0.246±0.020	0.370±0.010	0.293±0.015	0.336±0.020	1.246±0.118
12	Jul	0.336±0.025	0.373±0.015	0.346±0.020	0.396±0.025	1.453±0.092
Total = 1 year		0.289±0.100	0.422±0.130	0.345±0.107	0.391±0.118	1.448±0.466

In December, Chukar sightings were minimum, with 0.173±0.011 birds/ha in Zone – I, 0.236±0.020 birds/hae in Zone – III, and 0.263±0.015 birds/ha in Zone – IV, in various habitat types. Sightings greatly increased from July to November then began to fall in December. Study site at the lower temperate region Zone – II had the highest abundance (0.666±0.015 birds/ha), followed by Zone – IV (0.626±0.015 birds/ha) having shrubby habitat and proximity to agriculture land and Zone – III (0.563±0.015 birds/ha) with abundant shrubs and cultivated land. As compared to the other Zones the population of Zone – I was slightly less abundant even at the peak breeding season (0.186±0.020 birds/ha) which is a similar habitat as the other study area, possibly due to hunting, trapping and harsh environmental condition.

Breeding behavior

Late January marked the start of Chukar breeding season in Malakand division. The birds were seen in groups of 4–8 or occasionally 9–12 during this period (from late January to July). Pair-bonds started to form from February to March, whereas the breeding period lasted from March to June, marked by breeding calls, nesting, egg laying and incubation. The

Table 3. Chronology of Chukar Partridge breeding behavior

3. táblázat A csukár költésének kronológiája

Breeding behavior	Months
Breeding call begin	Late January
Frequent call heard	February
Pairing formation	February
Nesting	March/April
Incubation	April/ May
Hatching	May/June
Fledgling	July

maximum number of breeding calls were recorded in the months of February and March, indicating its breeding period *(Table 3).* During the breeding season, the difference between the first and second calls was 8–12 seconds, while in the non-breeding season, it was 1–2 minutes or even longer than 2 minutes. Similar to the rest of the months, calling frequency (number of calls per minute) was maximum in February (0.3) and March (0.26), indicating the start of breeding season. Males and females started to form pairs during this time. One way (ANOVA) showed a significant difference in call counts during various months of the year (df=6, F 2.60, P< 0.05).

Nest site selection

Forty-two active nests with a mean 10.5±3.354 were found in the study area at twelve different sites including six nests at Zone – I, fifteen at Zone – II, nine at Zone – III and twelve nest at Zone – IV *(Table 4)*. 27 out of forty two nests were built on slopes, with dense vegetation of *Dodonea viscosa, Calotropis procera* and *Zizyphus oxyphylla* along the walking trail *(Figure 2a),* 9 nests were created on the ground with a shallow depression, a scrape with dead leaves of *Desmodium elegans, Heliotropium strigosum, Solanum xanthocarpum,* soft stems and downy feathers. The remaining 6 nests were found in small depressions on the ground with dense vegetation of *Stellaria media, Dodonaea viscosa, Poa annua* and *Stachys parviflora* under the stones. All the related morphometry and nesting materials were measured only when the nests were left after hatching. Twelve nests were measured one in each transects. Shapes of the nests were round to partial round. The average diameter (cm) of the nest (n=12) (25.43±3.4) with cup diameter (23.32±3.1) and cup depth (6.20±1.2).

Incubation and hatching success

During the incubation period, only twelve nests were observed. The remaining nests were detected during incubation or after hatching. Wherever possible, all nests were monitored regularly at various habitats using either camera traps or binoculars *(Table 4).* The average clutch size was 94.5±30.187 eggs, (n=42 nests) with a minimum of 6 and a maximum of 14 eggs per nest. The incubation period was between 22–24 days (22.75±0.830). At the

Table 4. Breeding performance of Chukar Partridge at different sites
4. táblázat A csukár költési adatai az egyes vizsgált területeken

Study area	Nests (n)	Month	Average eggs hatched/nest (n=9)	Incubation period (days)	Hatching success (n) (%)	Chick mortality rate (n) %	Fledgling success (n) (%)
Zone – I	6	April	54	24	(41) 76%	(11) 26%	(30) 73%
Zone – II	15	March	135	22	(119) 88%	(14) 12%	(105) 88%
Zone – III	9	April	81	22	(67) 82%	(9) 13%	(58) 87%
Zone – IV	12	April	108	23	(88) 81%	(16) 18%	(72) 82%
Mean±SD	10.5±3.35		94.5±30.18	22.75±0.83	78.75±28.59	12.5±2.69	66.25±27.04

Figure 2. (a) showing field photograph of Chukar Partridge active nest (b) Hatching stage and leftover nest of Chukar Partridge (c) unsuccessful nest after hatching
2. ábra (a) a csukár aktív fészke, (b) sikeres költés utáni elhagyott fészke, (c) sikertelen fészke

nest, only female was seen incubating eggs. After being incubated by a female, a minimum number of eggs were fail to hatch *(Figure 2b)*. The hatching success was 82% in Zone – III, 81% in Zone – IV, 76% in Zone – I and 88% in Zone – II respectively. Of total 315 hatched chicks, 265 survived, whereas others were observed dead for unknown reasons *(Figure 2c)*. Overall the fledging success was 83% (n=42 nests) *(Table 4)*.

Discussion

The IUCN classifies the Chukar Partridge as "Least Concern," indicating that its population trend is stable and its population size is also very large (BirdLife International 2016). According to Rich *et al.* (2004), the global population of the species is estimated to be around 2,000,000 individuals. However, there have been reports of a population decrease in some parts of the world; for example, a small population in Europe is projected to decline at a rate of nearly 30% in 11.7 years (three generations) (BirdLife International 2015). Recent telemetry studies provide real information, but are expensive and require more effort. Count calls and line transects are commonly used techniques, but they have biases and limitations

(Evans *et al*. 2007). The number of Chukar sightings was lower in December across all the study zones of Pakistan. Zone – I is the worst impacted region because of the surrounding residential areas, where residents hunt for Chukar and destroy its habitat by cutting the grass and pasture *(Figure 1)*. Overgrazing can alter vegetation structure and composition, which can lead to changes in biodiversity and predator-prey relationships (Blaum *et al*. 2007). Chukar population densities in the study area varied from month to month due to differences in trapping, hunting, predation and diseases. The highest population density was recorded in Zone – II in September and October. Vegetation cover provides a variety of essential components for population survival and reproduction. This could be the reason for the increase in the number of birds observed per trail as vegetation cover with sufficient ground cover increased gradually in April and beyond at all the study sites. Chukar population densities were mostly higher in cultivated grass, slopes and barren rocks. According to Roberts (1991), Chukars could live in Pakistan arid, rocky, and hilly terrain. From August to October, however, the highest population density was observed in Zones – III and IV. Predation, hunting, and harsh environmental conditions caused the population decline in Zone – III to start in December and reach its lowest density in May. Chukar are preys of raptors that are frequently reported throughout the Great Basin and has been observed in Golden Eagles nests *(Aquila chrysaetos)* (Bloom & Hawks 1982). But in Zone – IV the lowest population was recorded in December, as a result of very harsh cold climate and territorial and altitudinal migration. Our results are consistent with those of Roberts (1991) and Christensen (2020). Overall the population density showed a nearly growing trend in September and October. Since this bird breeds in the summer, this number may be due to its breeding success. Population decline in December may be attributed to changes in activity

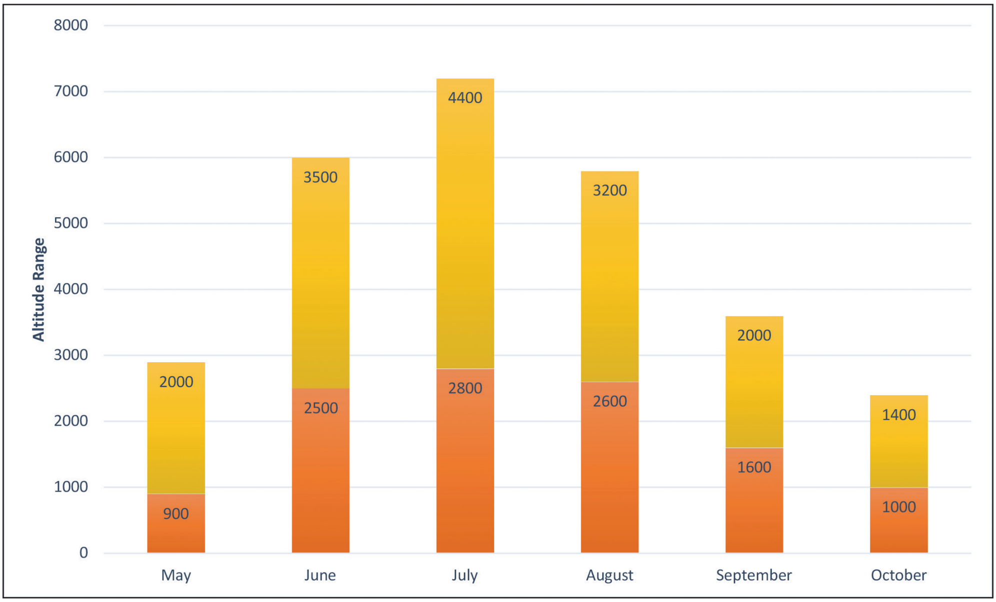

Figure 3. Altitudinal variation of Chukar Partridge in different months of the study area
3. ábra A csukár elterjedése tengerszintfeletti eloszlásának havi mintázata a vizsgált területen

patterns as the climate becomes harsher and most of the study areas is devoid of crops and other vegetation cover that could change the behavior and activities of these birds. In the study region, Chukar was found in a variety of altitudinal ranges *(Figure 3).* It can be found at elevations ranging from 900 to 4400 m (asl), in different months of the study area, though it can also be found at 600 m in Pakistan (Rasmussen & Anderton 2005). According to Roberts (1991), in summer it has been seen at 3353 m in Chitral and up to 3962 m in western Gilgit.

Breeding is an important phenomenon in the life stages of birds; the successful breeding results in stable populations of a species in an area. Most of the nests were spotted around *Dodonea viscosa, Calotropis procera* and *Zizyphus oxyphylla* on a sloping location with an average diameter of (26.42±3.3), with nearby water sources was preferred nest building sites. According to Olliver (2005) and Roberts (1991) Chukars build their nests under the rocks in dense scrub in a small depression on the ground with a diameter varying from 21–30.5 (cm). The building material of nest included grasses, dead leaves of *Poa annua, Cynodon dactylon, Dichanthium annulatum* and soft stems, mainly identified as *Zizyphus oxyphylla, Sarcococca saligna* were found abundantly. This has resulted in successful nesting of Chukars at various sampling locations. The nests were established successfully at different sites in the study area. According to Olliver (2005) Chukar Partridge builds a nest in a depression scratched in the ground under shrubberies or in particularly protected by rocks and fences in unpleasant areas. Our findings revealed that male Chukar started calling in January and early February, followed by pair formation. The months of March and April were the peak for breeding and nesting. These months indicate the spring season in the study area. Our results regarding Chukar Partridge pair formation and breeding records are consistent with earlier studies by del Hoyo *et al*. (1994) and Christensen (2020). We observed a maximum clutch size of 14 and a minimum of 6. The nests spotted in Zone – I, Zone – III and IV were initially found with 3, 2 and 4 eggs, then 12, 5 and 14 eggs were found at these nest. It can be assumed that the female lays eggs on different days, not on the same day.

Roberts (1991) reported a clutch size of 6 to 9 eggs in a drier environment, whereas Bates and Lowther (1952) found four clutch ranging between 15 and 19 eggs per clutch in Kashmir. Clutch size of Chukar was reported to be between 7–14 eggs and eggs hatched after an incubation period of about 23–25 days (Shahabuddin *et al*. 2016). Christensen (2020) reported that the incubation period was 24 days which is also the finding of our study. Using camera traps and binoculars wherever possible to achieve closer distances, the incubation period in the twelve nests was observed regularly and lasted about 23 days (22.75±0.830). Our findings show that the eggs hatched from May to July which is the summer season in the study area. This was also reported by Shahabuddin *et al*. (2016) from Totalai Game Reserve, Buner, Pakistan that summer is the breeding season of Chukar Partridge. Furthermore, the Chukar Partridge is nesting in the ground and is more frequently affected by birds of prey than by mammals (Angelstam 1986). In this study 265 of the 315 hatched chicks survived, while the others were found dead. However, no nests were found to be threatened by predation, so the chicks found dead were most likely the result of a parasitic attack or harsh environmental condition. The study of abundance or distribution estimates is important for the management and conservation of wildlife populations (Bibby *et al.* 1998). To increase the understanding on the status of a particular species, information about its distribution, habitat occupancy, population size and possible threats are very important. Unfortunately for chukar species in the region, quality information is not readily available. Although the current data reflects a 'snapshot' of the population, it does provide some valuable information on population status, habitat preference and some aspects of Chukar breeding biology in the Malakand division, Pakistan.

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References

Ahmad, B., Nabi, G., Hayat, M. & Muhammad, J. 2017. Habitat study of Chukar Partridge *(Alectorus chukar)* in district Malakand, A case study of town Thana, Pakistan. – Journal of Biodiversity and Environmental Sciences 11(4): 11–17.

Angelstam, P. 1986. Predation on ground nesting birds' nests in relation to predator densities and habitat edge per Angelstam. – Oikos 47: 365–373.

Bates, R. S. P. & Lowther, E. H. N. 1952. The history of bird photography in India. – Journal of Bombay Natural History Society 50: 779–784.

Besnard, A., Novoa, C. & Jimenez, O. 2010. Hunting impact on the population dynamics of Pyrenean Grey Partridge *Perdix perdix hispaniensis*. – Wildlife Biology 16(2): 135–143. DOI: 10.2981/08-077

Bhattacharya, T., Sathyakumar, S. & Rawat, G. S. 2009. Distribution and abundance of Galliformes in response to anthropogenic pressures in the buffer zone of Nanda Devi Biosphere Reserve. – International Journal of Galliformes Conservation 1: 78–84.

Bibby, C. J., Jones, M. & Marsden, S. 1998. Bird Surveys. – London: Expedition Advisory Centre

BirdLife International 2015. European Red List of Birds. – Office for Official Publications of the European Communities, Luxembourg

BirdLife International 2016. The IUCN red list of threatened species 2016: e.T22678691A89355978. DOI: 10.2305/IUCN.UK.2016-3.RLTS. T22678691A89355978.en. Downloaded on 06 November 2018.

Blaum, N., Rossmanith, E., Schwager, M. & Jeltsch, F. 2007. Responses of mammalian carnivores to land use in arid savanna rangelands. – Basic and Applied Ecology 8(6): 552–564. DOI: 10.1016/j.baae.2006.09.003

Bloom, P. H. & Hawks, S. J. 1982. Food habits of nesting Golden Eagles in northeast California and northwest Nevada. – Journal of Raptor Research 16: 110–115.

Buckland, S. T., Anderson, D. R., Burnham, K. P., Laake, J. L., Borchers, D. L. & Thomas, L. 2001. Introduction to Distance Sampling-estimating Abundance of Biological Populations. – Oxford University Press, Oxford

Çetin, O., Kırıkçı, K. & Gülşen, N. 1997. Some productivity characteristics of Chukar Partridges *(A. chukar)* in different management conditions. – Eurasian Journal of Veterinary Sciences 13(2): 5–10.

Christensen, G. C. 2020. Chukar *(Alectoris chukar)*, version 1.0. – In: Poole, A. F. & Gill, F. B. (eds.) Birds of the World. – Cornell Lab of Ornithology, Ithaca, NY, USA. DOI: 10.2173/bow.chukar.01.

del Hoyo, J., Elliott, A. & Sargatal, J. (eds.) 1994. Handbook of Birds of the World, Vol 2. New world vultures to guinea fowl. – Lynx Edicions, Barcelona, p. 234.

Evans, S. A., Mougeot, F., Redpath, S. M. & Fiona, L. 2007. Alternative methods for estimating density in an upland game bird: the Red Grouse *Lagopus lagopus scoticus*. – Wildlife Biology 13(2): 130–139. DOI: 10.2981/0909-6396(2007)13

Gaudioso, V. R., Alonso, M. E., Robles, R., Garrido, J. A. & Olmedo, J. A. 2002. Effects of housing type and breeding system on the reproductive capacity of the Red-legged Partridge *(Alectoris rufa)*. – Poultry Science 81(2): 169–172. DOI: 10.1093/ps/81.2.169

Grimmett, R., Roberts, T. & Inskipp, I. 2008. Birds of Pakistan. – Christopher Helm, London

Olliver, N. 2005. Birds of New Zealand: Chukar. 271 – http://www.nzbirds.com/birds/chukor. html (Accessed March 2011).

Rasmussen, P. C. & Anderton, J. C. 2005. Birds of South Asia: The Ripley Guide, Vol. 2. – Smithsonian Institution and Lynx Edicions

Rich, T. D., Beardmore, C. J., Berlanga, H., Blancher, P. J., Bradstreet, M. S. W., Butcher, G. S., Demarest, D. W., Dunn, E. H., Hunter, W. C., Inigo-Elias, E. E., Martell, A. M., Panjabi, A. O., Pashley, D. N., Rosenberg, K. V., Rustay, C. M., Wendt, J. S. & Will, T. C. 2004. Partners in flight: North American land bird conservation plan. – Cornell Lab of Ornithology, Ithaca, NY

Roberts, T. J. 1991. The Birds of Pakistan: Regional Studies and Non-passerines. – Oxford University Press, Karachi, Pakistan

Shahabuddin, S. K., Naveed, A., Adil, K., Waheed, A. & Basit, A. 2016. Exploring the population status of family Phasianidae in Totalai Game Reserve, District Buner, Khyber Pukhtunkhwa, Pakistan. – British Journal of Poultry Sciences 5(1): 13–20. DOI: 10.5829/idosi.bjps.2016.5.1.101200

WA. 2011. Fact sheet: Chukar Partridge. – Department of Agriculture and Food, Western Australia

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Human activities can hinder the breeding of a top avian predator: preliminary results

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Abstract The Eastern Imperial Eagle (EIE) is a large-sized, globally threatened species with a wide distribution. The species is extremely vulnerable and sensitive to human disturbance and activities in the vicinities of its nests. We analyzed the effect of human disturbance in two territories of Eastern Imperial Eagles from Bulgaria in 2008–2009. We recorded 375 cases of different types of human activity in the vicinities of the two surveyed nests – 60 cases in nest A and 315 in nest B. The most common activity around the studied nests was the passing of light motor vehicles (n=100 cases). Our results highlight that the probability of EIE's reaction is dependent on the type of activity, distance from the nest and the duration of the activity. However, eagles' reaction is independent from the number of intruders. We found that with the decrease of the distance to the nest, the reaction progresses and is more acute. We found statistical differences between the distance belts and the majority of alert and flight reactions that were recorded at distances up to 300 m from the nests. We reported that humans walking around nests (mainly hunters, fishermen, tourists, people illegally extracting sand in the close vicinities of the nests) result in a large number of reactions of flight off by the eagles thus, leaving the nest unattended. More research on a large scale on this topic is needed including more accurate measures to address human disturbance in EIE territories. The findings will be applied to ensure higher breeding rates and species conservation.

Keywords: Eastern Imperial Eagle, Bulgaria, disturbance, breeding, alteration, population

Összefoglalás A parlagi sas nagy testméretű, globálisan veszélyeztetett, széles elterjedési területtel rendelkező madárfaj, amely rendkívül érzékeny a fészke közelében végzett emberi tevékenységekre, zavarásra. A vizsgálatban ez utóbbiak hatását elemeztük Bulgáriából származó parlagi sasoknál, két területen 2008–2009-ben. A két vizsgált fészek környezetében 375 különböző típusú emberi tevékenységet rögzítettünk – 60 esetet az A, és 315-öt a B fészeknél. Ezek közül a leggyakoribb a könnyű gépjárművek áthaladása volt (n=100 eset). Eredményeink rámutatnak arra, hogy a parlagi sasok reakciója függ a tevékenység típusától, időtartamától és a fészektől való távolságtól, nem függ azonban a „betolakodók" számától. Megállapítottuk, hogy a fészektől való távolság csökkenésével a sasok reakciója egyre hevesebb. Statisztikai különbségeket találtunk az egyes távolságokra levő zónák, illetve a legtöbb riasztási és repülési reakció között, amelyeket a fészkektől legfeljebb 300 m-es távolságban rögzítettünk. Beszámoltunk arról, hogy az emberek (főleg vadászok, halászok, turisták, földmunkások) a fészkek körül járva gyakran zavarják fel a sasokat, így a fészkek sokszor felügyelet nélkül maradnak. A témában további, nagyobb léptékű kutatás szükséges annak érdekében, hogy a parlagi sasok elterjedési területein az ember okozta zavarás hatásait megfelelő intézkedésekkel ellensúlyozni lehessen. A kutatások eredményeit felhasználva biztosítható a parlagi sasok magasabb szaporodási aránya, illetve a faj megőrzése.

Kulcsszavak: parlagi sas, Bulgária, zavarás, költés, változás, populáció

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Introduction

The Eastern Imperial Eagle *(Aquila heliaca)*, hereafter EIE, is a long-lived, large-sized territorial raptor with a wide range spanning from Central Europe and the Balkans to South Siberia, China and Mongolia (BirdLife International 2021). The species global population is estimated to exceed 10,000 mature individuals (BirdLife International 2021), whereas its European population is estimated to 1,800–2,200 pairs during the period 2000–2010 (Demerdzhiev *et al.* 2011a). EIE faces severe threats, such as high adult and juvenile mortality due to persecution, poisoning, and hazardous powerlines, therefore the species is listed as vulnerable on a global scale (BirdLife International 2021). In Bulgaria, the EIE was a widespread species in the past (Patev 1950). Recovery of the population started from 2000 onwards and the species gradually increased reaching 35–40 pairs nowadays (authors data), distributed mainly in the southeastern part of the country (Demerdzhiev *et al.* 2014).

Human activities have severely affected biodiversity and raptor populations worldwide (Heath & Evans 2000, McClure *et al.* 2018). In this respect, numerous studies explore the relationship between human activities and their effect on various breeding rates in birds and raptors (White & Thurow 1985, Jenny 1992, Watson 1994, McGrady 1997, Ruhlen *et al.* 2003, Ruddock & Whitfield 2007). Human activities in the close vicinity of the nest during incubation and early stages of chick development can alter breeding in raptors (Grier & Fyve 1987, Grubb *et al.* 1992, Holmes *et al.* 1993, Steidl & Anthony 1996, Richardson & Miller 1997, Swarthout & Steidl 2001). Disturbance of breeding birds can increase energy costs and decrease hunting success and/or lead to abnormal distribution of nest attendance (Grier & Fyve 1987). Moreover, human disturbance can affect also parental care in various dimensions (Fernandez & Ackona 1993, Verhulst *et al.* 2001, Bautista *et al.* 2004), although behavioral response varies individually (Richardson & Miller 1997). Various studies from Spain have found a considerable negative effect of human activities over the demography of the Spanish Imperial Eagle *(Aquila adalberti)* (hereafter SIE) (Gonzalez *et al.* 1990, Gonzalez *et al.* 1992, Ferrer & Harte 1997, Bisson *et al.* 2002, Gonzalez *et al.* 2006). Results show that habitat selection and breeding success are affected by urbanization (for example distance from the nearest settlement) and inaccessibility of the breeding territories to humans that suggests that the SIE avoids disturbance (Gonzalez *et al.* 1992, Castano & Guzman 1995). As a result of the surveys conducted in Spain, some authors recommended a ban of human activities in a radius of 500–800 m around nests of Spanish Imperial Eagle during the breeding season (Gonzalez *et al.* 2006). Contrastingly, such measures might not improve fecundity and lead to a negative attitude towards eagles. Thus, others argue that the acquaintance of the SIE to human disturbance might increase the potential breeding habitat and adult survival of the species and improve its plasticity and tolerance to humans (Ferrer *et al.* 2007).

Studies on human disturbance and its effects on the breeding rates of the EIE are scarce. However, human disturbance was listed as one of the main threats for the EIE in Bulgaria, altering species breeding rates, especially in plain areas (Demerdzhiev *et al.* 2011a). A survey that analyses the effects of urbanization, transport and power lines over the current and potential distribution of the EIE in the Pannonian Plain, suggests that eagles avoid human activities at a certain distance and that settlements are the most important factor in terms of territories selection and establishment (Horváth *et al.* 2009).

In Bulgaria, the impact of human activities on EIE's behavior in the breeding season is barely studied, not quantified, except for some characteristics of parental care (Dobrev 2009). Based on the considerable changes in species breeding and foraging habitats, the conservation status of the EIE and species vulnerability to habitat alterations (Demerdzhiev *et al.* 2011a, Demerdzhiev *et al.* in press), a detailed study on the relationship between human activities and the response of the EIE is needed. The current study aims to reveal the impact of different human activities on the reactions of the EIE in the breeding season and investigate the tolerance of the species towards such disturbances, giving a first preliminary assessment. Hence, we set the following objectives: (1) to reveal the most common human activities and their magnitude in EIE territories; (2) to define and measure eagle's reactions and (3) suggest primary conservation measures to avoid disturbance and nest/clutch abandonment.

Materials and Methods

Study area

The study was conducted in the Sakar Mountain, Bulgaria. This is a low-mountain region landlocked in the southeastern corner of the country *(Figure 1)*. The area is characterized by a continental-Mediterranean climate. It contains patches of mixed deciduous forests of several oak species (*Quercus* sp.) and oriental hornbeam *(Carpinus orientalis)*, agricultural areas, pastures and open areas overgrown with shrubs of Jerusalem thorn *(Paliurus spina-christi)*, and xerothermic grass communities (Bondev 1991).

Survey design

Observations were carried out in April–July during the period 2008–2009. We monitored three breeding attempts of two EIE pairs during the breeding season, 20% of the species population in the studied area (n=10) (Demerdzhiev *et al.* 2011a). Observations lasted between 7.00 and 20.00 h each monitoring day (n=67) and were performed from vantage points at a distance of around 1,000 m from the nest using binoculars 10x50 and spotting scopes 20x60 (Bibby *et al.* 1999). To avoid disturbance, alarming the birds and register species natural reactions, we followed recommendations of Gonzalez *et al.* (2006) and Zuberogoitia *et al.* (2008). Birds were not provoked to study their reactions (White & Thurow 1985). The age of the partners within each pair was determined following Forsman (2005). Pair A consisted of fully adult birds, whereas in pair B, the male partner was fully adult in sixth plumage, while the female was in fifth plumage. During the second year, however, pair B consisted of the same female in its sixth plumage and a new male in fifth plumage, because of a replacement within the pair. Observations covered the period from the laying of the eggs until fledging. For each breeding attempt, we differentiated two periods:

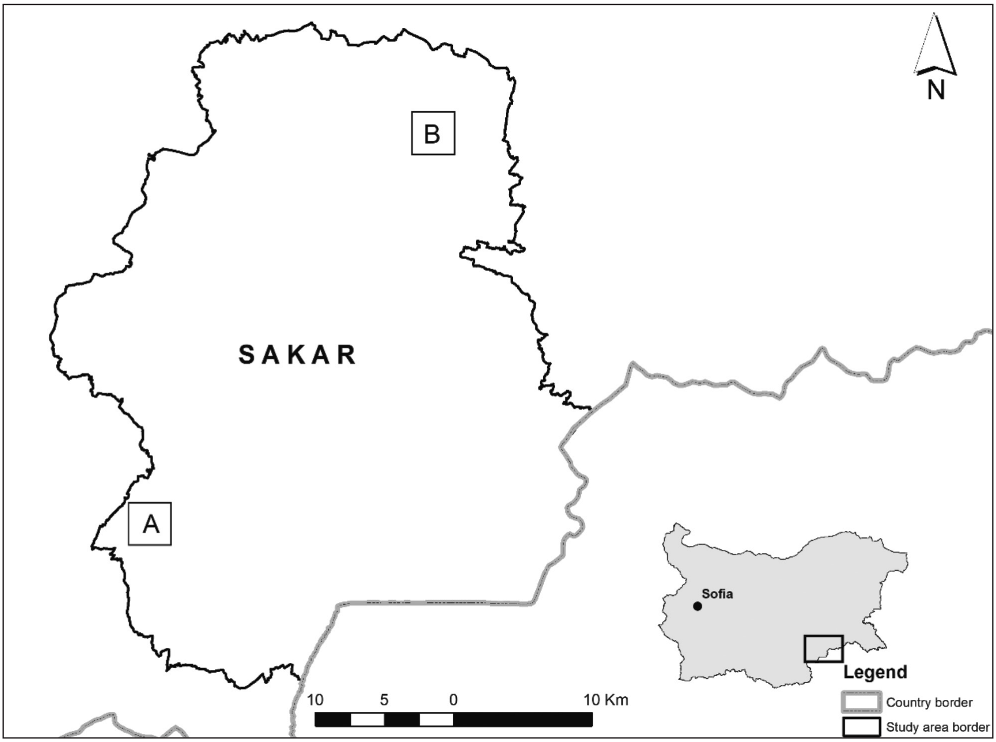

Figure 1. Location of the study area and the surveyed nests A and B
1. ábra A vizsgálati terület és a vizsgált fészkek (A és B) elhelyezkedése

(a) incubation (from the laying of the first egg until hatching of the first chick) and (b) chick rearing period (since the hatching of the first egg until fledging).

We recorded and categorized human activities to assess their quantity, magnitude and impact on the reactions of the species when they occurred in the vicinity of the nests. Whenever such activity occurred less than 1 km of the nest (Gonzalez *et al.* 2006), we recorded the following information: (1) exact starting time, (2) type and (3) duration of the activity, (4) linear distance to the nest, (5) number of people or vehicles involved in the activity and (6) the eagles' response. The type of activity was defined according to the following seven categories: A1 – light vehicles (cars, 4wd cars, microbuses, motorcycles); A2 – agricultural machinery (tractors, reaping machines); A3 – freight motor vehicles (trucks, buses); A4 – non-mechanized vehicles (carriages, bicycles); A5 – illegal, small quarries for sand extraction around nests; A6 – locals and/or shepherds regularly present in the area; A7 – casual people non-regularly present in the area (hunters, fishermen). The duration of the activity was categorized into the following time intervals: T1 (short-term activity) – 1–2 min; T2 (medium-term activity) – 3–10 min and T3 (long-term activity) – over 10 min. The linear distance to the nest was categorized into the following distance belts: 0–100 m; 101–300 m; 301–500 m and over 500 m (Ruddock & Whitfield 2007). The number of people or vehicles involved in the activity where: N1 – single people/vehicles and N2 – groups of two and/or more people/vehicles.

The reaction of the birds was categorized as follows: (NR) No reaction, whenever birds do not react to the particular activity; (AR) Alarm reaction, whenever a bird changes its behavior, is abused by the activity and exhibits alarming signs (staring, head signals) without leaving the nest unattended; (FR) Flight reaction when the bird is scared away and leaves the nest. We further measured the intrusion frequency, calculated as the ratio of the total number of human activities registered for each breeding attempt to the total number of observation days (Gonzalez *et al*. 2006).

Statistical procedures

Data were analyzed using Statistica for Windows, Release 10.0 (Statsoft Inc 2011). We employed descriptive statistics to calculate the observation time where means are presented ± Standard Deviation (SD). We used Break down & one way ANOVA to study the effect of different human activities on the reactions of the eagles. We set the reaction of the birds as a dependent variable and the number, the distance, the type and the duration of the activities as predictor variables. All tests were considered significant when $p \leq 0.05$.

Results

During our study, we spent 672.15 h of observations (192±17 h for nest A and 480±10 h for nest B). The observation time during incubation for nest A was 104.65±2 h and 87.35±3 h during the chick-rearing period. We observed nest B for 273.15±5 h during incubation and for 207.05±6 h during the chick-rearing period. We recorded 375 cases of different types of human activity in the vicinities of the two surveyed nests. The majority of the activities were recorded during the chick-rearing season (n=335, 89%) and the rest during incubation (n=40, 11%). We recorded 60 activities in nest A and 315 in nest B. The most common activities around the studied nests were A1 (n=100 cases, 26.6%), A6 (n=96, 25.6%) and A4 (n=85, 22.7%). In 2008, the intrusion frequency for nest A was 1.46 and 6.35 for nest B. In 2009 the intrusion frequency for nest B was 10.4.

In 313 cases (83%) we did not record any reaction of the eagles. Whilst, in 40 cases (11%) eagles showed signs of alarming reaction towards the given activity and in 22 cases (6%) eagles left the nest without attendance because of it. Our results highlight that the probability of reaction in the EIE is dependent on the type of the activity ($F=10.09$, $P=0.002$), distance from the nest ($F=8.39$, $P=0.004$) and the duration of the activity ($F=7.29$, $P=0.007$). Our results show that eagles' reaction is independent from the number of intruders ($F=0.00$, $P=0.99$).

We registered a statistical significance of the reactions of the eagle towards locals (A6) ($t=-5.15$, $P<0.0001$) and casual people (A7) ($t=28.88$, $P<0.0001$). We found that humans walking around nests (mainly hunters, fishermen, tourists, people illegally extracting sand in the close vicinities of the nests) result in a large number of reactions of flight off thus, leaving the nest unattended. We, however, noted that EIE could adjust to local people and routine activities as an adaptation to breeding successfully, especially in highly intensified agricultural fields and rural areas.

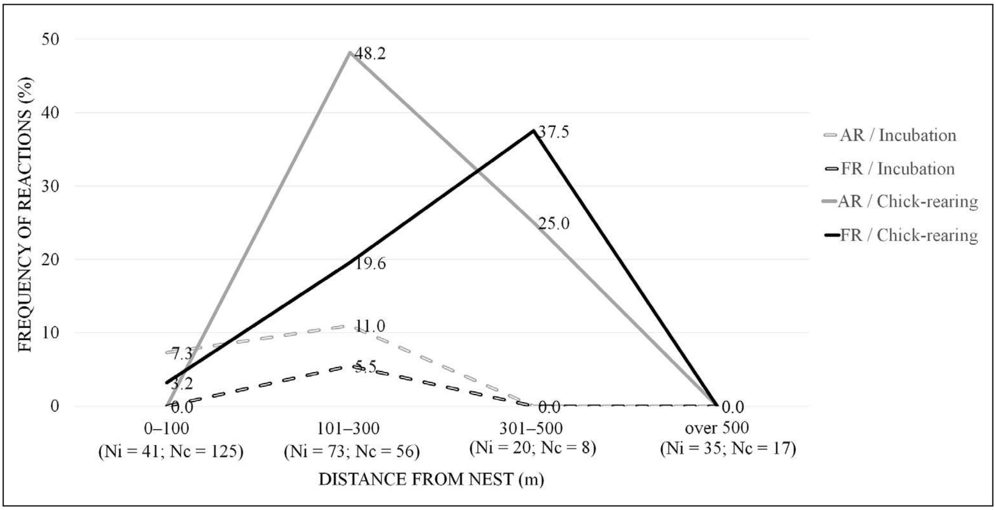

Figure 2. Frequency of alert (AR) and flight reactions (FR) caused by human activities according to their distance from the nest and breeding stage. Number of registered human activities in the different distance belts during incubation (Ni) and chick-rearing (Nc) are indicated respectively in brackets

2. ábra A riasztási (AR) és repülési reakciók (FR) aránya az emberi tevékenység hatására a fészektől való távolság és a költési stádium függvényében. A regisztrált emberi tevékenységek száma az egyes távolsági kategóriákban a kotlási (Ni) és a fiókanevelési (Nc) időszakokban zárójelben található

Eagles start to respond to any intruder when it approaches at distances of 301–500 m from the nest. However, we found that with the decrease of the distance to the nest, the reaction progresses and its frequency is most pronounced at a distance belt 101–300 m (t=-2.66, P=0.008) when the majority of the reactions were registered during incubation and chick-rearing period (n=50). At this distance, we recorded 5.5% of the flight reactions during incubation (n=4) and 19.6% of the flight reactions during the chick-rearing period (n=11). At distance 301–500 m, we recorded 2 alert reactions (25%) during chick-rearing period and 3 flight reactions (37.5%). We did not register reactions at distances greater than 500 m *(Figure 2)*.

Discussion

Human disturbance to wildlife is a growing topic of concern that can deviate animals from their normal behavior and result in different biological and ecological responses (Martínez-Abraín *et al.* 2010, Pauli *et al.* 2016). We recorded a significant number of human activities in the close vicinities of the nests whereas passing cars accounted for the majority of the cases. Similar results of numerous human activities in the vicinities of eagles nests were reported from Spain where more than 2,000 observations of different activities were recorded (Gonzalez *et al.* 2006). This can be explained by species ecology, inhabiting open and human-managed landscapes, namely arable lands, pastures and rural areas in general. Hence, some authors suggested that habituation of breeding pairs to human activities can potentially increase species nesting habitat in the human-dominated landscapes (Ferrer *et al.* 2007).

We found a statistical significance of the reactions of the eagle towards humans walking around nests (mainly hunters, fishermen, tourists) that resulted in a large number of reactions of flight off. The most common characteristics of such people are the lack of a behavioral model. They stop at random positions, spend different times not moving or staring doubtfully. Such behavior is untypical for eagle-known models of routine activities, for example, by shepherds or vehicles that simply cross the territory, in a discrete interval of time during similar day hours, etc. (Gonzalez *et al.* 2006). Human disturbance of bird species (Fernandez-Juricic 2002) can urge birds to occupy lower quality territories (McGarigal *et al.* 1991), decrease nest attendance time (Martínez-Abraín *et al.* 2010) or compromise breeding (Zuberogoitia *et al.* 2008, Zuberogoitia *et al.* 2014). Some authors found that EIE responds to long-term disturbance by selecting nesting habitats away from human infrastructures (Gonzalez *et al.* 1992, Bisson *et al.* 2002, Horváth *et al.* 2009). In our study, light vehicles were the most frequent in EIE territories, similarly to the SIE (Gonzalez *et al.* 2006). We did not record any reaction of the eagles to this activity, but also towards non-mechanized vehicles and agricultural machinery. These types of activities are familiar and temporally predictable around nests (passing periodically, tilting, mowing, sowing). Nevertheless, they can affect eagles when they are executed very close to the nests and/or last long.

We did not find any relationship between the number of intruders and eagles' reaction. Conversely, a study from Spain revealed that the more people are involved, the stronger the reaction of the eagles was (Gonzalez *et al.* 2006). A larger number or groups of people would be easily spotted by the eagles at a greater distance. However, even a single person can disturb eagles when approaches too close to the nest similarly to a group of people, which might explain the lack of significance in our study. Nonetheless, the popularity of countryside tourism and rural areas among people living in urbanized areas is growing (Martínez-Abraín *et al.* 2010, Perona *et al.* 2019). As a result, the concentration of people during weekends and holidays might become an issue in some of the EIE territories in the future, considering also the species distribution in Bulgaria (Demerdzhiev *et al.* 2011b).

Our results are in agreement with previous studies in terms of the average reaction distance in the Golden Eagle *(Aquila chrysaetos)* (Holmes *et al.* 1993), the SIE (Gonzalez *et al.* 2006) and the Bald Eagle *(Haliaetus leucocephalus)* (Stalmaster & Newman 1978, Fraser *et al.* 1985, Grubb & King 1991, Steidl & Anthony 2000, Fernandez-Juricic 2002). In SIE, the average alert distance was 252 m (range 50–580), and the average flight distance was 261 m (Gonzalez *et al.* 2006). Golden Eagle exhibits alarming reaction at 400 m during incubation and at 625 m during chick-rearing period. The species leaves the nest at an average distance of 225 m during incubation and 400 m during the chick-rearing period (Ruddock & Whitfield 2007). These results confirm that a minimum of 250–300 m is required to buffer species reactions. Moreover, this implies that similar to other raptor species, the EIE, can detect hazards without showing any signs of a reaction. Therefore, early alarming reactions (before the flight of the bird) can remain unnoticed by the observer until the human activity is close enough to provoke the eagle to take off (Gonzalez *et al.* 2006). Based on our findings and other studies, we suggest that significant and/or regular human activities likely to cause frequent flight reactions from the nests of the EIE should be restricted at around 500 m to all active nests as a precaution measure during the breeding season. Nevertheless, such decisions must be considered cautiously and accommodated individually to every pair to avoid unnecessary negative reactions from the local communities (Ferrer *et al.* 2007, Zuberogoitia *et al.* 2014).

In the current survey, we found a statistically significant dependence between the reaction of the eagles and the duration of the activity in all time frames. We found that eagles react to some activities that last short, however, are unfamiliar an unusual to the eagles. For example, casual people, tourists, motorcycles that pass quickly through the territory but are not a routine around the nest. This is in contrast with the duration of activities that are not random, but regular around nests (for example ploughing, shepherds passing) but eagles are prone to them and recognize it. Therefore, eagles would react stronger to sudden and not typical activities around nests as a response to the long-term persecution from humans (Bijleveld 1974).

We recommend further and detailed research on this topic where the effect of different human activities is assessed to breeding success, productivity, age of birds and occupancy in much broader population scale. Thereby, a more robust estimate on human activities and their impact will be received and precise conservation measures (nest-guarding, adaptive management, restriction zones, etc.) might be applied accordingly at each territory.

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References

Bautista, L. M., García, J. T., Calmaestra, R. G., Palacin, C., Martín, C. A., Morales, M. B., Bonal, R. & Viñuela, J. 2004. Effect of weekend road traffic on the use of space by raptors. – Conservation Biology 18(3): 1–7. DOI: 10.1111/j.1523-1739.2004.00499.x

Bibby, C., Jones, M. & Marsden, S. 1999. Expedition Field Techniques. – Bird Surveys. Royal Geographical Society, London

Bijleveld, M. 1974. Birds of Prey in Europe. – The MacMillan Press Ltd., London

BirdLife International 2021. Species factsheet: *Aquila heliaca*. – http://www.birdlife.org on 06/03/2021.

Bisson, I. A., Ferrer, M. & Bird, D. M. 2002. Factors influencing nest-site selection by Spanish Imperial Eagles. – Journal of Field Ornithology 73(3): 298–302. DOI: 10.1648/0273-8570-73.3.298

Bondev, Z. 1991. Flora in Bulgaria. – Publishing House “Kliment Ohridsky”, Sofia (in Bulgarian)

Castano, J. P. & Guzman, J. A. 1995. Reproductive aspects of *Aquila adalberti* and *Aquila chrysaetos* in eastern Sierra Morena. – Ardeola 42(1): 83–89.

Demerdzhiev, D., Horváth, M., Kovács, A., Stoychev, S. & Karyakin, I. 2011a Status and population trend of the Eastern Imperial Eagle *(Aquila heliaca)* in Europe in the period 2000–2010. – Acta Zoologica Bulgarica Suppl. 3: 5–14.

Demerdzhiev, D., Dobrev, D., Popgeorgiev, G., Stoychev, S. in press. Landscape alteration affecting the demography of an endangered avian predator by reducing the habitat quality. – Journal of Avian Research

Demerdzhiev, D. A., Gradev, G. Zh., Stoychev, S. A., Ivanov, I., Petrov, Tz. & Marin, S. A. 2011b Increase of the population of the Eastern Imperial Eagle *(Aquila heliaca)* in Bulgaria. – Acta Zoologica Bulgarica Suppl. 3: 41–54.

Demerdzhiev, D., Stoychev, S., Dobrev, D., Spasov, S. & Terziev, N. 2014. Conservation measures undertaken to improve the population status of Eastern Imperial Eagle *(Aquila heliaca)* in Bulgaria. – Slovak Raptor Journal 8(1): 27–39. DOI: 10.2478/srj-2014-0007

Dobrev, D. 2009. Some aspects of nest etology of the Eastern Imperial Eagle *(Aquila heliaca)* (Aves: Accipitridae) in Sakar mountain. – Ecologia Balkanica 1: 60–66.

Fernandez, C. & Azkona, P. 1993. Human disturbance affects parental care of Marsh Harriers and nutritional status of nestlings. – Journal of Wildlife Management 57(3): 602–608. DOI: 10.2307/3809289

Fernandez-Juricic, E. 2002. Can human disturbance promote nestedness? A case study with breeding birds in urban habitat fragments. – Oecologia 131(2): 269–278. DOI: 10.1007/s00442-002-0883-y

Ferrer, M. & Harte, M. 1997. Habitat selection by immature Spanish Imperial Eagles during the dispersal period. – Journal of Applied Ecology 34(6): 1359–1364. DOI: 10.2307/2405253

Ferrer, M., Negro, J., Casado, E., Muriel, R. & Madero, A. 2007. Human disturbance and the conservation of the Spanish Imperial Eagle: A response to Gonzalez *et al.* 2006. – Animal Conservation 10(3): 293–294. DOI: 10.1111/j.1469-1795.2006.00090.x

Forsman, D. 2005. Eastern Imperial Eagle plumages. – Alula 4: 147–152.

Fraser, J. D., Frenzel, L. & Mathisen, J. E. 1985. The impact of human activities on breeding Bald Eagles in north-central Minnesota. – Journal of Wildlife Management 49: 585–592. DOI: 10.2307/3801676

Gonzalez, L. M., Arroyo, B., Margalida, A., Sanchez, R. & Oria, J. 2006. Effect of human activities on the behaviour of breeding Spanish Imperial Eagle *(Aquila adalberti)*: management implications for the conservation of a threatened species. – Animal Conservation 9(1): 85–93. DOI: 10.1111/j.1469-1795.2005.00016.x

Gonzalez, L. M., Bustamante, J. & Hiraldo, F. 1990. Factors influencing the present distribution of the Spanish Imperial Eagle *(Aquila adalberti)*. – Biological Conservation 51(4): 311–319. DOI: 10.1016/0006-3207(90)90116-7

Gonzalez, L. M., Bustamante, J. & Hiraldo, F. 1992. Nesting habitat selection by the Spanish Imperial Eagles *Aquila adalberti*. – Biological Conservation 59(1): 45–50. DOI: 10.1016/0006-3207(92)90712-V

Grier, J. W. & Fyfe, R. W. 1987. Preventing research and management disturbance. – In: Pendleton, B. A. G., Millsap, B. A., Cline, K. W. & Bird, O. M. (eds.) Raptor Management Techniques Manual. – Scientific Technical Ser. No. 10, National Wildlife Federation, pp. 173–182.

Grubb, T. G. Bowerman, W. W., Giesy, J. P. & Dawson, G. A. 1992. Responses of breeding Bald Eagles *Haliaetus leucocephalus* to human activities in northcentral Michigan. – Canadian Field-Naturalists 106(4): 443–453.

Grubb, T. G. & King, R. M. 1991. Assessing human disturbance of breeding Bald Eagles with classification tree models. – Journal of Wildlife Management 55(3): 500–511. DOI: 10.2307/3808982

Heath, M. F. & Evans, M. I. (eds.) 2000. Important Bird Areas in Europe: Priority Sites for Conservation. – BirdLife Conservation Series No. 8., BirdLife International, Cambridge, UK

Holmes, T. L., Knight, R. L., Stegall, L. & Craig, G. R. 1993. Responses of wintering grassland raptors to human disturbance. – Wildlife Society Bulletin 21: 461–468.

Horváth, M., Fehérvári, P., Szitta, T., Harnos, A., Kabai, P. & Moskát, Cs. 2009. Effects of infrastructural network on the distribution of Imperial Eagles in the Hungarian Plain. – In: Horváth, M. Habitat and prey selection in Imperial Eagles *(Aquila heliaca)*. – PhD thesis chapter, Faculty of Science, Biological Institute, Eötvös Loránd University

Jenny, D. 1992. Bruterfolg und Bestandsregulatione iner alpinen Population des Steinadlers *Aquila chrysaetos* [Breeding success and population regulation in the alpine population of the Golden Eagle *Aquila chrysaetos*]. – Der Ornithologische Beobachter 89: 1–43. (in German)

Martínez-Abraín, A., Oro, D., Jiménez, D., Stewart, J. & Pullin, A. 2010. A systematic review of the effect of recreational activities on nesting birds of prey. – Basic and Applied Ecology 11(4): 312–319. DOI: 10.1016/j.baae.2009.12.011.

McClure, Ch. J. W., Westrip, J. R. S., Johnson, J. A., Schulwitz, S. E., Virani, M. Z., Davies, R., Symes, A., Wheatley, H., Thorstrom, R., Amar, A., Buij, R., Jones, V. R., Williams, N. P., Buechley, E. R. & Butchart,

S. H. M. 2018. State of the world's raptors: Distributions, threats, and conservation recommendations. – Biological Conservation 227: 390–402. DOI: 10.1016/j.biocon.2018.08.012

McGarigal, K., Anthony, R. G. & Isaacs, F. B. 1991. Interactions of human and Bald Eagles on the Columbia Riverestuary. – Wildlife Monographs 115: 3–47.

McGrady, M. 1997. *Aquila chrysaetos* Golden Eagle. – BWPUpdate 1: 99–114.

Patev, P. 1950. Birds in Bulgaria. – BAS, Sofia (in Bulgarian)

Pauli, B., Spaul, R. & Heath, J. 2016. Forecasting disturbance effects on wildlife: Tolerance does not mitigate effects of increased recreation on wildlands. – Animal Conservation 20(3): 251–260. DOI: 10.1111/acv.12308

Perona, A., Urios, V. & López-López, P. 2019. Holidays? Not for all. Eagles have larger home ranges on holidays as a consequence of human disturbance. – Biological Conservation 231: 59–66. DOI: 10.1016/j.biocon.2019.01.010

Richardson, C. T. & Miller, C. K. 1997. Recommendations for protecting raptors from human disturbance: a review. – Wildlife Society Bulletin 25: 634–638.

Ruddock, M. & Whitfield, D. 2007. A review of disturbance distances in selected bird species. – A report from Natural Research (Projects) Ltd. to Scottish Natural Heritage

Ruhlen, T. D., Abbott, S., Stenzel, L. E. & Page, G. W. 2003. Evidence that human disturbance reduces Snowy Plover chick survival. – Journal of Field Ornithology 74(3): 300–304. DOI: 10.1648/0273-8570-74.3.300

Stalmaster, M. V. & Newman, J. R. 1978. Behavioral responses of wintering Bald Eagles to human intrusion. – Journal of Wildlife Management 42(3): 506–513. DOI: 10.2307/3800811

Statsoft 2011. STATISTICA (data analysis software system), version 10. – www.statsoft.com.

Steidl, R. J. & Anthony, R. G. 1996. Responses of Bald Eagles to human activity during the summer in interior Alaska. – Ecological Applications 6(2): 482–491. DOI: 10.2307/2269385

Swarthout, E. C. H. & Steidl, R. J. 2001. Flush responses of Mexican Spotted Owls to recreationists. – Journal of Wildlife Management 65(2): 312–317. DOI: 10.2307/3802910

Verhulst, S., Oosterbeek, K. & Ens, B. J. 2001. Experimental evidence of effects of human disturbance on foraging and parental care in Oystercatchers. – Biological Conservation 101(3): 375–380. DOI: 10.1016/S0006-3207(01)00084-2

Watson, J. 1994. Golden Eagle *Aquila chrysaetos*. – In: Tucker, G. M. & Heath, M. F. (eds.) Birds in Europe: Their Conservation Status. – BirdLife Conservation Series No 3., Cambridge, U.K., pp. 180–181.

White, C. & Thurow, T. L. 1985. Reproduction of Ferruginous Hawks exposed to controlled disturbance. – Condor 87: 14–22.

Zuberogoitia, I., Zabala, J., Martínez, J. A., Martínez, J. E. & Azkona, A. 2008. Effect of human activities on Egyptian Vulture breeding success. – Animal Conservation 11(4): 313–320. DOI: 10.1111/j.1469-1795.2008.00184.x.

Zuberogoitia, I., Zabala-Albizua, J., Martínez, J., González-Oreja, J. & López-López, P. 2014. Effective conservation measures to mitigate the impact of human disturbances on the endangered Egyptian Vulture. – Animal Conservation 17(5): 410–418. DOI: 10.1111/acv.12107.

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Seasonal and age-specific dynamics of the Griffon Vulture's home range and movements in the Eastern Rhodopes

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Abstract The spatial ecology of the Eurasian Griffon Vulture *(Gyps fulvus)* has been a subject of scientific interest for long due to its conservation status, critical ecosystem role, gregarious lifestyle and complex foraging behavior. The trans-border Eastern Rhodope Mountain in Bulgaria and Greece holds an increasing population of the species and one of the largest on the Balkan Peninsula. We used high-frequency GPS data from 13 Griffon Vultures from this population to study their movements, home range size and its seasonal or age specific dynamics. The overall foraging home range (95% kernel) was 3,204 km^2 and the core area of activity (50% kernel) was 256.5 km^2. We found high seasonal variation of the home range size. Vultures were foraging over larger areas in the summer and spring but their activity was limited to four times smaller areas in winter. We found no age specific variation in the home range sizes but the non-adult vultures showed tendency to conduct exploratory movements far from the breeding colony. Our results can be used for planning conservation efforts in the areas of high importance for the species.

Keywords: *Gyps fulvus*, GPS-tracking, foraging, movement, raptor conservation

Összefoglalás Az eurázsiai fakó keselyű *(Gyps fulvus)* térökológiája régóta a tudományos érdeklődés tárgya a faj védettségi állapota, kritikus ökoszisztéma-szerepe, közösségi életmódja és összetett táplálkozási viselkedése miatt. A határokon átnyúló, Bulgáriában és Görögországban található Kelet-Rodope-hegységben a faj populációja növekvő, és az egyik legnagyobb a Balkán-félszigeten. Ebből a populációból 13 egyed nagyfrekvenciás GPS-adatait használtuk fel mozgásuk, mozgáskörzetük és szezonális vagy korspecifikus dinamikájuk tanulmányozására. A teljes táplálkozási terület (95% valószínűségi elterjedés) 3204 km^2, a fő tevékenységi terület (50% valószínűségi elterjedés) 256,5 km^2 volt. A mozgáskörzet méretében nagy szezonális ingadozást találtunk. A keselyűk nyáron és tavasszal nagyobb területeken kerestek táplálékot, ehhez képes télen tevékenységük negyedannyi területre korlátozódott. Nem találtunk korspecifikus eltérést a mozgáskörzet méretében, de a fiatal keselyűk hajlamosak voltak felfedező mozgást végezni a költőkolóniától távol is. Eredményeink felhasználhatóak a faj szempontjából kiemelt jelentőségű területek védelmi intézkedéseinek tervezésére.

Kulcsszavak: fakó keselyű, GPS-követés, táplálkozás, mozgás, ragadozómadár-védelem

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Introduction

The Eurasian Griffon Vulture *(Gyps fulvus)* is a large long-lived scavenger with a breeding distribution extending from Portugal in the west to the Himalayas in the east (Ferguson Lees & Christie 2001). It is listed as 'Least Concern' by the IUCN and is the most common vulture species in Europe (BirdLife International 2021). The Iberian Peninsula holds over 85% of its European population (Botha *et al.* 2017, Del Moral & Molina 2018). In the Balkan Peninsula, the species has undergone a decline during the 20th century (Andevski 2013). However, over the past 30 years it showed recovery in Serbia, Bulgaria and Croatia (Dobrev *et al.* 2021a). Its population in Bulgaria has steadily increased as a result of intensive conservation actions and successful reintroduction programs (Demerdzhiev *et al.* 2014, Kmetova-Biro 2021, Dobrev *et al.* 2021a). The adult and immature Griffon Vultures are mostly resident, while some juveniles are migratory and winter in the Middle East and Africa (del Hoyo *et al.* 1994, McGrady & Gavashelishvili 2006, Arkumarev *et al.* 2019).

The Griffon Vulture is an obligate scavenger, exclusively feeding on carcasses of livestock and wild ungulates (Cramp & Simmons 1980), thus providing critical ecosystem services (Pain *et al.* 2003, Grilli *et al.* 2019). It relies on food resources, which are spatially and temporally unpredictable. Therefore, vultures evolved various adaptations to increase their foraging efficiency and minimize the energy costs related to flight (Spiegel *et al.* 2013a, Duriez *et al.* 2014). The social foraging strategy reduces the uncertainty regarding food location (Harel *et al.* 2017). Vultures almost exclusively use soaring-gliding flight supported by thermal updrafts to travel long distances in search of carcasses minimizing the energy costs (Ruxton & Houston 2004). Griffon Vultures forage over wide range of open habitats and have large home ranges (García-Ripollés *et al.* 2011). However, their ranging movements are affected by the availability of food and they tend to concentrate near predictable food sources (Deygout *et al.* 2010, Monsarrat *et al.* 2013, Dobrev *et al.* 2021b). In the Balkans, vultures largely depend on extensively grazed livestock and supplementary feeding stations, which shape their home ranges (Azmanis 2009, Arkumarev *et al.* 2021a). The spatial ecology of the Griffon Vulture has been a subject of scientific interest for long due to its conservation status, critical ecosystem role and complex foraging behavior. GPS-based telemetry has been used as an effective tool to study the movements of vultures and other raptors and to identify the most frequently used foraging areas, the threats in these areas and to inform the conservationists (Alarcon & Lambertucci 2018). The movement ecology is now providing the knowledge needed to incorporate movements of species into management planning (Allen & Singh 2016). Vulture movement data is used to assess the effectiveness of protected areas, the risk of collision with human-made infrastructure, detect poisoning incidents and identify mortality hotspots and priority areas for conservation (Alarcon & Lambertucci 2018). Studies on the Griffon Vulture's movements were already conducted in Spain (García-Ripollés *et al.* 2011), Israel (Bahat *et al.* 2001), Crete (Xirouchakis *et al.* 2021), on the Balkans (Peshev *et al.* 2021), Bulgaria (Peshev *et al.* 2018), Serbia (Hribsek *et al.* 2021), Eastern Alps and Croatia (Genero *et al.* 2020). However, the increasing Griffon Vulture population in the trans-border Rhodope Mountains in Bulgaria and Greece has not yet been studied. The growing number of planned new energy infrastructure in the area

(Kafetzis *et al.* 2017) requires detailed studies on vulture's movements and foraging. We determined the home range size of adult and non-adult vultures from the Eastern Rhodopean population and studied the seasonal foraging dynamics.

Material and Methods

Between 2017 and 2019, 13 Griffon vultures were trapped and equipped with solar-powered GPS/GSM transmitters in the Eastern Rhodopes, Bulgaria. Vultures were captured using a walk-in trap, located at a vulture feeding station. At the time of transmitter deployment, 7 birds were adults (>6 calendar years) and 6 birds were non-adults (2–5 calendar years) (Clark 2004, Zuberogoitia *et al.* 2013). However, as two non-adults aged into adults over the course of the study our final dataset included 141 non-adult bird months and 171 adult bird months. The vultures were fitted with 57 g high-resolution GPS/GSM and accelerometer (ACC) tags (E-Obs GmbH; Munich, Germany). Transmitters were attached with a backpack or leg-loop harness configuration with 11.2 mm Teflon ribbon (Anderson *et al.* 2020). The transmitter harness, rings and wing tag did not exceed 3% of the bird's body mass in accordance with the recommended limits to avoid adverse effects (Bodey *et al.* 2018). The transmitters were programmed to acquire a GPS fix and ACC data every 5 min during the day (between 03:00 and 19:00 UTC) with dormancy periods during the night. In winter due to insufficient solar charging GPS fixes were acquired once every 20 min. All data were stored in and accessed from Movebank (www. movebank.org, Kranstauber *et al.* 2011). Prior analyses, the data were inspected and visualized to check for outliers. Using the Movebank data filters, we removed erroneous GPS fixes (Walter *et al.* 2011).

Between May 2017 and March 2021 we received 1,638,251 GPS locations, on average 94.650±67.350 per individual. However, to estimate the Griffon Vulture's foraging home range we removed all stationary points by filtering the GPS locations with recorded ground speed <3 km/h. We conducted all further analyses with the remaining 415,434 GPS locations. We estimated vultures' foraging home range by applying the kernel density approach (Kenward 2001). We used iterative plug-in bandwidth selection (Amstrup *et al.* 2004, Gitzen *et al.* 2006). Data processing and estimations were completed in R v4.0.2 (R Core Team 2020) using package 'plugdensity' (Herrmann 2011). Fixed 99%, 95%, 75% and 50% kernel density contours were calculated to estimate home ranges. The 50% isopleth contour was defined as the vulture's core area and the 95% isopleth contour as the foraging home range. We also calculated the overall foraging range as the Minimum Convex Polygon (MCP) encompassing all GPS locations (Worton 1989). We pooled the data of all individuals for defining the foraging home range of the entire population. Furthermore, we defined the home ranges for each of the two age groups and for each season. The seasons were defined by the winter and summer solstices and spring and autumn equinox dates. The differences among the age groups and the seasonal home range size were assessed using one-way ANOVA and LSD post-hoc tests. For the aim of these comparisons and to comply with the sample independence rule, we excluded the adult home ranges of the two individuals, which aged into adults during the study.

We calculated the daily travelled distances for each tracked vulture by summing the distance between all successive GPS points for the day (Spiegel *et al.* 2013a). This parameter was considered as a proxy of vulture's foraging effort. Means are presented ± Standard Deviation (SD). The statistical processing of the data was carried out using the program Statistica 10 for Windows and R v4.0.2 (StatSoft Inc. 2010, R Core Team 2020). Spatial data were mapped and processed using QGIS software v3.12 (QGIS Development Team 2020).

Results

We tracked 13 Griffon Vultures for 1,093±244 tracking days (range 692 – 1,398). The average number of GPS fixes per day was 89.5±24.3. The overall foraging range, calculated as MCP, was 98,785 km^2. The foraging home ranges based on 95% and 75% kernel contours were 3,204 and 924.2 km^2, respectively. The core area (50% kernel) of the population in the Eastern Rhodopes was 256.5 km^2 *(Figure 1)*.

We found significant seasonal variation in the size of the core area ($F_{3,52}$ = 20.001, P = 0.002) of the Griffon Vulture activity *(Figure 2)*. The post-hoc analyses revealed that in summer it was larger than in autumn (P = 0.01) and winter (P = 0.0002), but not significantly larger than in spring (P = 0.10). The core area in summer was 249.1±169.5 km^2 and in winter only 64.2±54.2 km^2. It was 173.4±117.7 km^2 in spring and 127.1±117.3 km^2 in autumn

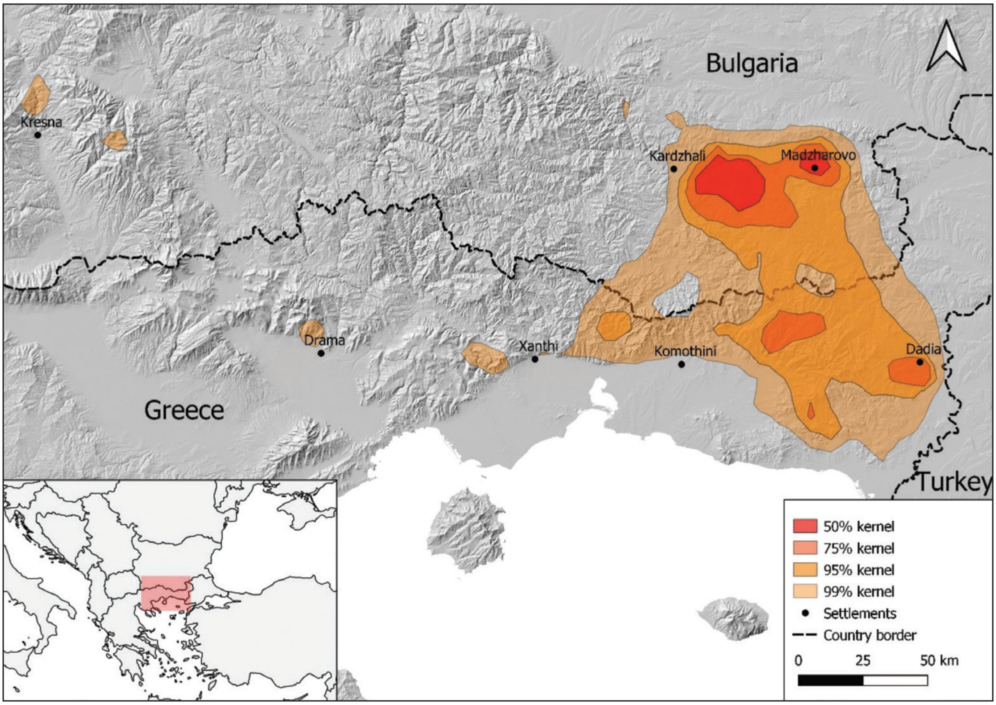

Figure 1. Home range of the Griffon Vulture population from the Eastern Rhodopes
1. ábra A fakó keselyű populáció mozgáskörzete a Keleti-Rodopéból

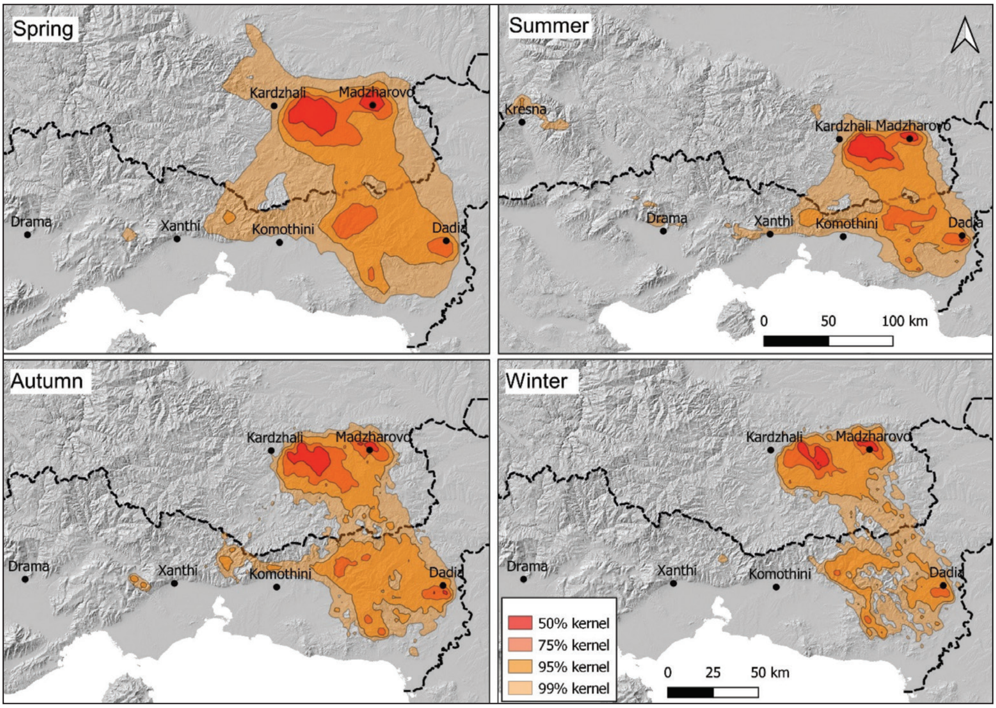

Figure 2. Seasonal variation in the home ranges of the Griffon Vulture population in the Eastern Rhodopes
2. ábra A fakó keselyű populáció mozgáskörzetének évszakonkénti változása Kelet-Rodope területén

(Figure 2). We found also significant difference in the 95% kernel home range among the seasons ($F_{3,52} = 5.93$, $P = 0.001$). The home range (95% kernel) was similar in size in spring and summer – 2,383.6±1737 km^2 and 2,687.6±1,949.7 km^2, respectively ($P = 0.57$). However, in autumn and winter it was significantly smaller than in summer ($P = 0.009$; $P = 0.0006$) – 1,241.6±850.9 km^2 and 764.8±592.6 km^2, respectively. No significant differences

Table 1. Home range (95% kernel), core area (50% kernel) size and daily travelled distances of Griffon Vultures per season and age groups
1. táblázat A fakó keselyű mozgáskörzete és napi megtett távolsága évszakonként és korcsoportonként

Age	Season	50% kernel (km^2)	95% kernel (km^2)	Distance travelled per day
Adult	Spring	167.3±80	2,016.3±1,233.4	91.3±62.8 km
	Summer	223.3±94.8	2,097.3±1,059.4	122.1±66.4 km
	Autumn	123.3±97.9	1,030.1±578.4	45.3±43.6 km
	Winter	70.8±50.5	746.5±435	34.9±39.2 km
Non-adult	Spring	182.5±157.8	2,934.5±2,180.7	78.3±62 km
	Summer	287.8±236.2	3,573.1±2,552	111.1±65.5 km
	Autumn	132.9±141.3	1,558.8±1,067.9	48.2±46.8 km
	Winter	54.5±57.9	792.3±769.9	41.9±40.4 km

were found in the size of the core area or the home range (95% kernel) between the adult and non-adult Griffon Vultures ($F_{1,13}$ = 0.056, P = 0.82; $F_{1,13}$ = 2.31, P = 0.16). Furthermore, no significant differences were found between the two age groups among the seasons *(Table 1)*.

Vultures travelled on average 75 ± 64.5 km per day. The longest recorded distance for a day was 369.7 km. Adult and non-adult individuals travelled similar daily distances – 76.3 ± 65.6 and 72.5±62.3 km, respectively. Vultures travelled longer distances in summer (118.2 ± 66.3 km) and spring (87.5±62.8 km) than in autumn (46.4±44.8 km) and winter (37.2±39.7 km) *(Table 1)*.

Discussion

In our study, we used spatial data from 13 Griffon Vultures to determine the foraging home range size of the population inhabiting the trans-border Eastern Rhodope Mountain. We found that the core of the home range encompasses the species breeding sites along the Arda river valley in Bulgaria, the central feeding station and two small feeding stations, and the Boynik ridge. Along that ridge occur most of the vulture feedings on natural carcasses found in the field (Arkumarev *et al.* 2021a). The Griffon Vultures frequently visited also the central feeding station near the village of Dadia, which is managed by the Management Body of Dadia-Soufli-Lefkimi Forest National Park (DSLFNP). Vultures foraged also along the Filiouri river and the villages of Esochi and Ragada in Greece where they also frequently find carcasses (Arkumarev *et al*. 2021a). The Kompsatos river valley, northeast from Komothini, was mostly visited by the vultures from spring to autumn. It is a mountainous area with rugged terrain where high numbers of sheep and goats are raised. Three small supplementary feeding stations provide additional food to vultures there since 2019. However, in winter most of the livestock is moved to the lowlands which can explain the exclusion of this area from the vulture's foraging range during this period. The areas near the country border are used by the vultures mostly to move between the main foraging areas in Bulgaria and Greece. Vultures rarely feed in this border area and mostly fly through (Arkumarev *et al*. 2021a). In the east, vultures follow Byala reka river valley and cross the border over the lower parts of Maglenik ridge to reach the central feeding station in DSLFNP. The same area is used as a main movement corridor by the Cinereous Vultures *(Aegypius monachus)* breeding in DSLFNP but foraging also in Bulgaria (Noidou & Vasilakis 2011, Arkumarev *et al.* 2020). Vultures also use the thermal updrafts over the ridges Irantepe and Maglenik to gain height and glide towards the main foraging areas in Greece along the Filiouri river valley. Less frequently, vultures follow the Varbitsa river valley and cross the border over Gyumyurdzhinski Snezhnik ridge to reach the foraging areas along the Kompsatos river valley in Greece. The foraging home range of the species encompasses mostly low mountainous areas with high numbers of livestock where the main husbandry practices are extensive grazing or semi-intensive grazing. The population in the Eastern Rhodopes feeds mostly on livestock carcasses found in the field (Arkumarev *et al.* 2021a, 2021b).

The size of the home range in our study is similar to the figures reported by Peshev *et al.* (2021) derived from GPS tracking of exogenous vultures (reintroduced and wild immatures) which have visited the Eastern Rhodopes. We consider that as a result of the social and gregarious lifestyle of the species the exogenous birds which arrive in the area are foraging and roosting together with the local groups of vultures, hence using the same areas. Looking at a broader scale, we found considerable variations in the home range estimations and daily travelled distances among the different populations on the Balkans, in Italy, France, Spain and Israel (Bahat & Kaplan 1995, Xirouchakis & Andreou 2009, García-Ripolles *et al.* 2011, Monsarrat *et al.* 2013, Genero *et al.* 2020, Hribsek *et al.* 2021, Peshev *et al.* 2021, Xirouchakis *et al.* 2021). These differences could be a result of different tracking methods and intensity, analytical approach, age and season variations. Despite that, it might also reflect the differences in the availability and fragmentation of suitable foraging habitats, presence of supplementary feeding stations and availability of natural food in the inhabited areas. The foraging home ranges of the Cape Vultures *(Gyps coprotheres)* and White-backed Vultures *(Gyps africanus)* in Africa are over 5 times larger than those of the Griffon Vultures in Europe (Phipps *et al.* 2013, Kane *et al.* 2016). Furthermore, the White-backed Vultures and the Lappet-faced Vultures *(Torgos tracheliotus)* travel longer daily distances and have larger daily displacement than the vultures in Europe (Spiegel *et al.* 2013b). Tucker *et al.* (2019) reported that birds travel much longer distances in areas with homogenous environments e.g. in Africa. The patchy, fragmented habitats and the availability of supplementary feeding stations in Europe might play an important role in shaping the foraging range of vultures.

We found high seasonal variation in the size of the home range and the daily distances travelled by the vultures. In spring and summer vultures forage over larger areas. The favorable weather conditions in terms of daily temperatures and formation of thermal updrafts allow vultures to travel far from the roosting and breeding cliffs. Previous study on the same population revealed that the daily temperatures, precipitation and wind speed are the main factors influencing vulture's foraging behavior and the probability to find food (Arkumarev *et al.* 2021a). During that period some non-adult vultures were undertaking long distance travels reaching other colonies on the Balkans. One individual visited the reintroduction site near Sliven in the Balkan Mountain. Another individual made a trip to Serbia and then to Kresna gorge where supplementary feeding station operates and a breeding colony was established as result of a reintroduction program (Peshev *et al.* 2018). In summer, some non-adults visited also the area near Drama and the Nestos river gorge west of Xanthi, Greece where the breeding colony has vanished but recently showed signs of recovery (Dobrev *et al.* 2021). These foraging trips of the non-adult vultures show the connectivity of the different Griffon Vulture populations on the Balkans, which are well documented also in other studies (Peshev *et al.* 2021). However, the adult Griffon Vultures we tracked did not undertake any long trips outside the main foraging and breeding areas in the Eastern Rhodopes. Immature vultures in other species are known to have larger home ranges due to their exploratory behavior and lack of attachment to breeding sites (Yamac & Bilgin, 2012, Kruger *et al.* 2014, Lopez-Lopez *et al.* 2014). However, we did not find significant difference in the home range size between the immature and adults Griffon Vultures in the Eastern Rhodopes. Similar results were reported for the Grands Causses in France where the increase of the population was followed also by increase of the number of supplementary feeding station (Monsarrat *et al.* 2013). Thus, the authors suggest that when the food availability in the area is high, the intraspecific competition is relatively low, which presumably decreases the need for the young vultures to forage over larger areas than adults. Our results are in line with this hypothesis considering that the availability of food, both natural and provided at supplementary feeding stations, in the Eastern Rhodopes currently exceeds the needs of the population (Arkumarev *et al.* 2021a). In winter, when the breeding season starts, the adult Griffon Vultures travelled shorter daily distances. In that time of the year the adults prepare the nests for the breeding, attend more time at the nest and start the incubation (Xirouchakis 2010). The immature vultures travelled slightly longer distances probably because they are not attached to a particular site and invest more time and efforts in foraging. In winter, the Griffon Vultures increase their reliance on the food provided at the supplementary feeding stations near the breeding cliffs (Arkumarev *et al.* 2021a). The unfavorable flight conditions and the decreased amount of natural carcasses in the landscape turn the feeding stations into preferred source of food (Arkumarev *et al.* 2021a). The higher attendance at the feeding stations probably increases the intraspecific competition. As adults are dominant during feeding, the immature birds are probably forced to search for additional sources of food, hence to travel longer distances. In spring and summer, Griffon Vultures travelled long daily distances as a result of the improving weather condition. In addition, the adults raise their chicks in that period and have to increase the foraging time and efficiency to find enough food for themselves and the growing hatchlings (Xirouchakis & Mylonas 2007).

Implications for conservation

Our study defined the foraging areas of the Griffon Vulture population in the trans-border Eastern Rhodope Mountain. It may serve as a baseline for planning and applying direct conservation measures to mitigate the threats and hence to increase the survival of the individuals of this increasing population. The construction and operation of wind farms is among the main threats for the vulture species and other soaring birds. The Greek part of Thrace, which falls within the Griffon Vulture home range, is designated as a priority area for development of wind farms (Kafetzis *et al.* 2017). Hundreds of wind turbines have been already erected in the area and proposals for the construction of new wind farms are underway. Systematic field studies have confirmed that the wind farms cause mortality of Griffon, Cinereous Vultures and many other birds of prey species (Carcamo *et al.* 2011, Doutau *et al.* 2011, author's unpubl. data). A comprehensive collision risk model suggested that 5 – 11% of Thrace's Cinereous Vulture population may collide annually with the wind turbines in the area (Vasilakis *et al.* 2016). Proposals for construction of wind farms have also been made in the Bulgarian part of the mountain. Here we provide valuable information for the movement patterns and important foraging areas of the Griffon Vulture population in the Eastern Rhodopes. We urge the policy makers to consider this and other relevant studies when planning the development of wind farms in the area to minimize the collision risk and avoid vulture mortality incidents.

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References

Alarcon, P. A. E. & Lambertucci, S. A. 2018. A three-decade review of telemetry studies on vultures and condors. – Movement Ecology 6: 13. DOI: 10.1186/s40462-018-0133-5.

Allen, A. M. & Singh, N. J. 2016. Linking movement ecology with wildlife management and conservation. – Frontiers in Ecology and Evolution 3: 155. DOI: 10.3389/fevo.2015.00155.

Amstrup, S. C., McDonald, T. L. & Durner, G. M. 2004. Using satellite radiotelemetry data to delineate and manage wildlife populations. – Wildlife Society Bulletin 32(3): 661–679. DOI: 10.2193/0091-7648(2004)032[0661:USRDTD]2.0.CO;2

Anderson, D., Arkumarev, V., Bildstein, K., Botha, A., Bowden, C. G. R., Davies, M., Duriez, O., Forbes, N. A., Godino, A., Green, R. E., Krüger, S., Lambertucci, S. A., Orr-Ewing, D., Parish, C. N., Parry-Jones, J. & Weston, E. 2020. A practical guide to methods for attaching research devices to vultures and condors. – Vulture News 78a: 1–72.

Andevski, J. (ed.) 2013. Vulture conservation in the Balkan Peninsula and adjacent regions. 10 years of vulture research and conservation. – Action plan for the recovery and conservation of vultures on the Balkan Peninsula and adjacent regions. Vulture Conservation Foundation & Frankfurt Zoological Society, Skopje, North Macedonia

Arkumarev, V., Dobrev, D. & Stamenov, A. 2019. First record of Eurasian Griffon Vulture *(Gyps fulvus)* from the Balkans migrating to South Sudan revealed by GPS tracking. – Scopus 39(2): 27–35.

Arkumarev, V., Dobrev, D., Stamenov, A., Angelova, V., Delchev, A., Dyulgerska, V., Zakkak, S., Kapsalis, E. & Stoychev, S. 2020. Occurences of the Cinereous Vulture *(Aegypius monachus)* in the Eastern Rhodopes, Bulgaria. – Ecologia Balkanica 12(2): 55–63.

Arkumarev, V., Dobrev, D., Stamenov, A., Terziev, N., Delchev, A. & Stoychev, S. 2021a Seasonal dynamics in the exploitation of natural carcasses and supplementary feeding stations by a top avian scavenger. – Journal of Ortnithology 162: 723–735. DOI: 10.1007/s10336-021-01865-1.

Arkumarev, V., Dobrev, D., Stamenov, A., Terziev, N., Delchev, A. & Stoychev, S. 2021b Using GPS and accelerometry data to study the diet of a top avian scavenger. – Bird Study 67(3): 300–310. DOI: 10.1080/00063657.2020.1864285.

Azmanis, P. N. 2009. Regulations of transmissible spongiform encephalopathies (TSEs) and management of feeding sites for birds of prey: a comparative approach between Western Europe (Iberian Peninsula and France) and the Balkan Peninsula. – In: Donazar, J. A., Margalida, A. & Campion, D. (eds.) Buitres, muladares y legislation sanitaria: perspectives de un conflicto y sus consecuencias desde la biologia de la conservation. – Aranzadi, Donostia, Spain, pp. 521–525.

Bahat, O., Hatzofe, O., Kaplan, A. & Woodley, B. 2001. Foraging range and movements of Griffon Vulture *(Gyps fulvus)* in Israel, as determined by satellite tracking. – In: Proceedings of the 4th Eurasian Congress on Raptors. – IV. Eurasian Congress on Raptors, Seville, Spain, Donana Biological Station & Raptor Research Foundation, Seville, pp. 11–12.

Bahat, O. & Kaplan, A. 1995. Foraging behaviour in Griffon Vultures. – Torgos 25: 18–26.

BirdLife International 2021. Species factsheet: *Gyps fulvus*. Downloaded from http://www.birdlife.org on 13/09/2021.

Bodey, T. W., Cleasby, I. R., Bell, F., Parr, N., Schultz, A., Votier, S. C. & Bearhop, S. 2018. A phylogenetically controlled meta-analysis of biologging device effects on birds: deleterious effects and a call for more standardized reporting of study data. – Methods in Ecology and Evolution 9(4): 946–955. DOI: 10.1111/2041-210X.12934.

Botha, A. J., Andevski, J., Bowden, C. G. R., Gudka, M., Safford, R. J., Tavares, J. & Williams, N. P. 2017. CMS Multi-species Action Plan to conserve African-Eurasian Vultures. – Abu Dhabi, United Arab Emirates: Coordinating Unit of the CMS Raptors MOU. CMS Raptors MOU Technical Publication No. 5. CMS Technical Series No. 35.

Carcamo, B., Kret, E., Zografou, C. & Vasilakis, D. 2011. Assessing the impacts of 9 wind farms on the birds of prey of Thrace. – Technical report of WWF-Greece. http://www.wwf.gr/images/pdfs/WWF-aiolika-arpaktika2011.pdf (in Greek)

Clark, W. S. 2004. Wave molt of the primaries of Accipitrid raptors, and its use in ageing. – In: Chancellor, R. D. & Meyburg, B.-U. (eds.) Raptors worldwide: Proceedings of the V. World Conference on Birds of Prey. – World Working Group on Birds of Prey, Berlin, Germany, pp. 795–804.

Cramp, S. & Simmons, K. 1980. The birds of the Western Palearctic, Vol. 2. – Oxford University Press, London

del Hoyo, J., Elliott, A. & Sargatal, J. 1994. Handbook of the Birds of the World. Vol. 2: New World Vultures to Guineafowl. – Lynx Edicions, Barcelona

Del Moral, J. C. & Molina, B. (eds.) 2018. El buitre Leonado en Espana. Poblacion reproductora en 2008 y metodo de censo. The Griffon Vulture in Spain. Breeding population in 2018 and census methods. – Madrid: SEO/BirdLife (in Spanish)

Demerdzhiev, D., Hristov, H., Dobrev, D., Angelov, I. & Kurtev, M. 2014. Long-term population status, breeding parameters and limiting factors of the Griffon Vulture *(Gyps fulvus)* population in Eastern Rhodopes, Bulgaria. – Acta Zoologica Bulgarica 66(3): 373–384.

Deygout, C., Gault, A., Duriez, O., Sarrazin, F. & Bessa-Gomes, C. 2010. Impact of food predictability on social facilitation by foraging scavengers. – Behavioral Ecology 21(6): 1131–1139. DOI: 10.1093/beheco/arq120

Dobrev, D., Tsiakiris, R., Skartsi, T., Dobrev, V., Arkumarev, V., Stara, K., Stamenov, A., Probonas, N., Kominos, T., Galanaki, A., Kret, E., Hallmann, B., Grubac, B., Susic, G., Marinkovic, S., Hribsek, I., Skoric, S., Jerrentrup, H., Lucic, V., Kapelj, S., Stoyanov, G., Zakkak, S., Hristov, H., Stoychev, S., Sidiropoulos, L., Bino, T. & Demerdzhiev, D. 2021a Long-term size and range changes of the Griffon Vulture *Gyps fulvus* population in the Balkans: a review. – Bird Conservation International, First View, pp. 1–16. DOI: 10.1017/S0959270921000198

Dobrev, D., Arkumarev, V., Dobrev, V., Stamenov, A. & Demerdzhiev, D. 2021b Use and selection of roost sites by Eurasian Griffon Vultures *Gyps fulvus* in Bulgaria. – Bird Study 67(4): 496–504. DOI: 10.1080/00063657.2021.1950122.

Doutau, B., Cafkaletou-Diaz, A., Carcamo, B., Vasilakis, D. & Kret, E. 2011. Impacts of wind farms on the birds of prey of Thrace. – Annual Technical Report August 2009–August 2010, WWF Greece. http://www.wwf.gr/images/pdfs/WWF-aiolika-arpaktika2011-etisio.pdf (in Greek)

Duriez, O., Kato, A., Tromp, C., Dell'Omo, G., Vyssotski, A. L., Sarrazin, F. & Ropert-Coudert, Y. 2014. How cheap is soaring flight in raptors? A preliminary investigation in freely-flying vultures. – PLoS ONE 9(1): e84887. DOI: 10.1371/journal.pone.0084887

Fergusson-Lees, J. & Christie, D. 2001. Raptors of the World. – Christopher Helm, London

García-Ripollés, C., López-López, P. & Urios, V. 2011. Ranging behaviour of non-breeding Eurasian Griffon Vultures *Gyps fulvus*: a GPS-telemetry study. – Acta Ornithologica 46(2): 127–134. DOI: 10.3161/000164511x625892

Genero, F., Franchini, M., Fanin, Y. & Filacorda, S. 2020. Spatial ecology of non-breeding Eurasian Griffon Vultures *Gyps fulvus* in relation to natural and artificial food availability. – Bird Study 67(1): 53–70. DOI: 10.1080/00063657.2020.1734534

Gitzen, R. A., Millspaugh, J. J. & Kernohan, B. J. 2006. Bandwidth selection for fixed-kernel analysis of animal utilization distributions. – Journal of Wildlife Management 70(5): 1334–1344. DOI: 10.2193/0022-541X(2006)70[1334:BSFFAO]2.0.CO;2

Grilli, M. G., Bildstein, K. L. & Lambertucci, S. A. 2019. Nature's clean-up crew: Quantifying ecosystem services offered by a migratory avian scavenger on a continental scale. – Ecosystem Services 39: 100990. DOI: 10.1016/j.ecoser.2019.100990

Harel, R., Spiegel, O., Getz, W. M. & Nathan, R. 2017. Social foraging and individual consistency in following behaviour: testing the information centre hypothesis in free-ranging vultures. – Proceedings of the Royal Society B 284: 20162654. DOI: 10.1098/rspb.2016.2654

Herrmann, E. 2011. R interface etc. by Martin Maechler plugdensity: Plug-in Kernel Density Estimation. – R package version 0.8-3. https://CRAN.R-project.org/package=plugdensity

Hribsek, I., Plecas, M., Skoric, S. & Marinkovic, S. 2021. First description of movement and ranging behavior of the Griffon Vulture *(Gyps fulvus)* from Serbia using GPS satellite tracking. – Archives of Biological Sciences 73(2): 185–195. DOI: 10.2298/ABS201210013H

Kafetzis, A., Kret, E., Skartsi, T., Vasilakis, D. & Christopoulou, I. 2017. – Wind farms in areas of high ornithological value-conflicts, solutions, challenges: The case of Thrace, Greece. – In: Koppel, J. (ed.) Wind Energy and Wildlife Interactions. – Presentations from the CWW 2015 Conference pp. 191–205 – Springer

Kane, A., Wolter, K., Neser, W., Kotze, A., Naidoo, V. & Monadjem, A. 2016. Home range and habitat selection of Cape Vultures *Gyps coprotheres* in relation to supplementary feeding. – Bird Study 63(3): 387–394. DOI: 10.1080/00063657.2016.1214105.

Kenward, R. E. 2001. A Manual for Wildlife Radio Tagging. – London: Academic Press

Kmetova-Biro, E., Stoynov, E., Ivanov, I., Peshev, H., Marin, S., Bonchev, L., Stoev, I. P., Stoyanov, G., Nikolova, Z., Vangelova, N., Parvanov, D. & Grozdanov, A. 2021. Re-introduction of Griffon Vulture *(Gyps fulvus)* in the Eastern Balkan Mountains, Bulgaria - completion of the establishment phase 2010–2020. – Biodiversity Data Journal 9: e66363. DOI: 10.3897/BDJ.9.e66363

Kranstauber, B., Cameron, A., Weinzerl, R., Fountain, T., Tilak, S., Wikelski, M. & Kays, R. 2011. The Movebank data model for animal tracking. – Environmental Modelling and Software 26(6): 834–835. DOI: 10.1016/j.envsoft.2010.12.005

Kruger, S., Reid, T. & Amar, A. 2014. Differential range use between age classes of Southern African Bearded Vultures *Gypaetus barbatus*. – PLoS One 9(12): e114920. DOI: 10.1371/journal.pone.0114920

Lopez-Lopez, P., García-Ripolles, C. & Urios, V. 2014. Food predictability determines space use of endangered vultures: implications for management of supplementary feeding. – Ecological Applications 24(5): 938–949. DOI: 10.1890/13-2000.1

McGrady, M. & Gavashelishvili, A. 2006. Tracking vultures from the Caucasus into Iran. –Podoces 1(1/2): 21–26.

Monsarrat, S., Benhamou, S., Sarrazin, F., Bessa-Gomes, C., Bouten, W. & Duriez, O. 2013. How predictability of feeding patches affects home range and foraging habitat selection in avian social scavengers? – PLoS One 8(1): e53077. DOI: 10.1371/journal.pone.0053077

Noidou, M. & Vasilakis, D. 2011. Characterizing Eurasian Black Vulture's *(Aegypius monachus)* flight movement corridors in Thrace: a need for conservation on a landscape-level scale. – Report of WWF Greece pp 16. https://www.contentarchive.wwf.gr/images/pdfs/WWF-Flight-movement-corridors-2011.pdf

Pain, D. J., Cunningham, A. A., Donald, P. F., Dukworth, J. W., Houston, D. C., Katzner, T., Parry-Jones, J., Poole, C., Prakash, V., Round, P. & Timmins, R. 2003. Causes and effects of temporal and spatial declines of *Gyps* vultures in Asia. – Conservation Biology 17(3): 661–671. DOI: 10.1046/j.1523-1739.2003.01740.x

Peshev, H., Grozdanov, A., Kmetova-Biro, E., Ivanov, I., Stoyanov, G., Tsiakiris, R., Marin, S., Marinković, S., Sušić, G., Lisichanets, E., Hribšek, I., Karić, Z., Kapelj, S., Bonchev, L. & Stoynov, E. 2021. New insight into spatial ecology of Griffon Vulture *(Gyps fulvus)* on the Balkans provides opportunity for focusing conservation actions for a threatened social scavenger. – Biodiversity Data Journal 9: e71100. DOI: 10.3897/BDJ.9.e71100

Peshev, H., Stoynov, E., Parvanov, D. & Grozdanov, A. 2018. Seasonal and spatial dynamics of the population of the Griffon Vulture *Gyps fulvus* (Hablizl, 1783) (Aves: Accipitridae) in Southwestern Bulgaria. – Acta Zoologica Bulgarica Suppl. 12: 67–75.

Phipps, W. L., Willis, S. G., Wolter, K. & Naidoo, V. 2013. Foraging ranges of Immature African White-Backed Vultures *(Gyps africanus)* and their use of protected areas in Southern Africa. – Plos One 8(1): e52813. DOI: 10.1371/journal.pone.0052813.

QGIS Development Team 2020. QGIS Geographic Information System. – Open Source Geospatial Foundation Project. http://qgis.osgeo.org

R Core Team 2020. R: A language and environment for statistical computing. – R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org

Ruxton, G. D. & Houston, D. C. 2004. Obligate vertebrate scavengers must be large soaring fliers. – Journal of Theoretical Biology 228(3): 431–436. DOI: 10.1016/j.jtbi.2004.02.005

Spiegel, O., Harel, R., Getz, W. M. & Nathan, R. 2013a Mixed strategies of Griffon Vultures' *(Gyps fulvus)* response to food deprivation lead to a hump-shaped movement pattern. – Movement Ecology 1(1):5. DOI: 10.1186/2051-3933-1-5

Spiegel, O., Getz, W. M. & Nathan, R. 2013b Factors influencing foraging search efficiency: Why do scarce Lappet-faced Vultures outperform ubiquitous White-backed Vultures? – The American Naturalist 181(5): 102–115. DOI: 10.1086/670009

Statsoft 2010. STATISTICA (data analysis software system), version 10. – www.statsoft.com

Tucker, M. A., Alexandrou, O., Bierregaard, R. O., Bildstein, K. L., Böhning-Gaese, K., Bracis, C., Brzorad, J. N., Buechley, E. R., Cabot, D., Calabrese, J. M., Carrapato, C., Chiaradia, A., Davenport, L. C., Davidson, S. C., Desholm, M., DeSorbo, C. R., Domenech, R., Enggist, P., Fagan, W. F., Farwig, N., Fiedler, W., Fleming, C. H., Franke, A., Fryxell, J. M., García-Ripollés, C., Grémillet, D., Griffin, L. R., Harel, R., Kane, A., Kays, R., Kleyheeg, E., Lacy, A. E., LaPoint, S., Limiñana, R., López-López, P., Maccarone, A. D., Mellone, U., Mojica, E. K., Nathan, R., Newman, S. H., Noonan, M. J., Oppel, S., Prostor, M., Rees, E. C., Ropert-Coudert, Y., Rösner, S., Sapir, N., Schabo, D., Schmidt, M., Schulz, H., Shariati, M., Shreading, A., Paulo Silva, J., Skov, H., Spiegel, O., Takekawa, J. Y., Teitelbaum, C. S., van Toor, M. L., Urios, V., Vidal-Mateo, J., Wang, Q., Watts, B. D., Wikelski, M., Wolter, K., Žydelis, R. & Mueller, T. 2019. Large birds travel farther in homogeneous environments. – Global Ecology and Biogeography 28(5): 576–587. DOI: 10.1111/geb.12875.

Vasilakis, D. P., Whitfield, D. P., Schindler, S., Poirazidis, K. S. & Kati, V. 2016. Reconciling endangered species conservation with wind farm development: Cinereous Vultures *(Aegypius monachus)* in South-Eastern Europe. – Biological Conservation 196: 10–17. DOI: 10.1016/j.biocon.2016.01.014

Walter, W. D., Fischer, J. W., Baruch-Mordo, S. & VerCauteren, K. C. 2011. – What is the proper method to delineate home range of an animal using today's advanced GPS telemetry systems: the initial step. – In: Krejcar, O. (ed.) Modern Telemetry. – Tech Open Access Publisher, Rijeka, Croatia, pp. 249–268.

Worton, B. J. 1989. Kernel methods for estimating the utilization distribution in home-range studies. – Ecology 70(1): 164–168.

Xirouchakis, S. 2010. Breeding biology and reproductive performance of Griffon Vultures *Gyps fulvus* on the island of Crete (Greece). – Bird Study 57(2): 213–35. DOI: 0.1080/00063650903505754

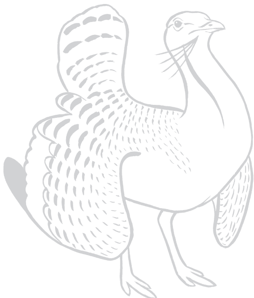
Xirouchakis, S. M., Grivas, C., Andreou, G. & Georgopoulou, E. 2021. Home range size, space use and resource selection of Griffon Vultures in an insular environment. – Journal of Zoology 314(2): 116–131. DOI: 10.1111/jzo.12868

Xirouchakis, S. M. & Andreou, G. 2009. Foraging behavior and flight characteristics of Griffon Vultures *(Gyps fulvus)* on the island of Crete (Greece). – Wildlife Biology 15(1): 37–52. DOI: 10.2981/07-090

Xirouchakis, S. M. & Mylonas, M. 2007. Breeding behavior and parental care in the Griffon Vulture *Gyps fulvus* on the island of Crete (Greece). – Ethology Ecology & Evolution 19: 1–26. DOI: 10.1080/08927014.2007.9522578

Yamac, E. & Bilgin, C. 2012. Post-fledging movements of Cinereous Vultures *Aegypius monachus* in Turkey revealed by GPS telemetry. – Ardea 100(2): 149–156. DOI: 10.5253/078.100.0206

Zuberogoitia, I., De La Puente, J., Elorriaga, J., Alonso, R., Palomares, L. & Martínez, J. 2013. The flight feather molt of Griffon Vultures *(Gyps fulvus)* and associated biological consequences. – Journal of Raptor Research 47(3): 292–303. DOI: 10.3356/jrr-12-09.1



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The wing phalanges *(Phalanx proximalis digiti majoris)* of European Accipitriformes and Falconiformes

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Abstract The authors compared the first phalanx of the second wing finger of 33 European diurnal raptors. The importance of studying this bone lies in the fact that, although it has diagnosable characteristics, it was practically neglected by osteologists and paleontologists. Thus, fossil materials can be identified through them, as well as those from owl pellets. The comparison was made possible by the comparative avian skeleton collection of the Hungarian Natural History Museum. In a preliminary investigation we examined the morphological diversity of the first phalanx of the second wing finger among the different species. We used principal component (PC) analyses on measurements. The PC described the curvature of the anterior surface of the bone and the relative size of the distal and proximal epiphyses. The principal component analysis showed slightly overlapping in shape between the taxons but the accipitriform and falconiform birds diverged in the morphospace. The attributes and geometry of the first phalanx of the second wing finger reflects more on taxonomic background than flying behaviour. The avian wing is a complex and highly modulable structure, therefore, probably body mass and size affect flying performance than the other morphological features of this bone. The text is supplemented by 6 figures and one size table.

Keywords: Europe, diurnal raptors, *phalanx proximalis digiti majoris*, osteology

Összefoglalás A szerzők 33 európai nappali ragadozómadár faj szárnyának második ujja első ujjpercét hasonlították össze. A vázrész tanulmányozásának jelentősége abban áll, hogy bár jól meghatározható jellegekkel rendelkezik, gyakorlatilag el lett hanyagolva a csonttannal és őslénytannal foglalkozók részéről. Általuk mind a fosszilis anyagok, mind a bagolyköpetek anyagai jól meghatározhatóak. A vizsgálatot a MTM összehasonlító madárcsont gyűjteménye tette lehetővé. Egy előzetes vizsgálatban tanulmányoztuk a szárny második ujjának első ujjpercének morfológiai diverzitását. A vizsgálat során kétdimenziós landmarkok használatával főkomponens-analíziseket végeztünk. A főkomponensek a csont anterior irányú görbületét, valamint a proximális és a disztális epifízisek egymáshoz viszonyított nagyságát magyarázzák. Az analízis eredményei szerint némi átfedés látható a fajok között, de a vágómadár-alakúak és a sólyomalakúak elkülönülnek egymástól a morfotérben. Az eredmények azt sugallják, hogy az ujjperc alakja és tulajdonságai sokkal inkább tükrözik a fajok rendszertani hovatartozását, semmint a repülés módját. A madarak szárnya egy komplex modulálható struktúra, így feltehetőleg a testtömegnek és a méretnek nagyobb szerepe van a repülést illetően, mint az ujjperc alaktani sajátságainak. A szöveget 6 ábra és egy mérettáblázat egészíti ki.

Kulcsszavak: Európa, nappali ragadozómadarak, *phalanx proximalis digiti majoris*, csonttan

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Introduction

In this study, the authors discuss a lesser-known part of the avian skeleton used for description. The relatively simple first *phalanx* of the second wing finger is rather small, even in the largest avian species – only 6–7 centimetres long, and even though it bears excellent diagnostic characteristics, it has never been the focus of experts studying avian bones. There are virtually no study results, or most of the time not even a mention.

Of the four wing *phalanges* only one bears such morphological characteristics, namely the first *phalanx* of the second wing finger. Its morphometrical analysis results can be used for comparative anatomical, paleontological, archaeological, taxonomical or ornithological examinations of owl pellets.

The morphological terminology and measurement method of the skeletal part was used based on the works of Baumel *et al.* 1979 and Kessler 2013. The wing phalanx can primarily

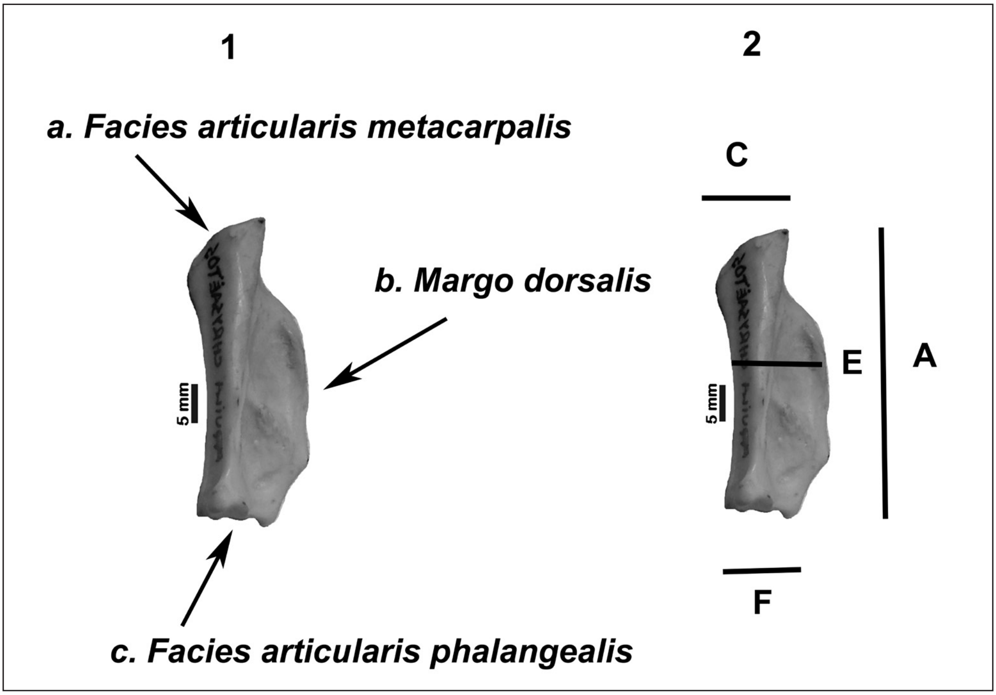

Figure 1. Aquila chrysaetos (Linnaeus, 1758)
1. *Phalanx proximalis digiti majoris* osteology characteristics:
a. proximal end – *Facies articularis metacarpalis*; b. the dorsal side – *margo dorsalis*; c. distal end – *Facies articularis phalangealis*
2. Measurements methods of bone:
A. total length; C. breadth of the proximal end; E. breadth of the *corpus*; F. breadth of the distal end
1. ábra Aquila chrysaetos (Linnaeus, 1758)
1. A nagy (középső) kézujj első ujjperce csonttani jellegei: a. proximális vég – *facies articularis metacarpalis*; b. dorsalis oldal – m*argo dorsalis*; c. disztális vég – *facies articularis phalangealis*;
2. A csont mérési mintái:
A. teljes hossz; C. proximális vég szélessége; E. a test szélessége; F. a disztális vég szélessége

be found within the literature in the works of Milne-Edwards 1867–1868, Solti 1980, 1981a, 1981b, 1996, Gilbert *et al*. 1981, Cohen & Serjeantson 1996, Kessler 2015, 2016a, 2016b, 2019, 2020 discussed or illustrated, but usually only in the case of a few species.

The measured sizes given in *Table 1* are for information purposes only, since in the case of species of this order, a size difference in favour of females is quite frequent, and we do not have the corresponding skeletal parts from both sexes in all species.

In this study, we also made an attempt to investigate the morphological diversity of this bone among the different species. Our objective was to increase our knowledge on the possible relationship between shape and flying behaviour of diurnal raptors. The differences in force acting on this skeletal part during flying and gliding may be related to shape and geometry of wing finger. To investigate the morphological diversity, we used landmark-based morphometric methods.

Abbreviations

MTM/NHMUS: Natural History Museum of Hungary Budapest, Department of Paleontology and Geology
A – total length
C – the width of the proximal end
E – the largest width of the middle phalanx
F – the width of the distal end
wingspan – the distance between the two wing tips
wing – half the wingspan
wing/phalanx – wing length to phalanx ratio
New genus, sp. name – Gill *et al*. 2020

We used the following anatomical terminology *(Figure 1)*:
Facies articularis metacarpalis – this is the proximal end
Margo dorsalis – this is the largest width of the middle phalanx
Facies articularis phalangealis – this is the distal end

Osteological description

In the case of vultures *(Figure 2a–d)*:

a) Shape of *facies articularis metacarpalis*:
- strongly protuberant, with a protruding ventral tip: *Gypaetus barbatus*;
- slightly convex, with a rounded ventral end: *Aegypius monachus*;
- obliquely cut, lifting from the ventral end towards the dorsal: *Gyps fulvus, Neophron percnopterus*;

b) The arc of the *margo dorsalis*:
- forms a regular arc: *Gypaetus barbatus*;
- the arch is protuberant: *Aegypius monachus*;

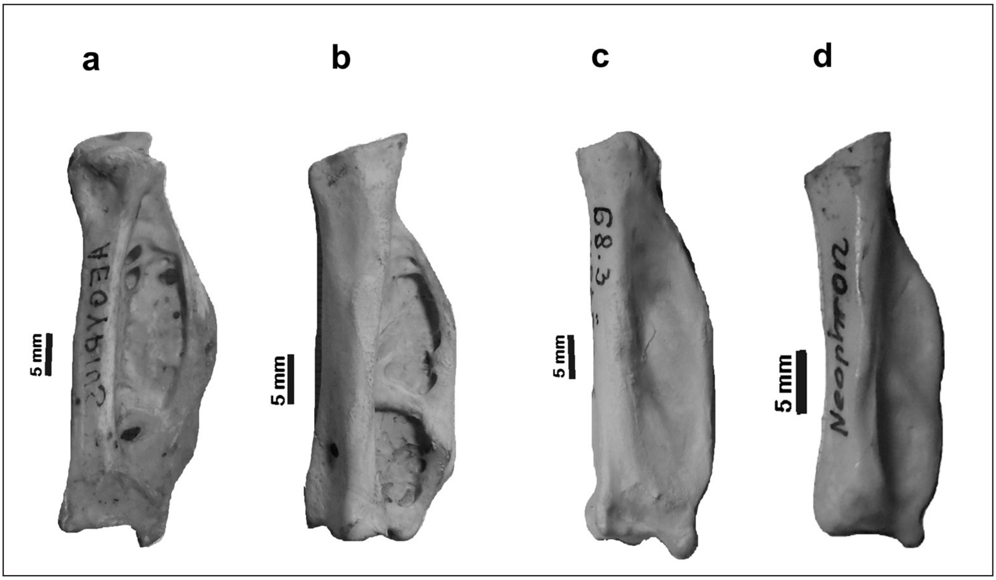

Figure 2. Left *phalanx proximalis digiti majoris* (ventral aspect)
2. ábra Bal oldali kézujjperc (1. ujjperc, II. ujj, alulnézet)
a. *Aegypius monachus* (Linnaeus, 1766); b. *Gypaetus barbatus* (Linnaeus, 1758); c. *Gyps fulvus* (Hablizl, 1783); d. *Neophron percnopterus* (Linnaeus, 1758)

- the middle of the arc is cut straight: *Gyps fulvus*;
- the slightly protuberant arc is wavy: *Neophron percnopterus*;

c) Shape of *facies articularis phalangealis*:
- oblique, with a strongly protruding dorsal projection: *Aegypius monachus*;
- wavy, with a strongly protruding dorsal projection: *Gypaetos barbatus*;
- wavy, with a slightly protruding dorsal projection: *Gyps fulvus*;
- concave, with a strongly protruding dorsal projection: *Neophron percnopterus*;

In the case of eagles *(Figure 3a–i)*:

a) Shape of *facies articularis metacarpalis*:
- obliquely convex, with a rounded ventral end: *Pandion haliaetus*;
- slightly convex: *Haliaeetus albicilla*;
- oblique straight rise from the ventral end: *Aquila chrysaetos, A. heliaca, A. fasciata, Clanga clanga, C. pomarina*;
- the dorsal projection is like a pointy cone: *Aquila chrysaetos, Clanga clanga, C. pomarina*;
- the dorsal projection is like a blunt cone: *Aquila heliaca*;
- straight waves: *Aquila rapax*;
- convex, with a dorsally protruding cone-like projection: *Hieraaetus pennatus*;

b) The arc of the *margo dorsalis*:
- convex, with a cut arch: *Aquila chrysaetos, A. fasciata, Clanga clanga, Haliaetos* spp.;
- slightly convex, with a symmetrical arch: *Clanga pomarina*;

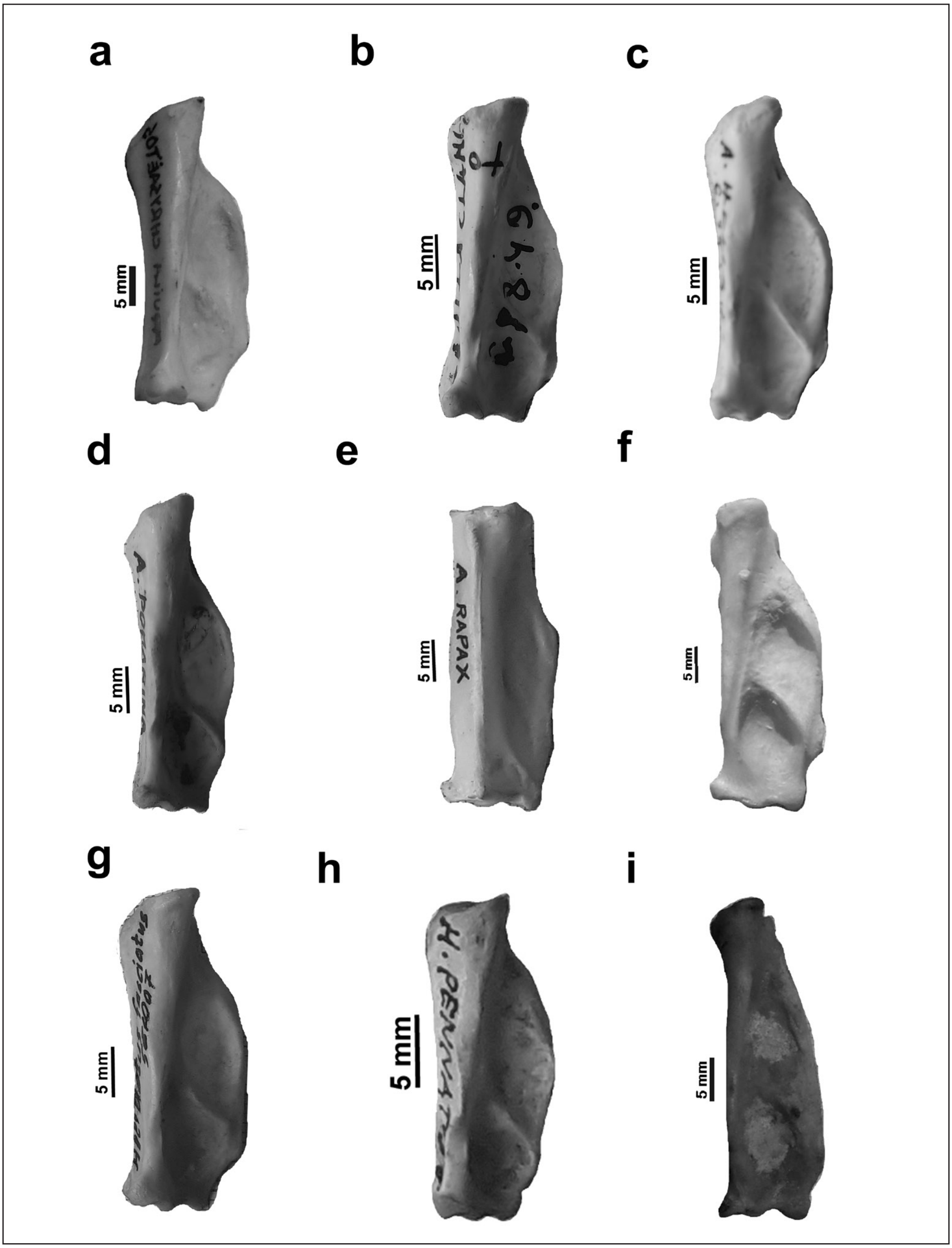

Figure 3. Left *phalanx proximalis digiti majoris* (ventral aspect)
3. ábra Bal oldali kézujjperc (1. ujjperc, II. ujj, alulnézet)
a. *Aquila chrysaetos* (Linnaeus, 1758); b. *Clanga clanga (*Pallas, 1811); c. *Aquila heliaca* (Savigny, 1809); d. *Clanga pomarina* (Brehm, 1831); e. *Aquila rapax* (Cabanis, 1854); f. *Haliaeetus albicilla* (Linnaeus, 1758); g. *Aquila fasciata* (Vieillot, 1822); h. *Hieraaetus pennatus* (Gmelin, 1788); i. *Pandion haliaetus* (Linnaeus, 1758)

- symmetrically and strongly arched: *Aquila heliaca, Hieraaetus pennatus*;
- asymmetrically and slightly arched: *Pandion haliaetus*;
- asymmetrical and cut: *Aquila rapax*;

c) Shape of *facies articularis phalangealis*:

- wavy, with a strongly protruding, blunt cone-like projection: *Aquila chrysaetos, Pandion haliaetus*;
- wavy, with a strongly protruding, pointy cone-like projection: *Clanga clanga, Aquila rapax*;
- wavy, with a slightly protruding projection: *Aquila heliaca, A. fasciata, Clanga pomarina, Haliaetos haliaetus, Hieraaetus pennatus*;

In the case of buzzards and kites *(Figure 4a–g)*:

a) Shape of *facies articularis metacarpalis*:

- slightly convex: *Buteo lagopus*;
- slightly obliquely cut, with a straight ventral end and a small dorsal point: *Buteo rufinus, Circaetus gallicus, Milvus migrans*;
- obliquely convex, with a pointed dorsal projection: *Milvus milvus*;
- obliquely rising, with a dorsal pointed projection: *Pernis apivorus*;
- oblique and slightly raised, with a dorsal pointed projection: *Buteo buteo*;

b) The arc of the *margo dorsalis*:

- the bulge is cut: *Buteo rufinus*, *Circaetus gallicus, Milvus migrans*;
- strongly protuberant, with a symmetrical arch: *Buteo buteo, Pernis apivorus*;
- slightly protuberant, with a symmetrical arch: *Buteo lagopus*;
- slightly protuberant, with an asymmetrical arch: *Milvus milvus*;

c) Shape of *facies articularis phalangealis*:

- wavy, with a slightly protruding projection: *Circaetus gallicus, Milvus milvus, Pernis apivorus*;
- convex, with a pointy dorsal projection: *Buteo buteo, B. lagopus*;
- convex, with a blunt dorsal projection: *Buteo rufinus, Milvus migrans*;

In the case of goshawks and harriers *(Figure 4h–l)*:

a) Shape of *facies articularis metacarpalis*:

- convex: *Accipiter gentilis, Circus pygargus*;
- obliquely cut, with a pointy dorsal point: *Circus aeruginosus, C. cyaneus, C. macrourus*;

b) The arc of the *margo dorsalis*:

- the arch is cut: *Accipiter gentilis, Circus aeruginosus, C. cyaneus, C. macrourus, C. pygargus*;

c) Shape of *facies articularis phalangealis*:

- straight and wavy: *Circus cyaneus*;
- oblique and wavy: *Accipiter gentilis, Circus pygargus*;
- concave: *Circus aeroginosus, C. macrourus*;

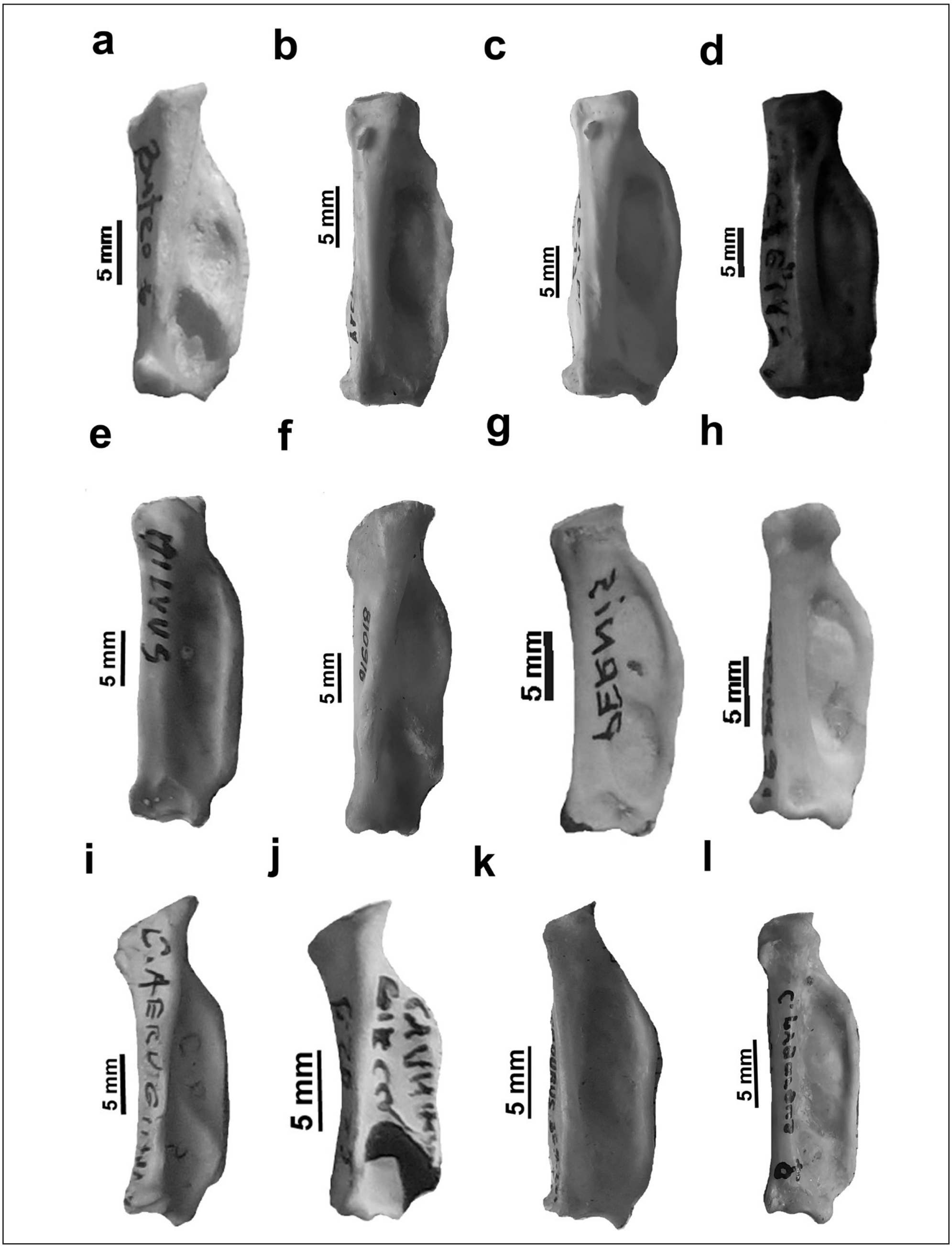

Figure 4. Left *phalanx proximalis digiti majoris* (ventral aspect)
4. ábra Bal oldali kézujjperc (1. ujjperc, II. ujj, alulnézet)
a. *Buteo buteo* (Linnaeus, 1758); b. *Buteo lagopus* (Pontoppidan, 1763); c. *Buteo rufinus* (Cretzschmar, 1827); d. *Circaetus gallicus* (Gmelin, 1788); e. *Milvus migrans* (Boddaeert, 1783); f. *Milvus milvus* (Linnaeus, 1758); g. *Pernis apivorus* (Linnaeus, 1758); h. *Accipiter gentilis* (Linnaeus, 1758); i. *Circus aeruginosus* (Linnaeus, 1758); j. *Circus cyaneus* (Linnaeus, 1758); k. *Circus macrourus* (Gmelin, 1771); l. *Circus pygargus* (Linnaeus, 1758)

In the case of falcons and kestrels *(Figure 5a–g)*:

a) Shape of *facies articularis metacarpalis*:
- protuberant: *Falco columbarius*;
- protuberant, with a well-developed blunt conelike dorsal projection; *Falco cherrug*;
- protuberant, with a well-developed pointed conelike projection: *Falco peregrinus*;
- protuberant, with a small pointed projection: *Falco subbuteo*;
- obliquely cut: *Falco rusticolus, F. vespertinus*;
- obliquely cut, with a long pointed conelike projection: *Falco tinnunculus*;

b) The arc of the *margo dorsalis*:
strongly and symmetrically arched: *Falco cherrug, F. peregrinus, F. columbarius, F. subbuteo, F. tinnunculus*;
slightly arched: *Falco rusticolus*;
the arch is cut: *Falco vespertinus*;

c) Shape of *facies articularis phalangealis*:
obliquely wavy, without a protruding dorsal projection: *Falco rusticolus, F. tinnunculus*;
obliquely wavy, with a protruding dorsal projection: *Falco cherrug, F. peregrinus, F. columbarius, F. subbuteo, F. vespertinus*

Examination of the first phalanx of the second wing finger

For the morphometric comparison, we examined the morphological variability of the first phalanx of the second wing finger. For each specimen, we used 300 sliding landmarks to examine the shape of the whole bone in ventral view.

These landmarks were allowed to slide along their corresponding curve. It is necessary in the minimalisation of the bending energy. The coordinates of the 2D landmarks were digitised using the software TpsDig 2.16 (Rohlf 2010). The resulted coordinates were transformed using the Procrustes superimposition method. The consensus configurations and relative warps were conducted. Variability in shape was assessed using the scores obtained for each individual on the first two relative warps. We conducted principal component analyses (PCA) on these morphological variables. The relative warps are corresponding to the principal components (PCs) and define the shape space in which individuals are replaced. We used PAST v.1.7 software (Hammer *et al*. 2001) for principal component analysis and to extract deformation grids. We only considered those PCs explaining individually >10% of the variance.

The first two PCs explained 53% and 21% of the variance in bone shape. The first PC axis described the curvature of the anterior surface of the proximal phalanx of the major digit (PC1). The second PC axis described the relative size of the distal and proximal epiphyses (PC2). The Osprey *(Pandion haliaetus)* and the Eastern Imperial Eagle *(Aquila heliaca)* showed the most convex shape and the Peregrine Falcon *(Falco peregrinus)* showed the most concave shape on the anterior curvature. The Red-footed Falcon *(F. vespertinus)* and the Eurasian Hobby *(F. subbuteo)* showed very narrow proximal epiphyses compared to the Red Kite *(Milvus milvus)* which showed prominent proximal epiphysis but the distal epiphysis was very narrow *(Figure 6)*.

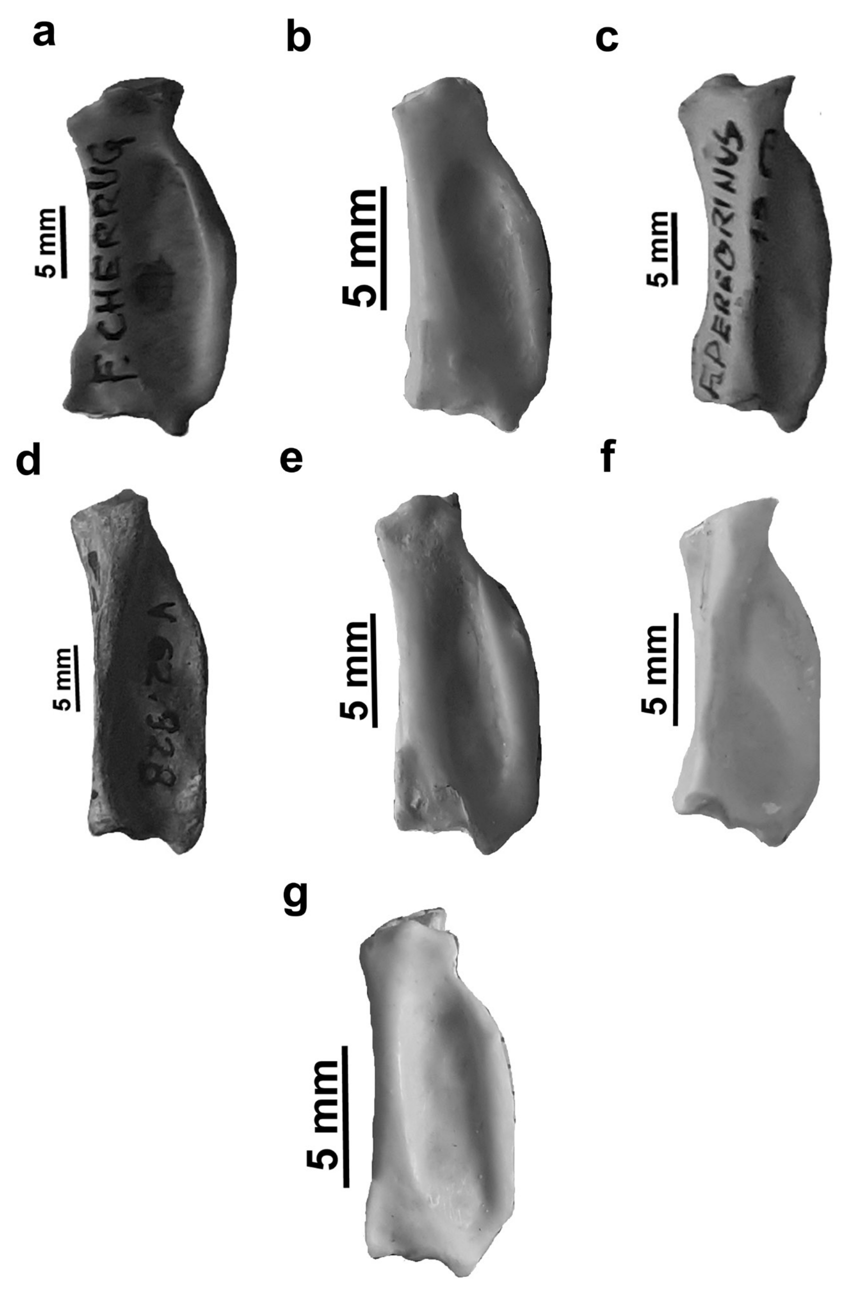

Figure 5. Left *phalanx proximalis digiti majoris* (ventral aspect)
5. ábra Bal oldali kézujjperc (1. ujjperc, II. ujj, alulnézet)
a. *Falco cherrug* (Gray, 1844); b. *Falco columbarius* (Linnaeus, 1758); c. *Falco peregrinus* (Tunstall, 1771)*;* d. *Falco rusticolus* (Linnaeus, 1758); e. *Falco subbuteo* (Linnaeus, 1758); f. *Falco tinunculus* (Linnaeus, 1758); g. *Falco vespertinus* (Linnaeus, 1766)

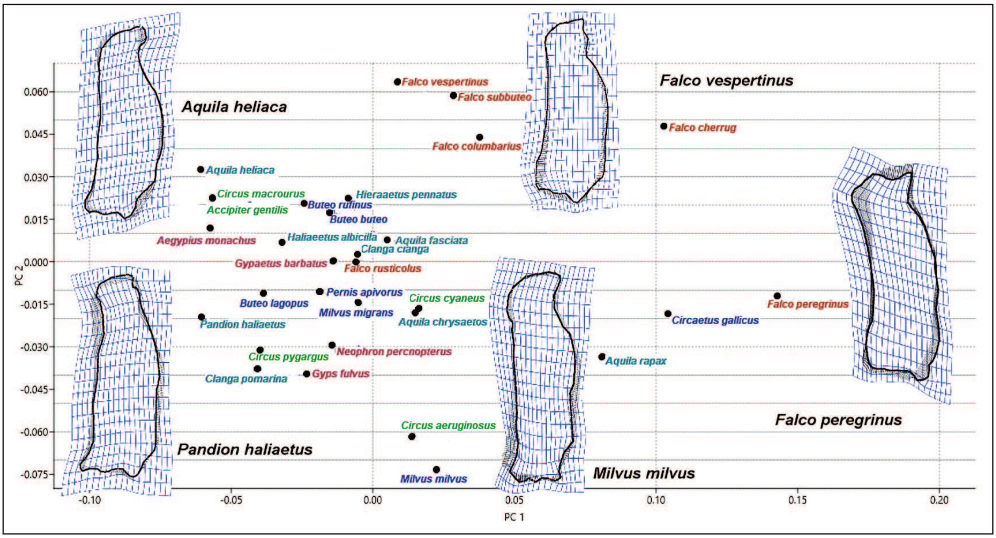

Figure 6. Graphical output of PCA performed on the two-dimensional landmark data (ventral view). PC1–PC2 biplot. The first PC axis described the curvature of the anterior surface of the proximal phalanx of the major digit (PC1). The second PC axis described the relative size of the distal and proximal epiphyses (PC2). Thick black areas show the differences compared to the computer generated mean shape

Purple: vultures; Cyan: eagles; Blue: buzzards and kites; Green: goshawk and harriers; Red: falcons and kestrels

6. ábra A PCA grafikus megjelenítése kétdimenziós landmark adatok alapján (alulnézet). A főkomponensek a csont anterior irányú görbületét (PC1) valamint a proximális és a disztális epifízisek egymáshoz viszonyított nagyságát magyarázzák (PC2). A fekete, vastagított terület a komputergenerált átlagformától való eltérést mutatja

Lila: keselyűk; Világoskék: sasok; Kék: ölyvek és kányák; Zöld: héja és rétihéják; Piros: sólymok és vércsék

Conclusions

In the case of examined diurnal raptor species living in wet habitats, the shape and size of the phalanx is quite diverse. What is more, we see sexual dimorphism in the case of most species. This can be quite substantial in buzzards, goshawks and falcons in favour of females.

The changes in form of the studied characteristics are worth comparing in terms of size differences, since the difference between vultures and kestrels is quite significant even if the similarities in form do exist. Representatives of other families and genera are between these two extremes in terms of size.

The first *phalanx* of the second wing finger of Accipitriformes is typically long and medium wide. The length to width ratio varies between 1/2 and 1/4. More specifically, it is around 1/2 at the Egyptian Vulture *(Neophron percnopterus)*, Golden Eagle *(Aquila chrysaetos)*, Booted Eagle *(Hieraaetus pennatus)*, Bonelli's Eagle *(Aquila fasciata)*, European Honney-buzzard *(Pernis apivorus)*, Common Kestrel *(Falco tinnunculus)*, Eurasian Hobby *(F,* *subbuteo)* and Red-footed Falcon *(F. vespertinus)*. This ratio is around 1/4 at the Bearded Vulture *(Gypaetus barbatus)*, Western Marsh Harrier *(Circus aeruginosus)*, while in the other species it is about 1/3.

Studying the ratio between wingspan / length phalanx x/2, from the size tables *(Table 1)* we found the followings:

In the case of the vulture – eagle group, we did not find an extraordinary relationship between the wingspan and the length of the phalanx, except of Egyptian Vulture, Bonelli's Eagle and Osprey. In the Egyptian Vulture and Bonelli's Eagle the ratio of wing length to phalanx length is much higher (39.47–47.37) than in the other species in the group (17.50–29.05). In the case of the Osprey, however, the situation is reversed (13.80–19.57).

In the case of the buzzard, kites, goshawks and harriers group, we did not find an extraordinary relationship between the wingspan and the length of the phalanx, except of European Honney-buzzard, where the ratio of wing length to phalanx length is much higher (35.53–39.47) than in the other species in the group (17.92–29.79).

In the case of the falcon and kestrel group, we found a difference between falcons (14.23–23.08) and kestrels in the wingspan and the length of the phalanx ratio, where the value is higher (22.58–27.88).

We can assume that the flight speed of these species is clearly consistent with these data but previous studies showed that the mechanism of flights in raptors is more complex (Hart *et al*. 2018, Krishnan *et al*. 2020). The effects of different attributes like body size and wing morphology on flight speed also very important, but the phylogenetic relationships accounted for an important part of the remaining variation in flight speed among species. Differences in flight apparatus and behaviour among species of different evolutionary origin, and with different ecology, behaviour and flight styles are likely to influence cruising flight in performance (Alerstam *et al*. 2007, Usherwood *et al*. 2020). Many raptors shared similar size and attributes, but the modularity of wing shape during locomotion and action has a key role because avian wings are not rigid parts of the body (Klaasen *et al*. 2016, Cheney *et al*. 2021). Previously, it was also observed that typical flight strategies of some migrating raptors that climbing rate in thermal circling did not differ between species, indicating that chiefly the strength of thermal updrafts determined the climbing rate and that morphological features were less relevant. Heavier species glided faster and had smaller gliding angles therefore body mass and size have more effect on flying performance. Eagles and buzzards are using mostly soaring and gliding flight, furthermore soaring in a straight line whilst gliding was extensively used by the Lesser Spotted Eagle *(Clanga pomarina)* and Booted Eagle and even more frequently by the species like the Griffon Vulture *(Gyps fulvus)* and Short-toed Snake Eagle *(Circaetus gallicus)* (Spaar 1997).

The tissues which are related to the metacarpophalangeal joint and the craniodorsal surface are mostly ligaments and muscles and affecting on the physical characteristics of the proximal phalanx of the major digit. On the anterior surface *tendo m. extensor longus digiti majoris* and *retinaculum dorsalis digiti majoris*, on the proximal epiphysis *tendo m. interosseus dorsalis* and *m. interosseus ventralis*, while on the distal epiphysis *m. interosseus ventralis* have key roles in the appearance of the bone (Hyeronymus 2016). The principal component analysis showed slightly overlapping in shape between the taxa but the accipitriform and

Table 1. Size table of *phalanx proximalis digiti majoris*
A – total length of bone (mm); C – breadth of proximal end; E – breadth of corpus; F – breadth of distal end. New name (Gill *et al.* 2020)

1. táblázat A *phalanx proximalis digiti majoris* mérettáblázata
A – teljes hossz (mm); C – proximális vég szélessége; E – a test szélessége; F – disztális vég szélessége. Új név – New name (Gill *et al.* 2020)

Species	**New genus, sp. name**	**A**	**C**	**E**	**F**	**wing**	**wing/ phalanx**
Aegypius monachus		60.00	11.50	22.00	12.00	1250-1500	20.83-25.00
Gypaetus barbatus		66.00	12.00	15.00	15.00	1155-1415	17.50-21.44
Gyps fulvus		58.00	14.50	20.00	13.50	1200-1400	20.69-24.14
Neophron percnopterus		18.00	8.00	12.00	10.00	775-850	**43.06-47.22**
Aquila chrysaetos		39.00	11.00	16.50	9.50	950-1100	24.36-28.21
Aquila clanga	*Clanga*	32.00	7.00	11.00	9.00	800-900	25.00-28.16
Aquila heliaca		39.00	6.50	11.50	7.00	900-1075	23.08-27.56
Aquila pomarina	*Clanga*	33.00	7.00	10.50	7.50	675-800	20.45-24.24
Aquila rapax		41.00	11.00	14.50	13.00	795-930	19.39-22.31
Haliaeetus albicilla		42.00	8.00	15.50	14.50	965-1240	22.98-29.05
Hieraaetus fasciatus	*Aquila fasciata*	19.00	8.50	11.00	9.50	750-900	**39.47-47.37**
Hieraaetus pennatus		22.00	7.00	9.50	5.50	500-605	22.73-27.50
Pandion haliaetus		46.00	8.00	14.00	12.00	635-900	**13.80-19.57**
Buteo buteo		23.50	5.50	7.50	6.50	545-700	23.19-29.79
Buteo lagopus		27.50	6.50	10.00	7.50	600-750	21.80-27.27
Buteo rufinus		31.50	7.00	11.00	8.50	625-740	19.84-23.49
Circaetus gallicus		34.00	9.00	13.00	12.00	810-975	23.82-28.68
Milvus migrans		30.50	9.00	10.00	7.00	800-900	26.63-29.51
Milvus milvus		34.00	8.50	10.00	8.50	875-895	25.74-26.32
Pernis apivorus		19.00	5.00	8.00	5.00	675-750	**35.53-39.47**
Accipiter gentilis		26.50	5.00	7.50	5.00	475-625	17.92-23.58
Accipiter nisus		15.50	3.00	5.50	4.00	295-400	19.03-25.81
Circus aeruginosus		28.00	7.50	7.50	5.00	575-650	20.54-23.21
Circus cyaneus		21.00	5.50	7.00	5.00	500-600	23.81-28.57
Circus macrourus		23.00	4.00	7.50	5.50	475-600	20.65-26.09
Circus pygargus		25.00	5.00	7.50	5.00	525-600	21.00-24.00
Falco cherrug		28.00	8.00	11.00	9.00	510-600	18.21-21.43
Falco columbarius		14.50	4.00	5.50	4.50	250-310	17.24-21.38
Falco peregrinus		26.00	8.50	9.00	8.00	370-600	14.23-23.08
Falco rusticolus		30.00	7.00	9.50	7.50	525-675	17.50-22.50
Falco subbuteo		16.50	4.50	6.50	4.00	410-460	**24.85-27.88**
Falco tinnunculus		15.50	5.50	6.00	4.50	350-400	**22.58-25.81**
Falco vespertinus		13.50	3.50	5.50	3.50	350-375	**25.93-27.78**

falconiform birds diverged in the morphospace. These results suggested that the attributes and geometry of the proximal phalanx of the major digit are affected rather by phylogenetic factors than physical or mechanical demands of different flight modes. Moreover, these characteristics can be used for identify the remains of raptor species.

In further studies, the phylogenetic control would be necessary. We should also increase our knowledge about the effect of growth factors and age on these parts of the avian skeletal system.

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References

Alerstam, T., Rosén, M., Bäckman, J., Ericson, P. G. P. & Hellgren, O. 2007. Flight speeds among bird species: allometric and phylogenetic effects. – PLoS Biology 5(8): e197. DOI: 10.1371/journal.pbio.0050197

Baumel, J. J., King, A. S., Lucas, A. M., Breazile, J. E. & Evans, H. E. 1979. Nomina anatomica avium. – Academical Press, London

Cheney, J. A., Stevenson, J. P., Durston, N. E., Maeda, M., Song, J., Megson-Smith, D. A., Windsor, S. P., Usherwood, R. P. & Bomphrey, R. J. 2021. Raptor wing morphing with flight speed. – Journal of the Royal Society Interface 18(180): 20210349. DOI: 10.1098/rsif.2021.0349

Cohen, A. & Serjeantson, D. 1996. A manual for the identification of bird bones from archaeological sites (revised ed.). – Archetype Publications, London

Gilbert, B. M., Martin, L. D. & Savage, H. G. 1981. Avian Osteology. – Library of Congress, Wyoming

Gill, F., Donsker, D. & Rasmussen, P. (eds.) 2020. IOC World Bird List. – DOI: 10.14344/IOC.ML.10.2.

Hammer, Ø., Harper, D. A. T. & Ryan, P. D. 2001. PAST: paleontological statistics software package for education and data analysis. – Palaeontologia Electronica 4: 1–9.

Hieronymus, T. L. 2016. Flight feather attachment in Rock Pigeons *(Columba livia)*: covert feathers and smooth muscle coordinate a morphing wing. – Journal of Anatomy 229(5): 631–656. DOI: 10.1111/joa.12511

Kessler, E. 2013. A Kárpát-medence madárvilágának őslénytani kézikönyve [Paleontological Handbook of Birdlife in Carpathian Basin]. – Könyvműhely, Miskolc (in Hungarian)

Kessler, J. (E.) 2015. Osteological guide of songbirds from Central Europe. – Ornis Hungarica 23(2): 62–156. DOI: 10.1515/orhu-2015-0009

Kessler, J. (E.) 2016a Picidae in the European fossil, subfossil and recent bird faunas and their osteological characteristics. – Ornis Hungarica 24(1): 96–114. DOI: 10.1515/orhu-2016-0009

Kessler, J. (E.) 2016b Evolution and skeletal characteristics of European owls. – Ornis Hungarica 24(2): 65–103. DOI: 10.1515/orhu-2016-0009

Kessler, J. (E.) 2019. Pigeons, sandgrouse, cuckoos, nightjars, rollers, bee-eaters, kingfishers and swifts in the European fossil avifauna and their osteological characteristics. – Ornis Hungarica 27(1): 132–165. DOI: 10.2478/orhu-2019-0009

Kessler, J. (E.) 2020. Evolution of Corvids and their presence in the Neogene and the Quaternary in the Carpathian Basin. – Ornis Hungarica 28(1): 121–168. DOI: 10.2478/orhu-2020-0009

Kessler, J. (E.) & Horváth, I. 2020.The wing phalanges *(phalanx proximalis digiti majoris)* of Gaviiformes, Podicipediformes, Pelecaniformes, Ardeiformes, Anseriformes, Gruiformes, Ralliformes, Charadriiformes, and Galliformes. – Ornis Hungarica 29(1): 149–169. DOI: 10.2478/orhu-2021-0012

Klaassen van Oorschot, B., Mistick, E. A. & Tobalske, B. W. 2016. Aerodynamic consequences of wing morphing during emulated take-off and gliding in birds. – Journal of Experimental Biology 219(19): 3146–3154. DOI: 10.1242/jeb.136721

Krishnan, K., Ben-Gida, H., Morgan, G., Kopp, G. A., Guglielmo, C. G. & Gurka, R. 2020. Turbulent wake-flow characteristics in the near wake of freely flying raptors: a comparative analysis between an owl and a hawk. – Integrative and Comparative Biology 60(5): 1109–1122. DOI: 10.1093/icb/icaa106

Milne-Edwards, A. 1867–1868. Recherches anatomiques et paléontologiques pour servir àl'histoire des oiseaux fossiles de la France,Vol. 1. [Anatomical and paleontological research to be used in the history of the fossil birds of France, Vol. 1.]. – Paris: Victor Masson et Fils (in French)

Rohlf, F. J. 2010. TpsDig, version 2.16. – Department of Ecology and Evolution, State University of New York, Stony Brook, USA

Solti, B. 1980. Beiträge zur Kenntnis der Osteologie des Gerfalken (*Falco rusticolus* L., 1758) [Contributions to the knowledge of the osteology of the Gyrfalcon (*Falco rusticolus* L., 1758)]. – Folia Historico-Naturalia Musei Matraensis 6: 189–204. (in German)

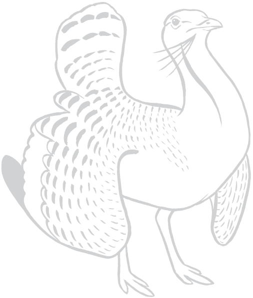
Solti, B. 1981a Vergleichend-osteologische Untersuchungen am Skelettsystem der Falkenarten *Falco cherrug* Gary und *Falco peregrinus* Tunstall [Comparative osteological studies on the skeletal system of the falcon species *Falco cherrug* Gary and *Falco peregrinus* Tunstall]. – Vertebrata Hungarica 20: 75–125. (in German)

Solti, B. 1981b Osteologische Untersuchungen an *Falco biarmicus* Temminck, 1825 [Osteological investigations on *Falco biarmicus* Temminck, 1825]. – Folia Historico-Naturalia Musei Matraensis 7: 135–151. (in German)

Solti, B. 1996. The comparative osteomorphological study of the European small-statured falcons (Aves: Falconidae). – Folia Historico-Naturalia Musei Matraensis 21: 5–282.

Spaar, R. 1997. Flight strategies of migrating raptors; a comparative study of interspecific variation in flight characteristics. – Ibis 139(3): 523–535. DOI: 10.1111/j.1474-919X.1997.tb04669.x

Usherwood, J. R., Cheney, J. A., Song, J., Windsor, S. P., Stevenson, J. P., Dierksheide, U., Nilla, A. & Bomphrey, R. J. 2020. High aerodynamic lift from the tail reduces drag in gliding raptors. – Journal of Experimental Biology 223(3): jeb214809. DOI: 10.1242/jeb.214809



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The orientation of nestboxes influences their occupation rates and the breeding success of passerine birds

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Abstract Nestboxes are widely provided as nesting sites for hole-nesting birds, yet the relative contribution of nestbox characteristics and habitat quality in determining the occupancy rates and breeding success of birds remains unclear. We provided nestboxes in deciduous woodlands in England and examined if those nestboxes were erected in random orientations and whether the orientation of nestboxes and habitat quality, in the form of tree density, influenced their occupation by, and breeding success of, Blue Tits *(Cyanistes caeruleus)*, Great Tits *(Parus major)* and Pied Flycatchers *(Ficedula hypoleuca)*. We found that first, the nestboxes were erected non-randomly orientated towards the north and east, and away from the south and west. Second, the occupation rates of none of the species was related to nestbox orientation or tree density. Third, the breeding success of neither Blue Tits nor Great Tits varied with tree density but did vary with nestbox orientation. Blue Tit hatching success and fledging success was higher in nestboxes facing south than in other directions whilst in Great Tits, clutch sizes, hatching success and fledging success was higher in nestboxes facing south than nestboxes facing other directions. Our results suggest that nestbox characteristics, such as orientation, have more influence on the reproductive success of passerines than habitat quality. This further suggests that conservationists should orientate nestboxes southwards in order to maximise their benefit to birds in temperate climates during the breeding season.

Keywords: conservation tool; habitat quality; nest site selection; reproductive success; woodland birds

Összefoglalás Az odúban fészkelő madarak költési lehetőségeinek támogatására széles körben használnak mesterségesen kihelyezett költőládákat. A költőládák tulajdonságainak és az élőhely minőségének az elfoglalás gyakoriságára és a madarak költési sikerére gyakorolt hatása azonban további kérdések megválaszolását igényli. Ezért költőládákat helyeztünk ki Anglia több lombhullató erdejébe és megvizsgáltuk, hogy a költőládák vajon véletlenszerű irányokban lettek-e kihelyezve, valamint tájolásuknak és az élőhely minőségének, a környező fák sűrűségében mérve, volt-e hatása azok elfoglalásának arányára és a kék cinege *(Cyanistes caeruleus)*, a széncinege *(Parus major)*, illetve a kormos légykapó *(Ficedula hypoleuca)* költési sikereire. Eredményeink azt mutatják, hogy a költőládák nem véletlenszerűen, hanem többnyire északi és keleti tájolással lettek kihelyezve, ugyanakkor az elfoglalás aránya egyik fajnál sem mutatott összefüggést a tájolással és a fák sűrűségével. A költési sikert nem befolyásolta a környező fák sűrűsége, sem a kék cinegénél, sem pedig a széncinegénél, viszont a költőládák tájolása igen. Míg a kék cinegék kikelési és kirepülési sikere a délre néző költőládákban magasabb volt más tájolású költőládákhoz képest, addig a széncinegéknél ezeken felül még a fészekalj mérete is nagyobb volt a déli tájolású költőládákban. Eredményeink alapján elmondható, hogy a költőládák tájolásának nagyobb hatása van a madarak költési sikerére, mint az élőhely minőségének. Ez azt is alátámasztja, hogy a természetvédelmi tevékenységek során a költőládákat déli irányba érdemes kihelyezni, hogy a mérsékeltövi madarak számára minél több előny származzon ebből a költési időszakban.

Kulcsszavak: erdei madarak, élőhelyminőség, kék cinege, kormos légykapó, költési siker, költőládák, széncinege, tájolás

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Introduction

Nestboxes are widely provided for hole-nesting birds to support their nesting opportunities in temperate woodlands (Lambrechts *et al*. 2010, Culina *et al*. 2021). The frequent occupation of nestboxes by passerine birds such as Blue Tits *(Cyanistes caeruleus)*, Great Tits *(Parus major)*, Collared Flycatchers *(Ficedula albicollis)* and Pied Flycatchers (*Ficedula hypoleuca*) in Europe and species such as Black-capped Chickadees *(Poecile atricapillus)* and Tree Swallows *(Tachycineta bicolor)* in North America has resulted in them becoming some of the most popular model species in ecological and evolutionary research (Lambrechts *et al*. 2010, Culina *et al*. 2021). The provision of nestboxes also helps to provide nesting sites for species of conservation concern. Pied Flycatcher populations have declined (Hewson *et al*. 2007), at least partly, because of climate change induced lethal competition with sedentary Great Tits (Samplonius & Both 2019) and nestboxes help to provide nest sites for the Flycatchers (Goodenough *et al*. 2009).

Nestboxes are not uniformly effective in providing nest sites for birds, though, because occupancy rates vary with respect to their shape (Bellrose *et al*. 1964), age (Mazgajski 2007), colour (Browne 2006), exposure (Charter *et al*. 2010), the size of the entrance holes (Le Roux *et al*. 2016, Stanback *et al*. 2019) and their orientation. In the temperate zone of the northern hemisphere, birds are expected to preferentially occupy nestboxes facing southwards towards the sun in order to increase nest temperatures (Goodenough *et al*. 2008a, 2008b, Butler *et al*. 2009, Lambrechts *et al*. 2020). Illustratively, Tree Swallows in North America occupied warmer nestboxes orientated south and east during the cooler first half of the breeding season yet showed no preference for nestboxes in relation to their orientation in the warmer second half of the breeding season when nest temperatures were unrelated to nestbox orientation (Ardia *et al*. 2006). This agrees with studies showing that passerine birds in temperate woodlands in the northern hemisphere preferentially select nestboxes facing south (Lumsden 1986, Sacilotto & Anderson 2005). However, other studies show that passerine birds preferentially occupy nestboxes facing east (Ardia *et al*. 2006) and the north-west (Navara & Anderson 2011), whilst others show that nestboxes are randomly occupied with respect to orientation (Lumsden 1986, Mennill & Ratcliffe 2004, Goodenough *et al*. 2008a, 2008b). This suggests that further studies are required to understand how nestbox orientation influences their occupation rates.

Birds may also differentially occupy nestboxes with respect to their surrounding habitat. For example, European Roller's *(Coracias garrulus)* preferentially occupied nestboxes located in olive groves, irrigated crops and holm oaks, whilst actively avoiding nestboxes located in cereals (Avilés *et al*. 2000). Meanwhile, House Wrens *(Troglodytes aedon)* preferentially occupied nestboxes located in areas with sparser woodland understories and were more likely to fledge offspring from nestboxes in sparsely foliated areas (Finch

1989), whilst Bearded Reedlings *(Panurus biarmicus)* preferentially occupied nestboxes located in reed beds over water than over dry land (Wilson 2015). Birds are therefore able to differentiate between nestboxes in relation to the habitat in which they are located (Bloom & Hawks 1983, Belles-Isles & Picman 1986, Finch 1989, Rohrbaugh & Yahner 1997, Monti *et al*. 2019, Mueller *et al*. 2019, but see Le Roux *et al*. 2016). Miller (2010) provided nestboxes for passerine birds in temperate woodlands and suggested that the very low occupancy rates of those nestboxes indicated that habitat structure played a greater role in determining passerine bird populations than the availability of nestboxes. This suggests that habitat quality may well influence nestbox occupancy rates, although our understanding of the relative contributions of nestbox characteristics and habitat quality in determining the occupancy rates of nestboxes and the breeding success of the occupants remains incomplete.

We used data from a long-running nestbox scheme (Smith *et al*. 2011, Briggs & Deeming 2016, 2021, Briggs & Mainwaring 2017, 2019) to examine whether nestboxes were provided randomly with respect to their orientation, whether breeding Blue Tits, Great Tits and Pied Flycatchers occupied nestboxes based on their orientation and/or the density of trees surrounding nestboxes and whether the breeding success of Blue Tits and Great Tits varied in relation to nestbox orientation and tree density. We predict that (1) nestboxes will be placed randomly with respect to their orientation. Meanwhile, given that south-facing nestboxes receive more sunlight than nestboxes facing in other directions (Goodenough *et al*. 2008b, Butler *et al*. 2009, Lambrechts *et al*. 2020), we predict that (2) nestboxes facing south will be preferentially occupied because of the thermal benefits they receive from being exposed to solar radiation. We also predict that (3) birds breeding in nestboxes facing southwards will have higher levels of breeding success than conspecifics in nestboxes facing in other directions because of their high exposure to direct sunlight.

Methods

Study sites and their characteristics

We studied breeding populations of Blue Tits, Great Tits and Pied Flycatchers occupying nestboxes in seven largely deciduous woodlands, two of which contained small areas of coniferous trees. The woodlands were in Lancashire, north-west England (for more information see Briggs & Mainwaring 2017, 2019, Briggs & Deeming 2016, 2021). The woodlands in the Forest of Bowland Area of Outstanding National Beauty (54˚05’N, 02˚ 36’W; altitude 100–250 metres above sea level) on calcium poor gritstone rock types had many oak (*Quercus* spp.) and birch (*Betula* spp.) trees and had an understorey of wavy hairgrass *(Deschampsia flexuosa)*, which was classified as ‘W10’ (*Quercus robur* - *Pteridium aquilinum* - *Rubus fruticosus* woodland) in the National Vegetation Classification (Rodwell 1991).

A total of 187 identical wooden nestboxes with small circular entrance holes were provided for small passerine birds throughout the woodlands by one person (KBB) in 1977, many years prior to the initiation of this present study. Each year, all the nestboxes were checked for their suitability as breeding sites for passerine birds and were repaired or replaced if they were damaged, so as to ensure that all the nestboxes were available for birds in each breeding season. The orientation of all the nestbox holes was then quantified in April 2021 using a 'Silva Explorer III compass' and in each instance, a bearing of the degrees from due north was recorded.

We quantified the number of trees and shrubs, identified to species, surrounding all the 187 nestboxes in order to give an indication of the number of caterpillars that are likely to have been available for parent birds to feed to their offspring (Wilkin *et al*. 2009). In September 2019, one observer (KBB) quantified the number of trees, shrubs and bushes within a 25-meter radius of all of the nestboxes and identified them to a total of 26 different species including English oak *(Quercus robur)*, ash, sycamore *(Acer pseudoplatanus)*, Scots pine *(Pinus sylvestris)* and birch trees and hazel *(Corylus avellana)*, hawthorn *(Crataegus monogyna)*, bird cherry *(Prunus padus)*, and plum *(Prunus domestica)* bushes as outlined in the guide provided by Mitchell (1974).

Quantifying reproductive parameters

Small passerine birds were provided with wooden nestboxes of identical dimensions (see Briggs & Mainwaring 2017) throughout the study woodlands by one observer (KBB) in order to minimise variation in nestbox design between breeding pairs (Lambrechts *et al*. 2010, Møller *et al*. 2014, Mainwaring *et al*. 2015). All nestboxes were checked at roughly weekly intervals from late March onwards, to establish the beginning of nest building behaviours. After nests were found, regular checks established the date on which the first eggs were laid and if multiple eggs were present in nests, we therefore back-calculated egg laying dates assuming that one egg was laid per day as is usual in the three study species (Lundberg & Alatalo 1992, Cramp & Perrins 1993).

Further visits established the onset of incubation and close to the date on which eggs were due to hatch, clutch sizes were determined as the number of eggs within nests at that time. The nests were then routinely checked every few days in order to quantify hatching success, which was defined as the number of eggs that hatched. Then, when the eldest offspring within broods were 14 days old, all nestlings were weighed to the nearest 0.1 grams to quantify their mass at pre-fledging (Mainwaring & Hartley 2016). The nests were then left alone in order to prevent the premature fledging of nestlings but were again checked when the eldest nestling/s within broods would have been about 20 days old. It was at this point that we checked for dead nestlings inside the nestboxes, which therefore allowed us to quantify the fledging success of breeding pairs.

Statistical analyses

The data were analysed in the SPSS version 25 (SPSS, Chicago, IL, USA) statistical package. The distribution of the orientation of nestboxes was examined using a one-sample Chi-square test to examine if their orientation differed from a random distribution.

Specifically, the proportion of nestboxes orientated towards the north (315–45 degrees), east (45–135 degrees), south (135–225 degrees) or west (225–315 degrees) was tested against an expected proportion of 0.25 for each direction.

The occupation rates of nestboxes by Blue Tits, Great Tits, Pied Flycatchers and all bird species were examined using general linear models. The ‘all bird’ species data refers to the occupation of nestboxes by all bird species because in addition to Blue Tits, Great Tits and Pied Flycatchers, we also found very small numbers of Common Redstarts *(Phoenicurus phoenicurus)*, Eurasian Nuthatches *(Sitta europaea)*, Marsh Tits *(Poecile palustris)* and Coal Tits *(Periparus ater)* breeding inside the nestboxes. The assumptions of normality were examined using Kolmogorov-Smirnov one sample tests before separate general linear models for Blue Tits, Great Tits, Pied Flycatchers and all bird species examined the role of nestbox orientation and tree density in determining occupation rates. In each of the models, the dependent variable was occupancy rate (0–6 years occupied), and the explanatory variables were ‘orientation’ (north, east, south or west: factor), ‘tree density’ (number of trees: covariate) and their two-way interaction term. The fixed terms, as well as their two-way interaction term, were initially included in the ‘full’ initial models, before they were sequentially ‘removed’ in a stepwise manner. For each of the terms, their significance was assessed by their Type I sum of squares and their values were only reported when they were the final terms in the models (Crawley 1993). Terms were dropped from models when their P values were non-significant, meaning that the final minimal models contained only those fixed terms whose P values were significant, or instead were included in a significant two-way interaction term.

Linear mixed models with a Gaussian error structure were used to analyse the breeding success data because the inclusion of random effects allowed us to consider spatially autocorrelated data and so here, ‘clutch identity’ was included as a random effect in all models. The reproductive success of the Blue Tits and Great Tits in relation to the orientation of nestboxes and tree densities were examined as follows, whilst we had no data on Pied Flycatchers and so they were not included in these analyses. The dependent variables were first egg date, clutch size, hatching success, nestling masses at pre-fledging and fledging success and the explanatory variables were ‘orientation’ (north, east, south or west: factor), ‘tree density’ (number of trees: covariate) and their interaction term, whilst ‘clutch identity’ was fitted as a random term in all models. The full models were simplified as outlined above. Throughout the manuscript, means are presented ± 1 standard error and a critical P value of 0.05 is applied throughout.

Results

A one-sample Chi-square test showed that nestboxes were non-randomly orientated towards the north and east and away from the south and west and so were not randomly distributed with respect to their orientation ($\chi^2 = 9.624$, $P = 0.005$) *(Figure 1).* The occupation rates of nestboxes by Blue Tits, Great Tits, Pied Flycatchers and all bird species was not generally related to the orientation of nestboxes, the density of trees within a 25-meter radius

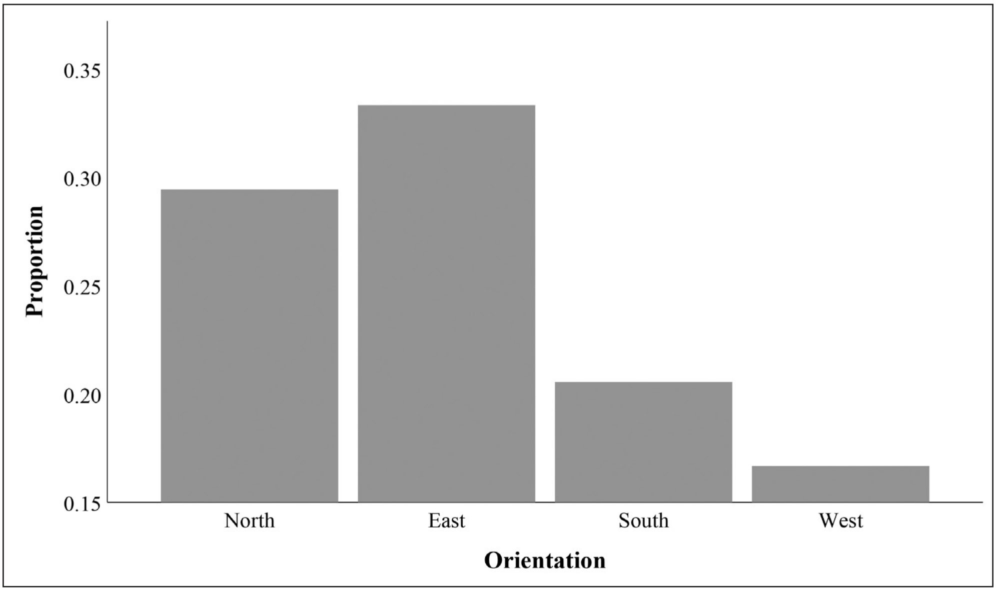

Figure 1. The non-random orientation of nestboxes
1. ábra A költőládák tájolásának nem véletlenszerű megoszlása

Table 1. Summary of general linear models examining variation in the occupancy rate of nestboxes by Blue Tits, Great Tits, Pied Flycatchers and all bird species combined in relation to the orientation of nestboxes and tree densities. The dependent variables were occupancy rate (0–6 years occupied), and the explanatory variables were 'orientation' (north, east, south or west: factor), 'tree density' (number of trees: covariate) and their interaction term. Note that significant terms are highlighted in bold

1. táblázat Az általános lineáris modellek eredményeinek összefoglalása, amelyekkel a költőládák elfoglalási arányának változatosságát (függő változó, 0–6 éven keresztül használt) és azok tájolását (észak, kelet, dél, nyugat – főhatás), valamint a környező fák sűrűségének (a fák száma – kovariáns) és a két magyarázó változó kölcsönhatásának összefüggéseit vizsgáltuk kék cinegék, széncinegék, kormos légykapók és az összes vizsgált faj esetében. A statisztikailag szignifikáns eredményeket félkövér betűtípussal emeltük ki

Species	Fixed effects	d.f.	F-value	P value
Blue Tit	Orientation	4,186	1.156	0.332
	Tree density	1,186	2.310	0.130
	Orientation × tree density	4,186	0.113	0.978
Great Tit	Orientation	4,186	0.720	0.579
	Tree density	1,186	1.665	0.200
	Orientation × tree density	4,186	0.269	0.897
Pied Flycatcher	Orientation	4,186	0.786	0.535
	Tree density	1,186	0.002	0.963
	Orientation × tree density	4,186	1.449	0.220
All bird species	Orientation	4,186	1.176	0.323
	Tree density	**1,186**	**4.055**	**0.046**
	Orientation × tree density	4,186	0.469	0.759

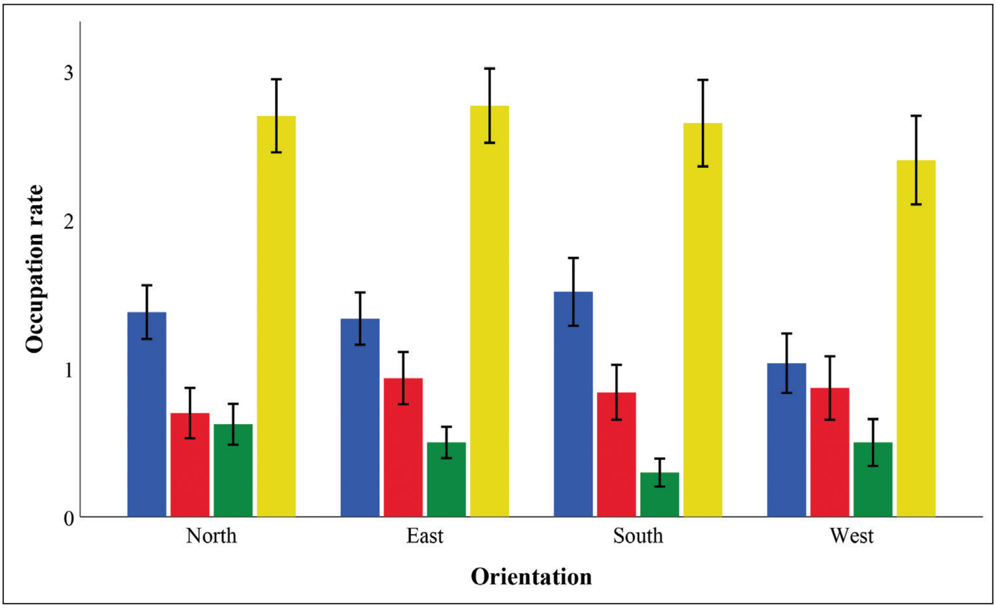

Figure 2. The occupation rates of nestboxes in relation to their orientation. Note that Blue Tits are shown by blue bars, Great Tits by red bars, Pied Flycatchers by green bars and all bird species by yellow bars

2. ábra A költőládák elfoglalásának aránya a tájolásuk szerinti csoportosításban. Kékkel a kék cinegék, pirossal a széncinegék, zölddel a kormos légykapók és sárgával az összes madárfaj adatai láthatóak

of nestboxes or their interaction terms *(Table 1, Figure 2)*, although there was a weakly significant trend for the occupation rates of all bird species to be positively correlated with tree density (P = 0.046) *(Table 1)*.

In terms of the breeding success of Blue Tits, there was no variation in first egg dates or clutch sizes in relation to the orientation of nestboxes or tree density *(Table 2, Figures 3a, 3b)*. Meanwhile, hatching success was higher in nestboxes facing south than in all other directions (P = 0.014) *(Table 2, Figure 3c)*, but it did not vary in relation to tree density *(Table 2)*. The mass of fledglings at pre-fledging did not vary with the orientation of nestboxes or with tree density *(Table 2, Figure 3d)*. Finally, fledging success never varied in relation to tree density *(Table 2)* but did in relation to orientation (P = 0.007) *(Table 2, Figure 3e)*, being lowest in nestboxes facing north and west and higher for nestboxes facing east and higher still for nestboxes facing south.

In terms of the breeding success of Great Tits, there was no variation in first egg dates in relation to the orientation of nestboxes or with tree density *(Table 2, Figures 3a)*. Meanwhile, clutch sizes never varied in relation to the density of trees *(Table 2)* but did vary in relation to orientation (P = 0.041) *(Table 2, Figure 3b)*, being higher in nestboxes facing south than in nestboxes facing other directions. Similarly, hatching success was higher in nestboxes facing south than in nestboxes facing all other directions (P = 0.017) *(Table 2 Figure 3c)*, but did not vary with tree density *(Table 2)*. There was no variation in the mass of fledglings at

Table 2. Summary of linear mixed models examining variation in the reproductive success of Blue Tits and Great Tits in relation to the orientation of nestboxes and tree densities. The dependent variables were 'first egg date', 'clutch size', 'hatching success', 'nestling masses at pre-fledging' and 'fledging success' and the explanatory variables were 'orientation' (north, east, south or west: factor), 'tree density' (number of trees: covariate) and their interaction term, whilst 'clutch identity' was fitted as a random term. Note that significant terms are highlighted in bold

2. táblázat A lineáris kevert modellek eredményeink összefoglalása, amelyekkel a kék cinege és a széncinege költési sikereinek és a költőládák tájolásának (észak, kelet, dél, nyugat – főhatás), valamint a környező fák sűrűségének (a fák száma – kovariáns) és a két magyarázó változó kölcsönhatásának összefüggéseit vizsgáltuk. Függő változóként az első tojás lerakásának dátuma, a fészekalj mérete, a kikelési siker, a kirepülés előtti fiókák tömege, valamint a kirepülési siker szerepelt, míg a fészekaljak azonosítóit véletlen tényezőként tartalmazták a modellek. A statisztikailag szignifikáns eredményeket félkövér betűtípussal emeltük ki

Species	Breeding parameter	Fixed effects	d.f.	F-value	P value
Blue Tit	First egg date	Orientation	3,40	0.522	0.670
		Tree density	1,40	0.529	0.471
		Orientation × tree density	3,40	0.456	0.715
	Clutch size	Orientation	3,40	0.164	0.920
		Tree density	1,40	0.442	0.510
		Orientation × tree density	3,40	0.481	0.697
	Hatching success	Orientation	3,40	5.662	0.014
		Tree density	1,40	0.654	0.423
		Orientation × tree density	3,40	0.350	0.789
	Nestling mass	Orientation	3,40	0.413	0.302
		Tree density	1,40	0.551	0.477
		Orientation × tree density	3,40	0.469	0.644
	Fledging success	**Orientation**	**3,40**	**8.010**	**0.007**
		Tree density	1,40	0.037	0.849
		Orientation × tree density	3,40	1.351	0.271
Great Tit	**First egg date**	**Orientation**	**3,47**	**2.935**	**0.094**
		Tree density	1,47	1.835	0.173
		Orientation × tree density	3,47	1.582	0.180
	Clutch size	**Orientation**	**3,47**	**4.189**	**0.041**
		Tree density	1,47	0.274	0.694
		Orientation × tree density	3,47	0.124	0.838
	Hatching success	**Orientation**	**3,47**	**6.183**	**0.017**
		Tree density	1,47	0.821	0.573
		Orientation × tree density	3,47	1.744	0.152
	Nestling mass	Orientation	3,47	0.022	0.914
		Tree density	1,47	0.028	0.926
		Orientation × tree density	3,47	0.173	0.727
	Fledging success	**Orientation**	**3,47**	**7.835**	**0.012**
		Tree density	1,47	1.647	0.315
		Orientation × tree density	3,47	1.274	0.239

Figure 3.
A. First egg date

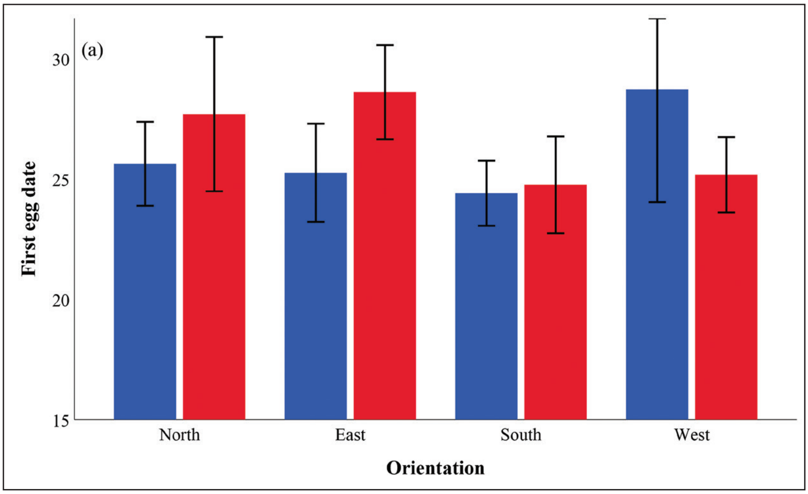

Figure 3.
B. Clutch size

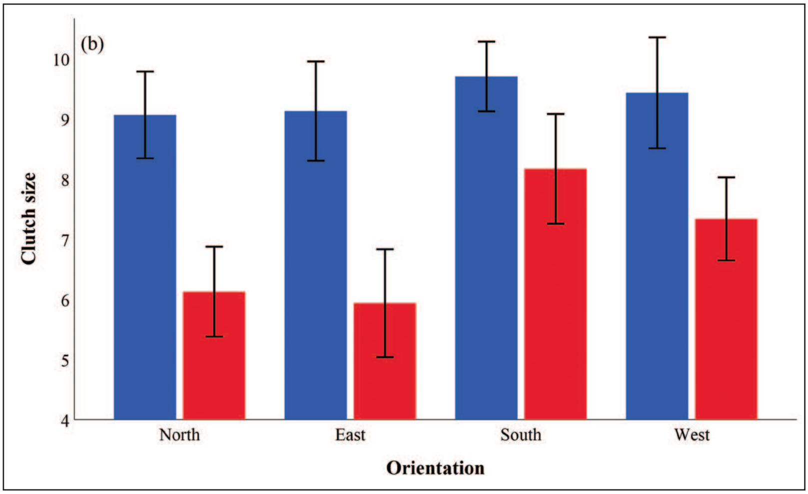

Figure 3.
C. Hatching success

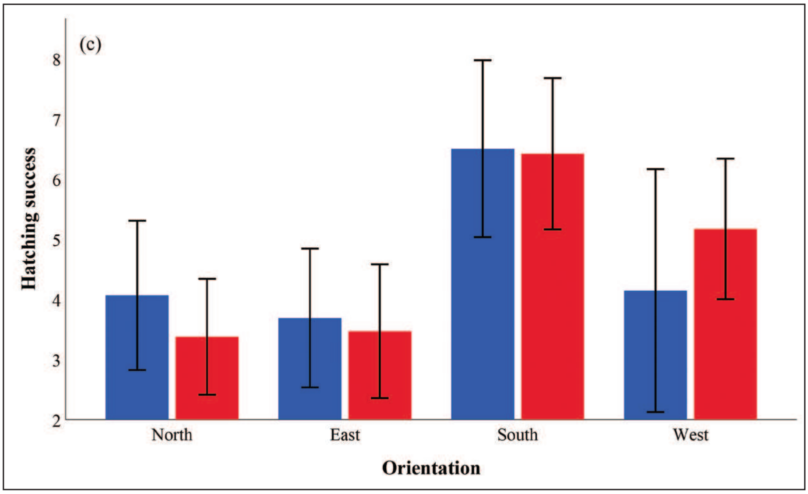

Figure 3.
D. Mass at day 14

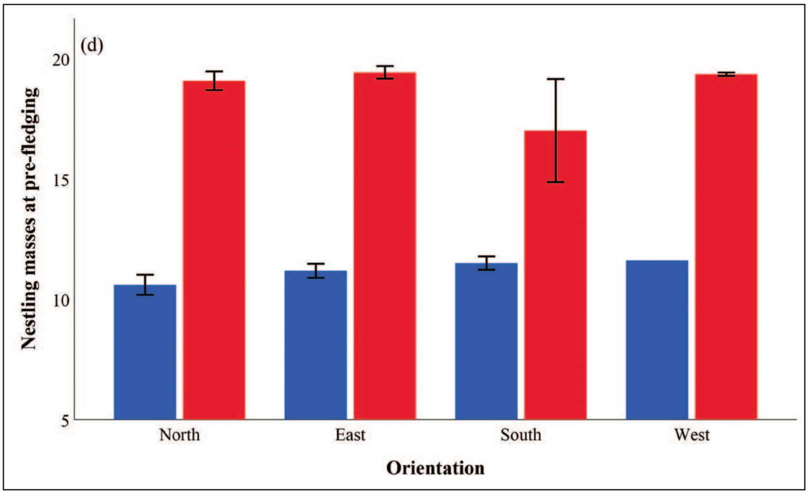

Figure 3.
E. Fledging success

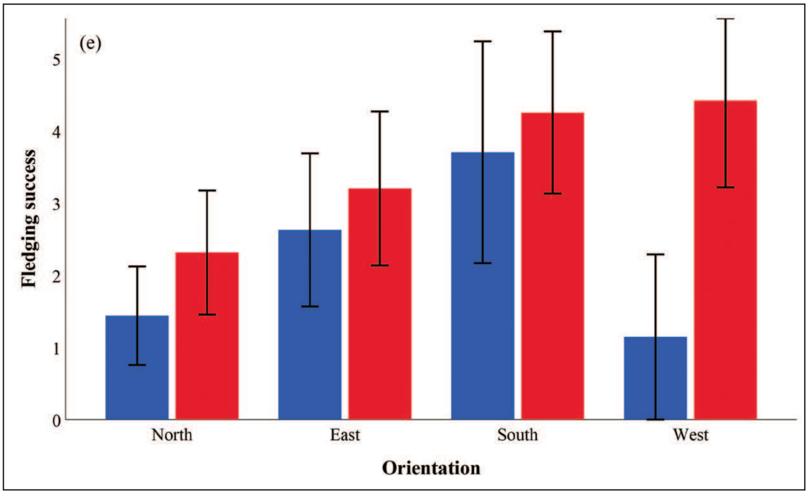

Figure 3. The reproductive success of Blue Tits and Great Tits, in terms of their (a) first egg date, (b) clutch size, (c) hatching success, (d) nestling masses at pre-fledging and (e) fledging success in relation to the orientation of nestboxes. Note that Blue Tits are shown by blue bars and Great Tits are shown by red bars

3. ábra A kék cinegék (kék) és a széncinegék (piros) költési sikereinek alakulása a költőládák tájolása szerinti csoportosításban: (a) az első tojás lerakásának dátuma, (b) fészekalj-méret, (c) kikelési siker, (d) a kirepülés előtti fiókák tömege, valamint (e) a kirepülési siker pre-fledging in relation to orientation or tree density *(Table 2, Figure 3d)*. Finally, fledging success never varied in relation to tree density *(Table 2)* but did vary in relation to orientation (P = 0.012) *(Table 2, Figure 3e)*, being lowest in nestboxes facing north and getting ever higher in nestboxes facing east, south and west.

Discussion

The main findings of this study were that nestboxes were non-randomly positioned on trees in relation to their orientation, being orientated towards the north and east and away from the south and west. Meanwhile, although the occupation rates of nestboxes by all three study species was unrelated to the orientation of nestboxes or tree density, Blue Tit and Great Tit breeding success was affected by the orientation of nestboxes, but not to tree density. Blue Tit hatching success and fledging success was higher in nestboxes facing south than in nestboxes facing any other directions. However, Great Tit clutch sizes and hatching success were higher in nestboxes facing south than in nestboxes facing other directions, whilst fledging success was higher in nestboxes facing west and south than in nestboxes facing east and north. Our results suggest that the orientation of nestboxes have more influence on the reproductive success of passerine birds than tree density. They further suggest that nestboxes in the temperate zone of the northern hemisphere should be preferentially orientated southwards.

The orientation of nestboxes never influenced their occupation rates by Blue Tits, Great Tits or Pied Flycatchers, but did affect the reproductive success of Blue Tits and Great Tits. Fledging success in both species was highest in nestboxes facing southwards, presumably because they received more sun than nestboxes facing in other directions, which thus maximised internal nest temperatures (Lumsden 1986, Sacilotto & Anderson 2005, Ardia *et al*. 2006, Goodenough *et al*. 2008a, 2008b, Butler *et al*. 2009, Mainwaring & Hartley 2012, Lambrechts *et al*. 2020). Birds inhabiting temperate environments in the northern hemisphere experience ambient temperatures below the threshold at which embryos develop optimally (Salaberria *et al*. 2014), and so they are likely to prefer nestboxes orientated towards the sun to reduce the energy required to keep the embryos warm (Goodenough *et al*. 2008b, Butler *et al*. 2009). Whilst birds breeding in nestboxes that faced south had the highest levels of breeding success, the majority of nestboxes were positioned on trees so that they faced north or east. This suggests that those nestboxes facing north may have been acting as an ecological trap because whilst the passerine birds readily occupied them, they ultimately suffered reduced levels of reproductive success by occupying them, when compared to conspecifics occupying nestboxes facing southwards (O'Connor 1984).

The density of trees surrounding nestboxes had no effect on the occupation rates of nestboxes or on the reproductive success of Blue Tits and Great Tits. This is perhaps surprising because previous studies show that the habitat surrounding nestboxes influences their occupation rates (Bloom & Hawks 1983, Belles-Isles & Picman 1986, Finch 1989, Rohrbaugh & Yahner 1997, Avilés *et al*. 2000, Wilson 2015, Mueller *et al*. 2019, but see Le Roux *et al*. 2016) and the reproductive success of the occupants (Finch 1989, Goodenough *et al*. 2008a). It is unclear why tree density never influenced occupancy rates or breeding success, but it may be that caterpillar abundance was unrelated to tree density. Alternatively, territories with fewer trees are likely to have more open canopies than territories with more trees, meaning that they may well have received a higher amount of solar radiation (Wachob 1996). The temperature inside nestboxes is positively correlated with ambient temperatures (Lambrechts *et al*. 2020) and so birds occupying nestboxes with fewer trees may therefore have benefitted from having relatively warm nest microclimates. Determining the mechanisms underlying this pattern deserves further research attention.

We have shown that hole-nesting passerine birds occupying nestboxes facing south had higher levels of breeding success than conspecifics occupying nestboxes facing in other directions. This is likely to be a result of those nestboxes facing south receiving a greater amount of solar radiation than nestboxes facing other directions, which thus increased the temperatures inside nestboxes and enabled birds to save energy keeping offspring warm. Our findings suggest that ornithologists should provide nestboxes that face south in temperate woodlands in the northern hemisphere. Further research is needed to examine the patterns described here. First, studies show that nestboxes provide less insulation that natural cavities (Maziarz *et al*. 2017) and so it would be interesting to examine if these patterns occur in species breeding in natural cavities (Lundberg & Alatalo 1992). Second, the patterns we have reported here are observational and thus may have been confounded by alternative factors. It would be useful to experimentally disassociate the relationship between the occupation of nestboxes and the density of trees by cutting trees to examine the influence of tree density on nestbox occupation rates. Third, south facing nestboxes provide thermal benefits for birds and research that quantifies how nest microclimates are influenced by the orientation of nestboxes in a changing climate may help ornithologists provide nesting sites for birds in the future.

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References

Ardia, D. R., Pérez, J. H. & Clotfelter, E. D. 2006. Nestbox orientation affects internal temperature and nest site selection by Tree Swallows. – Journal of Field Ornithology 77(3): 339–344. DOI: 10.1111/j.1557-9263.2006.00064.x

Avilés, S. M., Sánchez, J. M. & Parejo, D. 2000. Nest-site selection and breeding success in the Roller *(Coracias garrulus)* in the Southwest of the Iberian peninsula. – Journal of Ornithology 141(3): 345–350. DOI: 10.1007/BF02462244

Bellrose, F. C., Johnson, K. L. & Meyers, T. U. 1964. Relative value of natural cavities and nesting houses for Wood Ducks. – Journal of Wildlife Management 28(4): 661–676. DOI: 10.2307/3798781

Belles-Isles, J. C. & Picman, J. 1986. Nesting losses and nest preferences in House Wrens. – Condor 88(4): 483–486. DOI: 10.2307/1368275

Bloom, P. H. & Hawks, S. J. 1983. Nest box use and reproductive biology of American Kestrel in Lassen County, California. – Raptor Research 17(1): 9–14.

Briggs, K. B. & Deeming, D. C. 2016. Use of materials in nest construction by Pied Flycatchers *Ficedula* *hypoleuca* reflects localised habitat and geographical location. – Bird Study 63(4): 516–524. DOI: 10.1080/00063657.2016.1238867

Briggs, K. B. & Deeming, D. C. 2021. Localized habitat affects size and materials used in the construction of Common Redstart *Phoenicurus phoenicurus* nests. – Bird Study 68(1): 9–20. DOI: 10.1080/00063657.2021.1958197

Briggs, K. B. & Mainwaring, M. C. 2017. Habitat geology influences intraspecific variation in the speckling patterns of Blue Tit *Cyanistes caeruleus* and Great Tit *Parus major* eggs. – Acta Ornithologica 52(1): 11–20. DOI: 10.3161/00016454AO2017.52.1.002

Briggs, K. B. & Mainwaring, M. C. 2019. Experimental evidence of non-random nest material selection in Pied Flycatchers. – Behavioural Processes 164: 59–64. DOI: 10.1016/j.beproc.2019.04.008

Browne, S. J. 2006. Effect of nestbox construction and colour on the occupancy and breeding success of nesting tits *Parus* spp. – Bird Study 53(2): 187–192. DOI: 10.1080/00063650609461432

Butler, M. W., Whitman, B. A. & Dufty, A. M. Jr. 2009. Nest box temperature and hatching success of American Kestrels varies with nest box orientation. – Wilson Journal of Ornithology 121(4): 778–782. DOI: 10.1676/08-124.1

Charter, M., Meyrom, K., Leshem, Y., Aviel, S., Izhaki, I. & Motro, Y. 2010. Does nest box location and orientation affect occupation rate and breeding success of Barn Owls in a semi-arid environment? – Acta Ornithologica 45(1): 115–119. DOI: 10.3161/000164510X516164

Cramp, S. & Perrins, C. M. 1993. The birds of the Western Palearctic, Vol. 7: Flycatchers to Shrikes. – Oxford University Press, Oxford

Crawley, M. J. 1993. GLIM for Ecologist. – Blackwell Scientific Publications, Oxford

Culina, A., Adriaensen, F., Bailey, L. D., Burgess, M. D., Charmantier, A., Cole, E. F., Eeva, T., Matthysen, E., Nater, C. R., Sheldon, B. C., Sæther, B-E., Vriend, S. J. G., Adamík, P., Aplin, L. M., Angulo, E., Artemyev, A., Barba, E., Barišić, S., Belda, E., Can Bilgin, C., Bleu, C., Both, C., Bouwhuis, S., Branston, C., Broggi, J., Burke, T., Bushuev, A., Camacho, C., Campobello, D., Canal, D., Cantarero, A., Caro, S. P., Cauchoix, M., Chaine, A., Cichoń, M., Ćiković, D., Cusimano, C. A., Deimel, C., Dhondt, A., Dingemanse, N. J., Doligez, B., Dominoni, D. M., Doutrelant, C., Drobniak, S. M., Dubiec, A., Eens, M., Erikstad, K. E., Espín, S., Farine, D. R., Figuerola, J., Gülbeyaz, P. K., Grégoire, A., Hartley, I. R., Hau, M., Hegyi, G., Hille, S., Hinde, C. A., Holtmann, B., Ilyina, T., Isaksson, C., Iserbyt, A., Ivankina, E., Kania, W., Kempenaers, B., Kerimov, A., Komdeur, J., Korsten, P., Král, M., Krist, M., Lambrechts, M., Lara, C. E., Leivits, A., Liker, A., Lodjak, J., Mägi, M., Mainwaring, M. C., Mänd, R., Massa, B., Massemin, S., Martínez-Padilla, J., Mazgajski, T. D., Mennerat, A., Moreno, J., Mouchet, A., Nakagawa, S., Nilsson, J-Å., Nilsson, J., Norte, A. C., van Oers, K., Orell, M., Potti, J., Quinn, J. L., Réale, D., Reiertsen, T. K., Rosivall, B., Russel, A. F., Rytkönen, S., Sánchez-Virosta, P., Santos, E. S. A., Schroeder, J., Senar, J. C., Seress, G., Slagsvold, T., Szulkin, M., Teplitsky, C., Tilgar, V., Tolstoguzov, A., Török, J., Valcu, M., Vatka, E., Verhulst, S., Watson, H., Yuta, T., Zamora-Marín, J. M. & Visser, M. E. 2021. Connected data landscape of long-term ecological studies: the SPI-Birds data hub. – Journal of Animal Ecology 90(9): 2147–2160. DOI: 10.1111/1365-2656.13388

Finch, D. M. 1989. Relationships of surrounding riparian habitat to nest-box use and reproductive outcome in House Wrens. – Condor 91(4): 848–859. DOI: 10.2307/1368069

Goodenough, A. E., Elliot, S. L. & Hart, A. G. 2009. The challenges of conservation for declining migrants: are reserve-based initiatives during the breeding season appropriate for the Pied Flycatcher *Ficedula hypoleuca*? – Ibis 151(3): 429–439. DOI: 10.1111/j.1474-919X.2009.00917.x

Goodenough, A. E., Hart, A. G. & Elliot, S. L. 2008a Variation in offspring quality with cavity orientation in the Great Tit. – Ethology Ecology and Evolution 20(4): 375–389. DOI: 10.1080/08927014.2008.9522518

Goodenough, A. E., Maitland, D. P., Hart, A. G. & Elliot, S. L. 2008b Nestbox orientation: a species-specific influence on occupation and breeding success in woodland passerines. –Bird Study 55(2): 222–232. DOI: 10.1080/00063650809461526

Hewson, C. M., Amar, A., Lindsell, J. A., Thewlis, R. M., Butler, S., Smith, K. & Fuller, R. J. 2007. Recent changes in bird populations in British broadleaved woodland. – Ibis 149(s2): 14–28. DOI: 10.1111/j.1474-919X.2007.00745.x

Lambrechts, M. M., Adriaensen, F., Ardia, D. R., Artemyev, A. V., Atiénzar, F., Bańbura, J., Barba, E., Bouvier, J-C., Camproden, J., Cooper, C. B., Dawson, R. D., Eens, M., Eeva, T., Faivre, B., Garamszegi, L. Z., Goodenough, A. E., Gosler, A. G., Grégoire, A., Griffith, S. C., Gustafsson, L., Johnson, L. S., Kania, W., Keišs, O., Llambias, P. E., Mainwaring, M. C., Mänd, R., Massa, B., Mazgajski, T. D., Møller, A. P., Moreno, J., Naef-Daenzer, B., Nilsson, J-Å., Norte, A. C., Orell, M., Otter, K. A., Park, C. R., Perrins, C. M., Pinowski, J., Porkert, J., Potti, J., Remes, V., Richner, H., Rytkönen, S., Shiao, M-T., Silverin, B., Slagsvold, T., Smith,

H. G., Sorace, A., Stenning, M. J., Stewart, I., Thompson, C. F., Tryjanowski, P., Török, J., van Noordwijk, A. J., Winkler, D. W. & Ziane, N. 2010. The design of artificial nestboxes for the study of secondary hole-nesting birds: a review of methodological inconsistencies and potential biases. – Acta Ornithologica 45(1): 1–26. DOI: 10.3161/000164510X516047

Lambrechts, M. M., Caizergues, A. E., Perrier, C., Charmantier, A. & Caro, S. P. 2020. Surface temperatures of non-incubated eggs in Great Tits *(Parus major)* are strongly associated with ambient temperature. – International Journal of Biometeorology 64(10): 1767–1775. DOI: 10.1007/s00484-020-01958-1

Le Roux, D. S., Ikin, K., Lindenmayer, D. B., Bistricer, G., Manning, A. D. & Gibbons, P. 2016. Effects of entrance size, tree size and landscape context on nest box occupancy: Considerations for management and biodiversity offsets. – Forest Ecology and Management 366(1): 135–142. DOI: 10.1016/j.foreco.2016.02.017

Lumsden, H. G. 1986. Choice of nestboxes by Tree Swallows, *Tachycineta bicolor*, House Wrens, *Troglodytes aedon*, Eastern Bluebirds, *Sialia sialis*, and European Starlings, *Sturnus vulgaris*. – Canadian Field-Naturalist 100(3): 43–49.

Lundberg, A. & Alatalo, R. V. 1992. The Pied Flycatcher. – T & AD Poyser, London

Mainwaring, M. C. & Hartley, I. R. 2012. Causes and consequences of differential growth in birds: a behavioral perspective. – Advances in the Study of Behavior 44: 22–277. DOI: 10.1016/B978-0-12-394288-3.00006-X

Mainwaring, M. C. & Hartley, I. R. 2016. Local weather conditions have complex effects on the growth of Blue Tit nestlings. – Journal of Thermal Biology 60(1): 12–19. DOI: 10.1016/j.jtherbio.2016.05.005

Mainwaring, M. C., Reynolds, S. J. & Weidinger, K. 2015. The influence of predation on the location and design of nests. – In: Deeming, D. C. & Reynolds, S. J. (eds.) Nests, eggs, and incubation: new ideas about avian reproduction. – Oxford University Press, Oxford, pp. 50–64.

Mazgajski, T. D. 2007. Nest hole age decreases nest site attractiveness for the European Starling *Sturnus vulgaris*. – Ornis Fennica 84(1): 32–38.

Maziarz, M., Broughton, R. K. & Wesołowski, T. 2017. Microclimate in tree cavities and nest-boxes: Implications for hole-nesting birds. – Forest Ecology and Management 389(1): 306–313. DOI: 10.1016/j.foreco.2017.01.001

Mennill, D. J. & Ratcliffe, L. M. 2004. Nest cavity orientation in Black-capped Chickadees *Poecile atricapillus*: do the acoustic properties of cavities influence sound reception in the nest and extra-pair matings? – Journal of Avian Biology 35(6): 477–482. DOI: 10.1111/j.0908-8857.2004.03351.x

Miller, K. E. 2010. Nest-site limitation of secondary cavity-nesting birds in even-aged southern pine forests. – Wilson Journal of Ornithology 122(1): 126–134. DOI: 10.1676/07-130.1

Mitchell, A. 1974. A Field Guide to the Trees of Britain and Northern Europe. – Collins, London

Møller, A. P., Adriaensen, F., Artemyev, A., Bańbura, J., Barba, E., Biard, C., Blondel, J., Bouslama, Z., Bouvier, J-C., Camprodon, J., Cecere, F., Chaine, A., Charmantier, A., Charter, M., Cichoń, M., Cusimano, C., Czeszewik, D., Doligez, B., Doutrelant, C., Dubiec, A., Eens, M., Eeva, T., Faivre, B., Ferns, P. N., Forsman, J. T., García-del-Ray, E., Goldshtein, A., Goodenough, A. E., Gosler, A. G., Góźdź, I., Grégoire, A., Gustafsson, L., Hartley, I. R., Heeb, P., Hinsley, S. A., Isenmann, P., Jacob, S., Järvinen, A., Juškaitis, R., Kania, W., Korpimäki, E., Krams, I., Laaksonen, T., Leclercq, B., Lehikoinen, E., Loukola, O., Lundberg, A., Mainwaring, M. C., Mänd, R., Massa, B., Mazgajski, T. D., Merino, S., Mitrus, C., Mönkkönen, M., Morales-Fernaz, J., Moreno, J., Morin, X., Nager, R. G., Nilsson, J-Å., Nilsson, S. G., Norte, A. C., Orell, M., Perret, P., Perrins, C. M., Pimentel, C. S., Pinxten, R., Priedniece, I., Quidoz, M-C., Remeš, V., Richner, H., Robles Díez, H., Russell, A., Rytkönen, S., Senar, J. C., Seppänen, J. T., Pascoal da Silva, L., Slagsvold, T., Solonen, T., Sorace, A., Stenning, M. J., Török, J., Tryjanowski, P., van Noordwijk, A. J., von Numers, M., Walankiewicz, W. & Lambrechts, M. M. 2014. Clutch size variation in Western Palearctic secondary hole-nesting passerine birds in relation to nestbox design. – Methods in Ecology and Evolution 5(4): 353–362. DOI: 10.1111/2041-210X.12160

Monti, F., Nelli, L., Catoni, C. & Dell'Omo, G. 2019. Nest box selection and reproduction of European Rollers in Central Italy: a 7-year study. – Avian Research 10(1): 13. DOI: 10.1186/s40657-019-0150-0

Mueller, A. J., Twedt, D. J. & Bowers, E. K. 2019. Rapid adoption of nest boxes by Prothonotary Warblers *(Protonotaria citrea)* in mesic deciduous forest. – Canadian Journal of Zoology 97(12): 1109–1115. DOI: 10.1139/cjz-2019-0059

Navara, K. J. & Anderson, E. M. 2011. Eastern Bluebirds choose nestboxes based on box orientation. – Southeast Naturalist 10(4): 713–720. DOI: 10.1656/058.010.0410

O'Connor, R. J. 1984. The Growth and Development of Birds. – John Wiley and Sons, London

Rodwell, J. S. 1991. British Plant Communities, Volume 1: Woodlands and Scrub. – Cambridge University Press, Cambridge

Rohrbaugh, R. W. Jr. & Yahner, R. H. 1997. Effects of macrohabitat and microhabitat on nest box use and nesting success of American Kestrels. – Wilson Bulletin 109(1): 410–423.

Sacilotto, K. A. & Anderson, J. T. 2005. Avian nest box use on islands in the Ohio River. – Northeastern Naturalist 12(4): 403–410. DOI: 10.1656/1092-6194(2005)012[0403:ANBUOI]2.0.CO;2

Salaberria, C., Celis, P., López-Rull, I. & Gill, D. 2014. Effects of temperature and nest heat exposure on nestling growth, dehydration and survival in a Mediterranean hole-nesting passerine. – Ibis 156(2): 265–275. DOI: 10.1111/ibi.12121

Samplonius, J. M. & Both, C. 2019. Climate change may affect fatal competition between two bird species. – Current Biology 29(2): 327–331. DOI: 10.1016/j.cub.2018.11.063

Smith, K. W., Smith, L., Charman, E., Briggs, K., Burgess, M., Dennis, C., Harding, M., Isherwood, C., Isherwood, I. & Mallord, J. 2011. Large-scale variation in the temporal patterns of the frass fall of defoliating caterpillars in oak woodlands in Britain: implications for nesting woodland birds. – Bird Study 58(4): 506–511. DOI: 10.1080/00063657.2011.616186

Stanback, M. T., Niemasik, E., Millican, D. & McGovern, P. 2019. Pairing nest boxes does not promote coexistence of Eastern Bluebirds *(Sialia sialis)* and subordinate cavity nesters. – Wilson Journal of Ornithology 131(2): 422–427. DOI: 10.1676/18-93

Wachob, D. G. 1996. A microclimate analysis of nest-site selection by Mountain Chickadees. – Journal of Field Ornithology 67(4): 525–533.

Wilkin, T. A., King, L. E. & Sheldon, B. C. 2009. Habitat quality, nestling diet, and provisioning behaviour in Great Tits *Parus major*. – Journal of Avian Biology 40(2): 135–145. DOI: 10.1111/j.1600-048X.2009.04362.x

Wilson, J. 2015. A nest-box and colour-ringing study of Bearded Tits *Panurus biarmicus* at Leighton Moss, Lancashire. – Ringing and Migration 30(2): 75–80. DOI: 10.1080/03078698.2015.1128247

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Breeding phenology of Common Redstart *(Phoenicurus phoenicurus)* and its reproduction biology with artificial nests in Northeastern Ukraine

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Abstract The paper describes investigations on the reproduction biology (nesting, clutching, hatching, fledglings` departure) of the Common Redstart *(Phoenicurus phoenicurus)* in artificial nest boxes (AN) in Northeastern Ukraine. There were three sites of research: Hetman NNP, NPP "Gomilshansky Forests", and RLP "Feldman Ecopark". The research was performed during the nesting period from the first week of April to the first week of July in 2015–2020. Annually, 5–8 bird counts were conducted at each site. The first complete egg clutches at Hetman NNP were observed from 08.05 to 17.05 (2015–2020) and at NPP "Gomilshansky Forests" from 02.05 to28.05 (2017–2020). Dates of the first egg laying, at various conditions, had inter-annual variability because of unstable weather conditions in May. The average parameters of nests in AN at Hetman NNP were the following: diameter of nests (D) – 124.1±6.3 mm; diameter of trays (d) – 61.5±1.7 mm; nest height (H) – 63.5±9.4 mm; depth of trays (h) – 48.6±2.7 mm; nest mass (m) – 43.7±3.8 mm. The size of complete clutches in Northeastern Ukraine was calculated when eggs were incubated. According to the average indicators, during 2015–2020, the average size of the clutch was 6.9±0.3 (5–8) eggs at Hetman NNP, 6.2±0.4 (6–8) eggs at NPP "Gomilshansky Forests" and 8.5±0.5 (8–9) eggs at RLP "Feldman Ecopark". Incubation period of *Ph. phoenicurus* lasted on average for 15–20 days.

Keywords: Common Redstart, Hetman NNP, NPP "Gomilshansky Forests", RLP "Feldman Ecopark", artificial nestboxes (AN), reproduction biology

Összefoglalás A cikk a kerti rozsdafarkú *(Phoenicurus phoenicurus)* szaporodásbiológiájával (fészkelés, kikelés, fiókák kirepülése) kapcsolatos kutatásokat ismerteti, melyeket kihelyezett mesterséges fészkek alkalmazásával végeztek Északkelet-Ukrajnában. Három helyszínen történtek vizsgálatok, a Hetman Nemzeti Természeti Parkban, a „Gomilshansky Forests" Nemzeti Természeti Parkban és a „Feldman Ökopark" Regionális Tájvédelmi Parkban. A kutatást a fészkelési időszakban, április első hetétől július első hetéig végezték 2015–2020-ban. Évente 5–8 madárszámlálást végeztek minden helyszínen. A Hetman NTP területén május 8-tól május 17-ig (2015–2020 között), a „Gomilshansky Forests" NTP területén pedig május 2-től május 28-ig (2017–2020 között) figyelték meg az első teljes fészekaljakat. Az első tojásrakás időpontja évenként változott a májusi kiszámíthatatlan időjárás miatt. A Hetman NTP területén kihelyezett mesterséges fészekodúk esetén a fészkek átlag paraméterei a következők voltak: fészekátmérő (D) – 124,1±6,3 mm; fészekcsésze átmérője (d) – 61,5±1,7 mm; fészekmagasság (H) – 63,5±9,4 mm; fészekcsésze mélysége (h) – 48,6±2,7 mm; fészek tömege (m) – 43,7±3,8 mm. A teljes fészekaljméretet Északkelet-Ukrajnában a kotlás során számították ki. Az átlagos mutatók szerint 2015 és 2020 között az átlagos fészekaljméret a Hetman NTP területén 6,9±0,3 (5–8) tojás, a „Gomilshansky Forests" NTP területén 6,2±0,4 (6–8) tojás, a „Feldman Ökopark" RTP területén pedig 8,5±0,5 (8–9) tojás volt. A kerti rozsdafarkú inkubációs periódusa átlagosan 15–20 napig tartott.

Kulcsszavak: kerti rozsdafarkú, Hetman Nemzeti Természeti Park, „Gomilshansky Forests" Nemzeti Természeti Park, „Feldman Ecopark" Regionális Tájvédelmi Park, mesterséges fészkodúk, szaporodásbiológia

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Introduction

Progressive anthropogenic impact, climate change and the subsequent transformation of natural biogeocenoses significantly alter the number of species or whole complexes and affect the restructuring of bird populations, their phenology and reproductive biology (Kuranov 2009, Arikan & Turan 2020). Birds are most sensitive to anthropogenic changes during their reproductive period. That is when they must have the most stable topical connections and unlimited opportunities for nesting and feeding (Batary *et al.* 2014, Chaplygina 2015, Shupova & Chaplygina 2016).

Insectivorous hollow-nesting birds may be involved in the restoration of transformed areas by administering artificial nests, thus, the number of invertebrates can be regulated naturally (Erdoğan *et al.* 2005). In Northeastern Ukraine, many species of hollow-nesting birds inhabit artificial nests (Chaplygina 2018). Due to its small numbers and sporadic nesting, the Common Redstart *(Phoenicurus phoenicurus)* is insufficiently and fragmentarily studied in the region (Matvienko 2009, Savynska & Chaplygina 2016, Knysh 2017, Yuzyk & Chaplygina 2018), as well as in Ukraine in general (Shkaran 2008, Stankevich-Volosyanchuk 2008, Blinkova & Shupova 2018, Kovalska & Parkhomenko 2019, Dubovyk *et al.* 2020).

The Common Redstart is a representative of the European avifauna and one of the functionally important insectivorous birds inhabiting forests of various types (Stoyan *et al.* 2007). The species is distributed throughout Europe (Gatter 2007, Milchev 2010, Martinez 2012, Willemoes 2013), Turkey (Erdoğan *et al.* 2005) and Asia (Kuranov 2009) down to Northwest China (Poludy 2018). It is included in Annex II of the Bonn and Annex II of the Bern Conventions, and in the IUCN European Red List under the category "Least Concern" (LC).

In recent decades, there has been a tendency for expansion of its nesting area (Ilyinsky 2008, Yevtushenko & Lytvynenko 2009, Pidlatiuk 2013, Podobivskyi & Kotiv 2016, Redinov 2016, Korenieva *et al*. 2019). In Crimea, particularly, the birds have been found in mountainous forests, as well as on the southern coast of the peninsula (Appak *et al*. 2013). An increase in numbers of the species is also observed in Northeastern Ukraine, which served as one of the reasons for a close investigation (Chaplygina 2018).

In artificial nests, the breeding ecology of the Common Redstart has been studied in Belarus (Abramova *et al.* 2012), Switzerland (Martinez 2012), the Czech Republic (Porkert *et al.* 2005), Finland (Veistola *et al.* 1996), Poland (Zawadzki *et al.* 2019), Asia (Koryakina 2018) and Northwestern Africa (Mullarney *et al.* 2000).

The objective of this research is to identify the peculiarities of phenology and reproduction biology of the Common Redstart in artificial nests to preserve its populations in Northeastern Ukraine. This included the investigation of the birds` arrival time frame, the peculiarities of reproductive indexes (timing of the reproductive period, construction and composition of nests, the size of the clutch, breeding success, death factors) in the recreational areas of the Nature Reserve Fund of Ukraine (Hetman NNP, NPP "Gomilshansky Forests", and RLP "Feldman Ecopark").

Material and Methods

The investigation area is located in Northeast Ukraine within the Dnieper lowland and Poltava plain *(Figure 1)*. According to physical and geographical zoning, the territory belongs to the Forest-Steppe Zone, Eastern Ukrainian Forest-Steppe Region, Kharkiv and Sumy Slope-Upland Regions.

The vegetation of the investigation area is represented by pine forests. Hetman NNP near Klementove village has *Pinus sylvestris* as the tree stand of the first tier; *Quercus robur, Tilia cordata, Prunus padus, Sambucus racemosa, Robinia pseudoacacia, Ulmus glabra* in the second tier; *Sorbus aucuparia, Corylus avellana, Acer tataricum* as the undergrowth, and *Polygonatum odoratum, Milium effusum, Convallaria majalis, Peucedanum oreoselinum, Stellaria holostea, Carex pilosa,* and *Poa angustifolia* as the lowest herbal-bushy tier. NPP "Gomilshansky Forests" near Zadonetske village has *P. sylvestris* as the dominant species of the tree stand in the first and second tiers. Its undergrowth consists of *Padus avium, C. avellana, A. tataricum, S. racemosa*, and its herbal tier contains *Festuca beskeri, Centaurea jacea, Knautia arvensis, Hypericum perforatum, Anthericum ramosum,* and *Euphorbia nicaeensis*.

"Feldman Ecopark" Regional Landscape Park is located in the zone of Maple-linden-oak forests. The tree stand of the first tier has *Q. robur* with an admixture of *T. cordata*, and *Acer platanoides* occurring singly, the second tier is usually unexpressed. *Acer campestre* occurs in the third tier. The undergrowth is of medium density with *C. avellana, A. tataricum, Euonymus europaeus, Cornus sanguinea* in its composition. The herbaceous tier is dominated by *C. pilosa, Anemone ranunculoides, P. multiflorum, Aegopodium podagraria, Asarum europaeum,* and *Scilla siberica*.

Artificial nests (AN) were of standard sizes made of plank material, with a notch diameter of 3 cm, and were placed at a height of 1–3 m from the ground. The front wall of the AN

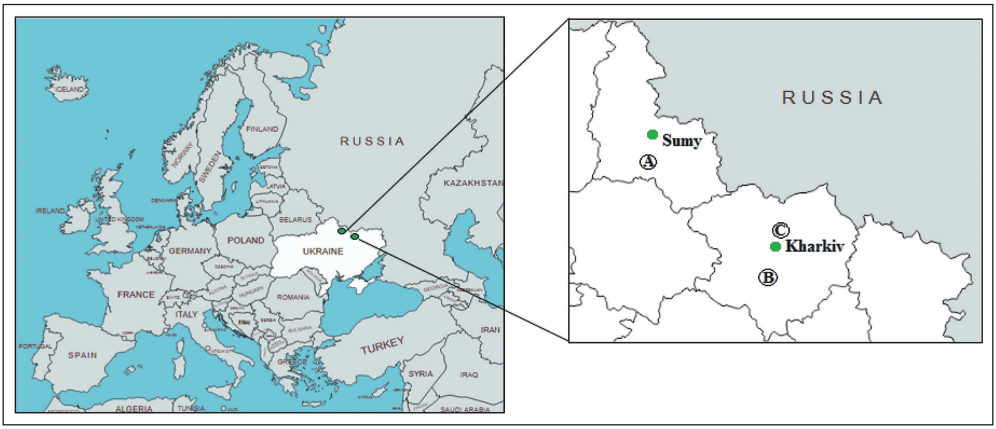

Figure 1. Map of Ukraine showing the location of the investigation areas. Regions: Hetman NNP (A), NPP "Gomilshansky Forests" (B) and Regional Landscape Park "Feldman Ecopark" (C) in Northeastern Ukraine

1. ábra Ukrajna térképe, a vizsgálati terület megjelölésével. Régiók: Hetman NNP (A), "Gomilshansky Forests" NTP (B) és "Feldman Ecopark" RTP (C) Ukrajna északkeleti részén was removable, which facilitated their inspection. Over time, ANs may lose their front wall, which increases the diversity of species of different groups of animals and birds that use them optionally.

One hundred ANs were distributed on the territories of Hetman NPP (50° 22’57” N 34° 55’34” E), NPP “Gomilshansky Forests” (49° 38’12” N 36° 18’27” E), and RLP “Feldman Ecopark” (50° 06’09” N 36° 17’00” E) each.

The research was performed during the nesting period from the first week of April to the first week of July, during 2015–2020. Annually, 5–8 bird counts were conducted at each site. A total of 43 nests of the Common Redstart were described. Egg-laying was defined in 40 cases: in 2015 in Hetman NPP (3 nests), 2017 – (3), 2018 – (4), 2019 – (4), 2020 – (9); in 2017 in NPP “Gomilshansky Forests” (3), in 2018 – (4), 2019 – (5), 2020 – (5). In the RLP “Feldman Ecopark”, nests were registered without any breeding activity. The terms of incubation of clutches are established at the laying of eggs Hetman’s NNP (17 nests), NPP “Gomilshansky Forests” – (15); at the hatching of chicks (17 nests) in Hetman NPP; and in NPP “Gomilshansky Forests” (15); and by the age of the chicks (14). The sizes of 13 clutches were defined. Reproductive success and causes of death of eggs and chicks in Hetman NNP were determined for 24 nests containing 157 eggs; in NPP “Gomilshansky Forests” there were 18 nests and 103 eggs. The sizes of eggs and nests were measured according to generally accepted methods with calipers (Kostin 1977, Hoyt 1979, Schönwetter 1979, Myand 1988). Egg volume was determined by following the formula $V = 0.51LB^2$, sphere index (roundness) – $B / L100\%$. In the above formulas, L is the length and B is the width of the egg. The arithmetic mean of eggs (M) is one of the main characteristics of the variation, which is the center of distribution around which all variants of the statistical population are grouped; Limits (Lim) – an indicator of variation, the values of the minimum (Xmin) and maximum (Xmax) options, between which are all the options of this set located. To compare the variability of the characteristics expressed by different units, we used the relative index of variation (CV) – a percentage of the standard deviation (Sx) to the arithmetic mean (M).

The success of hatching chicks was determined by the formula: $np / ne \cdot 100\%$; where ne is the number of eggs, np is the number of hatched chicks. The success of the post-embryonic nesting period was determined by the formula: $np2 / np1 \cdot 100\%$, where np1 is the number of hatched chicks, np2 is the number of feathered chicks. Concerning statistical parameters, a standard deviation is given in most cases, excluding the size of a clutch, the size of eggs and parameters of nests, where the standard error is used. Birds were caught with mist nets and ringed with standard aluminum rings of the Ukrainian Bird Ringing Center. Public weather diaries from the website www.gismeteo.com and the weather archive from the website rp5.ua were used to calculate average air temperatures over the years of research. The beginning of the arrival of migrants was determined by the date of the first encounter.

Statistical analysis of the data was performed using the software STATISTICA 12. Differences in the average values of multidimensional traits in our research were considered significant if $P \leq 0.05$. Statistical processing of the results was performed using one-way analysis of variance (differences between the mean values were considered probable at $p<0.05$). Mean values and standard deviation ($x \pm SD$) were determined. Data were verified using the Kruskal-Wallis distribution normality criterion and the Fisher F-test before using

ANOVA in Past 4.03. Pearson correlation was used to determine the variability between the laying of the first egg of *Ph. phoenicurus* with the May temperature in Northeastern Ukraine, the Pearson correlation coefficient was calculated by the formula:

$$r = \frac{\sum(x-\overline{x})(y-\overline{y})}{\sqrt{\sum(x-\overline{x})^2 \sum(y-\overline{y})^2}};$$ where x and y are variables of the compared series.

Results and Discussion

Biotope

At the conditions of Northeast Ukraine, *Ph. phoenicurus* sporadically inhabits different types of forests. The main nesting places are well-lit areas in light or moderately closed tree stands with uneven fragmentary undergrowth. Birds avoid areas with tall and thick grass. They nest in Maple-linden-oak forests, near recreation centers and camps, where their numbers reach 3–4 pairs/km^2. In mature pine forests, their density sometimes amounts up to 10–12 pairs/km^2. In the last five years, it even tends to increase to 18 pairs/km^2.

Phenology of migration and nesting

At Hetman NNP, the arrival of *Ph. phoenicurus* was as follows: 16.04.2015; 23.04.2017; 21.04.2018; 16.04.2019; 10.05.2020. The average for the 5 years is 21.04±2.8 days. The dates of arrival at NPP "Gomilshansky Forests" were 22.04.2017; 30.04.2018; 18.04.2019; 01.05.2020; on average for the 4 years – 25.04±3.1 days. At the conditions of RLP "Feldman Ecopark", arrival dates were registered on 20.04.2019 and 03.04.2020. The average date for the 2 years was 26.04±6.5 days.

When selecting nesting sites, *Ph. phoenicurus* readily chooses ANs placed at on average 2.4 m (1.5–3.0). Construction of a nest in AN in the investigated areas began in late April 20–30 The first fully built nests in Hetman NNP were found during the first, second weeks of May (03.05–11.05), in NPP "Gomilshansky Forests" – the third week of May (17.05–25.05), while in RLP "Feldman Ecopark" – in 2019 – in the third week of April (30.04); in 2020 – in the second week of May (11.05). After the construction of the nest, the females immediately began to lay their eggs, which usually lasted 15–20 days on average. The first complete clutches were laid from 08.05 to 17.05 (2015–2020) at Hetman NNP and at NPP "Gomilshansky Forests" – 02.05–28.05 (2017–2020). Dates of the beginning of the first egg laying, at various conditions, have inter-annual variability because of unstable weather conditions in May.

In the investigated areas, *Ph. phoenicurus* was registered to usually lay a second clutch. We recorded the chicks of the second brood at Hetman NNP and NPP "Gomilshansky Forests" in 2020 (05.06–19.07). During 2015–2020, at Hetman NNP, the chicks began to hatch in the fourth week of May and the first week of June (26.05–03.06), and at NPP "Gomilshansky Forests" – from the third week of May to the middle of June (19.05–15.06). Mass hatching of chicks occurred in the fourth week (26.05–29.05) of May and the second (11.06–15.06)

Table 1. Phenology of reproduction of *Phoenicurus phoenicurus* at Hetman NNP during 2015–2020. Note: n is the number of nests, μ is the average value; m is the standard error of the mean

1. táblázat A kerti rozsdafarkú *(Phoenicurus phoenicurus)* szaporodásának fenológiája a Hetman NNP-ben 2015–2020 között. Megjegyzés: n a fészkek száma, μ az átlagos érték; m az átlag standard hibája

Year	n	Nesting beginning μ ± m	First egg laying μ ± m	Chicks hatching μ ± m	Fledglings departure μ ± m
2015	3	03.05±4.3	09.05±4.3	26.05±3.2	08.06±3.2
2017	3	09.05±4.3	15.05±4.4	31.05±4.3	23.05±14.7
2018	4	02.05±0.3	08.05±0.3	26.05±0.8	08.06±0.8
2019	4	07.05±4.1	13.05±4.1	28.05±1.0	10.06±1.0
2020	9	11.05±4.7	17.05±4.7	03.06±8.4	17.06±8.2
Together	23	06.05±3.5	12.05±3.6	29.05±3.5	07.06±5.6

Table 2. Phenology of reproduction of *Phoenicurus phoenicurus* at NPP "Gomilshansky Forests" during 2017–2020 Note: n is the number of nests, μ is the average value; m is the standard error of the mea.

2. táblázat A kerti rozsdafarkú szaporodásának fenológiája a NPP "Gomilshansky Forests" területén, 2017–2020 között. Megjegyzés: „n" a fészkek száma, „μ" az átlagérték, „m" az átlag standard hibája

Year	n	Nesting beginning μ ± m	First egg laying μ ± m	Chicks hatching μ ± m	Fledglings departure μ ± m
2017	3	18.05±3.4	24.05±3.4	11.06±3.2	24.06±3.2
2018	4	25.05±9.1	02.05±7.9	19.05±8.1	02.06±8.1
2019	6	17.05±7.1	28.05±8.3	15.06±8.5	28.06±17.5
2020	5	18.05±7.0	24.05±7.2	11.06±12.5	11.06±1.2
Together	23	19.05±6.7	19.05±6.7	06.06±8.1	16.06±7.5

week of June. The last year-on-year hatching of chicks was registered in 2019 in the second week of June (15.06). The mass departure of fledglings at Hetman NNP fell on the third and fourth weeks of June (02.06–28.06) *(Table 1, 2)*. After leaving the nests, the young birds, together with the adults, stayed near their artificial nests for two weeks.

The phenology of *Ph. phoenicurus* reproduction at RLP "Feldman Ecopark" during 2019–2020 was uneven. For the whole period of research, the following were registered (two nests): nesting (06.05±6.0), laying of the first egg (15.05±2.0). During the weekly inspections of ANs, it was found that the birds had left the two clutches that were observed.

Structure of nests

By the nature of the location of nests and their distance from each other, *Ph. phoenicurus* belongs to a group of single (territorial) birds that tolerates areas with high anthropogenic influence. The choice of nesting sites is determined by various parameters of the plant group, including the species composition of the tree stand, its spatial structure and the degree of shading and humidity. These factors determine the food available for chicks, the temperature of the nest and its protection from adverse weather conditions. The birds mainly inhabited ANs, which were located in the *P. sylvestris* tree stand at Hetman NNP and NPP "Gomilshansky Forests". At RLP "Feldman Ecopark", the birds occupied ANs in the *Q. robur* tree stand. Less often, birds inhabited ANs placed on *A. platanoides* and *T. cordata*. The size of the nest depends on the size of the artificial nest and its type, but the most important requirement for the population of *Ph. phoenicurus* is a wide notch of ANs. The nest of *Ph. phoenicurus* is a dense cup, a regular hemispherical shape with thick sidewalls and a rounded recessed tray. The inside of the nest is lined up with thin dry grass of *Poaceae* (50%). The litter consists of last year's leaves (6%), tree bast (10%), needles (8%), moss (3%), *Equus* sp. (4%), fur (4%), maple lionfish (4%), feathers (6%) of various species of birds *(Parus major, Ficedula albicollis, Dendrocopos major)*. Sometimes the nests contained materials of anthropogenic origin (pieces of threads, ropes, bags) – 5%.

The average indexes of nests in ANs at Hetman NNP showed their maximum parameters to be as follows: diameter of nests (D) – 124.1±6.3 mm; diameter of trays (d) – 61.5±1.7 mm; nest height (H) – 63.5±9.4 mm; depth of trays (h) – 48.6±2.7 mm; nest mass (m) – 43.7±3.8 mm *(Table 3)*.

Over the years of research in Hetman NPP and NPP "Gomilshansky Forests", the length (L) of eggs was found not to differ significantly: from 18.6±0.1 to 19.1±0.4 (CV=0.9%), and from 18.6±0.2 to 18.9±0.16 (CV=0.9%). Diameter (D) of eggs varied from 13.5±0.05 to 14.4±0.4 (CV=2.4%) in Hetman NPP, and from 13.4±0.09 to 14.3±0.09 (CV=1.0%) in

Table 3. Nidological parameters of *Phoenicurus phoenicurus* nests in ANs in Northeastern Ukraine during 2015–2020. Note: μ is the arithmetic mean of the value, m is the error of the mean, Lim is the minimum and maximum value of the sign, σ is the standard deviation, and CV is the coefficient of variation

3. táblázat A kerti rozsdafarkú fészkek paraméterei Északkelet-Ukrajna mesterségesen telepített fészkelőhelyein, 2015–2020 között. Megjegyzés: μ az érték számtani közepe, m az átlag hibája, Lim az előjel minimális és maximális értéke, σ a szórás, CV a variációs együttható

Parameters	**Hetman National Nature Park (n=6)**			**National Nature Park "Gomilshansky Forests" (n=6)**			**Landscape Park "Feldman Ecopark" (n=2)**		
	$\frac{\mu \pm m}{Lim}$	σ	CV, %	$\frac{\mu \pm m}{Lim}$	σ	CV, %	$\frac{\mu \pm m}{Lim}$	σ	CV, %
Nest diameter (D)	$\frac{124.1 \pm 6.3}{101.2 - 141.3}, mm$	14.0	12.05	$\frac{116.1 \pm 2.2}{109.0 - 123.0}, mm$	4.8	4.2	$\frac{116.3 \pm 2.9}{115.6 - 119.2}, mm$	3.0	2.7
Tray diameter (d)	$\frac{61.5 \pm 1.7}{57.7 - 68.2}, mm$	3.8	6.3	$\frac{60.2 \pm 7.7}{56.5 - 88.05}, mm$	17.4	30.0	$\frac{1.0 \pm 0.9}{58.0 - 59.1}, mm$	1.0	1.5
Nest height (H)	$\frac{63.5 \pm 9.4}{46 - 105}, mm$	21.0	33.0	$\frac{59.2 \pm 6.6}{48 - 92}, mm$	14.3	25.0	$\frac{55.2 \pm 0.2}{55.1 - 55.3}, mm$	0.1	0.3
Tray depth (h)	$\frac{48.6 \pm 2.7}{41.5 - 60.0}, mm$	6.0	12.4	$\frac{42.1 \pm 1.7}{36.0 - 46.6}, mm$	3.8	9.1	$\frac{42.2 \pm 2.0}{42.1 - 44.1}, mm$	2.0	4.6
Nest mass (m)	$\frac{43.7 \pm 3.8}{37.7 - 56.4}, g$	8.5	18.7	$\frac{26.1 \pm 2.8}{13.0 - 32.2}, g$	6.3	24.2	$\frac{30.3 \pm 0.8}{29.5 - 31.1}, g$	0.8	2.6

NPP "Gomilshansky Forests". There was no significant difference in volume of eggs (V) between the territories. In Hungary, the size of 68 eggs varied the length from 17.0 to 20.5 mm; the diameter from 12.9 to 14.6 mm, average is 18.9x13.98 mm (Solti 2010). The annual difference in Sphere index (Sph) of eggs ranged at Hetman NPP from 71.2±0.62 to 76.5±4.8 (CV=2.7%), and at NPP "Gomilshansky Forests" from 71.2±0.62 to 75.7±0.68 (CV=0.6%). The Sphere index (Sph) during 2015–2020 indicates that the morphometric parameters of *Ph. phoenicurus* eggs do not show significant differences. However, compared to Sph, calculations of the Elongation index (Iel) prove otherwise. At Hetman NPP, the indexes vary from 33.1±0.7 to 40.4±1.19 (CV=8.3%); in case of NPP "Gomilshansky Forests", from 32.3±1.18 to 33.8±1.3 (CV=2.2%). In our opinion, such variations can lead to the deterioration of the egg`s incubation properties in general. E. Peebles (2004) proved that the Elongation index (Iel) could significantly affect the position of the embryo during the development, which ultimately determines the effectiveness of its survival. Among the measurements we have taken, the most distinctive and important parameter is the weight of the egg. According to our data, at Hetman NPP, the average indexes vary from 1.7±0.02 to 1.9±0.02 (CV=5.0%), and at NPP "Gomilshansky Forests", from 1.9±0.04 to 1.9±0.05 (CV = 0.1%).

The return of *Ph. phoenicurus* to the nesting areas in Northeastern Ukraine, on average, falls to 24.04±1.5 days (2015–2020). Other reports of first arrival in Ukraine include: in the vicinity of Lake Pisochne Shatsky NNP in 2004 (06.04), 2005 (09.04), 2007 (14.04) (Shkaran 2008); at the ponds of Stanychno-Luhansk fish farm during 1986–2007 (15.04–21.04) (Yevtushenko & Lytvynenko 2009); in 21.04.1997 in the Alexandria district of Kirovograd region (Shevtsov 2008); in Southern part of Rivne region in 02.04.2012, 04.04.2014 (Ilchuk 2015); in the Nikolaev area on 23.04.1997, 25.04.2000, 22.04.2003 (Redinov 2016). In the

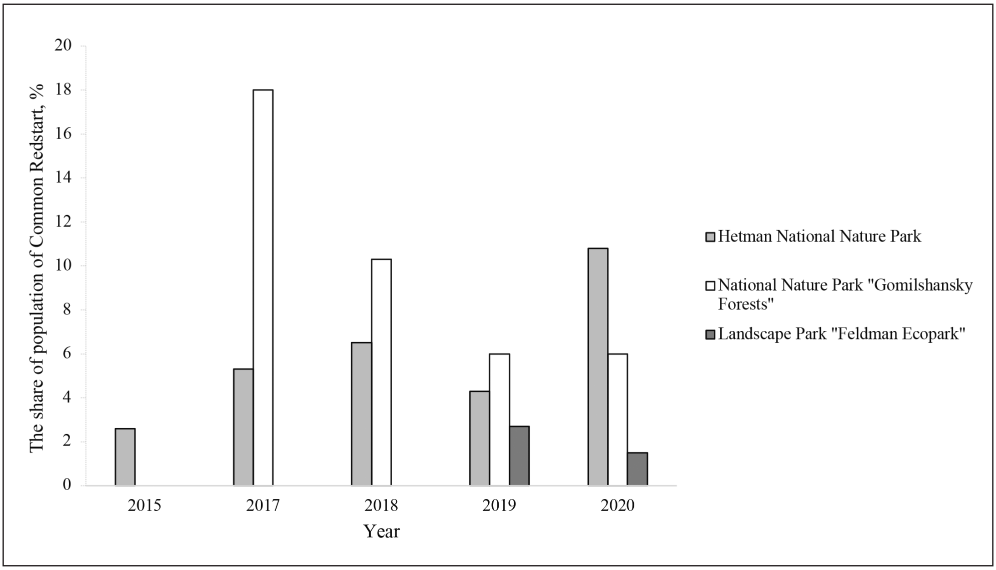

Figure 2. Population of *Phoenicurus phoenicurus* in ANs in Northeastern Ukraine during 2015–2020
2. ábra A kerti rozsdafarkú populációja Ukrajna északkeleti részén telepített mesterséges fészkelőhelyein, 2015–2020 között

Sumy region, for the last 50 years *Ph. phoenicurus* arrived from 14.04.1967 to 07.05.1983, with the average for 7 years: 27.04 (Knysh 2017).

The average populations in ANs at Hetman NPP (during 2015–2020) varied from 2.6% to 10.8%, at NPP "Gomilshansky Forests" (2017–2020) from 18.0% to 6.0%, and at the RLP "Feldman Ecopark" (2019–2020) from 2.7% to 1.5% *(Figure 2)*. In the Shatsk NNP in the biotope of *P. sylvestris* monoculture, the presence of Common Redstart population in 2005 was 6.2% (Shkaran 2009).

The obtained data of interannual population of *Ph. phoenicurus* in ANs at different conditions confirms its growth in the territory of Hetman NNP. The decrease of populations in ANs at NPP "Gomilshansky Forests" can be explained by the influence of predation by *Martes martes* and competition with *Bombus* spp. and *Vespa* spp. In RLP "Feldman Ecopark", the population of *Ph. phoenicurus* remains low due to the small Redstart numbers in the urban landscape.

Incubation period of *Ph. phoenicurus* lasts on average for 15–20 days. The first clutches at Hetman NNP can be seen between 08.05–17.05 and at NPP "Gomilshansky Forests" between 02.05–28.05. The earliest clutches in Central Europe are laid, on average, on 06.05

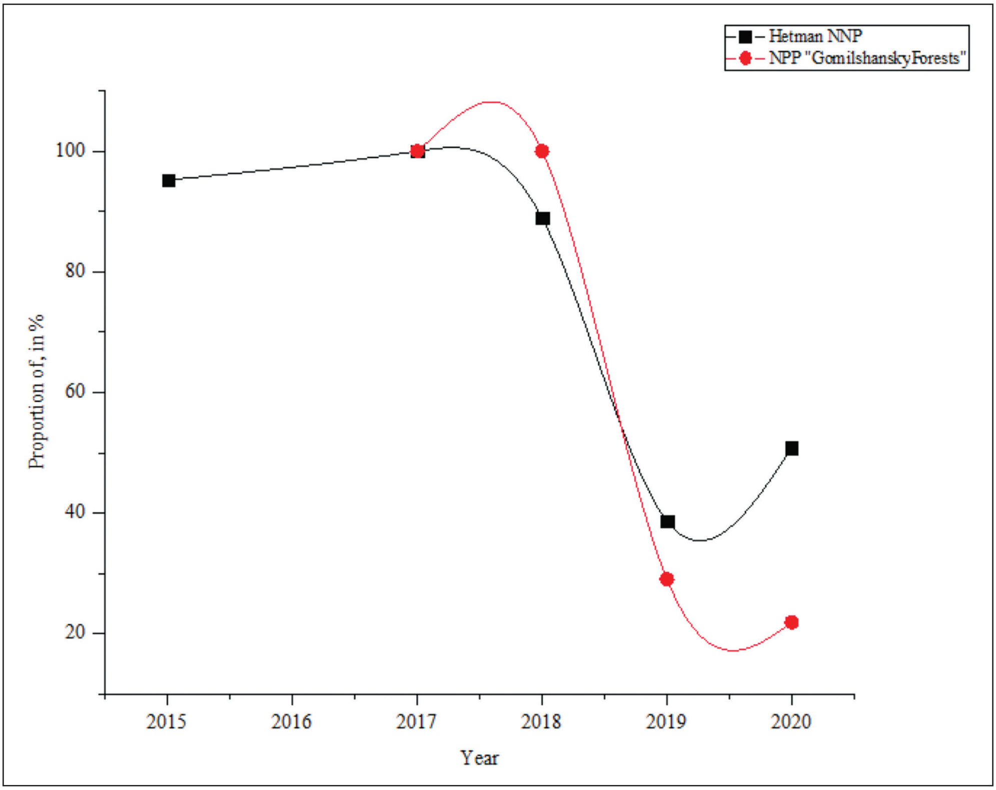

Figure 3. Dependence of the clutch size of *Phoenicurus phoenicurus* on temperature in Northeastern Ukraine during 2015–2020.

3. ábra A kerti rozsdafarkú fészekalj méretének alakulása Északkelet-Ukrajna hőmérsékletének függvényében, 2015–2020 között

Table 4. Indexes of average clutch size of *Phoenicurus phoenicurus* in Northeastern Ukraine during 2015–2020. Note. µ Arithmetic mean of the value, m error of the mean

4. táblázat A kerti rozsdafarkú átlagos fészekalj-méretei Ukrajna északkeleti részén, 2015–2020 között. Megjegyzés: µ Az érték számtani közepe, az átlag m hibája

Year	**Hetman National Nature Park (n=23 clutches)**	**National Nature Park “Gomilshansky Forests” (n=18 clutches)**	**Landscape Park “Feldman Ecopark” (n=2 clutches)**
	Size of egg clutch		
	µ±m	**µ±m**	**µ±m**
2015	7.0±0.1	–	–
2017	5.7±0.7	6.3±0.3	–
2018	6.7±0.3	5.3±0.3	–
2019	8.0±0.1	6.7±0.3	8.5±0.5
2020	7.3±0.7	6.7±0.9	

at 14.9 °C. In Finland, early egg-laying begins on 11.05–15.05, at an average temperature of 9.5 °C in the South, indicating that egg-laying is related to habitat interaction and spring weather, rather than just one of the investigated factors (Porkert *et al.* 2005). Data in the Utsjoki Valley (80–200 m above sea level) in Finland during 1982–1994 (Veistola *et al.* 1996) show the onset of clutching in ANs in mixed deciduous and coniferous forests on average on 04.06 (SD=6.89, n=136). Analysis of long-term data in Northeastern Ukraine shows a gradual increase in air temperature in May against the background of its strong year-on-year fluctuations of 19.5±1.14 ($P \leq 0.05$) *(Figure 3)*.

We found a significant correlation between the beginning of the first egg-laying of *Ph. phoenicurus* and the May temperature in Northeastern Ukraine ($r=0.051$, $p \leq 0.01$). With a slight increase in air temperature, egg-laying in 2019 increased at RLP “Feldman Ecopark” by 8.5±0.5, at Hetman NPP by 6.9±0.3 and at NPP “Gomilshansky Forests” by 6.7±0.3, which was similar to the average of 2020 (6.7±0.9).

In pine forests of the Northeast of Czech Republic and Central Europe, the average size of clutches in the “first” broods was 6.3 eggs (SD=0.88, n=100 nests), and in the “second” it was 5.4 eggs (SD=0.94, n=72 nests), and the difference was significant (Porkert 2005). The number of eggs in the clutch ranged from three to eight. The most common size of the first clutches was seven eggs that of the second was five. The size of the first clutches during 2011–2013 in Switzerland was 6.3±0.3 eggs, and that of the second was 5.0±0.3 eggs (Martinez & Zingg 2014). Studies in North Karelia recorded the laying of 6–7 eggs (µ=6.66, SD=0.95) (Avil *et al.* 2005). In the South-east of Western Siberia at the border between the southern taiga and subtaiga, in the central areas of Tomsk suburbs (1986–1990) and Seversk (1995–2007), Kuranov (2009) determined the average size of the first clutches to be 7.3±0.08 (CV=11.6±0.8), and that of the second ones to be 6.9±0.08 (CV=13.8±0.8). In the Hungarian oological collection are all together 52 cluch with 5.8 eggs/nest (6x4, 15x5, 14x6, 15x7 and 2x8 eggs) (Haraszthy 2019).

During 2015–2020 at Hetman NNP, the chicks began to hatch in the third decade of May and the first decade of June (26.05–03.06); in different years at NPP “Gomilshansky

Table 5. Parameters of *Phoenicurus phoenicurus* eggs in Northeastern Ukraine during 2018–2020. Note: μ is the arithmetic mean of the value, m is the error of the mean, Lim is the minimum and maximum value of the sign, σ is the standard deviation, and CV is the coefficient of variation

5. táblázat A kerti rozsdafarkú tojások paraméterei Északkelet-Ukrajnában 2018–2020 között. Megjegyzés: μ az érték számtani közepe, m az átlag hibája, Lim az előjel minimális és maximális értéke, σ a szórás, CV pedig a variációs együttható

Parameter	**Hetman National Nature Park**														
	2015			**2017**			**2018**			**2019**			**2020**		
	$\frac{\mu \pm m}{Lim}$	**σ**	**CV, %**	$\frac{\mu \pm m}{Lim}$	**σ**	**CV, %**	$\frac{\mu \pm m}{Lim}$	**σ**	**CV, %**	$\frac{\mu \pm m}{Lim}$	**σ**	**CV, %**	$\frac{\mu \pm m}{Lim}$	**σ**	**CV, %**
Mass, g	–	–	–	$\frac{1.7 \pm 0.02}{1.6 - 1.7}$	0.04	2.4	$\frac{1.7 \pm 0.3}{1.6 - 1.9}$	0.1	0.1	$\frac{1.9 \pm 0.02}{1.8 - 2.0}$	0.1	3.2	$\frac{1.7 \pm 0.02}{1.7 - 1.8}$	0.1	3.2
Length (L), mm	$\frac{19.0 \pm 0.26}{18.15 - 20}$	0.6	3.3	$\frac{18.9 \pm 0.14}{18.5 - 19.2}$	0.3	1.5	$\frac{18.9 \pm 0.13}{18.45 - 19.4}$	0.3	1.6	$\frac{18.6 \pm 0.1}{18.3 - 19.0}$	0.2	1.3	$\frac{19.1 \pm 0.4}{17.1 - 19.8}$	0.8	4.3
Diameter (D), mm	$\frac{14.1 \pm 0.15}{13.65 - 14.8}$	0.4	2.7	$\frac{13.5 \pm 0.05}{13.3 - 13.6}$	0.1	0.8	$\frac{13.8 \pm 0.1}{13.5 - 14.3}$	0.3	1.9	$\frac{14.0 \pm 0.1}{13.7 - 14.3}$	0.2	1.6	$\frac{14.4 \pm 0.4}{14.0 - 16.2}$	0.7	4.9
Sphere index (Sph), %	$\frac{74.2 \pm 1.04}{70.4 - 78.8}$	2.6	3.4	$\frac{71.2 \pm 0.62}{70.2 - 73.5}$	1.2	1.7	$\frac{73.3 \pm 0.52}{70.8 - 75.6}$	1.5	2.1	$\frac{75.3 \pm 0.4}{73.6 - 76.7}$	1.0	1.4	$\frac{76.5 \pm 4.8}{70.7 - 99.1}$	9.0	11.4
Volume index (V), mm^3	$\frac{19.4 \pm 0.57}{17.7 - 21.8}$	1.4	7.2	$\frac{17.5 \pm 0.19}{16.8 - 17.8}$	0.4	2.1	$\frac{18.4 \pm 0.35}{17.3 - 19.9}$	0.8	4.2	$\frac{18.6 \pm 0.3}{17.6 - 19.9}$	0.8	4.1	$\frac{20.1 \pm 0.5}{19.4 - 22.5}$	1.0	5
Elongation index (Iel)	$\frac{34.8 \pm 1.87}{26.9 - 42.1}$	4.6	13.2	$\frac{40.4 \pm 1.19}{36.0 - 42.5}$	2.4	5.9	$\frac{36.6 \pm 0.94}{32.4 - 41.5}$	2.9	7.8	$\frac{33.1 \pm 0.7}{30.5 - 36.3}$	1.9	5.6	$\frac{33.6 \pm 4.8}{10.7 - 41.9}$	8.5	28.6

Parameter	National Nature Park “Gomilshansky Forests”									Landscape Park “Feldman Ecopark”		
	2018			2019			2020			2019		
	$\frac{\mu \pm m}{Lim}$	σ	CV, %	$\frac{\mu \pm m}{Lim}$	σ	CV, %	$\frac{\mu \pm m}{Lim}$	σ	CV, %	$\frac{\mu \pm m}{Lim}$	σ	CV, %
Mass, g	$\frac{1.9 \pm 0.04}{1.4 - 2.0}$	0.1	3.7	$\frac{1.9 \pm 0.05}{1.8 - 2.1}$	0.1	5.3	–	–	–	–	–	–
Length (L), mm	$\frac{18.9 \pm 0.12}{14.0 - 19.3}$	0.2	1.3	$\frac{18.6 \pm 0.2}{18.1 - 19.2}$	0.4	1.9	$\frac{18.9 \pm 0.1}{18.3 - 19.2}$	0.3	1.4	$\frac{18.9 \pm 0.16}{18.3 - 19.7}$	0.4	2.2
Diameter (D), mm	$\frac{14.3 \pm 0.09}{10.6 - 14.5}$	0.2	1.3	$\frac{14.0 \pm 0.1}{13.7 - 14.2}$	0.2	1.4	$\frac{14.1 \pm 0.1}{13.8 - 14.4}$	0.2	1.7	$\frac{13.4 \pm 0.09}{13.1 - 13.9}$	0.2	1.7
Sphere index (Sph), %	$\frac{75.7 \pm 0.68}{55.3 - 77.7}$	1.4	1.8	$\frac{75.2 \pm 0.4}{73.6 - 76.5}$	1.0	1.3	$\frac{74.8 \pm 0.7}{72.5 - 76.9}$	1.6	2.1	$\frac{71.0 \pm 0.62}{67.5 - 72.7}$	1.6	2.3
Volume index (V), mm^3	$\frac{19.8 \pm 0.29}{14.3 - 20.7}$	0.6	2.8	$\frac{18.7 \pm 0.4}{17.3 - 19.6}$	0.8	4.2	$\frac{19.2 \pm 0.3}{18.4 - 20.1}$	0.6	3.6	$\frac{17.3 \pm 0.3}{16.3 - 18.9}$	0.8	4.6
Elongation index (Iel)	$\frac{32.3 \pm 1.18}{21.6 - 35.7}$	2.4	7.5	$\frac{33.0 \pm 0.8}{30.7 - 36.0}$	1.8	5.4	$\frac{33.8 \pm 1.3}{30.1 - 38.0}$	2.8	8.2	$\frac{40.8 \pm 1.3}{37.6 - 48.1}$	3.3	8.1

Forests” it fell to the second decade of May and June (19.05–15.06). Mass-hatching of chicks in Northeastern Ukraine occurred in the third (26.05–29.05) decade of May and the second (11.06–15.06) decade of June. The last year-on-year hatching of chicks was registered in 2019 in the second decade of June (15.06) in Finland (Veistola *et al.* 1996) on 24.06 (SD=6.63, n=124).

The mass departure of fledglings in Hetman NNP occurred in the second and third decades of June (June 2–June 28). We did not investigate the second broods by the same couples in detail due to the destruction of the nests during the egg-laying stage. Porkert *et al.* (2005) generally believe that further hatching of broods in one season, together with smaller clutches, can pose a threat in case of short daylight periods and high nest predation. Interpretations of the large number of second broods in Central Europe remain unclear and cannot be easily explained by different weather conditions or different estimation methods. Accordingly, data on the survival of second broods are needed for a more detailed analysis of the reproductive strategy of *Ph. phoenicurus*.

Based on these studies, we determined the success and reproductive productivity of *Ph. phoenicurus*. During 2015–2020, this species had high reproduction success rates *(Figure 4*, *Table 6)*.

We found that the overall success rate of reproduction at Hetman NNP varied from 95.2% (2015) to 50.8% (2020) with an average for the entire period of 74.2%. At NPP “Gomilshansky

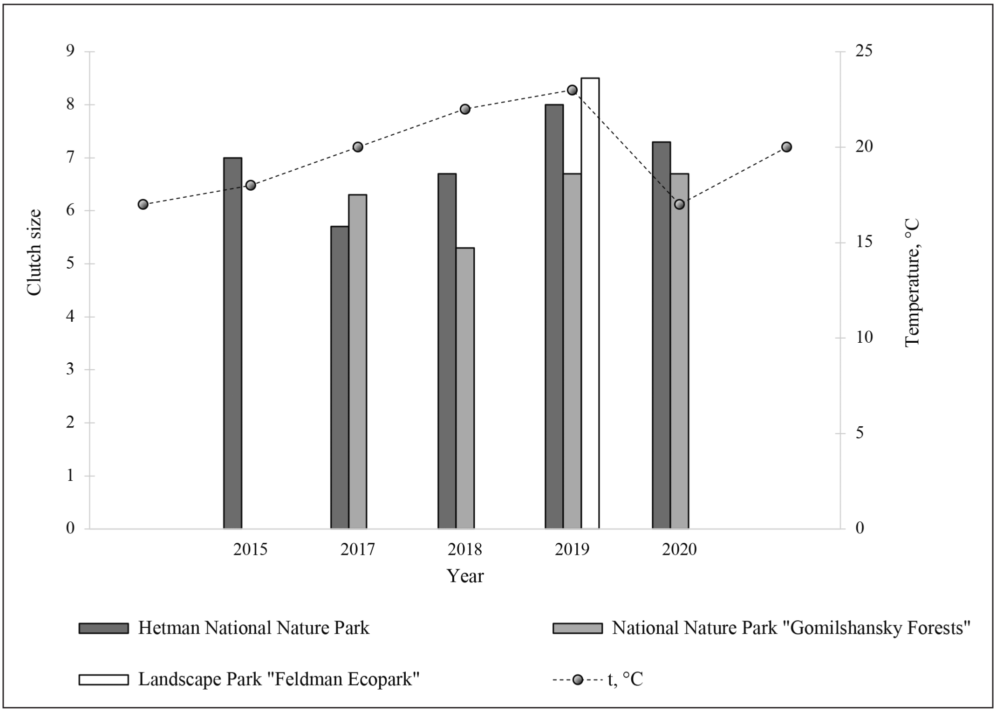

Figure 4. Success of reproduction of *Phoenicurus phoenicurus* in Northeastern Ukraine during 2015–2020.

4. ábra A kerti rozsdafarkú költési sikere Északkelet–Ukrajnában, 2015–2020 között

Table 6. Reproductive productivity of *Phoenicurus phoenicurus* at Hetman NNP during 2015–2020.
6. táblázat A kerti rozsdafarkú költési sikeressége a Hetman NTP területén, 2015–2020 között

							Newly born	Fledglings
Year	**n nests**	**n eggs**	**Quantity of dead eggs**	**n hatched chicks**	**n fledglings**	**Quantity of dead chicks**	**On average per one couple**	**On average per one couple**
2015	4	21	1	20	20	0	5.0	5.0
2017	3	17	0	17	17	0	5.7	5.7
2018	4	27	3	27	27	0	6.8	6.8
2019	4	31	19	31	31	0	7.8	7.8
2020	9	61	30	61	61	0	6.8	6.8

Forests”, the lowest success rate was registered at 21.9% (2020), the highest one was in the first year of the survey was 100% (2017), and the average for the entire period was 62.7%.

To establish the interannual reproductive productivity, we studied (n=24) nests. The maximum number of eggs was registered in 2020 (n=61), whereas the lowest numbere in 2017 (n=17). The main reasons for the low egg numbers in Hetman NNP are unfertilized eggs, embryonic mortality and destruction of clutches by predators (*Dryomys nitedula* and *Dendrocopos major*).

Table 7. Reproductive productivity of *Phoenicurus phoenicurus* at NPP “Gomilshansky Forests” during 2017–2020.
7. táblázat A kerti rozsdafarkú költési sikeressége a “Gomilshansky Forests” NTP területén, 2017–2020 között

							Newly born	Fledglings
Year	**n nests**	**n eggs**	**Quantity of dead eggs**	**n hatched chicks**	**n fledglings**	**Quantity of dead chicks**	**On average per one couple**	**On average per one couple**
2017	3	19	0	19	19	0	6.3	6.3
2018	4	21	0	21	21	0	5.3	5.3
2019	6	31	8	23	9	14	3.8	1.5
2020	5	32	16	16	7	9	3.2	1.4

At NNP "Gomilshansky forests", the largest share of the total success of reproduction was determined during 2017–2018 (100%) and the lowest in 2020 (21.9%) *(Table 7)*.

Nests (n=18) were studied in the pine forest of NNP "Gomilshansky Forests". The number of dead eggs in 2019 were n=8, in 2020 n=16. The cause of death of chicks at NPP "Gomilshansky Forests" was mostly predation by *Martes martes* (n=23 chicks). The average number of chicks per couple during 2017–2020 was 3.6±1.2 (CV=60.8%). The main positive factor in the success of *Ph. phoenicurus* reproduction in Hetman NNP in comparison with NNP "Gomilshansky Forests" were the weakening of predation pressure and the reduction of the share of abandoned broods.

References

Abramova, I. V. & Gajduk, V. E. 2012. E`kologiya oby`knovennoj gorikhvostki *Phoenicurus phoenicurus* L. (Turdidae, Passeriformes) v yugo-zapadnoj Belarusi [Ecology of the Common Redstart *Phoenicurus phoenicurus* L. (Turdidae, Passeriformes) in Southwestern Belarus]. – Vesnik Bresczkaga universiteta. Seriya 5. Khimiya. Biyalogiya. Nauki ab zyamli 1: 1–5. (in Russian)

Appak, B. A. & Tsvelykh, A. N. 2013. Seasonal population dynamics of the Common Redstart, *Phoenicurus phoenicurus* (L.), in Mountain Crimea: Contributions of local and migrating populations. – Russian Journal of Ecology 44(4): 345–349.

Arikan, K. & Turan, S. L. 2020. Organochlorine pesticide residues in feathers of four bird species from western part of Turkey. – Turkish Journal of Zoology 44: 401–407. DOI: 10.3906/zoo-2005-52

Avil, J. M., Rutila, J. & Møller, A. P. 2005. Should the Redstart *Phoenicurus phoenicurus* accept or reject Cuckoo *Cuculus canorus* eggs? – Behavioral Ecology Sociobiology 58: 608–617. DOI: 10.1007/s00265-005-0941-7

Batáry, P., Fronczek, S., Normann, C., Scherber, C. & Tscharntke, T. 2014. How do edge effect and tree species diversity change bird diversity and avian nest survival in Germany's largest deciduous forest? – Forest Ecology and Management 319: 44–50. DOI: 10.1016/j.foreco.2014.02.004

Blinkova, O. & Shupova, T. 2018. Bird communities and vegetation composition in natural and semi-natural forests of megalopolis: correlations and comparisons of diversity indices (Kyiv city, Ukraine). – Ekológia (Bratislava) 37(3): 259–288. DOI: 10.2478/eko-2018-0021

Chaplygina, A. B. 2015. Ekoloho-faunistychnyi analiz i uspishnist rozmnozhennia dendrofilnykh ptakhiv na transformovanykh terytoriiakh Pivnichno-Skhidnoi Ukrainy [Ecological-faunistic analysis and reproduction success of dendrophylic birds in transformed territories of Northeastern Ukraine]. – Biologichni Studii 9(2): 133–146. DOI: 10.30970/sbi.0902.410 (in Ukrainian)

Chaplygina, A. B. 2018. Dendrofilni horobtsepodibni (Passeriformes) yak strukturno-funktsionalnyi element antropohenno transformovanykh lisovykh bioheotsenoziv Pivnichno-Skhidnoi Ukrainy [Dendrophilous sparrows (Passeriformes) as a structural and functional element of anthropogenically transformed forest biogeocenoses of North-Eastern Ukraine]. – Avtoreferat dysertatsii na zdobuttia naukovoho stupenia doktora biol. nauk 03.00.16. Dnipro (in Ukrainian)

Dubovyk, O., Kuzyo, H. & Bokotey, A. 2020. Density variation in "rare" breeding birds in native forests and urban parks. – Geo & Bio 19: 20–31. DOI: 10.15407 / gb1904

Erdoğan, A., Kaçar, M. S., Turan, L. S., Kiziroğlu, İ. & Öz, M. 2005. Forest pest control by Common Redstart (*Phoenicurus phoenicurus* L.) near Antalya, Turkey. – Journal of Pest Science 78: 243–245. DOI: 10.1007 / s10340-005-0102-9

Gatter, W. 2007. Bestandsentwicklung des Gartenrotschwanzes *Phoenicurus phoenicurus* in Wäldern Baden-Württembergs [Population development of the Common Redstart *Phoenicurus phoenicurus* in the forests of Baden-Württemberg]. – Ornithologischer Anzeiger 46: 19–36. (in German)

Haraszthy, L. 2019. Kerti rozsdafarkú *Phoenicurus phoenicurus* (Linnaeus, 1958) [Common Redstart]. – In: Haraszthy, L. (ed.) Magyarország fészkelő madarainak költésbiológiája. 2. kötet Sárgarigóféléktől a sármányfélékig (Passeriformes) [Breeding Biology of the Hungarian Birds, Vol. 2., Passeriformes]. – Pro Vértes Nonprofit Zrt., Csákvár 505–509. (in Hungarian)

Hoyt, D. F. 1979. Practical methods of estimating volume and fresh of bird eggs. – Ornithology 96(1): 73–77. DOI: 10.1093/auk/96.1.73

Ilchuk, V. P. 2015. Materialy po fenolohii mihratsii ptakhiv u pivdenii chastyni Rivnenskoi oblasti [Materials on phenology of bird migrations in the southern part of Rivne region (West Ukraine)]. – Avifauna of Ukraine 6: 66–72. (in Ukrainian)

Ilyinsky, S. V. 2008. Suchasnyi stan ornitofauny m. Khmelnytskyi [Current state of ornithofauna in Khmelnytskyi city]. – Mizhvidomchyi naukovyi tematychnyi zbirnyk «Ptakhivnytstvo» 61: 10. (in Ukrainian)

Knysh, N. P. 2017. Zametki o redkikh i maloizuchenny`kh pticzakh lesostepnoj chasti Sumskoj oblasti [Notes about rare and small bird species of Forest-Steppe part of Sumy state]. – Russkij Ornitologicheskij Zhurnal 26(1410): 754–779. (in Russian)

Korenieva, Zh., Hunich, V., Holovanova, A., Verkhova, M. & Dudkina, V. 2019. Monitorinh ornitofauny Odeskoho rehionu v suchasnykh ekolohichnykh umovakh [Ornithofauna monitoring in Odessa region at the current ecological conditions]. – Agrarian Bulletin of the Black Sea Littoral 95: 19. (in Ukrainian)

Koryakina, T. N. 2018. Features gastostomy and characteristics of nests of the Great Tit (*Parus major* L.) and the Common Redstart (*Phoenicurus phoenicurus* L.) in the urbanized landscape of the city of Monchegorsk. – V Mire Naučnyh Otkrytij 10(1): 2307–9428.

Kostin, Yu. V. 1977. O metodike oomorfologicheskikh issledovanij i unifikaczii opisaniya oologicheskikh materialov [The method of oomorphological investigation and description of oomorphological matherials]. – Metodiki issledovaniya produktivnosti i struktury` vidov pticz v predelakh ikh arealov. Vil`nyus: Mokslas, pp. 14–22. (in Russian)

Kovalska, A. M. & Parkhomenko, O. V. 2019. Vydove riznomanittia ptakhiv riadu Horobtsepodibni (Passeriformes) masyvu Obolon mista Kyieva [Species diversity of Passeriformes birds in Obolon district of Kiev city]. – Aktualni problemy rozvytku nauky v konteksti hlobalnykh transformatsii informatsiinoho suspilstva: Materialy II Mizhnarodnoi naukovo-praktychnoi konferentsii (m. Kyiv, 25–26 zhovtnia 2019 r.), pp. 121–124 (in Ukrainian)

Kuranov, B. D. 2009. Nest biology of urban populations of cavity-nesting birds. – Contemporary Problems of Ecology 2(3): 240–247. DOI: 10.1134/S1995425509030138

Martinez, N. 2012. Sparse vegetation predicts clutch size in Common Redstarts *Phoenicurus phoenicurus*. – Bird Study 59: 315–319. DOI: 10.1080/00063657.2012.672949

Martinez, N. & Zingg, S. 2014. Zweitbruten beim Gartenrotschwanz *Phoenicurus phoenicurus* in der Schweiz [Second broods of the Common Redstart *Phoenicurus phoenicurus* in Switzerland]. – Ornithologischer Beobachter 111(3): 239–246. (in German)

Matvienko, M. E. 2009. Ocherki rasprostraneniya i e`kologii pticz Sumskoj oblasti [Study of distribution and ecology of birds in Sumy state]. – Sumy`: Universitetskaya kniga (in Russian)

Milchev, B. 2010. Expansion of the breeding distribution of Common Redstart *Phoenicurus phoenicurus* in SE Bulgaria. – Vogelwelt 131: 257–259.

Myand, R. 1988. Vnutripopulyaczionnaya izmenchivost` ptich`ikh yaicz [Intrapopulational variability of birds eggs]. – Tallin: Izd-vo Valgus (in Russian)

Mullarney, K., Svensson, L., Zetterström, D. & Grant, P. J. 2000. Birds of Europe. – United States: Princeton University Press

Pidlatiuk, O. Yu. 2013. Vydovyi sklad ornitofauny okolytsi sela Smorodshchyny Chutivskoho raionu Poltavskoi oblasti [Species composition of ornithofauna in Smorodshchina village of Poltava state, Chutiv district]. – Bioriznomanittia Ukrainy v svitli noosfernoi kontseptsii akademika V. I. Vernadskoho. Materialy Vseukrainskoi naukovo-praktychnoi konferentsii / Za zahalnoiu redaktsiieiu prof. M. V. Hrynovoi, Poltava, Astraia, pp. 64–68. (in Ukrainian)

Podobivskyi, S. S. & Kotiv, A. V. 2016. Poperedni dani pro vydovyi sklad ta osoblyvosti biolohii rozmnozhennia synantropnykh ptakhiv riadu Horobtsepodibni (Passeriformes). – Ternopolia [Preliminary data about species composition and reproduction biology particularities of Passeriformes synanthropic birds in Ternopil city]. – Biological Systems 8(2): 246–249. (in Ukrainian)

Poluda, A. M. 2018. Entsyklopediia mihruiuchykh vydiv dykykh tvaryn Ukrainy [Encyclopedia of Migratory Species of Wild Animals of Ukraine]. – Kyiv, pp. 581–582. (in Ukrainian)

Porkert, J. & Zajíc, J. 2005. The breeding biology of the Common Redstart, *Phoenicurus phoenicurus*, in the Central European pine forest. – Folia Zoologica 54(1–2): 111–122.

Redinov, K. O. 2016. Materialy do fenolohii mihratsii ptakhiv na zakhodi Mykolaivskoi oblasti [Data of birds migration phenology in western part of Mykholayiv state]. – Avifauna Ukrainy 7: 69–77. (in Ukrainian)

Savinskaya, N. A. & Chapligina, A. B. 2010. Sovremennoe sostoyanie Muholovok rodov *Muscicapa*, *Ficedula* na Severo-Vostoke Ukrainy [Current state of Flycatchers of the genera *Muscicapa*, *Ficedula* in the North-East of Ukraine]. – Vidovye populyacii i soobshchestva v estestvennyh i antropogenno transformirovannyh landshaftah: sostoyanie i metody ego diagnostiki: materialy XI. Mezhdunar. nauch.-prakt. ekologicheskoj konf. (20–25 sentyabrya). Belgorod, 128–129. (Osobistij vnesok: zbirannya j opracyuvannya materialu) (in Ukrainian)

Schönwetter, M. 1979. Handbuch der Oologie [Handbook of Oology]. – Academic Verlag, Berlin, pp. 385–448. DOI: 10.5962/bhl.title.61353 (in German)

Shevtsov, A. O. 2008. Fenolohiia vesnianoi mihratsii ptakhiv v Oleksandriiskomu raioni Kirovohradskoi oblasti [Phenology of spring migration of birds in Alexandria district of Kirovograd region]. – Avifauna Ukrainy 4: 94–100. (in Ukrainian)

Shkaran, V. I. 2008. Fenolohiia vesnianykh pryrodnykh yavyshch v okolytsiakh ozera Pisochne Shatskoho natsionalnoho pryrodnoho parku [Phenology of spring nature phenomena of Pisochne lake whereabouts in Shatskyi NPP]. – Stan i bioriznomanittia ekosystem Shatskoho natsionalnoho pryrodnoho parku. Materialy naukovoi konferentsii (11–14 veresnia 2008 roku, smt. Shatsk). Lviv, 130–133. (in Ukrainian)

Shkaran, V. I. 2009. Zaselenist shtuchnykh hnizdivel ptakhamy-duplohniznykamy y dribnymy ssavtsiamy v lisovykh biotopakh Shatskoho natsionalnoho pryrodnoho parku [Population of artificial nests by hollow-nesting birds and small mammals in forest biotopes of Shatskyi NPP]. – Naukovyi visnyk Volynskoho natsionalnoho universytetu imeni Lesi Ukrainky. Ser.: Biolohichni Nauky 2: 214–216. (in Ukrainian)

Shupova, T. V. & Chaplygina, A. B. 2016. Ornitofauna zakaznika obshchegosudarstvennogo znacheniya «Luchkovskij» [The avifauna of the national reserve «Luchkovsky»]. – Visn. Kharkivskoho nats. un-tu imeni V. N. Karazina. Seriia «Biolohiia» 26: 148–156. (in Russian)

Solti, B. 2010. A Mátra Múzeum madártani gyűjteménye III. Németh Márton tojásgyűjtemény [Oological collection of Mátra Museum III. Collection of Márton Németh]. – Folia Historico Naturalia Musei Matraensis Supplementum 5: 5–275. (in Hungarian)

Stankevych-Volosianchuk, O. I. 2008. Rarytetna ornitofauna mista Uzhhoroda [Rare fauna of Uzhgorod city]. – Naukovyi visnyk Uzhhorodskoho universytetu: Seriia: Biolohiia. Uzhhorod: Vydavnytstvo UzhNU «Hoverla» 23: 133–138. (in Ukrainian)

Stoyan, C. N. & Dimitar, G. 2007. Common Redstart *(Phoenicurus phoenicurus)*. – Atlas of Breeding Birds in Bulgaria, pp. 434–435.

Veistola, S., Lehikoinen, E., Eeva, T. & Iso-Iivari, L. 1996. The breeding biology of the Redstart *Phoenicurus phoenicurus* in a marginal area of Finland. – Bird Study 43: 351–355. DOI: 10.1080/00063659609461028

Willemoes, M. K., Tøttrup, A. P. & Thorup, K. 2013. Migration of the Common Redstart *(Phoenicurus phoenicurus)*: A Eurasian songbird wintering in highly seasonal conditions in the West African Sahel. – The Auk 130(2): 259. DOI: 10.1525 / auk.2013.13001

Yevtushenko, H. O. & Lytvynenko, S. P. 2009. Osoblyvosti sezonnykh mihratsii ptakhiv na terytorii Stanychno-Luhanskoho rybkombinatu [Particularities of birds seasonal migration on the territory of Stanychno-Luganskii fish factory]. – Visnyk Luhanskoho natsionalnoho universytetu imeni Tarasa Shevchenka. Biolohichni Nauky 2(165): 53. (in Ukrainian)

Yuzyk, D. I. & Chaplygina, A. B. 2018. Topichni ta fabrychni zviazky horykhvistky zvychainoi (*Phoenicurus phoenicurus* L.) v umovakh seredno transformovanykh terytorii Hetmanskoho NPP [Topic and fabric links of the Common Redstart (*Phoenicurus phoenicurus* L.) at conditions of medium transformed territories in Hetman NPP]. – Problemy ekolohii ta evoliutsii ekosystem v umovakh transformovanoho seredovyshcha: zbirnyk materialiv naukovykh prats II Mizhnarodnoi naukovo-praktychnoi konferentsii, m. Chernihiv, 11–12 zhovtnia 2018 r. Chernihiv: Desna Polihraf, 136–140. (in Ukrainian)

Zawadzki, G., Zawadzki, J., Zawadzka, D. & Sołtys, A. 2019. Using nest-boxes in pine stands of the Augustów Forest. – Forest Research Papers / Leśne prace badawcze 80(2): 137–143. DOI: 10.2478 / frp-2019-0012

Ornis Hungarica 2021. 29(2): 139–150.
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Characteristics of nest cavities used by White-backed Woodpecker *(Dendrocopos leucotos)* in Hungary

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Abstract The objective of this study was to document characteristics of cavities used for breeding by White-backed Woodpeckers *Dendrocopos leucotos* in Hungary. Twenty-five nest cavities were examined over a period of ten years (2011–2020). Five different tree species were used. The cavities were excavated at a mean height of 4.6 m and 2.2 m standard deviation and ranged from 2 to 8 m. All cavities found were in the main trunks of trees with trunk diameters ranging between 35–55 cm with a mean of 41.6 cm and 5.6 cm standard deviation. A southerly orientation of cavity entrances prevailed with a mean direction of 189±74 clockwise from north (median 180° IQR = 113°). The results suggest that cavity entrance orientation was non-random.

Keywords: woodpecker, nest cavities, cavity location, cavity height, cavity entrance orientation, deadwood

Összefoglalás A tanulmány célja, hogy dokumentálja a fehérhátú fakopáncsok *(Dendrocopos leucotos)* fészekodúinak jellemzőit Magyarországon. A tíz év időtartam alatt (2011–2020) vizsgált huszonöt odú öt különböző fafajban volt, elhelyezkedésük magassága 2 és 8 méter között változott, átlagosan 4,6±2,2 m volt. A fák törzsátmérője 35–55 cm között változott, átlagosan 41,6±5,6 cm volt. A röpnyílások jellemzően déli tájolásúak voltak, az óra járásával megegyezően 189±74° irányban, ami arra utal, hogy az odúk bejárata nem véletlenszerűen helyezkedett el.

Kulcsszavak: fakopáncs, költőodú, az odú helye, odúmagasság, röpnyílás tájolása, holtfa

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Introduction

The White-backed Woodpecker *Dendrocopos leucotos* is an Old-World picid, distributed within the Palearctic. Its range covers three eco-climatic regions: the temperate, Mediterranean and boreal zones. It occurs from Spain and France (the Pyrenees) in the west, central Italy (the Abruzzi) and the Balkans in the south, Scandinavia in the north and eastwards through Russia and Asia to Japan (Gorman 2014). The species is polytypic, with 11 (del Hoyo & Collar 2014) or 12 (Gorman 2014) subspecies generally recognised. The nominate *leucotos* occurs in Hungary where this study was carried out. The south-western Palaearctic *lilfordi* subspecies is sometimes regarded as a valid phylogenetic species based on molecular species delimitation methods, gene flow analyses and differences in plumage (Pons *et al.* 2021). This taxon has been extensively studied and relevant data is included for comparison in this paper.

White-backed Woodpecker inhabits a variety of forested habitats across its range, usually deciduous (Wesołowski 1995, Grangé *et al.* 2002), but also mixed deciduous-coniferous (Fernández & Azkona 1996). In parts of Russia, forests dominated by conifers are sometimes inhabited (Dement'ev & Gladkov 1966), but in general stands of pure conifer, such as plantations of Norway spruce *Picea abies,* are avoided (Hämäläinen *et al.* 2020). Ultimately, White-backed Woodpeckers are strongly associated with mature deciduous trees and deciduous deadwood. Numerous studies from across the range have shown that reasonably large areas of unmanaged old-growth forest with a high proportion of deadwood, standing and fallen, are required for both foraging and breeding (Håland & Ugelvik 1990, Costantini & Melletti 1992, Grangé 1993, 2016, Virkkala *et al* 1993, Hogstad & Stenberg 1994, Bernoni 1995, Mikusiński & Angelstam 1998a, 1998b, Frank 2002, Håpnes 2003, Melletti & Penteriani 2003, Pavelka 2003, Garmendia *et al.* 2006, Czeszczewik 2009, Gerdzhikov *et al.* 2018, Schwaiger & Lauterbach 2019, Urkijo-Letona *et al.* 2020, Bühler 2021). In Hungary, the species generally only occurs in hill forests where there are native deciduous tree species older than 60–70 years and a minimum volume of 20 m^3/ha of deadwood (Szmorad *et al.* 2018). Nevertheless, White-backed Woodpeckers will persist in managed forests if there is a natural, or close to natural regeneration, with sufficient rotting timber, mostly in forests where management is minimal (Aulén 1988, Aulén & Carlson 1990).

The global population of the species is unknown but is considered to be large with the bulk of the population found in the eastern part of its range. European populations occupy around 35% of the global range, with a breeding population of 232,000–586,000 pairs estimated. A preliminary estimate of the total world population size has been made of 1,320,000–3,350,000 adult birds, although validation of this estimate is required (BirdLife International 2021). The European range is rather disjunct and in the west of the continent, this woodpecker is often localised and rare. Despite some local increases, populations have been declining for decades in several European countries, for example in Poland (Wesołowski & Tomiałojć 1986), Sweden (Aulén 1988), Germany (Scherzinger 1990), Norway (Håland & Ugelvik 1990), Finland (Virkkala *et al.* 1993, Martikainen *et al.* 1998), Spain (Fernández & Azkona 1996) and Latvia (Krams 1998). Yet, the overall European trend is considered to be stable (Lanz et al. 2020). The main reasons for the decline of the species are the loss of old-growth deciduous forests and inappropriate forestry management (Carlson 2000, Håpnes 2003, Sabatini *et al.* 2018). The Red List of the International Union for Conservation of Nature (IUCN) categorises White-backed Woodpecker as *Least Concern* (BirdLife International 2021). Owing to the forest habitat types in which it thrives, the White-backed Woodpecker can be considered an umbrella species, as its occurrence is linked to the presence of many other species (flora, fauna and fungi) of high conservation concern (Roberge *et al.* 2008).

White-backed Woodpecker is the rarest and most range-restricted member of the Picidae family in Hungary, the only species from the nine that occur to be assigned 'Strictly Protected Species' status (Gorman *et al.* 2021). In Hungary, as elsewhere, intensive forest management, which often involves clear-cutting, fragmentation, felling of mature trees, removal of deadwood and the introduction of non-native tree species is a common problem (Szmorad *et al.* 2018). Logging in supposedly protected areas and even in the spring during the breeding season (pers. obs.), has resulted in a reduction in woodpeckers' numbers. The size of the Hungarian breeding population remains unclear, a recent estimation of between 260–670 pairs (Gorman *et al.* 2021) will be refined once an ongoing national survey is completed. Some studies of White-backed Woodpeckers have been conducted in Hungary, but none have focused on nest cavities.

Although White-backed Woodpeckers will occasionally create cavities in autumn and winter (Grangé *et al.* 2020), data were collected in the months of March, April and May as this is the period when cavity excavation is most prevalent. All woodpeckers tend to nest earlier when spring temperatures are high, but in Europe, White-backed Woodpeckers are usually the earliest breeders (Grangé *et al.* 2002, Wesołowski *et al.* 2021). Pairs tend to work on several new cavities each spring, but not all are finished. Old cavities are rarely re-used as nest sites but are used as roosts (Aulén & Carlson 1990).

This study had five main aims as follows: (1) to determine tree species used; (2) to document cavity locations, whether on trunks or limbs and the health of nest trees; (3) to measure the height of cavity entrances above ground level; (4) to measure the trunk diameters at breast height (DBH) of trees with cavities; (5) to document the orientation of cavity entrances.

Methods

Woodpecker nest sites may be quantitatively described by examining the characteristics of the nesting tree (Hågvar *et al.* 1990). This study was restricted to nest cavities excavated and used by White-backed Woodpeckers in Hungary. The main aims of this study were to characterize cavity trees used by this species including the tree species, its state of decay,

Figure 1. The three study areas in north-east Hungary
1. ábra A három vizsgálati terület ÉK-Magyarországon

cavity locations (whether they were on trunks or limbs), cavity height, orientation of cavity entrances and the diameter of trees used. The study was carried out over a ten-year period, from 2011–2020 and conducted in three hill areas in the north-east of the country characterized by tracts of deciduous forested slopes, namely Aggtelek, Bükk and Zemplén *(Figure 1).* These hills are characterised by relatively low elevations, the highest peaks in each all being below 1,000 m above sea-level.

Cavities were searched for in forest habitats, visually searching known areas where the species had been observed previously. Cavities were located by observing the behaviour of the birds, such as courtship behaviour and seeing birds carrying food for nestlings, finding fresh woodchips below trees and by hearing persistent calling, drumming, tapping and the sounds made during excavation work. Cavity height was estimated using simple trigonometry. The diameter of cavity trees was calculated by means of the standard method of DBH, with measurements taken using callipers at 1.3 m above the ground from the base of the trunk. Cavity orientation was calculated with a compass, using sixteen standard points (N, NNE, NE, ENE, etc.). The randomness of orientation was assessed using a Rayleigh test of uniformity (Pewsey *et al.* 2013) as implemented in the package 'circular' (version 0.4-3, Lund & Agostinelli 2011) in R, (R Development Core Team 2015). This provides a statistic, r, with value between 0 and 1, where zero indicates a random distribution and 1 represents perfect alignment to one direction. The package also provides a mean orientation.

Results

A total of twenty-five active White-backed Woodpecker nest cavities were found during the study and their attributes and locations documented. All cavity trees were in forests dominated by deciduous trees. All cavities were newly excavated: none were from previous years that were being reused. New cavities can be recognized by their entrances having clean edges with no renewed tree growth, and light-coloured wood, and an abundance of fresh woodchips at the base of trees (Gorman 1995). Cavities were often located in trees close to those used in previous years. Cavity entrances were generally circular in shape, occasionally horizontally oval, however their dimensions were not measured.

Cavity trees: In this study, nests were found in just five tree species (number of occasions in brackets). All were broadleaved: beech *Fagus sylvatica* (12), hornbeam *Carpinus betulus* (10), wild cherry *Prunus avium* (1), ash *Fraxinus excelsior* (1) and pedunculate oak *Quercus robur* (1). While there was a clear dominance of two species *(Table 1)*, beech (48%) and hornbeam (40%), this was not tested for selection by White-backed Woodpeckers due to lack of data on relative tree species availability from within the study area. Across their vast range White-backed Woodpeckers excavate their nest cavities in a variety of trees, however factors such as wood condition, location and general availability are more important than actual tree species. They do not seem to show any significant preference for particular tree species, rather they are only associated with specific trees locally.

Cavity location: All cavities found were located in main trunks, positioned on foliage-free areas below canopy level, with an open fly path to the entrance. None were found in boughs or branches. All were in dead trees (snags) or in decaying sections of living trees. It was not always possible to verify the existence or extent of decay on all cavity trees, however visually it was obvious that twenty-two (88%) were affected. Twenty of the total of twenty-five cavities (80%) were located beneath a growth of tinder polypore *Fomes fomentarius,* twelve on beech, eight on hornbeam.

Cavity height: In this study, the heights of cavity entrances from the ground level varied between 2 m and 8 m, with a mean of 4.6 m and a standard deviation of 2.2 m.

Tree trunk width: The widest diameter (DBH) of a cavity tree was 55 cm and the narrowest 35 cm. The most frequently documented diameter was 40 cm (5 cavities: 20%), with a mean of 41.6 cm and standard deviation of 5.6 cm.

Cavity entrance orientation: In this study, a southerly orientation prevailed. Of the twenty-five cavities, thirteen (52%) faced southwards (SE, SSE, S, SSW, SW). The most frequent alignments were SE and SW with four (16%) cavities each. Five cavities faced westwards (WNW, W, WSW), four eastwards (E, ESE) and three northwards (NE, NW). The cavity orientation was non-random and appreciably biased toward the south (Rayleigh test r=0.37, P=0.03) with a mean direction of 189° clockwise from north and 74° standard deviation (median 180° IQR (inter-quartile range) = 113°) *(Figure 2).*

Table 1. Summary of cavities used by White-backed Woodpeckers

1. táblázat A fehérhátú fakopáncs odúk adatai

Year	Location	Tree species	Height (approx. metres)	Trunk width (DBH) in cm	Cavity-entrance orientation
2011	Aggtelek	Beech	4	38	SE
2011	Bükk	Hornbeam	2	35	SE
2011	Zemplén	Beech	4.5	50	SSE
2012	Aggtelek	Beech	6	48	WSW
2012	Bükk	Hornbeam	2.5	36	WSW
2013	Aggtelek	Beech	6	40	S
2013	Bükk	Beech	6	45	SW
2013	Zemplén	Ash	4	36	SE
2014	Aggtelek	Hornbeam	5	38	SW
2014	Bükk	Hornbeam	4	40	W
2015	Aggtelek	Beech	7	50	NE
2015	Bükk	Hornbeam	5.5	35	SW
2015	Zemplén	Beech	8	50	WNW
2016	Bükk	Hornbeam	3.5	36	SSW
2016	Zemplén	Pedunculate Oak	5	55	E
2016	Zemplén	Beech	4	40	NW
2017	Aggtelek	Beech	3	44	NW
2017	Bükk	Hornbeam	2	35	S
2017	Zemplén	Beech	8	45	W
2018	Aggtelek	Beech	7	44	SE
2018	Zemplén	Wild Cherry	3	40	ESE
2019	Bükk	Hornbeam	4	38	E
2019	Zemplén	Hornbeam	5	40	S
2020	Bükk	Beech	3.5	44	E
2020	Bükk	Hornbeam	2.5	38	SW

Conclusions

This paper summarises some characteristics of twenty-five nest cavities of White-backed Woodpeckers in Hungary. As the study was conducted over ten years and in several disjunct areas, it is unlikely that the cavities found were all created by the same individuals and thus the individual preferences of a few birds are unlikely to significantly bias the results presented here.

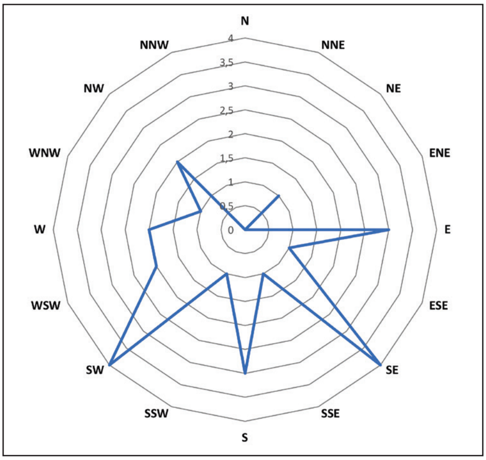

Figure 2. Orientation of White-backed Woodpecker cavity entrances as frequencies of cardinal points

2. ábra A fehérhátú fakopáncs odúk tájolása

Cavities were found in five different tree species with two (beech and hornbeam) dominating but owing to the relatively small sample size and lack of data on relative tree species availability within the study area, no firm conclusions could be made. However, trees in other parts of the species' range might be compared. In the Western Pyrenees, France, from 43 nests documented in one study 97% were situated in beech (Grangé *et al.* 2002) and in another in the same region all but one from 76 were in beech, the exception being a nest in a fir *Abies spp.* (Grangé 2009). In Navarra, Spain, all the nests in one study were also located in beech (Cárcamo *et al.* 2019). But in Fennoscandia, birches *Betula* spp. and particularly aspen *Populus tremula* are commonly used. Sometimes 90% of nest cavities are located in the latter soft wood species (Aulén 1988, Hogstad & Stenberg 1994). In the Białowieża forest in Poland, alders *Alnus* are used in boggy parts of the forest and hornbeam *Carpinus betulus* and oaks *Quercus* spp. in drier areas (Wesołowski 1995).

The extremely strong use of White-backed Woodpecker in this study for nest sites in dead trees or dead parts of trees was consistent with most studies on this species carried out elsewhere (Stenberg & Hogstad 1992, Hogstad & Stenberg 1994, Krams 1998, Pavlík 1999, Grangé *et al.* 2002). However, one study in France documented that 77% of nesting trees were in a healthy state (Grangé 2009), and in an Italian study 40% of pairs nested in trees without evidence of decay (Melletti & Penteriani 2003). Furthermore, in the same study, although the majority of nests (75%) were in trunks below the canopy, 25% were within the canopy. In a study in England of the nest sites of another woodpecker species, Great Spotted Woodpecker *Dendrocopos major*, it was found that about half were in live trees and half in dead ones, but because dead trees were scarce in the study areas this represented a very strong selection for dead trees (Smith 1997).

Woodpeckers, in general, invest significant time and effort in making cavities, however, despite being morphologically adapted to excavate wood, they regularly select trees (dead or alive) that have soft and weak sections of timber owing to fungal heart rot, insect infestation or wounds from weather events (Matsuoka 2010, Lorenz *et al.* 2015). The tinder polypore found on many of the nest trees in this study (80% were located just beneath this fungus) typically grows on snags and logs is often found in White-backed Woodpecker habitats. Indeed, some authors have suggested that it may have a certain relationship with the woodpecker (Urkijo-Letona *et al.* 2020). It seems highly likely that White-backed

Woodpeckers prefer dead or highly decayed trees for nest sites, but due to insufficient data about the share of such trees in the vicinity of cavity trees, it was not possible to test this statistically for this study. Such trees are easier to excavate than healthy ones and provide convenient opportunities for cavity placement. Studies of species which are often sympatric with White-backed Woodpeckers in Europe, such as Black Woodpecker *Dryocopus martius* (Zahner *et al.* 2012), Grey-headed Woodpecker *Picus canus* (Gorman 2019), Eurasian Green Woodpecker *Picus viridis* (Gorman 2020b) and Great Spotted and Middle Spotted Woodpeckers *Dendrocoptes medius* (Kosiński & Winiecki 2004), have all indicated that the hardness of the interior of trees is a major factor in cavity site selection. The White-backed Woodpeckers in this study were no exception, as ease of excavation was seemingly more important when selecting a cavity location rather than any relationship with specific tree species. Ultimately, providing the habitat surrounding potential nesting locations offers suitable foraging opportunities, almost any tree in a given area that is easy to excavate can be selected for cavities. This is in accordance with other studies on other European woodpeckers (Kosiński & Winiecki 2004).

The heights of cavities found during this study were substantially lower than those documented in most other similar studies on the species. In the French Pyrenees mean heights 10 m (Grangé 1993), 13.7 m (Grangé *et al.* 2002) and 14.2 m (Grangé 2009) have been documented, and in Navarra, Spain, 11.1 m (Cárcamo *et al.* 2019). In an Italian study, the lowest nest was 5 m and the highest 28 m (Melletti & Penteriani 2003). In Poland the lowest nest was 5 m and the highest 32 m (Wesołowski 1995), however, it might be noted that trees in Poland's old-growth forests can reach greater heights than in most other areas of Europe.

The widest diameter (DBH) of a cavity tree recorded in this study was 55 cm, the narrowest 35 cm, with a mean of 41.6 cm. This is remarkably similar to the mean of 42 cm recorded in a study of 43 nests in the Western Pyrenees, France (Grangé *et al.* 2002) and close to the 45.8 cm measured in a later study in the same region (Grangé 2009). Studies worldwide on the orientation of woodpecker cavity entrances have produced contrasting conclusions. A meta-analysis of cavity entrance orientation from 80 populations of 23 species of woodpecker throughout the Northern Hemisphere concluded that orientation was influenced by regional climatic conditions and was typically non-random (Landler *et al.* 2014). At present, there seems to have been no studies showing the adaptive advantages of particular nest orientations, but it is believed that they occur. Landler *et al.* (2014) included data from 12 studies on Great Spotted Woodpeckers and all but three of these studies found random orientation, the exceptions being studies in Great Britain (Tracy 1938), China (Wan *et al.* 2008) and Poland (Hebda 2009). In the present study of White-backed Woodpecker cavities across Hungary, orientation was non-random with a general alignment and southerly orientation of entrances prevailing. Similar studies on the species have either suggested that southward-facing cavity entrances are usual (Grangé 1993) or that compass direction was insignificant (Wesołowski 1995, Melletti & Penteriani 2003, Grangé 2009, Cárcamo *et al.* 2019).

A study on Syrian Woodpeckers *Dendrocopos syriacus* (also carried out in Hungary) likewise found that south-facing cavity entrances prevailed (Gorman 2020a). Indeed, there is some evidence that in hill forests compass direction influences the location of cavity entrances to a greater extent than topographical factors such as slope gradients for

White-backed Woodpeckers and *Dendrocopos* woodpeckers in general, birds tending to prefer south-facing slopes for nesting and also for foraging (Stenberg & Hogstad 1992). The reasons for the predominance of southerly orientated cavities in this study are not obvious and several factors are likely to be implicated.

Thermophilic benefits from the southerly orientation of nest cavities may be involved. Nest cavities facing southwards receive more sunlight in the morning hours, hence increased light and warmth. In the breeding period, morning is the time when adults emerge from a night of incubation or brooding inside the nest chamber and begin to forage. The predominance of cavity entrances facing southwards seems to suggest that early morning warming is favoured by woodpeckers. As an example, a study of the Northern Flicker *Colaptes auratus* in North America found such an orientation to be associated with increased egg productivity, although this did not convert into breeding success (Wiebe 2001). However, Landler *et al.* (2014) identified a stronger southerly orientation selection for North American over Eurasian woodpeckers, thus factors other than latitude may influence selection in Hungary. For instance, prevailing wind direction in Hungary, which is from the northwest (Hungarian Meteorological Service undated), may be an influencing component. Yet, as with the positioning of cavities on trees, a compromise might occasionally be at work with local conditions resulting in some factors overriding others. For example, the southern sides of trees may not always receive the most sun owing to the immediate environment, as other trees or structures may create shade. In addition, woodpeckers may disregard compass direction when excavating a cavity if doing so involves substantial energy output. A section of the tree where a cavity can be more easily excavated may be selected, although it might not be ideal in terms of orientation. Wood facing away from the south may be selected because it is easier to excavate. Studies in subtropical forests of South America by Schaaf (2020) identified non-random orientation of cavities made by Dot-fronted Woodpecker *Veniliornis frontalis* which was related to differences in vegetation cover. Two orientations were identified, north and most protected by vegetation and north-west unprotected, and suggested the former orientation in maximum exposure to the sun hastened drying out of the microclimate around the cavity following intense rainfall. Conversely, the unprotected north-westerly orientation, which was out of the most intense period of sunlight, benefitted from afternoon accumulation of heat within the cavity to enhance the thermal properties inside during low night-time temperatures. Clearly, studies which look at the interactions between orientation and proportion of foliage cover, and other habitat attributes surrounding nest cavities, may be valuable and inform forest management.

The White-backed Woodpecker is an insectivorous forest specialist which nests in dying or dead trees and forages mainly on wood-boring beetles in the deadwood of deciduous species. It is therefore extremely sensitive to the changes in forest structure and composition that result from intensive forestry management (Virkkala *et al.* 1993, Fernández & Azkona 1996, Martikainen *et al.* 1998, Mikusiński & Angelstam 1998a, 1998b, Carlson 2000, Melletti & Penteriani 2003). The White-backed Woodpeckers in this study were clearly dependent upon trees that were mature, dead, or in an advanced state of decay for nesting. Hence, as elsewhere across its range, the nesting and indeed overall habitat requirements of this species strongly indicate that intensive forest management methods should change.

Forest managers should adopt close-to-nature methods which retain a diverse mix of tree age classes including standing deadwood and limit harvesting in protected areas. The incorporation of woodpecker richness and diversity measures should be used to inform forest management bodies (Lõhmus *et al.* 2016).

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References

Aulén, G. 1988. Ecology and distribution history of the White-backed Woodpecker *Dendrocopos leucotos* in Sweden. – Report No 14. Department of Wildlife Ecology, Swedish University of Agricultural Sciences, Uppsala

Aulén, G. & Carlson, A. 1990. Demography of a declining White-backed Woodpecker population. – In: Carlson, A. & Aulén, G. (eds.) Conservation and Management of Woodpecker Populations. – Swedish University of Agricultural Sciences, Department of Wildlife Ecology, Report 17. Uppsala, Sweden

Bernoni, M. 1995. Densita della popolazione di Picchio dorsobianco *Picoides leucotos lilfordi* nel Parco Nazionale d'Abruzzo [Population density of White-backed Woodpecker in the Abruzzo National Park]. – Avocetta 19: 130. (in Italian)

BirdLife International 2021. Species factsheet: *Dendrocopos leucotos*. – http://www.birdlife.org on 01/09/2021.

Bühler, U. 2021. Vorkommen des Weissrückenspechts *Dendrocopos leucotos* und Waldnutzung in Nordbünden [Occurrence of White-backed Woodpecker *Dendrocopos leucotos* and forest use in northern Grisons]. – Ornithologischer Beobachter 118: 58–75. (in German with English Summary)

Cárcamo, S., Elosegi, M. M., Senosiain, A. & Arizaga, J. 2019. Nidotópica y parámetros reproductivos en el pico dorsiblanco *Dendrocopos leucotos lilfordi* Sharpe & Dresser, 1871 en Navarra [Nest site selection and reproduction parameters in the White-backed Woodpecker *Dendrocopos leucotos lilfordi* Sharpe & Dresser, 1871 in Navarre]. – Munibe Ciencias Naturales 67: 31–44. DOI: 10.21630/mcn.2019.67.06 (in Spanish with English Summary)

Carlson, A. 2000. The effect of habitat loss on a deciduous forest specialist: the White-backed Woodpecker *(Dendrocopos leucotos)*. – Forest Ecology and Management 131(1): 215–221. DOI: 10.1016/S0378-1127(99)00215-7

Costantini, C. & Melletti, M. 1992. The White-backed Woodpecker, *Picoides leucotos lilfordi*, and the Lesser Spotted Woodpecker, *Picoides minor*, in the beech woods of the Simbruini and Ernici Mountains (Central Italy). – Rivista Italiana di Ornitologia 62: 164–170.

Czeszczewik, D. 2009. Foraging behaviour of White-backed Woodpeckers *Dendrocopos leucotos* in a primeval forest (Białowieża National Park, NE Poland): Dependence on habitat resources and season. – Acta Ornithologica 44: 109–118. DOI: 10.3161/000164509X482687

del Hoyo, J. & Collar, N. J. 2014. HBW and BirdLife International Illustrated Checklist of the Birds of the World, Vol. 1. Non-passerines. – Lynx Edicions, Barcelona

Dement'ev, G. P. & Gladkov, N. A. 1966. (eds.) Birds of the Soviet Union. Vol. 1. – Israel Program for Scientific Translations, Jerusalem

Fernández, C. & Azkona, P. 1996. Influence of forest structure on the density and distribution of the White-backed Woodpecker *Dendrocopos leucotos* and Black Woodpecker *Dryocopus martius* in Quinto Real (Spanish western Pyrenees). – Bird Study 43: 305–313. DOI: 10.1080/00063659609461023

Frank, G. 2002. Brutzeitliche Einnischung des Weissrückenspechtes *Dendrocopos leucotos* im Vergleich zum Buntspecht *Dendrocopos major* in montanen Mischwäldern der nordlichen Kalkalpen [Habitat selection of

White-backed Woodpecker *Dendrocopos leucotos* during the breeding season in comparison to Great Spotted Woodpecker *Dendrocopos major* in mountainous mixed forests of the northern Austrian Chalk Alps]. – Vogelwelt 123: 225–239. (in German)

Garmendia, A., Cárcamo, S. & Schwendtner, O. 2006. Forest management considerations for conservation of Black Woodpecker *Dryocopus martius* and White-backed Woodpecker *Dendrocopos leucotos* populations in Quinto Real (Spanish Western Pyrenees). – Biodiversity and Conservation 15: 339–355. DOI: 10.1007/s10531-005-5410-0

Gerdzhikov, G. P., Georgiev, K. B., Plachiyski, D. G., Zlatanov, T. & Shurulinkov, P. S. 2018. Habitat requirements of the White-backed Woodpecker *Dendrocopos leucotos lilfordi* (Sharpe & Dresser, 1871) (Piciformes: Picidae) in Strandzha Mountain, Bulgaria. – Acta Zoologica Bulgarica 70: 527–534.

Gorman, G. 1995. Identifying the presence of woodpecker (Picidae) species on the basis of their holes and signs. – Aquila 102: 61–67.

Gorman, G. 2014. Woodpeckers of the World: The Complete Guide. – Helm, London, UK

Gorman, G. 2019. Characteristics of Grey-headed Woodpecker (*Picus canus*) cavities in Hungary. – Aquila 126: 33–39.

Gorman, G. 2020a Details of cavities used by Syrian Woodpeckers *Dendrocopos syriacus* in Hungary. – Bird Study 67(2): 251–255. DOI: 10.1080/00063657.2020.1808588

Gorman, G. 2020b Attributes of Eurasian Green Woodpecker *(Picus viridis)* nest cavities in Hungary. – Ornis Hungarica 28(2): 204–211. DOI: 10.2478/orhu-2020-0025

Gorman, G., Komlós, M., Ónodi, G. & Schmidt, A. 2021. Fehérhátú fakopáncs *Dendrocopos leucotos* White-backed Woodpecker. – In: Szép, T., Csörgő, T., Halmos, G., Lovászi, P., Nagy, K. & Schmidt, A. (eds.) Magyarország madáratlasza – Bird Atlas of Hungary. – Agrárminisztérium, Magyar Madártani és Természetvédelmi Egyesület, Budapest, pp. 428–429. https://mme.hu/birdatlas (in Hungarian with English Summary)

Grangé, J-L. 1993. Donnees preliminaires sur la biologie de reproduction du Pic a dos blanc pyreneen *(Dendrocopos leucotos lilfordi)* en Bearn [Preliminary data on the reproductive biology of the Pyrenian White-backed Woodpecker (*Dendrocopos leucotos lilfordi*) in Bearn]. – Nos Oiseaux 42: 17–28. (in French)

Grangé, J-L., Auria, J-C., André, C. & Navarre, P. 2002. Biologie de reproduction du Pic à dos blanc *Dendrocopos leucotos lilfordi* dans les Pyrénées occidentales (France) [Reproductive biology of the White-backed Woodpecker *Dendrocopos leucotos lilfordi* in the Western Pyrenees, France]. – Nos Oiseaux 49: 199–212. (in French with English Summary).

Grangé, J-L. 2009. Caractéristiques des arbres de nid chez le Pic à dos blanc *Dendrocopos leucotos lilfordi* dans les Pyrénées occidentales françaises [Characteristics of the trees in which the White-backed Woodpecker *Dendrocopos leucotos lilfordi* nests in the French western Pyrenees]. – Le Casseur d'Os 9: 92–110. (in French with English Summary).

Grangé, J-L. 2016. Breeding biology of the Lilford Woodpecker *Dendrocopos leucotos lilfordi* in the Western Pyrenees (SW France). – Denisia 36 Zugleich Kataloge des Oberösterreichischen Landesmus 164: 99–111.

Grangé, J-L., Senosiain, A., Marsaguet, P. & Navarre, P. 2020. Creusement de cavités en automne-hiver par les Picidés européens [Excavation of cavities by European woodpeckers in autumn-winter]. – Ornithos 27(6): 345–354. (in French with English Summary)

Hågvar, S., Hågvar, G. & Mønness, E. 1990. Nest site selection in Norwegian woodpeckers. – Holarctic Ecology 13:156–165. DOI: 10.1111/j.1600-0587.1990.tb00602.x

Håland, A. & Ugelvik, M. 1990. The status of the White-backed Woodpecker *Dendrocopos leucotos* in Norway. – In: Carlson, A. & Aulén, G. (eds.) Conservation and Management of Woodpecker Populations. – Swedish University of Agricultural Sciences, Department of Wildlife Ecology, Report 17. Uppsala, Sweden

Hämäläinen, K., Junninen, K., Halme, P. & Kouki, J. 2020. Managing conservation values of protected sites: how to maintain deciduous trees in White-backed Woodpecker territories. – Forest Ecology and Management 461: 117946. DOI: 10.1016/j.foreco.2020.117946

Håpnes, A. 2003. The White-backed Woodpecker, highly threatened by forestry. – WWF-Norway

Hebda, G. 2009. Nesting sites of the Great Spotted Woodpecker *Dendrocopos major* L. in Poland: analysis of nest cards. – Polish Journal of Ecology 57: 149–158.

Hogstad, O. & Stenberg, I. 1994. Habitat selection of a viable population of White-backed Woodpeckers *Dendrocopos leucotos*. – Journal für Ornithologie 138: 25–38.

Hungarian Meteorological Service. Undated. Wind conditions of Hungary. – https://www.met.hu/en/eghajlat/magyarorszag_eghajlata/altalanos_eghajlati_jellemzes/szel/. Accessed 01/09/2021.

Kosiński, Z. & Winiecki, A. 2004. Nest-site selection and niche partitioning among the Great Spotted Woodpecker *Dendrocopos major* and Middle Spotted Woodpecker *Dendrocopos medius* in riverine forest of Central Europe. – Ornis Fennica 81: 145–156.

Krams, I. 1998. Nest site selection of the White-backed Woodpecker *Dendrocopos leucotos* in the eastern part of Latvia. – Ornis Svecica 8: 11–16.

Landler, L., Jusino, M. A., Skelton, J. & Walters, J. R. 2014. Global trends in woodpecker cavity entrance orientation: latitudinal and continental effects suggest regional climate influence. – Acta Ornithologica 49: 257–266. DOI: 10.3161/173484714X687145

Lanz, M., Keiss, O. & Keller, V. 2020. *Dendrocopos leucotos* White-backed Woodpecker. – In: Keller, V., Herrando, S., Voříšek, P., Franch, M., Kipson, M., Milanesi, P., Martí, D., Anton, M., Klvaňová, A., Kalyakin, M. V., Bauer, H-G. & Foppen, R. P. B. 2020. European Breeding Bird Atlas 2: Distribution, Abundance and Change. – European Bird Census Council and Lynx Edicions, Barcelona

Lorenz, T. J., Vierling, K. T., Johnson, T. R. & Fischer, P. C. 2015. The role of wood hardness in limiting nest site selection in avian cavity excavators. – Ecological Applications 25(4): 1016–1033. DOI: 10.1890/14-1042.1

Lõhmus, A., Nellis, R., Pullerits, M. & Leivits, M. 2016. The potential for long-term sustainability in seminatural forestry: a broad perspective based on woodpecker populations. – Environmental Management 57(3): 558–571. DOI: 10.1007/s00267-015-0638-2

Lund, U. & Agostinelli, C. 2011. Package 'circular': circular statistics. – R package version 0.4-3.

Martikainen, P., Kaila, L. & Haila, Y. 1998. Threatened beetles in White-backed Woodpecker habitats. – Conservation Biology 12(2): 293–301. DOI: 10.1111/j.1523-1739.1998.96484.x

Matsuoka, S. 2010. Great Spotted Woodpeckers *Dendrocopos major* detect variation in wood hardness before excavating nest holes. – Ornithological Science 9(1): 67–74. DOI: 10.2326/osj.9.67

Melletti, M. & Penteriani, V. 2003. Nesting and feeding tree selection in the endangered White-backed Woodpecker, *Dendrocopos leucotos lilfordi*. – Wilson Bulletin 115(3): 299–306. DOI: 10.1676/03-022

Mikusiński, G. & Angelstam, P. 1998a Economic geography, forest distribution and woodpecker diversity in central Europe. – Conservation Biology 12: 200–208. DOI: 10.1111/j.1523-1739.1998.96310.x

Mikusiński, G. & Angelstam, P. 1998b European woodpeckers and anthropogenic habitat change: evidence of effects in different scales. – Vogelwelt 118: 277–283.

Pavelka, K. 2003. Strakapoud Belohrbety *(Dendrocopos leucotos)* v zapadnich Beskydech a Javornikach [White-backed Woodpecker in the Western Beskydy Mountains and in the Javorniky Mountains]. – Crex 21: 39–46. (in Czech with English Summary)

Pavlík, Š. 1999. Breeding ecology of the White-backed Woodpecker *Dendrocopos leucotos* in oak forest. – Biologia 54: 187–194.

Pewsey, A., Neuhauser, M. & Ruxton, G. D. 2013. Circular Statistics in R. – OUP, Oxford

Pons, J-M., Campión, D., Chiozzi, G., Ettwein, A., Grangé, J-L., Kajtoch, L., Mazgajski, T. D., Rakovic, M., Winkler, H. & Fuchs, J. 2021. Phylogeography of a widespread Palaearctic forest bird species: The White-backed Woodpecker (Aves, Picidae). – Zoologica Scripta 50: 155–172. DOI: 10.1111/zsc.12466

R Development Core Team 2015. R: A language and environment for statistical computing. – R Foundation for Statistical Computing, Vienna, Austria

Roberge, J. M., Mikusiński, G. & Svensson, S. 2008. The White-backed Woodpecker: Umbrella species for forest conservation planning? – Biodiversity and Conservation 17: 2479–2494. DOI: 10.1007/s10531-008-9394-4

Sabatini, F. M., Burrascano, S., Keeton, W. S., Levers, Ch., Lindner, M., Pötzschner, F., Verkerk, P. J., Bauhus, J., Buchwald, E., Chaskovsky, O., Debaive, N., Horváth, F., Garbarino, M., Grigoriadis, N., Lombardi, F., Duarte, I. M., Meyer, P., Midteng, R., Mikac, S., Mikoláš, M., Motta, R., Mozgeris, G., Nunes, L., Panayotov, M., Ódor, P., Ruete, A., Simovski, B., Stillhard, J., Svoboda, M., Szwagrzyk, J., Tikkanen, O-P., Volosyanchuk, R., Vrska, T., Zlatanov, T. & Kuemmerle, T. 2018. Where are Europe's last primary forests? – Diversity and Distributions 24: 1426–1439. DOI: 10.1111/ddi.12778

Schaaf, A. A. 2020. Effects of sun exposure and vegetation cover on Woodpecker nest orientation in subtropical forests of South America. – Journal of Ethology 38(1): 117–120. DOI: 10.1007/s10164-019-00617-2

Scherzinger, W. 1990. Is competition by the Great Spotted Woodpecker the cause for White-backed Woodpecker's rarity in Bavarian Forest National Park? – In: Carlson, A. & Aulén, G. (eds.) Conservation and management of woodpecker populations, report 17 (Swedish University of Agricultural Sciences, Department of Wildlife Ecology, Uppsala, Sweden) pp. 81–91.

Schwaiger, S. & Lauterbach, M. 2019. Verbreitung, Bestandssituation und Habitatansprüche von Dreizehenspecht *Picoides tridactylus* Und Weißrückenspecht *Dendrocopos leucotos* in den Bayerischen Alpen [Distribution, population density and habitat requirements of Three-toed Woodpecker *(Picoides tridactylus)* and White-backed Woodpecker *(Dendrocopos leucotos)* in the Bavarian Alps]. – Ornithologischer Anzeiger 57: 228–242. (in German with English Summary)

Smith, K. 1997. Nest site selection of the Great Spotted Woodpecker *Dendrocopos major* in two oak woods in southern England and its implications for woodland management. – Biological Conservation 80: 283–288. DOI: 10.1016/S0006-3207(96)00038-9

Stenberg, I. & Hogstad, O. 1992. Habitat use and density of breeding woodpeckers in the 1990s in Møre og Romsdal county, western Norway. – Fauna Norvegica, Serie C, Cinclus 19: 21–38.

Szmorad, F., Frank, T. & Korda, M. 2018. Erdőgazdálkodás és erdőkezelés Natura 2000 területeken. Natura 2000 kézikönyv erdőgazdálkodóknak [Forestry and forest management in Natura 2000 areas. Natura 2000 handbook for forest managers]. – Duna-Ipoly Nemzeti Park Igazgatóság, Budapest (in Hungarian)

Tracy, N. 1938. Der grosse Buntspecht [The Great Spotted Woodpecker]. – Beiträge zur Fortpflanzungsbiologie der Vögel 14: 41–48. (in German)

Urkijo-Letona, A., Cárcamo, S., Peña, L., Fernández de Manuel, B., Onaindia, M. & Ametzaga-Arregi, I. 2020. Key elements of the White-backed Woodpecker's *(Dendrocopos leucotos lilfordi)* habitat in its European South-Western limits. – Forests 11: 831. DOI: 10.3390/f11080831

Virkkala, R., Alanko, T., Laine, T. & Tiainen, J. 1993. Population contraction of the White-backed Woodpecker *Dendrocopos leucotos* in Finland as a consequence of habitat alteration. – Biological Conservation 66: 47–53.

Wan, T., Hu, J-F., Jiao, Z-B., Wen, J-B. & Luo, Y-Q. 2008. Nest-cavity characteristics of the Great Spotted Woodpecker *Dendrocopos major* in shelter plantations of west Inner Mongolia. – Forestry Studies in China 10: 36–40. DOI: 10.1007/s11632-008-0010-1

Wesołowski, T. & Tomiałojć, L. 1986. The breeding ecology of woodpeckers in a temperate primeaval forest - preliminary data. – Acta Ornithologica 22: 1–21.

Wesołowski, T. 1995. Ecology and Behaviour of White-backed Woodpecker *(Dendrocopos leucotos)* in a primaeval temperate forest (Białowieża National Park, Poland). – Vogelwarte 38: 61–75.

Wesołowski, T., Hebda, G. & Rowiński, P. 2021. Variation in timing of breeding of five woodpeckers in a primeval forest over 45 years: role of food, weather, and climate. – Journal of Ornithology 162: 89–108. DOI: 10.1007/s10336-020-01817-1

Wiebe, K. L. 2001. Microclimate of tree cavity nests: is it important for reproductive success in Northern Flickers? – The Auk 118: 412–421. DOI: 10.1093/auk/118.2.412

Zahner, V., Sikora, L. & Pasinelli, G. 2012. Heart rot as a key factor for cavity tree selection in the Black Woodpecker. – Forest Ecology and Management 271: 98–103. DOI: 10.1016/j.foreco.2012.01.041

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Migration of Firecrest *(Regulus ignicapilla)* in Hungary

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Abstract The Firecrest *(Regulus ignicapilla)* is a regular, but small-number breeder in spruce and coniferous forests in the mountainous and hilly areas of Hungary, but is found in all parts of the country during migration. Despite this, only sporadic field observation data and ringing results in Western Hungary have been published so far. The aim of this study was to investigate the migration of this species on a national scale. In our work, we processed field observation data collected on the Csanádi-hát and capture-recapture data from three bird ringing stations (Szalonna, Tömörd and Ócsa) from the second half of the 1980s to 2021. We also used archive published field observation data from different parts of Hungary. Our results show that the spring migration of the species in the study areas took place between mid-March and late April – early May, and its timing did not differ significantly between regions. In contrast, the autumn migration was earlier in mountainous areas than in lowland areas. The small numbers and short duration of recaptured birds suggested a rapid migration in both periods. In autumn, we obtained a significant difference between the annual number of birds captured in Tömörd and Szalonna. In both periods, males were caught in greater numbers than females. In southern Hungary, the timing of migration in spring has not changed, but in autumn the species migrated earlier than a few decades ago.

Keywords: Firecrest, passerine migration, Carpathian Basin, bird ringing, field observations

Összefoglalás A tüzesfejű királyka *(Regulus ignicapilla)* Magyarország hegy- és dombvidéki lucosainak és luccal elegyes fenyveseinek rendszeres, de csak kisszámú fészkelője. Vonuláskor azonban az ország egész területén előfordul. Ennek ellenére a fajról eddig csak szórványos terepi megfigyelési adatokat, illetve nyugat-magyarországi gyűrűzési eredményeket publikáltak. Jelen dolgozatban az volt a célunk, hogy országos szinten vizsgáljuk meg a faj vonulását. Munkánk során a Csanádi-háton gyűjtött terepi megfigyelési adatokat, valamint három madárgyűrűző-állomás (Szalonna, Tömörd és Ócsa) gyűrűzés–viszafogási adatait dolgoztuk fel az 1980-as évek második fele és 2021 közötti időszakból. Emellett ennél korábbi, Magyarország különböző részeiről származó, publikált megfigyelési adatokat is felhasználtunk. Eredményeink szerint a faj tavaszi vonulása a vizsgálati területeken március közepe és április vége – május eleje között zajlott, és annak időzítése nem különbözött szignifikánsan a régiók között. Ezzel szemben ősszel a hegyvidéki területeken korábban vonult át a faj, mint síkvidéken. A kisszámú visszafogás és a visszafogott madarak rövid tartózkodási ideje alapján mindkét időszakban gyors vonulásra következtettünk. Ősszel Tömördön és Szalonnán szignifikáns különbséget kaptunk az évente fogott madarak számában. A hímek mindkét időszakban gyakrabban akadtak hálóba, mint a tojók. Dél-Magyarországon tavasszal nem változott a faj vonulásának időzítése, ugyanakkor ősszel hamarabb vonult át a faj napjainkban, mint néhány évtizeddel korábban.

Kulcsszavak: tüzesfejű királyka, énekesmadár-vonulás, Kárpát-medence, madárgyűrűzés, terepi megfigyelések

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Introduction

The Firecrest – *Regulus ignicapilla* (Temminck, 1820) – is a polytypic species with four subspecies occurring from Western Europe to the Caucasus and North Africa (Martens & Päckert 2020). In Hungary, it is a regular, but only a small-number breeder (Hadarics & Zalai 2008). Its first breeding in Hungary was recorded in the Őrség region (Barbácsy 1978), and subsequently it has been recorded nesting in Börzsöny Hills (Varga 1980), Kőszeg Hills (Barbácsy 1981), Bükk Hills (Kasza 1981) and Sopron Hills (Varga 1982). It mainly breeds in homogeneous or mixed spruce forests (Haraszthy 2019), but it can also colonise urban parks (Barbácsy 2000). During the 2010s, probable or confirmed breeding was recorded in the Alpokalja, Zala Hills, Kemeneshát, Mecsek and the Northern Hills (Csörgő & Gyurácz 2021). In some years, the number of nesting pairs may increase in certain areas (Haraszthy 2019), but since the 2000s the population has declined mainly due to the destruction of spruce forests caused by climatic factors (Varga 2008, Gyurácz & Csörgő 2009). The Hungarian population was estimated at 400–500 pairs between 2014 and 2018 (Csörgő & Gyurácz 2021). In Italy, the number of ringed birds increased between 1982 and 2003 (Spina & Volponi 2009). During its expansion in the 20th century, the species colonised the British Isles, and the breeding population increased in Belgium, the Netherlands and Denmark (Martens & Päckert 2020). Over the past two decades, it has expanded significantly northwards into the British Isles, Sweden and the Baltic countries (Wilk & Knaus 2019).

Central and Eastern European populations are partial migrants, with a small proportion overwintering in mild winters (Cramp & Brooks 1992). They begin their autumn migration at the earliest 7–8 days after the summer molt is completed. In Central Europe they leave the breeding areas by the end of July. Their wintering areas are mainly on the Mediterranean coast and in Western Europe and the British Isles (Martens & Päckert 2020), where they may arrive as early as September, but typically only in November–December (Cramp & Brooks 1992). In Denmark, it is a rare breeding species and passage migrant in March – April and September–October (Bønløkke *et al.* 2006). In Italy, it migrates in spring, between mid-March and mid-April, when it is much more numerous than in autumn. The autumn migration takes place between September and November, with fairly extended movements in Northern and Central Italy based on recaptures (Spina & Volponi 2009). In Croatia, it migrates in March in spring and in September and October in autumn (Kralj 2013).

In Hungary, they can be found in almost any woody and shrubby habitat during migration, even in lowland areas (Hadarics & Zalai 2008). They typically move in small groups, often with Goldcrests *(Regulus regulus)* (Gyurácz & Csörgő 2009). The origin and wintering areas of migrants are unknown due to lack of recaptures (Gyurácz & Csörgő 2009). In spring, birds migrate between late February and early May, with a peak in late March and early April, while in autumn, migrating Firecrests can be seen between late August and early December, with a peak in early October (Gyurácz & Csörgő 2009). Based on 311 individuals ringed in Tömörd, Western Hungary, between 1998 and 2007, the average start of autumn migration was on 16 September, the median on 6 October and end on 5 November. The earliest date of capture was on 17 August and the latest on 7 November (Gyurácz & Bánhidi 2008). The birds migrate quickly, with medium fat reserves and developed pectoral muscles, and spend only a short time at the stopover sites (Gyurácz & Csörgő 2009). In the period between 1998–2007, birds spent 2–4 days in the stopover sites (Gyurácz & Bánhidi 2008).

In Hungary, regarding to their migration, only sporadic field observations are known, and some of these are from the known breeding sites. Csaba (1955), Anon. (1980), Varga and Király (1981), Varga (1982, 1996, 2003), Molnár (1983, 1985, 1988) and Kelemen (2003) mentioned data from the Őrség, Sopron Hills and Kőszeg Hills. Homoki-Nagy (1981) reported it as a rare, but regularly occurring species from the Börzsönyi Landscape Protection Area. From the Bükk, Moskát (1975), Barta (1978) and Anon. (1979a) published field observation data, while in the Karancs–Medves area two ringed individuals were reported by Varga (1978). Numerous published records are known from and near Budapest (Dénes 1979, Anon. 1981, Schmidt 1983, Molnár, 1988, Bajor 1994), as well as from the Gerecse (Beretzk 1963), Hévíz (Keve 1976), Hanság (Molnár 1982a) and Dombóvár (Nagy 1982).

In the lowlands, it was also seen regularly during the migration period. Most of the published data come from Szeged and its sorroundings, where it was observed during both the spring and autumn migration (Anon. 1978a, 1979b, Molnár 1982b, 1984b, 1985, 1988, Kasza 1981, 1989). Here, the species regulary overwintered (Kasza 1981, 1989). Overwintering individuals have also been recorded in Zsombó (Csongrád-Csanád County), and it is noteworthy that they were more frequently observed here in the 1980s than from the mid-1990s (Mészáros 2000). In the Fertő-Lapistó area of Szentes, it was a rare migrant in the period between 1968 and 1998 (Tóth 2000). Pásti (2000) describes it as an occasional migrant in the Tócó valley in Debrecen, and here was an overwintering individual too. It has also been recorded in Berettyóújfalu and Császárszállás (Anon. 1979b), Gödöllő (Molnár 1982a), Balmazújáros, Tömörkény and Tápiószecső (Molnár 1984a), Zagyvaróna, Mezőkövesd and Hajdúnánás (Molnár 1985). It is not rare in Békés County either. It is mentioned as a "sparse" species in the Biharugrai fishponds as early as the 1950s (Nagy 1961). In Biharugra, Vasas (1999) saw one individual on 22 October 1999, while Vasas and Zalai (1998) described it as a regular migrant in April. In 2002, a small number of migrants were seen in mid-March (Tőgye 2002), and in 2003, one observation was made on 5 April (Tőgye 2003). Two birds were seen in Körösladány on 2 April 1978 (Anon. 1978b), two in Sarkadremete on 26 November 1978 (Anon. 1979b), one in Békéscsaba in March 1979 (Molnár 1982d), and 2–4 in the Gulyagyep, Battonya on 9 April 2000 (Csathó 2009). An occasional migrant was observed along the Kettős-Körös river in 2005, where one individual was observed on 22 February 2009 (Durkó 2009). It was a regular spring (late March – mid-April) and autumn (late September – early November) migrant in Kevermes between 2004 and 2017 (Bozó 2017).

A detailed study of the migration of Firecrest in Hungary has only been carried out in Tömörd, and only sporadic data are known from other parts of the country (Gyurácz & Csörgő 2009).

In the present study, we investigated the migration of the species based on data collected in Tömörd in Western Hungary, Ócsa in Central Hungary, Szalonna in Northern Hungary and in the Csanádi-hát region in Southeastern Hungary. On the basis of data published in the literature, we also looked at when the species migrates through different parts of the country, and whether there are differences in the timing of migration between geographical regions? We also compared our own data with these datasets. Our work is intended to contribute to a better understanding of the migration of the Firecrest, of which migration, although not a rare species, is very poorly known in Central Europe.

Material and Methods

The dataset based on field observations is available from the area of the Csanádi-hát, a small geographical area in the southern part of Békés County. Regular observations were carried out in two settlements, Kevermes and Battonya, while data from the other settlements were collected only during occasional field visits. Data series are available from Battonya since 1996 and from Kevermes since 2004.

Today, the landscape is dominated by agricultural land, which typically accounts for around 90% of the total area with the exception of a few more forested settlements, such as Mezőhegyes (Hevesi 2005). Along the banks of the meandering Száraz-ér, along artificial drainage channels and mining lakes, and in populated areas, wooded shrub habitats are found, all of which are artificially planted, dominated by euro-american hybrid poplar *(Populus × euramericana)*, black locust *(Robinia pseudoacacia)*, common hackberry *(Celtis occidentalis)* and black elder *(Sambucus nigra)*, but also by elm (*Ulmus* spp.). Ash (*Fraxinus* spp.), wild pear *(Pyrus pyraster)*, walnut *(Juglans regia)* and common oak *(Querqus robur)* are also present (Bozó 2018). The observations were made by walking around the area, using binoculars and a camera. The presence of the birds was usually indicated by their distinctive call, but this was not always sufficient to distinguish them from the Goldcrest, so only those individuals that were seen with certainty were recorded as Firecrests. Given that in autumn the leaves are still on the trees when the species migrates, it was necessary to leave several *Regulus* species moving in the canopy undetermined. We considered spring to be between 15 February and 10 May and autumn between 1 September and 30 November.

In addition to the field observation data, we also used ringing data from 3 Actio Hungarica camps (Szalonna, Tömörd, Ócsa) in our analysis. For descriptions of the three sites and the ringing work see Farkas *et al.* (2014), Csörgő *et al.* (2016) and Gyurácz *et al.* (2017). When defining migration periods, only new ringings were considered, not recaptures (Szalonna: 920, Tömörd: 768, Ócsa: 167 individuals).

In the case of the autumn data for Tömörd and Szalonna, it was also possible to compare the differences in the number of birds captured between years. To do this, we used Kruskal–Wallis test. We also used the Kruskal–Wallis test to detect differences in the timing of migration between study sites.

The average of the start of autumn and spring migration was taken as the average of the dates on which the species was first observed in autumn and spring, respectively.

In addition to current data, we have also collected available archive data. For this, we searched for data in the journal Cinege (Kelemen 2005, Varga 1996, 2003), the Aquila (Csaba 1955, Beretzk 1963, Sterbetz 1972, Keve 1976, Schmidt 1976, Rékási 1992), the Madártani Tájékoztató (Anon. 1978a, 1978b, 1979a, 1979b, 1980, 1981, Barta 1978, Simig 1978, Varga 1978, 1982, Dénes 1979, Kasza 1981, 1989, Varga & Király 1981, Molnár 1982a, 1982b, 1982c, 1982d, 1983, 1984a, 1984b, 1985, 1988, Nagy 1982, Schmidt 1983, Bajor 1994), the

Túzok (Hadarics 1996a, 1996b, 1996c, 1996d, 1997a, 1997b, 1997c, 1997d, 1998a, 1998b, 1998c, 1999a, 1999b, 1999c, 2000a, 2000b, 2000c, 2001a, 2001b), and the A Puszta (Vasas 1999, Tőgye 2003, Csathó 2009, Durkó 2009, 2015). Data were collected at regional level (Southern Great Hungarian Plain, Northern Great Hungarian Plain, Northern Hills, Budapest area, Southern Transdanubia, Northern Transdanubia, Western Transdanubia), and they were compared with each other. Since the migration peak was defined by its median value, we performed a Kruskal-Wallis test for comparisons. If the Kruskal–Wallis test indicated significant differences, we applied Dunn's post hoc test to find the exact differences among groups. We also compared our present data from the Csanádi-hát with the archive data from the southeastern part of the Greater Hungarian Plain by the use of Mann-Whitney U test. For statistical analysis, we used the Past statistical program (Hammer *et al.* 2001).

Results

Csanádi-hát

In spring, we observed 102 individuals on 71 occasions, while in autumn, 42 individuals on 32 occasions. The spring migration took place between mid-March (earliest sighting: 15 March 2020, Battonya) and the second half of April (latest sighting: 27 April 2016, Kevermes), with a median date on 31 March *(Figure 1)*.

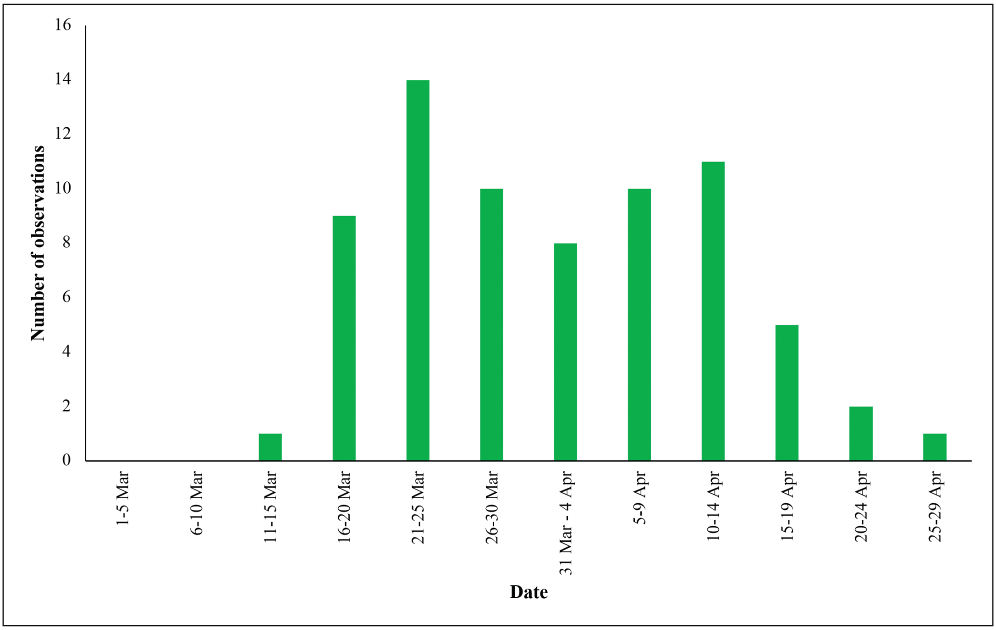

Figure 1. Numbers of Firecrest during the spring migration in the Csanádi-hát (SE Hungary)
1. ábra A tüzesfejű királyka megfigyelései a tavaszi vonulás alkalmával a Csanádi-háton (Délkelet-Magyarország)

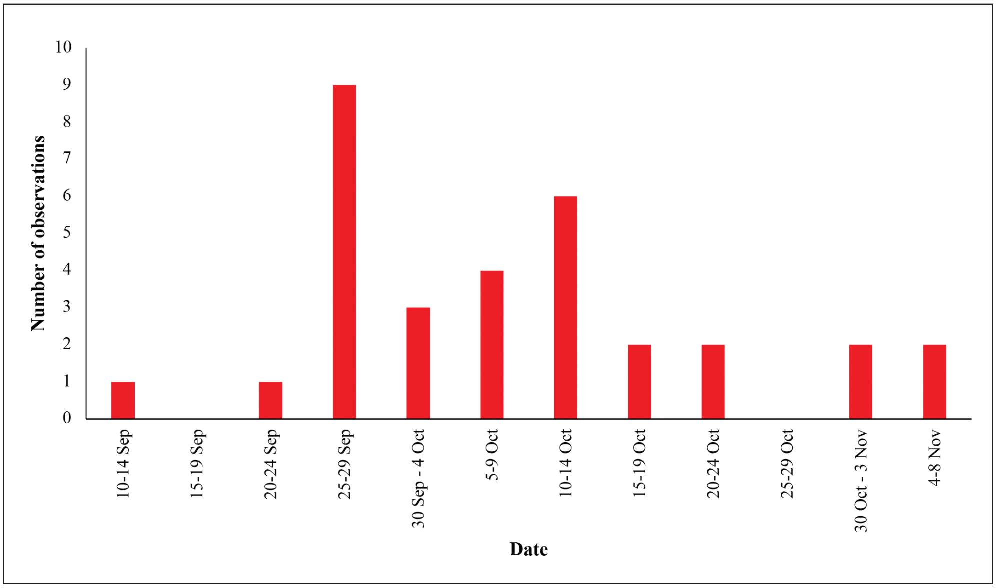

Figure 2. Numbers of Firecrest during the autumn migration in the Csanádi-hát (SE Hungary)

2. ábra A tüzesfejű királyka megfigyelései az őszi vonulás alkalmával a Csanádi-háton (Délkelet-Magyarország)

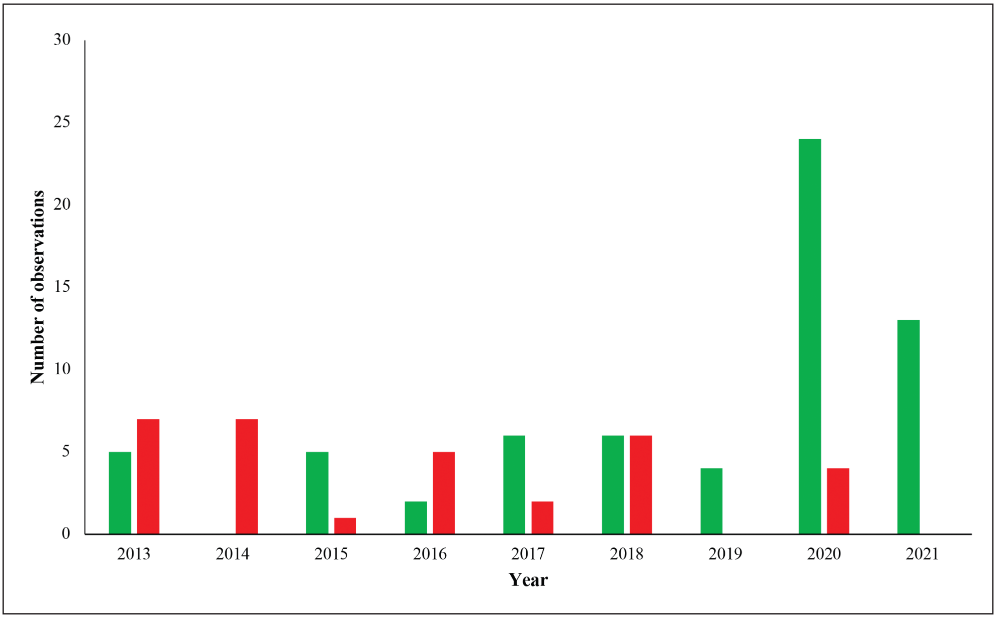

Figure 3. Number of annual observations of the Firecrests in the Csanádi-hát between 2013 and 2021. Green bars indicate spring data and red bars indicate autumn data

3. ábra A tüzesfejű királyka éves megfigyeléseinek száma a Csanádi-háton 2013 és 2021 között. A zöld szín a tavaszi, a piros szín az őszi adatokat jelöli

In autumn, the species migrated through the area between mid-September (earliest sighting: 11 September 2018, Kevermes) and early November (latest sighting: 8 November 2016, Kevermes), with the median date of migration being on 6 October *(Figure 2)*.

There is only one known winter record in the area: on 4 December 2016, a solitary individual was seen in the centre of Battonya, feeding on spruce trees.

It was not possible to compare the number of birds observed each year using statistical methods, but in the spring of 2020 and 2021 the number of observations increased compared to the previous years *(Figure 3)*.

Szalonna

Bird ringing was conducted only in autumn. Between 1986 and 2020, a total of 920 individuals were ringed and a total of 9 birds were recaptured on 11 different occasions.

The migration took place between 17 August and 3 November, with a median on 28 September. The migration was prolonged, but a peak was observed at the turn of September and October *(Figure 5)*.

Between 1996 and 2020, it was also possible to examine the differences between years, and it was a significant difference among the median values for the different years ($H=105.8$, $P< 0.001$).

The recaptured birds spent 1–6 days in the area, with an average of 3.2 days.

68.8% of the identified sexes were males (577 individuals), while only 31.2% were females (262 individuals).

Tömörd

Between 1984 and 2021, a total of 768 birds were ringed during the study period (spring: 30, autumn: 738), and a total of 59 birds were recaptured on 76 different occasions (spring: 6 birds 9 times, autumn: 53 birds 67 times). In one case, a bird was ringed in December (14 December 1997).

The spring migration took place between 16 March and 2 May, with a median on 27 March *(Figure 4)*. The six recaptured birds spent 1–5 days in the area, with an average of 2 days. In spring, 74.1% of the individuals with a defined sex were males (20 individuals), while only 25.9% were females (7 individuals).

The autumn migration took place between 7 August and 26 November, with a median on 6 October. The migration was prolonged, with similar numbers of birds caught between mid-September and mid-October *(Figure 5)*.

In the autumn period between 1998 and 2020, it was also possible to examine the differences between years regarding to the migration timing, with a significant difference in the median values for the years ($H=44.87$, $P=0.003$).

Recaptured birds spent 1–7 days in the area, with only two individuals staying longer than a week (28 and 55 days). Apart from these two outliers, the average stopover time was 2.4 days. There was also one bird ringed on 14 October 2018 and was trapped again one year later on 10 October 2019.

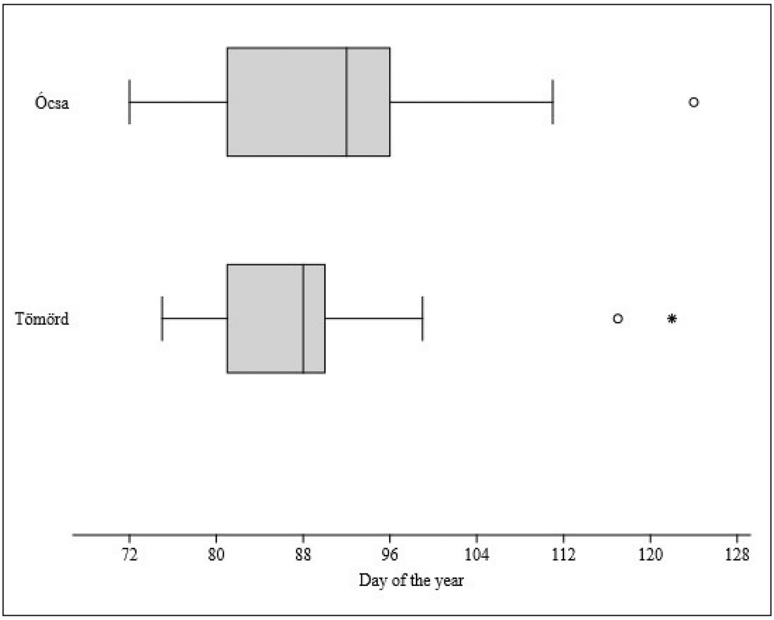

Figure 4. The timing of spring migration of Firecrest in Ócsa (CE Hungary) and Tömörd (W Hungary)

4. ábra A tüzesfejű királyka tavaszi vonulásának időzítése Ócsán (Közép-Magyarország) és Tömördön (Nyugat-Magyarország)

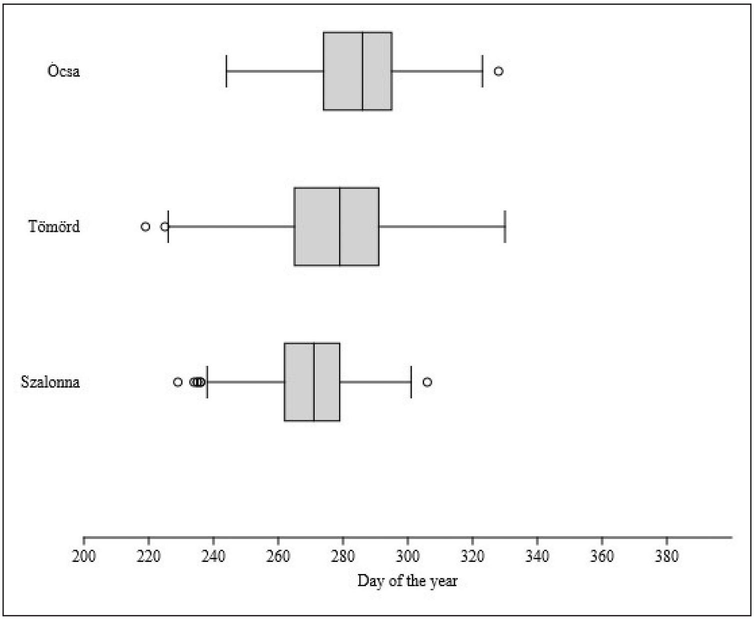

Figure 5. The timing of autumn migration of Firecrest in Ócsa (CE Hungary), Tömörd (W Hungary) and Szalonna (NE Hungary)

5. ábra A tüzesfejű királyka őszi vonulásának időzítése Ócsán (Közép-Magyarország), Tömördön (Nyugat-Magyarország) és Szalonnán (Északkelet-Magyarország)

In autumn, 69.1% of the individuals of the specified sex were males (466 individuals), while the proportion of females was only 30.9% (208 individuals).

Ócsa

Between 1984 and 2021, a total of 167 birds were ringed during the study period (spring: 49, autumn: 118), and a total of 21 birds were recaptured on 24 different occasions (spring: 10 birds 12 times, autumn: 10 birds 11 times). One was trapped in December (3 December 2009) and one in January (7 January 2010).

The spring migration took place between 13 March and 4 May, with a median on 2 April *(Figure 4)*. The 10 recaptured birds spent 1–5 days in the area, with an average of 2.3 days. In spring, 56.5% of the individuals with a defined sex were males (26 individuals), while the proportion of females was 43.5% (18 individuals).

The autumn migration took place between 1 September and 24 November, with a median on 13 October. The migration was prolonged, with similar numbers of birds caught between late September and late October *(Figure 5)*. Recaptured birds spent 1–5 days in the area, with an average of 2.1 days.

In autumn, 58.5% of individuals with a defined sex were males (62 individuals), while the proportion of females was 41.5% (45 individuals).

Among ringing sites, there was no significant difference between the migration timing of spring migration (H=3.373, P=0.185) (earliest in Tömörd and latest in Ócsa). In contrast, there was a significant difference in the migration timing in autumn (H=159.2, P < 0.0001) (earliest in Szalonna and latest in Ócsa). The sex ratios were 7:3 in favour of males in Tömörd and Szalonna in autumn and spring, while in Ócsa the ratio was much more equal, 6:4.

Archive data

Based on archive data collected from different regions of the country, the timing of the species' migration in spring differed significantly among regions (H=9.561, P=0.048). Namely, the timing of migration of birds migrated in Budapest differed significantly from

Table 1. The migration timing of the Firecrest in different regions of Hungary
1. táblázat A tüzesfejű királyka vonulásának időzítése Magyarország különböző régióiban

Region	Median		Minimum		Maximum		N	
	Spring	Autumn	Spring	Autumn	Spring	Autumn	Spring	Autumn
Northern Great Plain	2 Apr	15 Oct	15 Mar	16 Sep	3 May	5 Dec	26	26
Southern Great Plain	3 Apr	18 Oct	14 Feb	17 Sep	26 Apr	10 Dec	25	38
Northern Hills	8 Apr	1 Oct	17 Feb	9 Sep	10 May	14 Nov	18	10
South Transdanubia	–	5 Nov	–	14 Sep	–	30 Nov	5	16
North Transdanubia	–	–	–	–	–	–	1	5
West Transdanubia	3 Apr	6 Oct	8 Mar	14 Sep	30 Apr	8 Dec	52	31
Budapest	29 Mar	11 Oct	18 Feb	14 Sep	15 Apr	21 Nov	40	28

the Southern Great Plain (P=0.029), Northern Hills (P=0.033) and Western Transdanubia (P=0.005). The earliest time of return was in Budapest, while the latest time of return was in the Northern Hills (the difference between the median time of migration was 10 days). In autumn there was also a significant difference among regions (H=10.94, P=0.027), and the timing of migration differed significantly among most regions (P<0.05). At this time, the migration started in the Northern Hills and finished in the South Transdanubian region (difference between the median of migration was 35 days). The latter value may include data from overwintering birds, but the north-south temporal shift is also evident when considering the other areas *(Table 1)*.

Comparing the present data from the Csanádi-hát with the archive data from the Southern Great Plain, we found that there was no significant difference in the timing of migration in spring (Mann–Whitney U test, z=–1.88, P=0.59), but there was in autumn, as the species migrated through the Southern Great Plain earlier in the 1970s and 1980s than today (z= –2.91, P=0.003).

Discussion

The spring migration of the Firecrest in the study areas took place between mid-March and late April – early May, and the timing of the migration did not differ significantly among geographical regions. It means that the species appears almost simultaneously in different parts of the country in spring. The most probable reason for this is that the species migrates rapidly over long distances (Gyurácz & Csörgő 2009), with most birds spending only a short time in a given area. The migration of the Goldcrest is also similar in Northern Europe (Petterson & Hasselquist 1985) and in the Carpathian Basin (Miklay & Csörgő 1998, Gyurácz *et al.* 2003). In Ócsa and Tömörd, we caught far fewer birds in spring than in autumn, but on the Csanádi-hát both the number of observations and the number of birds were much higher in spring. One reason for this phenomenon is probably the detectability of the species, as in spring the species migrated before the trees were leafed out, which made it easier to observe and identify the birds. In autumn, however, more birds had to be left undetermined, as it is often very difficult to see the *Regulus* species at this time of year and identification by sound is not always straightforward. It is worth mentioning that a similar pattern was found in Italy in case of the annual numbers of ringed birds between 1982 and 2003 (Spina & Volponi 2009). In the context of spring migration, it is worth mentioning the late observation and capture data, as the members of the Hungarian populations are already on breeding areas in March and already engaged in nest building in early April (Haraszthy 2019). For this reason, it is likely that most of the migrating Firecrests belong to northern European populations rather than to the Carpathian Basin. In several species, such as the Peregrine Falcon *Falco peregrinus* (Bozó 2021), the Ural Owl *Strix uralensis* (Bozó 2019), the Coal Tit *Periparus ater* (Bozó 2020a) and the Eurasian Siskin *Spinus spinus* (Bozó 2020b) have been assumed that birds occured in the Csanádi-hát from autumn to spring originate from the Apuseni Mountains, and it cannot be excluded that Firecrests also come from that area. However, no evidence of this was found for this species, as the timing of migration does not differ from that observed in other parts of the country.

In autumn, there was a significant difference in the timing of migration, with migration typically taking place later in lowland areas than in mountainous areas. This was most noticeable in the timing of the occurrance of the first birds, as while the first birds were sometimes seen in Tömörd and Szalonna in mid-August, the first individuals were caught and observed in Ócsa and in the Csanádi-hát about a month later. The reason for this phenomenon is probably that while the species might nests near Tömörd and Szalonna, it does not breed in the Great Plain (Hadarics & Zalai 2008), so that migrating Firecrests may appear earlier in the former areas during post-breeding dispersal. A single long-term recapture was also recorded in Tömörd, which also confirms the hypothesis that the members of the breeding population were also captured during migration periods. Our results for the autumn migration are in agreement with Gyurácz and Csörgő (2009), who found that it is most intense in the first half of October. In autumn, as in spring, we have small numbers of recaptured birds at the ringing sites, and these individuals spent only a short time in the areas. As a result, their autumn migration was as rapid as in spring.

In winter, it was a rare, occasional visitor in each of the study areas, which is consistent with its previously known national status (Hadarics & Zalai 2008).

There was a significant difference in the timing of migration between years in the autumn in Tömörd and Szalonna. This may be mainly due to the weather in the respective years, as it influences both the timing of breeding and migration. If the weather was favourable in a given year, the birds could complete breeding and start migration on time, whereas in the opposite case, the end of breeding and the start of migration could be significantly delayed, even due to additional nesting. Of course, there may also be periods during migration that slow down fat accumulation, so that birds can only leave the stopover site later.

The male birds are generally more likley to be captured than females, but the differences in capture probability do not sufficiently explain why the sex ratio was male-biased (Lovász *et al.* 2018). In spring, male Firecrests were also more likely to be caught than females in both lowland and mountainous areas. The pattern in autumn is similar to that of the Goldcrest, with a twofold difference in favour of males found in this species in Ócsa and Ganda, Italy. In spring, however, the sex ratio was evenly balanced in this species (Miklay & Csörgő 1998), whereas in the present study, males were in a similar predominance in both Tömörd and Ócsa as in autumn. In the case of the Goldcrest, the equalisation of the sex ratio is due to higher mortality of males at wintering sites or on migration routes (Miklay & Csörgő 1998), but in the case of the Firecrest, it is likely to occur on migration routes towards breeding sites, further north of Hungary.

Based on field observations from different parts of the country in the 1970s and 1990s, a clear migration pattern emerged, with the species migrating first to the southern parts of the country in spring and then to the northern parts in autumn. This pattern was not supported by our own data, so it is possible that the migration pattern of the species has changed in recent decades. However, we do not have sufficient data to investigate this precisely.

In the 1970s and 1980s, the species migrated through the Southern Great Hungarian Plain earlier in the autumn than it does today, but there has been no change in the timing of its spring migration. Changes in migration timing are typically driven by climate change (e.g. Sparks *et al.* 2002, Lehikoinen *et al.* 2004, Van Buskirk *et al.* 2009, Bozó & Csörgő 2020). In this case, however, the pattern obtained may be more an artificial product of low sample size than an effect of climate change. A longer-term, larger sample-sized data set would be needed to determine this.

References

Anon. 1978a Faunisztika néhány sorban [Faunistics in short]. – Madártani Tájékoztató 1(6): 25–30. (in Hungarian)

Anon. 1978b Faunisztika néhány sorban [Faunistics in short]. – Madártani Tájékoztató 1(3): 20–27. (in Hungarian)

Anon. 1979a Faunisztika néhány sorban [Faunistics in short]. – Madártani Tájékoztató 2(2): 33–37. (in Hungarian)

Anon. 1979b Faunisztika néhány sorban [Faunistics in short]. – Madártani Tájékoztató 2(1): 19–23. (in Hungarian)

Anon. 1980. Faunisztika néhány sorban [Faunistics in short]. – Madártani Tájékoztató 3(3): 45–48. (in Hungarian)

Anon. 1981. Faunisztika néhány sorban [Faunistics in short]. – Madártani Tájékoztató 4(2): 111–115. (in Hungarian)

Bajor, Z. 1994. Megfigyelési adatok az Óbudai- (Hajógyári-) szigetről [Bird observation data from the Óbudai- (Hajógyári-) Island]. – Madártani Tájékoztató 17(2): 19–20. (in Hungarian)

Barbácsy, Z. 1978. Tüzesfejű királyka *(Regulus ignicapillus)* fészkelése hazánkban [Breeding of Firecrest *(Regulus ignicapillus)* in Hungary]. – Madártani Tájékoztató 2(5): 6–8. (in Hungarian)

Barbácsy, Z. 1981. Újabb adatok a tüzesfejű királyka *(Regulus ignicapillus)* előfordulásához Vas megyében 1979–1980-ban [Data to the occurences of Firecrest *(Regulus ignicaoillus)* in Vas County in 1979–1980]. – Madártani Tájékoztató 5(4): 211–212. (in Hungarian)

Barbácsy, Z. 2000. Tüzesfejű királyka *Regulus ignicapillus* [Firecrest *Regulus ignicapillus*]. – In: Haraszthy L. (ed.) Magyarország madarai. Második, javított kiadás [Birds of Hungary Second, revised edition]. – Mezőgazda Kiadó, Budapest, pp. 322–323. (in Hungarian)

Barta, Z. 1978. Adatok a Bükk-hegység madárfaunájához [Data to the bird fauna of the Bükk Hill]. – Madártani Tájékoztató 1(1): 7–12. (in Hungarian)

Beretzk, I. 1963. Adatok a Dunántúl madárvilágához [Data to the bird fauna of Transdanubia]. – Aquila 69–70: 260–266. (in Hungarian)

Bønløkke, J., Madsen, J. J., Thorup, K., Pedersen, K. T., Bjerrum, M. & Rahbek, C. 2006. Dansk Trækfugleatlas [The Danish Bird Migration Atlas]. – Rhodos, Humlebæk, Denmark, p. 685.

Bozó, L. 2017. Kevermes madárvilága [Birds of Kevermes]. – Dél-békési Természetvédelmi és Madártani Egyesület, Kevermes (in Hungarian)

Bozó, L. 2018. Dél-Békés természeti értékei [Nature assets of South-Békés County]. – Private edition, Kevermes (in Hungarian)

Bozó, L. 2019. Az uráli bagoly *(Strix uralensis)* újabb előfordulása a Csanádi-háton [Another occurrence of the Ural Owl *(Strix uralensis)* on the Csanádi-hát, SE Hungary]. – Aquila 126: 106–107. (in Hungarian and English)

Bozó, L. 2020a Néhány madárfaj inváziós mozgalmai egy délkelet-magyarországi síkvidéki területen [Invasion movements of certain bird species in a lowland study area in Southeast-Hungary]. – Aquila 127: 11–22. (in Hungarian with English Summary)

Bozó, L. 2020b A csíz *(Spinus spinus)* előfordulása Délkelet-Magyarországon [The occurrence of Eurasian Siskin *(Spinus spinus)* in Southeast Hungary]. – Állattani Közlemények 105(1–2): 59–69. (in Hungarian with English Summary)

Bozó, L. 2021. Adatok néhány ragadozómadár-faj dél-békési előfordulásához [Data to occurrence of some raptor species in southern part of Békés county]. – Heliaca 17: 101–107. (in Hungarian with English Summary)

Bozó, L. & Csörgő, T. 2020. Changes in spring arrival dates of Central European bird species over the past 100 years. – Acta Zoologica Academiae Scientiarum Hungaricae 66(3): 283–298.

Cramp, S. & Brooks, D. J. 1992. Handbook of the birds of Europe, the Middle East and North Africa: the birds of the Western Palearctic. Vol. VI. Warblers. – Oxford University Press, Oxford

Csaba, J. 1955. Ornithofaunisztikai adatok Csákánydoroszlóból [Ornithofunistical data from Csákánydoroszló]. – Aquila 59–62: 201–203. (in Hungarian)

Csathó, A. I. 2009. A battonyai Gulyagyep élővilága [The flora and fauna of the Gulyagyep meadow near Battonya town (SE Hungary)]. – A Puszta 23: 201–257. (in Hungarian with English Summary)

Csörgő, T. & Gyurácz, J. 2021. Tüzesfejű királyka [Firecrest]. – In: Szép, T., Csörgő, T., Halmos, G., Lovászi, P., Nagy, K. & Schmidt, A. (eds.) Magyarország madáratlasza [Bird Atlas of Hungary]. – Agrárminisztérium, Magyar Madártani és Természetvédelmi Egyesület, Budapest, pp. 583–584. https://mme.hu/birdatlas (in Hungarian with English Summary)

Csörgő, T., Harnos, A., Rózsa, L., Karcza, Zs. & Fehérvári, P. 2016. Detailed description of the Ócsa Bird Ringing Station, Hungary – Location, methods and overview of results (1984–2015). – Ornis Hungarica 24(2): 91–108. DOI: 10.1515/orhu–2016–0018

Dénes, J. 1979. További adatok Vác és környéke madárvilágához [Additional data to the avifauna of Vác and its surroundings]. – Madártani Tájékoztató 2(3): 20–22. (in Hungarian)

Durkó, L. 2009. Kettős-Körös madárállomány-mozgalmának vizsgálata 2005-ben [The study of the avifauna of the Kettős-Körös river in 2005]. – A Puszta 23. 115–173. (in Hungarian)

Farkas, R., Huber, A. & Gáti E. 2014. Fészkelő és vonuló madárfajok állományainak vizsgálata a Bódva-völgyben [A study on the populations of breeding and migrating birds species in the Bódva valley]. – ANP Füzetek XII. Aggteleki Nemzeti Park Igazgatóság, Jósvafő (in Hungarian with English Summary)

Gyurácz, J. & Bánhidi, P. 2008. Dynamics and spatial distribution of migratory birds. Results of Bird Ringing Project of Tömörd Bird Observatory 1998–2007. – Institute of Biology, Faculty of Sciences and Technics, Savaria Campus University of West Hungary, Szombathely

Gyurácz, J., Bánhidi, P., Góczán, J., Illés, P., Kalmár, S., Koszorús, P., Lukács, Z., Németh, C. & Varga, L. 2017. Bird number dynamics during the post-breeding period at the Tömörd Bird Ringing Station, western Hungary. – Ring 39(1): 23–82. DOI: 10.1515/ring-2017-0002

Gyurácz, J. & Csörgő, T. 2009. Tüzesfejű királyka [Firecrest]. – In: Csörgő, T., Karcza, Zs., Halmos, G., Magyar, G., Gyurácz, J., Szép, T., Bankovics, A., Schmidt, A. & Schmidt, E. (eds.) Magyar Madárvonulási Atlasz [Hungarian Bird Migration Atlas]. – Kossuth Kiadó Zrt., Budapest, pp. 529–530. (in Hungarian with English Summary)

Gyurácz, J., Góczán, J., Bánhidi, P. & Lepold, Á. 2003. Autumn migration of the Goldcrest *(Regulus regulus)* in western Hungary. – Ring 25(1–2): 37–46. DOI: 10.2478/v10050-008-0072-4

Hadarics, T. & Zalai, T. 2008. Magyarország madarainak névjegyzéke. Nomenclator avium Hungariae. An annotated list of the birds of Hungary. – Magyar Madártani és Természetvédelmi Egyesület, Budapest (in Hungarian and English)

Hadarics, T. 1996a Érdekes madármegfigyelések 1995/1996 tél (november–január) [Interesting bird observations in the winter of 1995/1996 (November–January)]. – Túzok 1(1): 49–53. (in Hungarian)

Hadarics, T. 1996b Érdekes madármegfigyelések 1996 tavasz (február–április) [Interesting bird observations in the spring of 1996 (February–April)]. – Túzok 1(2): 93–100. (in Hungarian)

Hadarics, T. 1996c Érdekes madármegfigyelések 1996. május–július [Interesting bird observations between May and July 1996]. – Túzok 1(3): 133–141. (in Hungarian)

Hadarics, T. 1996d Érdekes madármegfigyelések 1996. augusztus–október [Interesting bird observations between August and October 1996]. – Túzok 1(4): 168–181. (in Hungarian)

Hadarics, T. 1997a Érdekes madármegfigyelések 1996. november – 1997. január [Interesting bird observations between November 1996 and January 1997]. – Túzok 2(1): 38–47. (in Hungarian)

Hadarics, T. 1997b Érdekes madármegfigyelések 1997. február–április [Interesting bird observations between Februar and April 1997]. – Túzok 2(2): 71–82 (in Hungarian)

Hadarics, T. 1997c Érdekes madármegfigyelések 1997. május–július [Interesting bird observations between May and July 1997]. – Túzok 2(3): 115–126. (in Hungarian)

Hadarics, T. 1997d Érdekes madármegfigyelések 1997. augusztus–október [Interesting bird observations between August and October 1997]. – Túzok 2(4): 146–160. (in Hungarian)

Hadarics, T. 1998a Érdekes madármegfigyelések 1997. november – 1998. január [Interesting bird observations between November 1997 and January 1998]. – Túzok 3(1): 18–32. (in Hungarian)

Hadarics, T. 1998b Érdekes madármegfigyelések 1998. február–április [Interesting bird observations between Februar and April 1998]. – Túzok 3(2): 67–81. (in Hungarian)

Hadarics, T. 1998c Érdekes madármegfigyelések 1998. augusztus–október [Interesting bird observations between August and October 1998]. – Túzok 3(4): 168–185. (in Hungarian)

Hadarics, T. 1999a Érdekes madármegfigyelések 1998. november – 1999. január [Interesting bird observations between November 1998 and January 1999]. – Túzok 4(1–2): 33–45. (in Hungarian)

Hadarics, T. 1999b Érdekes madármegfigyelések 1999. február–április [Interesting bird observations between Februar and April 1999]. – Túzok 4(1–2): 46–56. (in Hungarian)

Hadarics, T. 1999c Érdekes madármegfigyelések 1999. augusztus–október [Interesting bird observations between August and October 1999]. – Túzok 4(4): 130–152. (in Hungarian)

Hadarics, T. 2000a Érdekes madármegfigyelések 1999. november – 2000. január [Interesting bird observations between November 1999 and January 2000]. – Túzok 5(1–2): 30–44. (in Hungarian)

Hadarics, T. 2000b Érdekes madármegfigyelések 2000. február–április [Interesting bird observations between Februar and April 2000]. – Túzok 5(1–2): 45– 56. (in Hungarian)

Hadarics, T. 2000c Érdekes madármegfigyelések 2000. augusztus–október [Interesting bird observations between August and October 2000]. – Túzok 5(3–4): 73–84. (in Hungarian)

Hadarics, T. 2001a Érdekes madármegfigyelések 2000. november – 2001. január [Interesting bird observations between November 2000 and January 2001]. – Túzok 6(1): 43–50. (in Hungarian)

Hadarics, T. 2001b Érdekes madármegfigyelések 2001. február–április [Interesting bird observations between Februar and April 2001]. – Túzok 6(2): 85–96. (in Hungarian)

Hammer, Ø., Harper, D. A. & Ryan, P. D. 2001. PAST: Paleontological statistics software package for education and data analysis. – Palaeontologia Electronica 4(1): 9.

Haraszthy, L. 2019. Tüzesfejű királyka [Firecrest]. – In: Haraszthy, L. (ed.) Magyarország fészkelő madarainak költésbiológiája. 2. kötet: Sárgarigóféléktől a sármányfélékig (Passeriformes) [Nesting birds of Hungary. Vol. 2. Oriolidae – Emberizidae (Passeriformes)]. – Pro Vértes Nonprofit Kft., Csákvár, pp. 537–541. (in Hungarian)

Hevesi, J. 2005. Határ mentén [Along the border]. – Békés Megyei Önkormányzat, Typografika Kft., Békéscsaba (in Hungarian and Romanian)

Homoki-Nagy, I. 1981. A Börzsönyi Tájvédelmi Körzet madárvilága az 1968–1979 közötti időszak megfigyelései alapján [Avifauna of the Börzsöny Landscape Protection Area between 1968–1979]. – Aquila 87: 23–25. (in Hungarian)

Kasza, F. 1981. Tüzesfejű királyka *(Regulus ignicapillus)* megfigyelések [Firecrest *(Regulus ignicapillus)* observations]. – Madártani Tájékoztató 5(2): 84. (in Hungarian)

Kasza, F. 1989. A tüzesfejű királyka *(Regulus ignicapillus)* áttelelési kísérlete [Overwintering Firecrest *(Regulus ignicapillus)*]. – Madártani Tájékoztató 12(1–2): 53. (in Hungarian)

Kelemen, T. 2003. Adatok a Vashegy madárvilágához [Data to the avifauna of Vashegy]. – Cinege 8: 36–37. (in Hungarian)

Kelemen, T. 2005. Madaras megfigyelések Szombathelyről [Bird observations from Szombathely]. – Cinege 10: 29–34. (in Hungarian)

Keve, A. 1976. Adatok a Kisbalaton madárvilágához II. [Data to the avifauna of Kisbalaton II.]. – Aquila 83: 191–226. (in Hungarian)

Kralj, J. 2013. Vatroglavi kraljić *(Turdis pilaris)* [Fieldfare *(Turdus pilaris)*]. – In: Kralj, J., Barišić, S., Ćiković, D. & Tutiš, V. (eds.) Atlas Selibde Ptica Hrvatske [Croatian Bird Migration Atlas]. – Croatian Academy of Sciences and Arts, Zagreb, pp. 169–170. (in Croatian with English Summary)

Lehikoinen, E., Sparks, T. H. & Zalakevicius, M. 2004. Arrival and departure dates. – Birds Climate Change 35: 1–31. DOI: 10.1016/S0065-2504(04)35001-4

Lovász, L., Roth, T., Karcza, Zs., Lukács, K. O., Gyurácz, J. & Amrhein, V. 2018. Survival, dispersal, and capture probability of male and female birds. – PeerJ Preprints 6: e27090v2. DOI: 10.7287/peerj.preprints.27090v2

Martens, J. & Päckert, M. 2006. Common Firecrest *(Regulus ignicapilla)*. – In: del Hoyo, J., Elliott, A., Sargatal, J., Christie, D. A. & de Juana, E. (eds.) Birds of the World. – Lynx Edition, Barcelona, p. 347.

Mészáros, Cs. 2000. A zsombói Ősláp-erdő madárvilága [Avifauna of the Ősláp-forest in Zsombó]. – A Puszta 17: 1–11. (in Hungarian)

Miklay, Gy. & Csörgő, T. 1998. What controls the sex ratio of Goldcrests *(Regulus regulus)*? – Ornis Hungarica 8 (Suppl. 1): 79–86.

Molnár, L. 1982a Faunisztika néhány sorban [Faunistics in short]. – Madártani Tájékoztató 5(1): 60–63. (in Hungarian)

Molnár, L. 1982b Faunisztika néhány sorban [Faunistics in short]. – Madártani Tájékoztató 5(4): 241–252. (in Hungarian)

Molnár, L. 1982c Adatok a Faunisztikai Szakosztály irattárából IV. [Data from the archives of the Department of Faunistics IV.]. – Madártani Tájékoztató 5(2): 64–74. (in Hungarian)

Molnár, L. 1982d Faunisztika néhány sorban [Faunistics in short]. – Madártani Tájékoztató 5(3): 176–181. (in Hungarian)

Molnár, L. 1983. Faunisztika néhány sorban [Faunistics in short]. – Madártani Tájékoztató 6(1): 22–25. (in Hungarian)

Molnár, L. 1984a Adatok a Faunisztikai Szakosztály irattárából VII. [Data from the archives of the Department of Faunistics VII.]. – Madártani Tájékoztató 7(1): 40–44. (in Hungarian)

Molnár, L. 1984b Adatok a Faunisztikai Szakosztály irattárából VIII. [Data from the archives of the Department of Faunistics VIII.]. – Madártani Tájékoztató 7(1): 44–53. (in Hungarian)

Molnár, L. 1985. Adatok a Faunisztikai Szakosztály irattárából IX. [Data from the archives of the Department of Faunistics IX.]. – Madártani Tájékoztató 8(1): 41–46. (in Hungarian)

Molnár, L. 1988. Adatok a Faunisztikai Szakosztály irattárából [Data from the archives of the Department of Faunistics]. – Madártani Tájékoztató 11: 56–78. (in Hungarian)

Moskát, Cs. 1975. A Karancs-Medves hegység madárvilága [Avifauna of the Karancs-Medves Hills]. – Aquila 82: 105–113. (in Hungarian)

Nagy, L. 1961. A volt Bihari Sárrét jelenlegi madárvilága [Data to the avifauna of the former Bihar-Sárrét]. – Aquila 67–68: 151–154. (in Hungarian)

Nagy, S. 1982. Adatok Dombóvár madárvilágához [Data to the avifauna of Dombóvár]. – Madártani Tájékoztató 5(2): 145–149. (in Hungarian)

Pásti, Cs. 2000. A Tócó-völgy madárvilága [Avifauna of the Tócó-valley]. – A Puszta 17: 158–192. (in Hungarian)

Pettersson, J. & Hasselquist, D. 1985. Fat deposition and migration capacity of Robins *Erithacus rubecula* and Goldcrests *Regulus regulus* at Ottenby, Sweden. – Ringing & Migration 6(2): 66–76.

Rékási, J. 1992. Adatok a dél-alföldi akácosok madárvilágához [Data to the avifauna of the acacia forest of the Southern Greatern Plain]. – Aquila 99: 137–148. (in Hungarian)

Schmidt, E. 1976. Faunisztikai jegyzetek 3. [Faunistical notes 3.]. – Aquila 83: 295–296. (in Hungarian)

Schmidt, E. 1983. Vonulási adatok a Gellérthegyről 1982 ősz [Bird migration data from the Gellérthegy in the autumn of 1982]. – Madártani Tájékoztató 6(1): 52–56. (in Hungarian)

Simig, L. 1978. Kiegészítő adatok a Budapesti Sas-hegy természetvédelmi terület madár faunájához [Additional data on the avifauna of the Sas-hegy Nature Reserve in Budapest]. – Madártani Tájékoztató 1(2): 5–8. (in Hungarian)

Sparks, T., Crick, H., Elkins, N., Moss, R., Moss, S. & Mylne, K. 2002. Birds, weather and climate. – Weather 57(11): 399–410. DOI: 10.1256/wea.142.02

Spina, F. & Volponi, S. 2009. Atlante della Migrazione degli Uccelli in Italia, Vol. 2. Passeriformi [Italian Bird Migration Atlas. Vol. 2.]. – ISPRA – MATTM, Roma, pp. 375–376. (in Italian with English Summary)

Sterbetz, I. 1972. A hódmezővásárhelyi Tisza-ártér természetvédelmi területeinek madárvilága [Avifauna of the nature reserves of the Tisza floodplain near Hódmezőváráshely]. – Aquila 78–79: 45–80. (in Hungarian)

Tóth, T. 2000. A szentesi Lapistó-Fertő tájtörténete, mai táji-természeti és kulturtörténeti értékei [The history of the Lapistó-Fertő in Szentes, its current landscape-natural and cultural-historical assets]. – A Puszta 17: 1–30. (in Hungarian)

Tőgye, J. 2002. A Biharugrai-halastavak madárállománya 2002-ben [Birds of the Biharugra Fishponds in 2002]. – A Puszta 19: 91–178. (in Hungarian)

Tőgye, J. 2003. Beszámoló a Biharugrai-halastavak madárállományáról és állapotáról 2003-ban [Birds of the Biharugra Fishponds in 2003]. – A Puszta 20: 81–134. (in Hungarian)

Van Buskirk, J., Mulvihill, R. S. & Leberman, R. C. 2009. Variable shifts in spring and autumn migration phenology in North American songbirds associated with climate change. – Global Change Biology 15(3): 760–771. DOI: 10.1111/j.1365-2486.2008.01751.x

Varga, F. 1978. Kiegészítés Moskát Csabának „A Karancs-Medves hegység madárvilága” c. dolgozatához [Supplement to the paper “The avifauna of the Karancs-Medves Hills” by Csaba Moskát]. – Madártani Tájékoztató 1(2): 8–11. (in Hungarian)

Varga, L. & Király, L. 1981. Faunisztikai adatok a Kőszegi Tájvédelmi Körzetből 1980 október – 1981 március [Avifaunistical data from the Kőszeg Landscape Protection Area between October 1980 and March 1981]. – Madártani Tájékoztató 4(3): 137–139. (in Hungarian)

Varga, L. 1982. Faunisztikai adatok Vas megyéből 1981. V–X. hó [Faunistical data from Vas County between May and October 1981]. – Madártani Tájékoztató 5(1): 45–47. (in Hungarian)

Varga, L. 1996. Néhány érdekes faunisztikai adat Vas megyéből [Some interesting faunistical data from Vas County]. – Cinege 1: 26–28. (in Hungarian)

Varga, L. 2003. A Vadása II. víztározón végzett madármegfigyelések összegzése [Summary of bird observations in the Vadása II. reservoir]. – Cinege 8: 29–35. (in Hungarian)

Varga, L. 2008. Átalakuló állami erdőgazdálkodás Vas megyében [Restructuring forest management in Vas county]. – Cinege 13: 7–13. (in Hungarian)

Varga, Zs. 1980. Tüzesfejű királykák a Börzsönyben [Firecrests in Börzsöny Hills]. – Madártani Tájékoztató 4(1): 23. (in Hungarian)

Vasas, A. & Zalai, T. 1998. Ritkább madarak előfordulása a Kis-Sárréten 1997–98 között [Rare birds in the Kis-Sárrét between 1997 and 1998]. – A Puszta 15: 291–294. (in Hungarian)

Vasas, A. 1999. Jelentés az 1999. évi madárállomány-változásról a Biharugrai, a Begécsi halastavakról és környékükről [Report on bird population changes in 1999 in and around the Biharugra and Begécs Fishponds]. – A Puszta 16: 32–83. (in Hungarian)

Wilk, T. & Knaus, P. 2019. *Regulus ignicapilla* Common Firecrest. – In: Keller, V., Hernando, S., Voříšek, P., Franch, M., Kipson, M., Milanesi, P., Martí, D., Anton, M., Klvaňová, A., Kalyakin, M. V., Bauer, H-G. & Foppen, R. P. B. 2020. European Breeding Bird Atlas 2. Distribution, Abundance and Change. – European Bird Census Council & Lynx Edicions, Barcelona, pp. 772–773.

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Algerian Nuthatch (*Sitta ledanti* Vielliard, 1976): Current challenges of an endemic species

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Abstract The Algerian Nuthatch *(Sitta ledanti)* is the emblematic bird species of Kabylia of Babors. It is a medium-sized passerine bird species of the Sittidae family. Jean-Paul Ledant discovered this species in October 1975, on the Babor Mountain. Twelve other forests in North-East Algeria has been discovered as breeding sites. These are the forests of the Babors region, in Kabylia. The Algerian Nuthatch was observed mainly in oak, cedar and pine habitats. It is mainly granivorous in winter and an insectivorous in spring/summer or breeding season, which starts in March and can last until July. The incubation period lasts from 14 to 17 days, with clutches of six eggs. The Algerian Nuthatch is listed as an endangered species by IUCN and is protected by the Algerian laws. However, many threats to the environment, such as tree felling and fires, are threatening the species.

Keywords: endemic species, *Sitta ledanti*, Algeria

Összefolglalás Az Atlasz-csuszka *(Sitta ledanti)* egy emblematikus faj Kabylia Baborsban. A Sittidae családba tartozó, közepes testméretű madárfajt Jean-Paul Ledant fedezte fel 1975 októberében a Babor-hegységben. Északkelet-Algéria 12 hegységében költ, főként tölgy-, cédrus- és fenyőerdőkben figyelték meg. Télen elsősorban növényi táplálékot fogyaszt, tavasszal és nyáron, vagyis a költési szezonban rovarokkal táplálkozik. Költési időszaka márciusban kezdődik, és július végéig tart. A kotlási idő 14–17 nap, a fészekalj általában 6 tojásból áll. Az Atlasz-csuszka az IUCN listáján veszélyeztetett fajként szerepel, valamint az algériai jogszabályok értelmében védettséget élvez. A fajt és élőhelyeit több tényező, de főként a fakitermelés és az erdőtüzek veszélyeztetik.

Kulcsszavak: endemikus fajok, *Sitta ledanti*, Algéria

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Introduction

Algeria, the second-largest country in Africa, is characterized by emblematic species, some of that are endemic to this country. Two examples are the Algerian Nuthatch *Sitta ledanti* Vielliard, 1976 which is regarded to breed strictly in the Kabylia of Babors (or Petite Kabylie region), and the Algerian fir *Abies numidica* which is solely located in the Babor forest of the same area. We can additionally distinguish the Atlas cedar *Cedrus atlantica* located in North Africa (Ledant 1977, Isenmann & Moali 2000).

The Algerian Nuthatch represents the flagship bird species of the Kabylia of Babors (north-eastern Algeria). Instead, knowledge about this species is limited. Few papers have mentioned the species only in general (Géroudet 1976, Bellatrèche 1999, Harrap 2002). However, data on breeding (Bougaham *et al.* 2017), feeding (Bellatrèche & Boubaker 1995, Mayache *et al.* 2020, Zemouri *et al.* 2021) and distribution (Bellatrèche & Chalabi 1990, Bougaham *et al.* 2018, Hamitouche *et al.* 2020) available in literature are summarised in this paper.

Description of the Kabylia of Babors

The Kabylia is a vast area of Northern Algeria stretching over more than 300 kilometres. It comprises four essential regions; the Grande Kabylie (or Djurdjurian Kabylia or Kabylia of Djurdjura), the Petite Kabylie (or Kabylia of Babors) in the North-East, the Kabylia of Collo (or Numidian Kabylia) in the extreme North-East, and the Kabylia of Bibans in the South-West (Ficheur 1890, Niox 1890, Dahmani 2004).

The Kabylia of Babors and of Collo are the wettest regions in Algeria due to their proximity to the Mediterranean Sea, their longitudinal gradient of rainfall, and their altitudes. The Kabylia of Babors region, indeed, receives about 1 meter of annual rainfall. Temperatures are particularly homogenous and vary from 0°C to 9 °C in winter and from 28 °C to 31 °C in summer (Ledant 1977, Vielliard 1978, De Smet & Bouaza 1984, Camps 1991, Benslimane *et al.* 2008). The Kabylia of Babors is thus rich in biodiversity, with a wide variety of plants and animals.

History of the Algerian Nuthatch's discovery

On 5 October 1975, Jean-Paul Ledant and his colleagues observed the Algerian Nuthatch for the first time on top of Babors forest in the country of Setif (in the Kabylia of Babors) (Géroudet 1976, Heim de Balsac 1976, Ledant 1977, Vielliard 1978). Independently of this discovery, Eric Burnier also located the species in the same place in June 1976 (Burnier 1976, Ledant 1977, Vielliard 1978). The Algerian Nuthatch was named *Sitta ledanti* in 1976 by the ornithologist Jacques Vielliard, as a tribute to the scientist who discovered it and reported it for the first time (Burnier 1976, Géroudet 1976, Vielliard 1976, Ledant 1977). It was only in 1989 that researchers resumed prospecting in the forests of the Kabylia of Babors. Indeed, Bellatrèche and Chalabi observed the species in the Guerouche national forest in the Taza National Park (Chalabi 1989, Bellatrèche & Chalabi 1990). After that, Bellatrèche, collectively with other researchers, have persisted their investigations and managed to observe the Algerian Nuthatch in the forests of Tamentout (Wilayas of Jijel, Setif and Mila) in June 1990, and of Djimla (Wilaya of Jijel) in July of the same year (Bellatrèche 1990, Harrap 2002). The study of the species has been stopped for almost 30 years, until a discovery in April 2018 when the Algerian Nuthatch was found in a fifth biotope; the Larbaâ forest (Moulai & Mayache 2018). Then, it was once contacted in seven other different forests located in the Kabylia of Babors; El Djarda in June 2018 (Haddad & Afoutni 2019), Tloudène and Tazegzeout in July 2019 (Bougaham *et al.* 2020), Sendouh and Coudia in November 2019 (Bougaham *et al.* 2020), Tababort in April 2020 (Bougaham *et al.* 2021), and Bouhanch in September 2021 (Mayache *et al.* 2021).

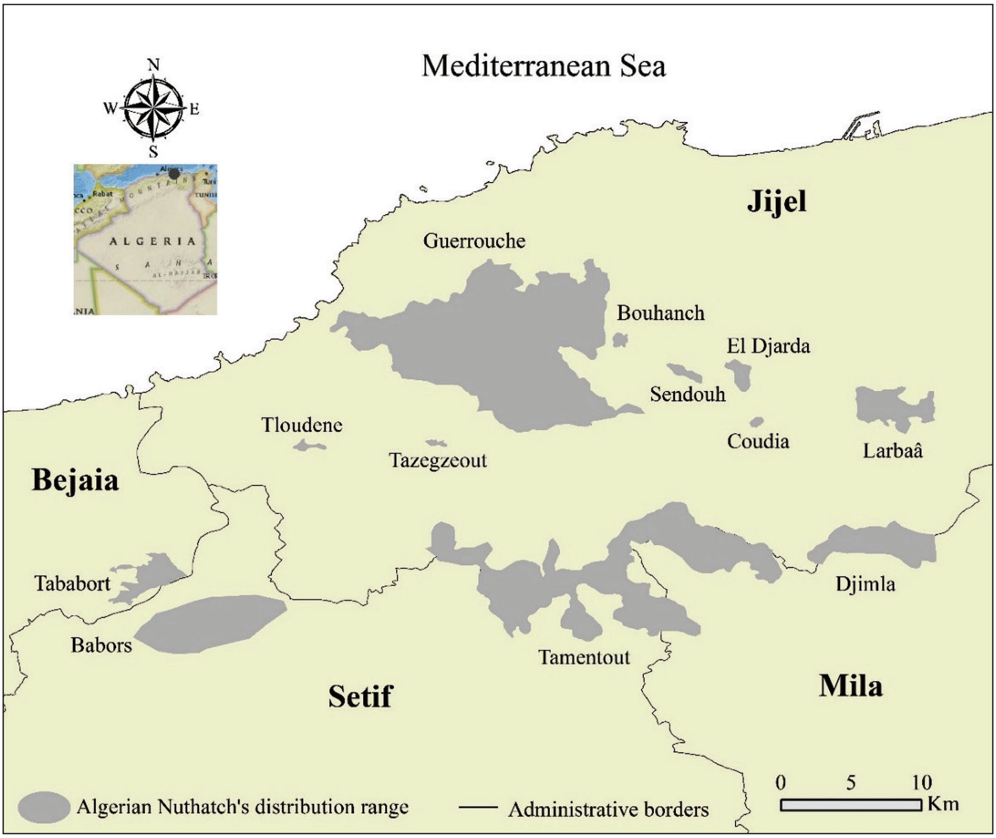

Figure 1. Map of the Algerian Nuthatch's distribution range. The grey colour highlights the breeding forests of the species discovered up to 2021 (© Souad Hamitouche)

1. ábra Az Atlasz-csuszka elterjedési térképe. Szürkével a 2021-ben felfedezett költőterületeit jelöltük (© Souad Hamitouche)

All the habitats that are home to the Algerian Nuthatch are located in the Kabylia of Babors (north-eastern Algeria). The distribution range of the Algerian Nuthatch is spread over a set of 12 forests separated by inhabitable plots (Harrap 2002). We have illustrated the species' distribution area with a map *(Figure 1)* using ArcGIS v10.5.1 software (Harder 2015, ESRI 2016).

Description of the Algerian Nuthatch

The Algerian Nuthatch is described as a medium-sized passerine measuring between 11.5 and 12.5 cm (Heinzel *et al.* 1996, Svensson *et al.* 1999, Harrap 2002). The male weighs 18 g and has a wing length of 81 millimetres, and is characterized by a grey-blue dorsal plumage and a beige to light red belly plumage. It also has a whitish throat and a black cap that is distinguished from the colour of the rest of its body. Its dark eye is crossed by a black line with a white eyebrow. The male has a long grey beak of 16 mm, slightly curved upwards, and its tail ends in a small white band (Burnier 1976, Vielliard 1976, Vielliard 1978, Fossé & Vaillant 1982, Chalabi 1989, Heinzel *et al.* 1996, Svensson *et al.* 1999, Harrap 2002, Monticelli & Legrand 2009, Bougaham *et al.* 2017).

The female Algerian Nuthatch weighs 16.5 g and has a wing length of 79 millimetres (mm), and is identical to the male but has a less pronounced pigmentation. Her cap is much smaller than the male's and is greyish which is similar to the rest of her body colour. She also has a finer beak than the male (Vielliard 1976, Vielliard 1978, Vielliard 1980, Fossé & Vaillant 1982, Chalabi 1989, Heinzel *et al.* 1996, Svensson *et al.* 1999, Harrap 2002, Monticelli & Legrand 2009, Bougaham *et al.* 2017).

The juveniles of Algerian Nuthatch have a yellowish beak which is shorter than that of their parents. They also have a greyish cap, their plumage colour is pale, and their headband is not very visible (Vielliard 1976, Ledant & Jacobs 1977, Jacobs *et al.* 1978, Vielliard 1978, Vielliard 1980, Fossé & Vaillant 1982, Harrap 2002). Algerian Nuthatches undergo moulting throughout the year. Therefore, between July and October, adults are subject to a post-breeding moult, and juveniles to a "post-juvenile" moult, which enables them to acquire characteristics similar to those of their parents (Jacobs *et al*. 1978, Vielliard 1980). Sexual dimorphism in the Algerian Nuthatch divided the opinions of researchers. Hence, there does not seem to be any sexual dimorphism affecting the individuals of the species for Vielliard (Vielliard 1976, Vielliard 1978, Vielliard 1980). Nevertheless, by referring to the definition of sexual dimorphism, and taking into account the differences which exist between individuals (male, female and juveniles), we are inclined to support opinion that sexual dimorphism does indeed affect the Algerian Nuthatch (Jacobs *et al.* 1978, Fossé & Vaillant 1982, Bougaham *et al.* 2017).

Phylogeny of the Algerian Nuthatch

Nuthatches are all sedentary and belong to a single genus called "*Sitta*" that has appeared in the Miocene (Vielliard 1978, Svensson *et al.* 2009). Among *Sitta* species, the Algerian Nuthatch *Sitta ledanti* is the only endemic bird species of Algeria (BirdLife International 2020). It belongs to the clade which contains the Yunnan Nuthatch *Sitta yunnanensis* as the common ancestor. Other species in this clade are the: Corsican *Sitta whiteheadi*, Krüper's *Sitta krueperi*, Chinese *Sitta villosa,* and North American *Sitta canadensis* nuthatches (Géroudet 1976, Pasquet *et al.* 2014).

Previous studies stated that the Algerian Nuthatch was phenotypically (or morphologically) similar to the Corsican and Krüper's nuthatches Heim de Balsac 1976, Vielliard 1978). However, according to current mitochondrial and genetic (DNA) analysis, the Algerian Nuthatch has been found to be more closely related to the Krüper's Nuthatch than to all the other nuthatches of its lineage (Pasquet 1998, Blondel 2018).

Habitat of the Algerian Nuthatch

The Algerian Nuthatch prefers mixed plant stands, even though it is also found in pure plant formations (Ledant & Jacobs 1977, Vielliard 1978, *Ledant et al.* 1985, Chalabi 1989, Bellatrèche & Boubaker 1995, Bougaham *et al.* 2018, Hamitouche *et al.* 2020). The tree species frequented by the Algerian Nuthatch are: (i) oaks: afares *Quercus afares*, zeen *Quercus canariensis*, cork *Quercus suber* and green *Quercus ilex*, (ii) Atlas cedar *Cedrus atlantica*, (iii) Numidian fir *Abies numidica*, (iv) maples *Acer obtusatum*, *Acer campestris*, etc. They are accompanied by a dense undergrowth in most of the forests sheltering the species (Ledant 1977, Ledant & Jacobs 1977, Vielliard 1978, Vielliard 1980, Ledant *et al.* 1985, Chalabi 1989, Bellatrèche 1990, Bougaham *et al.* 2017, Moulai *et al.* 2017, Bougaham *et al.* 2018, Moulai & Mayache 2018, Hamitouche *et al.* 2020).

The Algerian Nuthatch avoids the margins of forests of its distribution range, it then sets out in search of territories in the center of them. This preference ought to be detrimental to the survival of the species in case of important deforestation and/or devastating fires (Vielliard 1980, Bougaham *et al.* 2018, Hamitouche *et al.* 2020).

Vocalisation of the Algerian Nuthatch

The vocalisation of the Algerian Nuthatch is composed of 7 to 12 notations, repeated several times a minute. It is sometimes whispered, sometimes nasal, sometimes fast, and sometimes slow (Burnier 1976, Vielliard 1976, Heinzel *et al.* 1996, Harrap 2002). The species is characterized by two types of sounds. A territorial song (or species recognition song), which is of low intensity, and is emitted by male and female to communicate with each other, to look for food, etc. A whispered and aggressive call reminiscent of Eurasian Jay *Garrulus glandarius* which is emitted exclusively by the male to mark his presence and/or to defend his territory (Burnier 1976, Vielliard 1978, Heinzel *et al.* 1996, Svensson *et al.* 1999, Harrap 2002, Monticelli & Legrand 2009). The same reaction is observed using a recorder during field investigations (Ledant 1978).

Juveniles also emit a variety of sounds, no doubt trying to imitate their parents (Vielliard 1978). During searching food, the Algerian Nuthatch hardly manifests itself anymore, when so, it shouts more than it sings (Bellatrèche & Boubaker 1995).

Diet and foraging of the Algerian Nuthatch

The diet of the Algerian Nuthatch varies according to seasons. In the winter, the species is typically granivorous, then seeds and acorns of conifers, firs, and cedars are eaten. In spring (breeding) and summer seasons, the proliferation of insects allows the species to diversify its feeding. Thus, spiders, caterpillars, beetles, ants, hymenoptera and adult lepidoptera, which are abundant around its nests, are all in its diet, and *Forticula auricularia* is its most common prey, during its breeding period (Burnier 1976, Ledant & Jacobs 1977, Vielliard 1978, Ledant 1981, Bellatrèche & Boubaker 1995, Mayache *et al.* 2020, Zemouri *et al.* 2021). The Algerian Nuthatch does not seem to have any preference for the plant species of its environment when it is feeding.

The Algerian Nuthatch deploys various methods in its search for food. It digs, picks, gleans, and hollows out tree supports. The bird seems to opt for middle-aged trees with small and thin branches and explores more the foliage of trees than their trunks (Bellatrèche

& Boubaker 1995). The seeds it consumes are taken from the ground, they are eaten directly or are stored in the cavities of the trees (especially oaks). This strategy allows the species to conceal food for the summer supply and/or the feeding of its chicks (Ledant & Jacobs 1977, Vielliard 1978, Ledant 1981, Le Fur 1981).

Both parents are responsible for the feeding of the nestlings in the nest, but it is the male who mainly provides the energy needs of its family (Vielliard 1978, Bellatrèche & Boubaker 1995, Bougaham *et al.* 2017, Moulai *et al.* 2017). However, within the same brood, there is fraternal mutual aid where one nestling feeds another who is less developed (Ledant & Jacobs 1977). The feeding period for the nestlings lasts between 22 and 25 days. The parents continue to feed their young even after their fledging despite their independence (Ledant & Jacobs 1977, Vielliard 1978).

Breeding of the Algerian Nuthatch

The breeding season of the Algerian Nuthatch starts between March and May and can last until July, according to the weather. Mean clutch size is around 6, and only female incubates the eggs, which lasts from 14 to 17 days (Ledant & Jacobs 1977, Vielliard 1978, Bellatrèche & Chalabi 1990, Bougaham *et al.* 2017, Moulai *et al.* 2017). Fledglings leave the nest between June and July (Ledant & Jacobs 1977, Vielliard 1978, Fossé & Vaillant 1982, Moulai *et al.* 2017). Both males and females are taking out nestling feces. They still defend their territory even after a few days of their offspring's flight (Ledant & Jacobs 1977, Bougaham *et al.* 2017).

The Algerian Nuthatch digs its lodge and/or uses those abandoned by woodpeckers, mainly on dead trees still standing on their trunks (Ledant & Jacobs 1977, Bougaham *et al.* 2017, Moulai *et al.* 2017). The Algerian Nuthatch uses the Numidia fir, the Atlas cedar, the different oaks species, and the eucalyptus trees of its environment for nesting (Ledant & Jacobs 1977, Vielliard 1978, Bougaham *et al.* 2017, Moulai *et al.* 2017). Its nest is built of litter consisting of softwood shavings, dead leaf debris, moss, forest bird feathers, and wild boar bristles. Its depth varies from 15 to 20 cm and it is built approximately 5 to 15 metres above the ground (Ledant & Jacobs 1977, Vielliard 1978, Fossé & Vaillant 1982, Bougaham *et al.* 2017, Moulai *et al.* 2017).

Relation between Algerian Nuthatch and other forest birds

In the inter-breeding period, the Algerian Nuthatch makes flocks with the Coal Tit *Periparus ater*, the Great Tit *Parus major*, the Common Chiffchaff *Phylloscopus collybita,* and the Firecrest *Regulus ignicapilla* (Ledant & Jacobs 1977, Jacobs *et al.* 1978).

The European Pied Flycatcher *Ficedula hypoleuca* is among the species that competes for food with the Algerian Nuthatch (Ledant & Jacobs 1977). The Great Spotted Woodpecker *Dendrocopos major*, the Garden Dormouse *Eliomys quercinus,* and the Weasel *Mustela nivalis numidica* attack and destroy Algerian Nuthatch's broods and nests (Vielliard 1978, Bougaham *et al.* 2017).

Conservation status and numbers of the Algerian Nuthatch

The International Union for the Conservation of Nature (IUCN) has classified the Algerian Nuthatch as an endangered species in its Red List, and it can be found thanks to the code: "Endangered B1ab(iii,v);C2a(i) ver 3.1" (BirdLife International 2020).

In Algeria, the protection of the Algerian Nuthatch falls within the framework of the conservation of non-domestic wild species of the decree n° 83-509, established by the Algerian constitution since 1983 (JORA 1983). This law was updated in 2012 when more species were included (JORA 2012).

The solely complete inventories of the Algerian Nuthatch have been made in ten forests *(Table 1)*. Its numbers have been estimated between 250 and 999 individuals (BirdLife International 2020). However, the continuity of the prospections for counting the numbers of the Algerian Nuthatch could make a difference; the numbers of individuals could exceed the set interval, and its conservation status may be changed, giving hope that the species will not be longer threatened by extinction. This will be possible by lasting to study all aspects of this bird and by calling on all the authorities concerned to conserve its living environment.

Table 1. Number of individuals of the Algerian Nuthatch in eleven forest territories
1. táblázat Az Atlasz-csuszka egyedszámai tizenegy erdőterületen

Forests	Individuals	Pairs	Solitaries	References
Babor	275	133	9	(Zemouri & Bougaham in press)
Guerrouche	91	–	–	(Bellatrèche & Chalabi 1990)
Tamentout	187	80	27	(Hamitouche *et al.* 2020)
Djimla	60	27	6	(Bougaham *et al.* 2018)
Larbaâ	40	13	14	In press
El Djarda	10	3	4	Our own findings
Sendouh	9	4	1	Our own findings
Tloudène	7	3	1	Our own findings
Tazegzeout	3	1	1	Our own findings
Coudia	3	1	1	Our own findings
Tababort	6	3	0	Our own findings
Total	691	268	64	–

Threats

Man will remain an eternal danger for the forest. Among the many pressures on the forest, fires can damage the forest fauna, flora, and habitats. Illegal logging is another threat to the Algerian Nuthatch, especially when it is carried out to the detriment of dead trees. This disturbs the ecology of forest birds that use dead trees for their nesting as is the case of the Algerian Nuthatch (Vielliard 1978, Ledant 1981, Ledant *et al.* 1985, Bougaham *et al.* 2017, Bougaham *et al.* 2018, Hamitouche *et al.* 2020, Bougaham *et al.* 2021). Besides, residents over-exploit forest resources for their use and/or for sale on the market. They create tracks for the passage of transport vehicles within the forest itself, they gather edible wild fruits and known medicinal plants and exploit wood of forest trees, particularly cork oak *Quercus suber* (Ledant 1977, 1981, Ledant & Jacobs 1977, Ledant *et al.* 1985, Camps 1991, Madoui 2002, Ramade 2008, Bougaham *et al.* 2017, Moulai *et al.* 2017). In addition, they practice the breeding and the grazing of cattle, sheep, and goats, which causes soil erosion and flora diversity diminution. Therefore, the consequences of these human activities are harmful, threatening the forests of the Kabylia of Babors with degradation, reduction of their surface area, and even the disappearance of the species they shelter (Ledant 1977, 1981, Ledant *et al.* 1985, Camps 1991). The threats that we have summarised are a danger for all the fauna and flora of the Kabylia of Babors, in particular for the Algerian Nuthatch.

Conclusion

The Algerian Nuthatch is an endemic bird with small population size. Knowledge on the ecology and behaviour of this species is extremely limited. Here, we provide essential information about the species, to facilitate its presentation to the general public and scientists in particular. Finally, the ultimate aim of this article is to draw the attention of the authorities concerned, about the lack of studies undertaken on the Algerian Nuthatch and to the urgent need to establish means for the conservation of this endemic species still poorly known.

References

Bellatrèche, M. 1990. Découverte d'un troisième biotope de la Sittelle Kabyle (*Sitta ledanti* Vieilliard) en Algérie [Discovery of a third biotope of the Algerian Nuthatch (*Sitta ledanti* Vieilliard) in Algeria]. – Annales de l'Institut National Agronomique d'El Herrach 14: 13–20. (in French)

Bellatrèche, M. 1999. Diversité biologique et conservation : cas de l'avifaune forestière nicheuse de la Kabylie des Babors (Algérie) [Biological diversity and conservation: the case of the forest breeding avifauna of the Kabilia of Babors (Algeria)]. – Nature et Faune 15: 37–42. (in French)

Bellatrèche, M. & Boubaker, Z. 1995. Premières données sur le comportement alimentaire de la Sittelle Kabyle en période de reproduction [First data on the feeding behaviour of the Algerian Nuthatch *(Sitta ledanti)* duraing the breeding season]. – Annales de l'Institut National Agronomique d'El Harrach 16: 35–48. (in French)

Bellatrèche, M. & Chalabi, B. 1990. Données nouvelles sur l'aire de distribution de la Sittelle Kabyle *Sitta ledanti* [New data on the range of the Algerian Nuthatch (*Sitta ledanti*)]. – Alauda 58: 95–97. (in French)

Benslimane, M., Hamimed, A., El Zerey, W., Khaldi, A. & Mederbal, K. 2008. Analyse et suivi du phénomène de la désertification en Algérie du nord [Analysis and monitoring of the desertification phenomenon in Northern Algeria]. – VertigO – la revue électronique en sciences de l'environnement 8: 1–9. (in French)

BirdLife International 2020. Available at: http://datazone.birdlife.org/species/factsheet/algerian-nuthatch-sitta-ledanti

Blondel, J. 2018. Origins and dynamics of forest birds of the Northern Hemisphere. – In : Mikusiński, G., Roberge, J-M. & Fuller, R. (eds.) Ecology and Conservation of forest birds. – Cambridge University Press, England, pp. 11–50.

Bougaham, A-F., Announ, L., Aissat, L., Zemouri, M., Lillouch, S., Mirouh, A., Soukkou, W. & Bouchareb, A. 2018. Distribution et grandeur de la population de la Sittelle Kabyle *Sitta ledanti* dans la forêt de Djimla (Nord-Est algérien) [Distribution and population size of the Algerian Nuthatch's population in the Djimla forest (North-East Algeria)]. – Alauda 87: 299–304. (in French)

Bougaham, A-F., Benazouz, A. & Bouchareb, A. 2017. Reproduction et soins parentaux chez la Sittelle kabyle *Sitta ledanti* en forêt de Guerrouche (Jijel, Algérie) [Breeding and parental care in the Algerian Nuthatch *Sitta ledanti* in the Guerrouche forest (Jijel, Algeria)]. – Alauda 4: 269–274. (in French)

Bougaham, A-F., Hamitouche, S. & Bouchareb, A. 2020. Trois nouvelles localisations de la Sittelle kabyle *Sitta ledanti* en Algérie [Three new locations of the Algerian Nuthatch *Sitta ledanti* in Algeria]. – Alauda 88: 147–148. (in French)

Bougaham, A-F., Zemouri, M., Hamitouche, S. & Bouchareb, A. 2021. Découverte dans le Djebel Tababort (nord-est de l'Algérie) d'un nouveau site de reproduction de la Sittelle kabyle *Sitta ledanti* [Discovery in the Djebel Tababort (north-east Algeria) of a new breeding site of the Algerian Nuthatch *Sitta ledanti*]. – Bulletin de la Société Royale des Sciences de Liège 90: 241–248. (in French)

Burnier, E. 1976. Une nouvelle espèce de l'avifaune paléarctique : la Sittelle Kabyle, *Sitta ledanti* [A new species of Palearctic avifauna: The Algerian Nuthatch *Sitta ledanti*]. – Nos Oiseaux 33: 337–340. (in French)

Camps, G. 1991. Babor. – In: Camps, G. (ed.) Baal-Ben Yasla. – Edisud, Aix en Provence, pp. 1295–1296.

Chalabi, B. 1989. Du nouveau à propos de l'aire de distribution de la Sittelle Kabyle *(Sitta ledanti)* [News on the distribution area of the Algerian Nuthatch *Sitta ledanti*]. – Aves 26: 233–234. (in French)

Dahmani, M. 2004. Kabylie : Géographie [Kabylia: Geography]. – In: Chaker, S. (ed.) Judaisme-Kabylie. – Edisud, Aix en Provence, pp. 3986–3989. (in French)

De Smet, K. & Bouaza, F. 1984. La structure forestière du mont Babor [The forest structure of the Mount Babor]. – Silva Gandavensis 50: 65–84. (in French)

ESRI. 2016. ArcGIS Desktop: Release 10.5. – Environmental Systems Research Institute, Redlands, CA.

Ficheur, L-M. 1890. Description géologique de la Kabylie du Djurjura : étude spéciale des terrains tertiaires [Geological description of the Kabylia of Djurdjura; special study of the tertiary terrains]. – Imprimerie de Fontana-Bibliothèque municipale de Lyon, (in French)

Fossé, A. & Vaillant, G. 1982. A propos de la couleur de la calotte chez la Sittelle Kabyle *(Sitta ledanti)* [About the colour of the Algerian Nuthatch *(Sitta ledanti)*]. – Alauda 50: 228. (in French)

Géroudet, P. 1976. A propos de la Sittelle Kabyle [About the Algerian Nuthatch]. – Nos Oiseaux 33: 340–342. (in French)

Haddad, K. & Afoutni, L. 2019. La Sittelle Kabyle *Sitta ledanti* : nouvelle localité, répartition et habitat [The Algerian Nuthatch *Sitta ledanti*: new locality, repartition and habitat]. – Ornithos 26: 83–94. (in French)

Hamitouche, S., Bouchareb, A. & Bougaham, A-F. 2020. Status and distribution of the Algerian Nuthatch's population (*Sitta ledanti* Vielliard, 1976) in the Tamentout forest (north-eastern Algeria). – Avian Biology Research 13: 81–86. DOI: 10.1177/1758155920945842

Harder, C. 2015. The ArcGIS book : 10 big ideas about applying geography to your world. – Esri Press, California

Harrap, S. 2002. Little known West Palearctic birds: Algerian Nuthatch. – Birding World 5: 154–156.

Heim de Balsac, H. 1976. Commentaires sur la découverte d'un élément imprévu de la faune paléarctique [Comments on the discovery of an expected element of the Palearctic fauna]. – Alauda 44: 353–355. (in French)

Heinzel, H., Pitter, R. & Parslow, J. 1996. Guide Heinzel des oiseaux d'Europe, d'Afrique du Nord et du Moyen-Orient [Heinzel's Guide of the Birds of Europe, North Africa and the Middle East]. – Delachaux et Niestlé, Paris (in French)

Isenmann, P., Moali, A. 2000. Oiseaux d'Algérie [Birds of Algeria]. – SEOF, Paris

Jacobs, P., Mahler, F. & Ochando, B. 1978. A propos de la couleur de la calotte chez la Sittelle Kabyle *(Sitta ledanti)* [About the colour of the cap of the Algerian Nuthatch *(Sitta ledanti)*]. – Aves 15: 149–153. (in French)

JORA. 1983. Décret n°83-509 du 20 août 1983 relatif aux espèces animales non domestiques protégées [Decree No.83-509 of 20 August 1983 on protected non-domestic animal species]. – Journal Officiel de la République Algérienne Démocratique et Populaire 83: 1439–1440. (in French)

JORA. 2012. Décret exécutif n°12-235 du 24 mai 2012 fixant la liste des espèces animales non domestiques protégées [Decree No.12-235 of 24 May 2012 establishing the list of protected non-domestic animal species]. – Journal Officiel de la République Algérienne Démocratique et Populaire 12: 5–11. (in French)

Le Fur, R. 1981. Notes sur l'avifaune algérienne [Notes on the Algerian avifauna]. – Alauda 49: 295–299. (in French)

Ledant, J-P. 1977. La Sittelle Kabyle (*Sitta ledanti* Vielliard), espèce endémique montagnarde récemment découverte [The Algerian Nuthatch (*Sitta ledanti* Vielliard), endemic species recently discovered]. – Aves 14: 83–85. (in French)

Ledant, J-P. 1978. Données comparées sur la Sittelle Corse *(Sitta whiteheadie)* et sur la Sittelle Kabyle *(Sitta ledanti)* [Comparative data on the Corsican Nuthatch *(Sitta whiteheadie)* and the Algerian Nuthatch (*Sitta ledanti*)]. – Aves 78: 154–157. (in French)

Ledant, J-P. 1981. Conservation et fragilité de la forêt de Babor, habitat de la Sittelle Kabyle [Conservation and fragility of the Babor forest, habitat of the Algerian Nuthatch]. – Aves 18: 1–9. (in French)

Ledant, J-P. & Jacobs, P. 1977. La Sittelle Kabyle *(Sitta ledanti)* : Données nouvelles sur sa biologie [the Algerian Nuthatch *(Sitta ledanti)*: new data on its biology]. – Aves 14: 233–242. (in French)

Ledant, J-P., Jacobs, P., Ochando, B. & Renault, J. 1985. Dynamique de la forêt du Mont Babor et préférences écologiques de la Sittelle Kabyle *Sitta ledanti* [Mount Babor forest dynamics and ecological preferences of the Algerian Nuthatch *Sitta ledanti*]. – Biological Conservation 32: 231–254. (in French)

Madoui, A. 2002. Les incendies de forêt en Algérie : Historique, bilan et analyse [Forest fires in Algeria: History, assessment and analysis]. – Forêt Méditerranéenne 23: 23–30. (in French)

Mayache, M-E., Khider, B. & Moulai, R. 2021. Données récentes sur la distribution de la Sittelle Kabyle *Sitta ledanti* en Algérie [New data on the distribution of the Algerian Nuthatch *Sitta ledanti*]. – Alauda 89: 135–138. (in French)

Mayache, M-E., Temagoult, L., Bara, M. & Moulai, R. 2020. Diversity and dynamics of potential prey of the Algerian Nuthatch *Sitta ledanti* during the breeding season. – Studia Universitatis "Vasile Goldiş", Seria Ştiinţele Vieţii 30: 136–144. (in French)

Monticelli, D. & Legrand, V. 2009. Algerian Nuthatch: a photographic trip. – Dutch Birding 31: 247–251.

Moulai, R., Bouchareb, A., Gheribi, A. & Bougaham, A-F. 2017. Statut de la population et biologie de la reproduction de la Sittelle Kabyle dans la forêt de Guerrouche (Algérie) [Population status and breeding biology of the Algerian Nuthatch *Sitta ledanti* in the Guerrouche forest (Algeria)]. – Alauda 85: 101–107. (in French)

Moulai, R. & Mayache, M-E. 2018. Un nouveau site de reproduction pour la Sittelle kabyle *Sitta ledanti* [A new breeding site for the Algerian Nuthatch *Sitta ledanti*]. – Alauda 86: 73–74. (in French)

Niox, G-L. 1890. Géographie militaire VI : Algérie et Tunisie [Military geography VI: Algeria and Tunisia]. – Librairie militaire de L. Beaudoin et Cie, Paris (in French)

Pasquet, E. 2008. Phylogeny of the nuthatches of the *Sitta canadensis* group and its evolutionary and biogeographic implications. – Ibis 140(1): 150–156. DOI: 10.1111/j.1474-919X.1998.tb04553.x

Pasquet, E., Barker, F-K., Martens, J., Tillier, A., Cruaud, C. & Cibois, A. 2014. Evolution within the nuthatches (*Sittidae* : Aves, Passeriformes): molecular phylogeny, biogeography, and ecological perspectives. – Journal of Ornithology 155: 755–765. DOI: 10.1007/s10336-014-1063-7

Ramade, F. 2008. Dictionnaire encyclopédique des sciences de la nature et de la biodiversité [Encyclopaedic dictionary of natural sciences and biodiversity]. – Dunod, Paris (in French)

Svensson, L., Mullarney, K. & Zetterstrom, D. 2009. Le guide ornitho : le guide le plus complet des oiseaux d'Europe, d'Afrique du Nord et du Moyen-Orient (900 espèces) [The bird guide: the most complete guide of the birds of Europe, North Africa and the Middle East (900 species)]. – Delachaux et Niestlé, Paris (in French)

Svensson, L., Mullarney, K., Zetterstrom, D. & Grant, P-J. 1999. Le guide Ornitho [The bird guide]. – Delachaux et Niestlé, Paris (in French)

Vielliard, J. 1976. La Sittelle Kabyle [The Algerian Nuthatch]. – Alauda 44: 351–352.

Vielliard, J. 1978. Le Djebel Babor et sa Sittelle, *Sitta ledanti* Vielliard 1976 [The Djebel Babor and its Nuthatch, *Sitta ledanti* Vielliard 1976]. – Alauda 46: 1–42. (in French)

Vielliard, J. 1980. Remarques complémentaires sur la Sittelle Kabyle *Sitta ledanti* Vielliard 1976 [Additional comments on the Algerian Nuthatch *Sitta ledanti* Vielliard 1976]. – Alauda 48: 139–150. (in French)

Zemouri, M., Asloune, L., Adrar, A., Bouchareb, A. & Bougaham, A-F. 2021. Nestling diet of the Algerian Nuthatch *Sitta ledanti*, an endemic threatened bird in Babors' Kabylia region (north-eastern Algeria). – Ostrich 92: 1–9. DOI: 10.2989/00306525.2021.1923582

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Diet composition of the Pharaoh Eagle Owl, *Bubo ascalaphus* (Strigiformes, Strigidae) in a Steppe region of Algeria

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Abstract The diet composition of Pharaoh Eagle Owl *(Bubo ascalaphus)* was investigated in a semiarid area of North-western Algeria. A total of 65 pellets regurgitated by the *B. ascalaphus* were analysed, 288 food items were composed primarily of mammal remains (4 rodents, 1 bat and insectivore, 93.7%), and 1 bird species (passerine, 6.3%). The most frequent prey among the mammals were rodents (83.3%), which included *Mus musculus* (59.7%), *Meriones shawi* (11.1%), *Meriones libycus* (11.1%) and *Jaculus jaculus* (1.4%). The rodents were the most important prey items in biomass (91.4%), *M. shawi* made up to 41.9% of the total biomass. We may conclude that the Pharaoh Eagle Owl relies, in its feeding, very broadly on small mammals, completed by other groups.

Keywords: Pharaoh Eagle Owl, food habits, trophic ecology, Rechaïga area, Algerian steppe

Összefoglalás Az egyiptomi uhu *(Bubo ascalaphus)* táplálék összetételét vizsgáltuk Északnyugat-Algéria félszáraz területén. Összesen 65 köpetet elemeztünk és a 288 táplálékmaradvány alapján megállapítottuk, hogy a vizsgált területen a baglyok táplálléka elsősorban emlős maradványokból (4 rágcsáló, 1 denevér és rovarevő, 93,7%) és 1 madárfaj (verébalakú, 6,3%) maradványaiból álltak. Az emlősök közül leggyakrabban rágcsálókat (83,3%) ejtettek el, mint a *Mus musculus* (59,7%), *Meriones shawi* (11,1%), *Meriones libycus* (11,1%) és a *Jaculus jaculus* (1,4%). A biomassza jelentős része is rágcsálókból (91,4%) állt, a *M. shawi* a teljes biomassza 41,9%-át tette ki. Ebből arra lehet következtetni, hogy az egyiptomi uhu táplálkozása nagyon széles körben támaszkodik a kisemlősökre, egyéb más fajokkal kiegészítve.

Kulcsszavak: egyiptomi uhu, táplálkozási szokások, trofikus ökológia, Rechaïga környéke, algériai sztyepp

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Introduction

The Pharaoh Eagle Owl *Bubo ascalaphus* is largely distributed, its range extend from Northern Africa to the Middle East (Mohedano *et al.* 2014). This owl inhabits rocky mountain slopes in deserts and semi-deserts, but is found also in dry savannas, and locally endangered by human persecution, but its overall status is not known (Mikkola 2014).

The diet of the Pharaoh Eagle Owl is poorly known compared to its European counterpart, the Eurasian Eagle Owl *Bubo bubo* (Sándor & Orbán 2008). It has long been speculated that the Pharaoh Eagle Owl and Eurasian Eagle Owl differ in ecology and behaviour (Sibley

& Monroe 1990). However, unlike its northern counterpart, little is known about the food habits of *B. ascalaphus* (Mohedano *et al.* 2014). Generally, this owl eats mammals, birds and reptiles, and also scorpions and larger insects (Mikkola 2014).

Previous diet studies of owls in Algeria have primarily focused on the following owl species: the Little Owl *Athene noctua* (Baziz *et al.* 2005, Sekour *et al.* 2010, 2011, Chenchouni 2014), the Tawny Owl *Strix aluco* (Hamdine *et al.* 1999), the Long-eared Owl *Asio otus* (Sekour *et al.* 2010), the Western Barn Owl *Tyto alba* (Baziz *et al.* 1999, Sekour *et al.* 2010) and the Short-eared Owl *Asio flammeus* (Djilali *et al.* 2012). Few studies have focused on the diet of Pharaoh Eagle Owl *B. ascalaphus*, especially in Algerian steppe environments (Sellami & Belkacemi 1989, Boukhamza *et al.* 1994, Biche *et al.* 2001, Sekour *et al.* 2010) and listed small mammals, birds, reptiles and arthropods as prey. To better understand the basic ecological requirements of this species, we described its diet composition in a steppe region, North-western Algeria where no previous study has been carried out. We also provide an overview concerning the diet habits of *B. ascalaphus*.

Materials and Methods

The field study was conducted in Rechaïga area (35°19' N, 02°06' E) located in Tiaret province, North-western Algeria, where the presence of the Pharaoh Eagle Owl is confirmed. The presence of nests and pellets were the main criteria for selecting the study area. The habitat located in the steppe region, which is characterised by a semiarid Mediterranean climate. The mean annual temperature is 17.1 °C, while the mean annual rainfall varies from 170 to 260 mm. The vegetation is composed mainly of *Pinus halepensis* trees, part of the area is covered with shrubs, including *Thymelaea hirsuta*, *Ziziphus lotus* in association with *Pistacia atlantica*, and a few scattered tree individuals grow there. The forbs including *Noaea mucronata*, *Peganum harmala*, while grasses are e.g. *Poa bulbosa*, *Hordeum murinum*, *Bromus rubens*.

The diet composition of the Pharaoh Eagle Owl was studied based on pellets collected below the nests between June 2012 and March 2013. We determined remains of prey in the pellet samples by comparison of diagnostic body parts with material in private reference collections and information provided in works of Cuisin (1989) for bird species, and Bernard (1970), Barreau *et al.* (1991) and Aulagnier *et al.* (2009) for mammal species. We identified the prey items to the finest possible taxonomic category.

Diet composition was expressed as average relative numerical abundance (*N%*) for each prey item (Zaime & Gautier 1989) and biomass (*B%*) (Bayle 1996). In addition, to assess prey diversity in the diet, two measures were given: the Shannon-Weiner Diversity Index and Evenness ratio (Magurran 2004).

Results

A total of 65 pellets regurgitated by the Pharaoh Eagle Owl were analysed, 288 food items were composed primarily of mammal remains (4 rodents, 1 bat and insectivore, 93.7%), and

Table 1. Identified prey remains in Desert Eagle Owl pellets, Rechaïga area, North-western Algeria ni: Number of prey species; N%: Average relative numerical abundance (%); B%: Biomass (%).

1. táblázat Az egyiptomi uhu azonosított táplálékmaradványai Északnyugat-Algéria Rechaïga területéről

Groups	**Prey species**	***n***$_i$	***N%***	***B%***
Rodents	*Mus musculus*	172	59.72	25.23
	Meriones shawi	32	11.11	41.86
	Meriones libycus	32	11.11	19.18
	Jaculus jaculus	4	1.39	5.17
Insectivores	*Crocidura russula*	17	5.90	2.39
Bats	*Pipistrellus* sp.	13	4.52	1.81
Mammals		**270**	**93.75**	**95.64**
Passeriformes	*Passer* sp.	18	6.25	4.36
Birds		**18**	**6.25**	**4.36**

1 bird species (passerine, 6.3%) *(Table 1)*. Most pellets (71%) contained ≤ 3 prey items; the rest contained 5–10 prey items. Out of the total 65 pellets, 46 contained only rodents, 12 contained remains of rodents, insectivores and birds. 7 pellets contained remains of a bats and/or insectivores.

The most frequent prey among the mammals were rodents (83.3%), which included House Mouse *Mus musculus* (59.7%), Shaw's Jird *Meriones shawi* (11.1%), Libyan Jird *Meriones libycus* (11.1%) and Lesser Egyptian Jerboa *Jaculus jaculus* (1.4%) *(Table 1)*. The bat *Pipistrellus* sp. and Greater White-toothed Shrew *Crocidura russula*, both constituted mammal prey with 5.9% and 4.5%, respectively. The rodents were the most important prey items in biomass (91.4%), *M. shawi* made up to 41.9% of the total biomass *(Table 1)*. The *M. musculus* is the second most frequently recorded species (25.2%), whilst all the remaining prey species together are below 9% of the total biomass. The diet diversity and evenness index of *B. ascalaphus* were 1.34 and 0.69, respectively.

Discussion

The Pharaoh Eagle Owl's diet contained a wide variety of prey items that comprised mainly of small mammals (Abi-Said *et al.* 2020). With a diet that included 93.7% small mammals, our results corroborate those obtained by Denys *et al.* (1996), Boukhamza *et al.* (1994) and Biche *et al.* (2001) in Algeria, with 96%, 95.4% and 88.8%, respectively. Moreover, the small mammals were the most preyed species 92% in Marrakech, Morocco (Barreau & Bergier 2001a). Similar species of mammal rodents in the *B. ascalaphus* diet were found in the Mergueb Nature Reserve (Algeria) by Sekour *et al.* (2010). The *Mus musculus*, was the most preyed species support (Goodman 1990). The second species *Meriones shawi*, constituted a large proportion of biomass 41.9% (Boukhamza *et al.* 1994, Sekour *et al.* 2010). Rodent remains were the main food species of *B. ascalaphus,* as well as, the presence of bat species, suggests that hunting for food was mostly done at night for species of greater biomass and energetic yield, yet, the presence of *Meriones libycus*, suggests crepuscular and/or diurnal hunting activity (Rifai *et al.* 2000, Abi-Said *et al.* 2020). The Greater White-toothed Shrew accounted for 5.9% of the total prey remains, and contributed only 2.4% in biomass, similarly to previous findings (Sekour *et al.* 2010, Chenchouni 2014), this species has been reported with low proportions in the Little Owl diet. Birds are not frequently preyed by Pharaoh Eagle Owls (Mohedano *et al.* 2014). Our result is similar to that obtained by Biche *et al.* (2001), who reported that 6.4% of the total species were birds. In addition, insects were absent in the Pharaoh Eagle Owl diet (Denys *et al.* 1996), but constituted a very low proportion of the diet in each of the following studies: Rathgeber and Bayle (1997), Boukhamza *et al.* (1994) and Biche *et al.* (2001), with 1%, 3.5% and 4.8, respectively.

Diet diversity was very low, compared to other studies performed in steppe regions (Sekour *et al.* 2010). Likewise, the evenness index was low for *B. ascalaphus* in Rechaïga area. Since diet diversity often increases under conditions of food resource shortages (Gullion 1966), *B. ascalaphus* probably not selected more diverse diet because they were not under dietary stress. We may conclude that the Pharaoh Eagle Owl relies, in its feeding, very broadly on small mammals, completed by other groups. Further studies are needed to elaborate the seasonal fluctuation of the diet of the Pharaoh Eagle Owl in steppe regions as a function of the relative numerical abundance of the prey species in the environment.

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The authors declare that this study is in accordance with national legislation.

References

Abi-Said, M. R., Al Zein, M., Abu Baker, M. A. & Amr, Z. S. 2020. Diet of the Desert Eagle Owl, *Bubo ascalaphus*, in Eastern Saudi Arabia. – Pakistan Journal of Zoology 52(3): 1169–1171. DOI: 10.17582/journal.pjz/20190130180117

Aulagnier, S., Haffner, P., Mitchell-Jones, A. J., Moutou, F. & Zima, J. 2009. Mammals of Europe, North Africa and the Middle East. – A & C Black Publishers, London

Barreau, D., Roche, A. & Aulagnier, S. 1991. Eléments d'identification des crânes des rongeurs du Maroc [Identification elements of Rodent skulls from Morocco]. – Société Française Pour l'Étude et la Protection des Mammifères, Puceul (in French)

Barreau, D. & Bergier, P. 2001a L'avifaune de la région de Marrakech (Haouz et Hauts Atlas de Marrakech, Maroc) 2. Les espèces non passereaux [The avifauna of the Marrakech region (Haouz and High Atlas, Morocco). 2 non-passerine]. – Alauda 69(1): 167–202. (in French with English Summary)

Bayle, P. 1996. Régime alimentaire du grand-duc d'Europe *Bubo bubo* en période de reproduction dans le Parc National du Mercantour et ses environs (Alpes-Maritimes et Alpes-de-Haute-Provence, France) [Diet of the Eurasian Eagle Owl *Bubo bubo* during the breeding season in Mercantour National Park and its surroundings (Alpes-Maritimes and Alpes-de-Haute-Provence, France)]. – Avocetta 20: 12–25. (in French with English Summary)

Baziz, B., Doumandji, S. & Hamani, A. 1999. Adaptations trophiques de la Chouette effraie *Tyto alba* (Aves, Tytonidae) dans divers milieux en Algérie [Trophic adaptations of the Barn Owl *Tyto alba* (Aves, Tytonidae) in various environments in Algeria]. – Proceedings of the XXVth International Congress of the International Union of Game Biologists, Greece, pp. 217–227. (in French with English Summary)

Baziz, B., Sekour, M., Doumandji, S., Denys, C., Metref, S., Benjdabellah, S. & Nadji, F. Z. 2005. Données sur le régime alimentaire de la Chouette chevêche *Athene noctua* en Algérie [Data on the diet of the Little Owl *Athene noctua* in Algeria]. – Aves 42: 149–157. (in French with English Summary)

Bernard, J. 1970. Clef de détermination des Rongeurs de Tunisie [Key to determination of Rodents from Tunisia]. – Archives de l'Institut Pasteur de Tunis 47: 265–307. (in French)

Biche, M., Sellami, M., Libois, R. & Yahiaoui, N. 2001. Régime alimentaire du Grand-duc du désert, *Bubo ascalaphus* dans la réserve naturelle de Mergueb (M'Sila, Algérie) [Diet of the Pharaoh Eagle-owl *Bubo ascalaphus* in the Mergueb Nature Reserve (M'Sila, Algeria)]. – Alauda 69(4): 554–557. (in French with English Summary)

Boukhamza, M., Hamdine, W. & Thevenot, M. 1994. Données sur le régime alimentaire du Grand-duc ascalaphe *Bubo bubo ascalaphus* en milieu steppique (Aïn Ouessera, Algérie) [Data on the diet of the Eagle Owl *Bubo bubo ascalaphus* in Steppic Environment (Aïn Ouessera, Algeria)]. – Alauda 62(2): 150–152. (in French with English Summary)

Chenchouni, H. 2014. Diet of the Little Owl *(Athene noctua)* during the pre-reproductive period in a semi-arid Mediterranean region. – Zoology and Ecology 24(4): 314–323. DOI: 10.1080/21658005.2014.965919

Cuisin, J. 1989. L'identification des Cranes des Passereaux (Passeriformes: Aves) [Identification of Passerine Skulls (Passeriformes: Aves)]. – Bourgogne University, Bourgogne, Dijon (in French)

Denys, C., Dauphin, Y., Rzebik-Kowalski, B. & Kowalska, K. 1996. Taphonomic study of Algerian owl pellet assemblages and differential preservation of some rodents: palaentological implications. – Acta Zoologica Cracoviensia 39(1): 103–116.

Djilali, K., Sekour, M. & Bissati, S. 2012. Etude du régime alimentaire du hibou des marais, *Asio flammeus* (Pontoppidan, 1763) dans la région d'El-Golea [Study of the diet of Short-eared Owl, *Asio flammeus* (Pontoppidan, 1763) in the Region of El-Golea]. – Revue des Bioressources 2: 29–36. (in French with English Summary)

Goodman, S. M. 1990. The food habits of the Eagle Owl *(Bubo bubo ascalaphus)* in Kharga Oasis, Egyptian Western Desert. – Journal of Arid Environments 18: 217–220. DOI: 10.1016/s0140-1963(18)30855-3

Gullion, G. W. 1966. A viewpoint concerning the significance of studies of game bird habits. – Condor 68: 372–376. DOI: 10.2307/1365451

Hamdine, W., Bouhhemza, M., Doumandji, S. E., Poitevin, F. & Thévenot, M. 1999. Premières données sur le régime alimentaire de la Chouette hulotte *(Strix aluco mauritanica)* en Algérie [First Data on the diet of Tawny Owl *(Strix aluco mauritanica)* in Algeria]. – Ecologia Mediterranea 25: 111–123. (in French with English Summary)

Magurran, A. E. 2004. Measuring Biological Diversity. – Blackwell Publishing, Oxford

Mikkola, H., 2014. Owls of the World: A Photographic Guide. 2nd ed. – Firefly Books, Richmond Hill, Ontario

Mohedano, I., Abu Baker, M. A., Hunter, B., Buchan, J., Michaels, C. J. & Yamaguchi, N. 2014. On the diet of the Pharaoh Eagle Owl, *Bubo ascalaphus* (Savigny, 1809), in Qatar, with an overview of its feeding habits. – Zoology in the Middle East 60(2): 111–119. DOI: 10.1080/09397140.2014.914713

Rathgeber, C. & Bayle, P. 1997. Régime alimentaire du Grand-duc d'Europe *Bubo bubo*, en période de reproduction, dans la région de Menton (Alpes-Maritimes, France) [Diet of the Eurasian Eagle Owl *Bubo bubo*, during the breeding season, in menton region (Alpes-Maritimes, France)]. – Alauda 65(4): 351–356. (in French with English Summary)

Rifai, L. B., Al-Melhim, W. N., Gharaibeh, B. M. & Amr, Z. S. 2000. The diet of the Desert Eagle Owl, *Bubo bubo ascalaphus*, in the Eastern Desert of Jordan. – Journal of Arid Environments 44: 369–372. DOI: 10.1006/jare.1999.0601

Sándor, D. A. & Orbán, Z. 2008. Food of the Desert Eagle Owl *(Bubo ascalaphus)* in Siwa Oasis, Western Desert, Egypt. – Zoology in the Middle East 44: 107–110. DOI: 10.1080/09397140.2008.10638295

Sekour, M., Baziz, B., Denys, C., Doumandji, S., Souttou, K. & Guezoul, O. 2010. Régime alimentaire de la chevêche d'Athéna *Athene noctua*, de l'effraie des clochers *Tyto alba*, du hibou moyen-duc *Asio otus* et du grand-duc ascalaphe *Bubo ascalaphus*: Réserve naturelle de Mergueb (Algérie) [Diet of the Little Owl *Athene noctua*, the Barn Owl *Tyto alba*, the Long-eared Owl *Asio otus* and the Pharaoh Eagle Owl *Bubo ascalaphus*: Mergueb Nature Reserve (Algeria)]. – Alauda 78(2): 103–117. (in French with English Summary)

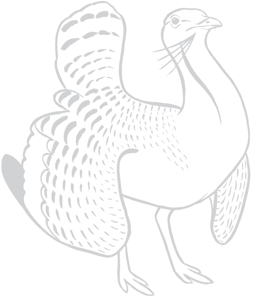
Sekour, M., Beddiaf, R., Souttou, K., Denys, C., Doumandji, S. E. & Guezoul, O. 2011. Variation saisonnière du régime alimentaire de la chouette chevêche *(Athene noctua)* (Scopoli, 1769) dans l'extrême sud-est du Sahara algérien (Djanet, Algérie) [Seasonal Variation in the diet of the Little Owl *(Athene noctua)* (Scopoli,

1769) in the Extreme Southeast of the Algerian Sahara (Djanet, Algeria)]. – Revue d'Ecologie (Terre et Vie) 66: 79–91. (in French with English Summary)

Sellami, M. & Belkacemi, H. 1989. Le régime alimentaire du Hibou grand-duc *Bubo bubo* dans une réserve naturelle d'Algérie: le Mergueb [The diet of the Eagle Owl *Bubo bubo* in a Nature Reserve in Algeria: The Mergueb]. – L'Oiseau et la Revue Française d'Ornithologie 59: 329–332. (in French with English Summary)

Sibley, C. G. & Monroe, B. L. 1990. Distribution and taxonomy of birds in the world. – Yale University Press, New Haven & London

Zaime, A. & Gautier, J. Y. 1989. Comparaison des régimes alimentaires de trois espèces sympatriques de Gerbillidae en milieu saharien au Maroc [Comparison of the diets of three sympatric species of Gerbillidae in the Saharan environment in Morocco]. – Revue d'Ecologie (Terre et Vie) 44(2): 153–163. (in French with English Summary)



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New cases of facultative interspecific brood parasitism in Black-winged Stilt *(Himantopus himantopus)* and Eurasian Coot *(Fulica atra)*

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Abstract Many hypotheses try to explain the evolution and possible relations between obligate and facultative brood parasitism in birds. To explore this, a large number of observations and data are needed. Our understanding based on the observations of facultative parasitic species published in the literature is less clear compared to the obligate parasitic species. This communication is about three cases of facultative interspecific brood parasitism. Two nests of Black-headed Gull *(Chroicocephalus ridibundus)* parasite by Eurasian Coot *(Fulica atra)* and one nest of Pied Avocet *(Recurvirostra avosetta)* parasite by Black-winged Stilt *(Himantopus himantopus)*. These observations are significant as long as interspecific brood parasitism was frequently described in Gruiformes (Rallidae) but has rarely observed within Charadriiformes.

Keywords: Rallidae, Charadriiformes, eggs, nests

Összefoglalás Az obligát és fakultatív fészekparazitizmus kialakulását és kapcsolatát számos feltevéssel próbálják magyarázni. Ennek további vizsgálatához még sok megfigyelésre és adatra van szükség. A fakultatív fészekparazitizmusról közölt adatok alapján ismereteink még kevésbé letisztultak, mint az obligát fészekparazita fajok esetében. Ebben a rövid közleményben három megfigyelést közlünk a fajok közötti fakultatív fészekparazitizmus esetére. Két dankasirály *(Chroicocephalus ridibundus)* fészekben egy-egy szárcsa *(Fulica atra)*, valamint egy gulipán *(Recurvirostra avosetta)* fészekben egy gólyatöcs *(Himantopus himantopus)* tojást azonosítottunk. Ezek a megfigyelések azért is fontosak, mert amíg a darualakúaknál, azon belül is a guvatfélék körében jól ismert a fészekparazitizmus, addig a lileféléknél ritka jelenség.

Kulcsszavak: guvatfélék, lilealakúak, tojások, fészkek

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Around 109 bird species (1% of all birds) are obligate brood parasites, never incubate their own eggs or raise their own young (Mann 2017). There are also known facultative brood parasite species, which usually take care for their own eggs and young but in some cases lay their eggs in other nests from the same species (conspecific parasites) and/or of another species (interspecific parasites) (Mann 2007, Lyon & Eadie 2008).

The evolution of brood parasitism in birds and the relations between obligate and facultative brood parasites, or conspecific and interspecific parasites, attracted the interest of evolutionary biologist, create a context for many hypotheses (Hamilton 1964, Hamilton & Orians 1965, Johnsgard 1997, Magali & Sorci 2001) and still remain poorly understood (Krakauer & Kimball 2009).

Figure 1. Black-headed Gull *(Chroicocephalus ridibundus)* nests with two eggs and one egg of Eurasian Coot *(Fulica atra)* (*) (Photo: A. N. Stermin)

1. ábra Két-két tojásos dankasirály *(Chroicocephalus ridibundus)* fészkek egy-egy idegen szárcsa *(Fulica atra)* tojással (*) (Fotó: A. N. Stermin)

It is assumed that brood parasitism evolved first as a facultative strategy in conspecific brood parasitism and developed into facultative or obligate interspecific parasitism (Hamilton & Orians 1965, Payne 1977, Yamauchi 1995). 234 species, about 2.4% of all birds are interspecific brood parasites (Yom-Tov 2001). Analysis in altricial birds revealed that obligate brood parasitism possibly arose directly from non-parasitic behaviour rather than from facultative parasitism (Sorenson & Payne 2001, 2002, Yom-Tov & Geffen 2006).

To understand the relations between parasitism types and to test the hypotheses, a large number of observations and data are needed. If the obligate parasitic species are better known than the facultative parasitic species, the observations published in the literature can be less evident, as long as the cases of facultative parasitism are more difficult to observe and to identify (Haraszthy 2019a). In this context, the publication of any observations related to cases of facultative parasitism are essential for understanding the extent of the phenomenon and its evolution.

Three cases of facultative interspecific brood parasitism were revealed during our field work in May 2021 in two ponds, Gruia (44.285426°; 22.689336°) and Gârla Mare (44.199930°; 22.775321°) from South Romania.

In a small colony (9 nests) of Black-headed Gull *(Chroicocephalus ridibundus),* on Gruia pond, two nests with two gull eggs were parasite by one egg of Eurasian Coot *(Fulica atra) (Figure 1)*. In the middle of the Black-headed Gull colony, about 2 m away from the gulls'

Figure 2. Pied Avocet *(Recurvirostra avosetta)* nest with 4 eggs and one egg of Black-winged Stilt *(Himantopus himantopus)* (*) (Photo: A. N. Stermin)

2. ábra Négytojásos gulipán *(Recurvirostra avosetta)* fészek egy idegen gólyatöcs *(Himantopus himantopus)* (*) tojással (Fotó: A. N. Stermin) nests, a European Coot nest with seven eggs was located. It is a high probability that the European Coot eggs from the Black-headed Gull nests belong to this female, as long as usually a coot lay between 6 – 10 eggs (Taylor & van Perlo 1998). The measurements of the eggs in the nests: (1) for Black-headed Gull: 52.83 × 36.84 mm, 52.47 × 36.71 mm and for Eurasian Coot: 54.31 × 37.45 mm, (2) for Black-headed Gull: 49.30 × 35.89 mm, 48.07 × 36.33 mm and for Eurasian Coot: 52.20 × 36.49 mm.

On Gârla Mare pond, a Pied Avocet *(Recurvirostra avosetta)* nest with 4 eggs was parasite by one egg of Black-winged Stilt *(Himantopus himantopus) (Figure 2)*. The measurements of the eggs in the nests for Pied Avocet: 52.81 × 35.64 mm, 51.18 × 36.04 mm, 52.05 × 34.32 mm, 52.95 × 35.45 mm and for Black-winged Stilt: 40.36 × 31.24 mm.

Other cases of Black-headed Gull nests with one European Coot egg ware also observed in other studies. It was found that Black-winged Stilt can lay eggs in nests of Common Tern *(Sterna hirundo)* and also Black-headed Gull (Haraszthy 2019b).

We mention that in the observed cases, they were involved nidifugous (European Coot, Pied Avocet and Black-winged Stilt) and also nidicolous (Black-headed Gull) species.

As long as interspecific brood parasitism was frequently described in Gruiformes (Rallidae) (Forman 2003, Haraszthy 2019b), the phenomenon has rarely been observed within Charadriiformes (Amat 1998, Yom-Tov 2001, Niemczynowicz *et al.* 2015). In this context, these observations are important and can be important points in the understanding of the occurrence and evolution of brood parasitism in birds.

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References

Amat, J. A. 1998. Mixed clutches in shorebirds nests: why are they so uncommon? – Wader Study Group Bulletin 85: 55–59.

Forman, D. 2003. Moorhen interspecific brood parasitism. – British Birds 96: 43–44.

Hamilton, W. D. 1964. The genetical evolution of social behaviour. – Journal of Theoretical Biology 7: 17–52.

Hamilton, W. J. & Orians, G. H. 1965. Evolution of brood parasitism in altricial birds. – Condor 67(5): 361–382. DOI: 10.2307/1365631

Haraszthy, L. 2019a New species on the list of species with intraspecific nest parasitism. – Ornis Hungarica 27(1): 166–206. DOI: 10.2478/orhu-2019-0010

Haraszthy, L. 2019b Cases of occasional interspecific brood parasitism and egg dumping in Hungary. – Ornis Hungarica 27(2): 115–141. DOI: 10.2478/orhu-2019-0020

Johnsgard, P. A. 1997. The Avian Brood Parasites. – Oxford University Press, Oxford

Krakauer, A. H. & Kimball, R. T. 2009. Interspecific brood parasitism in galliform birds. – Ibis 151(2): 373–381. DOI: 10.1111/j.1474-919X.2009.00916.x

Lyon, B. E. & Eadie, J. M. 2008. Conspecific brood parasitism in birds: a life-history perspective. – Annual Review of Ecology, Evolution, and Systematics 39: 343–363. DOI: 10.1146/annurev.ecolsys.39.110707.173354

Magali, R. & Sorci, G. 2001. The evolution of obligate interspecific brood parasitism in birds. – Behavioral Ecology 12(2): 128–133. DOI: 10.1093/beheco/12.2.128

Mann, C. F. 2017. A taxonomic review of obligate and facultative interspecific avian brood parasitism. – In: Soler, M. (ed.) Avian Brood Parasitism: Behaviour, Ecology, Evolution and Coevolution. – Springer, Berlin, Germany, pp. 61–92.

Niemczynowicz, A., Świętochowski, P., Zalewsk, A. & Chętnicki, W. 2015. Facultative interspecific brood parasitism in colonial breeding waterbirds in Biebrza National Park, Poland. – Waterbirds 38(3): 282–289. DOI: 10.1675/063.038.0308

Payne, R. B. 1977. The ecology of brood parasitism in birds. – Annual Review of Ecology and Systematics 8: 1–28. DOI: 10.1146/annurev.es.08.110177.000245

Sorenson, M. D. & Payne, R. B. 2001. A single ancient origin of brood parasitism in African finches: implications for hostparasite coevolution. – Evolution 55(12): 2550–2567. DOI: 10.1111/j.0014-3820.2001.tb00768.x

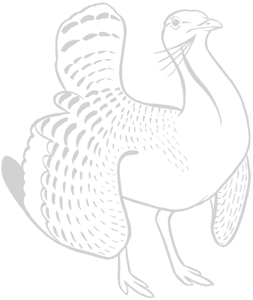
Sorenson, M. D. & Payne, R. B. 2002. Molecular genetic perspectives on avian brood parasitism. – Integrative & Comparative Biology 42(2): 388–400. DOI: 10.1093/icb/42.2.388

Taylor, B. & van Perlo, B. 1998. Rails: A Guide to the Rails, Crakes, Gallinules, and Coots of the World. – Yale University Press, New Haven

Yamauchi, A. 1995. Theory of evolution of nest parasitism in birds. – The American Naturalist 145(3): 434–456.

Yom-Tov, Y. & Geffen, E. 2006. On the origins of brood parasitism in altricial birds. – Behavioral Ecology 17(2): 196–205. DOI: 10.1093/beheco/arj013

Yom-Tov, Y. 2001. An updated list and some comments on the occurrence of intraspecific nest parasitism in birds. – Ibis 143: 133–143. DOI: 10.1111/j.1474-919X.2001.tb04177.x



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A call for observations of birds with sublingual oral fistulas in central and eastern Europe, and beyond

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Abstract Birds with major physical abnormalities do not live for extended periods and, therefore, are rarely observed in the wild. This is particularly the case for birds with defects in their feeding apparatus that succumb to mortality rapidly through precipitous declines in their foraging efficiency and body condition. Sublingual oral fistulas are such an abnormality and involve the development of an opening (or fistula) in the floor of the oral cavity through which the tongue extends, resulting in its permanent exclusion from the mouth. The tongue dehydrates and dies. First described in the 2000s in Stitchbirds *(Notiomystis cincta)* in New Zealand, it has rarely been reported in other species. However, following our recent discovery of two seabird species on Ascension Island in the South Atlantic displaying oral fistulas, in 2016 I launched a citizen science research project requesting reports of birds with the condition in the world's avifauna. To date, I have received 188 reports of birds of 82 different species with many contributed from western Europe. However, with only one report from central and eastern Europe, I am now requesting the assistance of birders in the region and in other parts of the world to contribute to this ongoing research project.

Keywords: abnormality, body condition, defect, foraging efficiency, predation and starvation risks

Összefoglalás A komoly fizikai elváltozásokat szenvedett madarak élettartama jelentősen csökken, ezért ilyen elváltozások ritkán figyelhetőek meg a vadon élő egyedek körében. Ez különösen igaz olyan madaraknál, ahol táplálkozást nehezítő hiányosságok jelennek meg, ezért az elhullás mértéke a csökkent táplálkozási hatékonyság és a gyorsan romló egészégi állapot következtében megnő. A nyelv alatt megjelenő szájtájéki sipolyok (oral fistulas) épp ilyen rendellenességek, amelyek egy nyílást képeznek az alsó csőrkáván, amelyen keresztül a nyelv kizáródik a szájüregből. Ennek következtében a nyelv kiszárad és elhal. A jelenséget először a 2000-es években írták le új-zélandi örvös maorimadaraknál *(Notiomystis cincta)*, míg más fajoknál ritkán jelentették. Azonban a dél-atlanti Ascension-szigeten további két madárfajnál figyelték meg a jelenséget, amit követően 2016-ban elindítottam azt a citizen science programot, amelynek keretében adatokat gyűjtök az ismertetett rendellenességről, a világ teljes madárfaunájára vonatkozóan. Eddig 192 bejelentés érkezett 82 madárfajról, főként Nyugat-Európából. Mivel csupán egyetlen adat származik Európa középső és keleti országaiból, ezért ösztönözni szeretném a régió és a világ más területeinek madarászait, hogy megfigyeléseikkel járuljanak hozzá ehhez a tudományos munkához.

Kulcsszavak: rendellenesség, egészségi állapot, hiányosság, táplálkozási hatékonyság, zsákmányolás és éhezés, kockázatok

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The physiological requirements of powered flight mean that compared with other classes of higher vertebrates, birds are heavily impacted by major physical abnormalities. Most of the world's 10,912 extant bird species (www.worldbirdnames.org) are volant and, therefore, most cannot carry excess (i.e. surplus to requirements) nutritional and energetic reserves because of the need to minimise body mass for effective and efficient flight (Ellington 1991). Excess body mass negatively impacts the ability of birds to evade predators in response to perceived mass-dependent predation risk (e.g. Witter *et al*. 1994, Gentle & Gosler 2001). Similarly, reduced body mass can lower survival of birds through a different (non-predatory) mechanism, namely that of starvation risk (Houston & McNamara 1993). Therefore, this delicately balanced trade-off between predation and starvation risks results in body mass dynamics in most bird species that are especially sensitive to any physical abnormalities that negatively influence their body condition (Brown 1996). Assessment of body condition is traditionally made by calculating mass per unit (structural) length in birds (Rising & Somers 1989), a derived metric that quantifies somatic reserves scaled by the overall body size of a bird. Although major physical abnormalities are seen in free-living birds (e.g. Sutton 1973, Durell *et al*. 1996), they are rare, perhaps because usually the body condition of birds declines rapidly as a result of deterioration in their foraging abilities (e.g. Sharp & Neill 1979, Murza *et al*. 2000). Precipitous declines in body condition inevitably

Figure 1. Two Western Jackdaws *(Corvus monedula)* in Woolacombe, North Devon, UK, where they were feeding at the edge of a woodland on the 4th of April 2018. The bird on the right displays an oral fistula where the tongue extends through an opening (fistula) in the floor of the bird's oral cavity from where it is permanently excluded (photo: Carl Bovis)

1. ábra Erdőszélen táplálkozó csókák *(Corvus monedula)* Woolacombe, North Devon, UK területén 2018. április 8-án. A jobb oldali madáron jól látható az a rendellenesség – szájtájéki sipoly (oral fistula) –, ami elsősorban az alsó csőrkávát érinti, azon egy nyílást képezve, melyen keresztül a madár nyelve a csőrön kívülre záródik (Fotó: Carl Bovis) result in mortality of birds, sometimes from increased susceptibility to predation (Temple 1987), rather than from starvation *per se*.

Body condition of birds declines dramatically as a result of abnormalities of their feeding apparatus (i.e. bill, tongue, oropharynx) (e.g. Clayton *et al*. 1999, Van Hemert *et al*. 2012). Given the rarity of free-living birds exhibiting such major abnormalities, we were surprised to observe a number of Sooty Terns *(Onychoprion fuscatus)* (Reynolds *et al*. 2009) and a Masked Booby *(Sula dactylatra)* (Hughes *et al*. 2013) with sublingual oral fistulas (hereafter referred to as 'oral fistulas') on Ascension Island in the South Atlantic. This is an abnormality of the oropharynx of birds in which an opening (or fistula) develops in the floor

Table 1. Details of species observed with oral fistulas and recorded in the research project database (as of the 23rd of October 2021). Taxonomy follows del Hoyo (2020)

1. táblázat A kutatási adatbázisban 2021. október 23-ig rögzített, szájtájéki sipollyal (oral fistula) megfigyelt madárfajok és rendszertani besorolásuk (del Hoyo 2020)

Order	**Family**	**Number of:**	
		species	**oral fistulas**
Galliformes	Megapodiidae	1	1
Anseriformes	Anatidae	7	9
Podicipediformes	Podicipedidae	2	2
Gruiformes	Rallidae	1	1
	Gruidae	1	2
Gaviiformes	Gaviidae	2	3
Sphenisciformes	Spheniscidae	2	2
Ciconiiformes	Ardeidae	9	16
	Pelecanidae	1	1
Suliformes	Sulidae	1	1
	Phalacrocoracidae	2	2
Charadriiformes	Scolopacidae	5	5
	Laridae	31	111
	Stercorariidae	1	1
Accipitriformes	Accipitridae	2	5
Falconiformes	Falconidae	1	1
Passeriformes	Tyrannidae	1	1
	Meliphagidae	2	2
	Corvidae	1	2
	Notiomystidae	1	12
	Sylviidae	1	1
	Certhiidae	1	1
	Mimidae	1	1
	Turdidae	2	2
	Muscicapidae	3	3

of the oral cavity *(Figure 1)*. As the condition progresses, the tongue of the bird protrudes through the fistula where eventually it remains permanently excluded from the oral cavity, drying and becoming necrotic with increasing exposure. We were not the first to observe and describe this condition in free-living birds because it had been seen initially in Stitchbirds (or 'hihi') *(Notiomystis cincta)* on Kapiti Island in New Zealand in 1992. The condition in free-living birds was first described in Stitchbirds in the scientific literature by Castro and Taylor (2001).

While the progression of oral fistulas in birds is well described in Stitchbirds (Low *et al.* 2007), its aetiology remains a mystery in this species and others. Since the publication of observations of oral fistulas in a Masked Booby, and in Sooty Terns and Stitchbirds, the condition has been reported in only a few other species to date: Audouin's Gull *(Larus audouinii)* (Greño *et al*. 2017), Franklin's Gull *(Larus pipixcan)* (Rintoul & Reynolds 2019), Griffon Vulture *(Gyps fulvus)* (Camiña & Guerrero 2013), and Swinhoe's Snipe *(Gallinago megala)* (Melville *et al*. 2019).

As a result of limited knowledge about the condition in the world's free-living birds, in May 2016 I launched a citizen science research project requesting global observations of birds with the condition. Its objectives are to document the occurrence of oral fistulas in temporal, spatial, phylogenetic and foraging niche 'spaces'. Through this global survey, it is hoped to understand: how long historically the condition has impacted birds; where in the world birds are most heavily impacted by the condition; whether the condition 'targets' particular avian taxa; and, through considering phylogenetic relationships between species, whether some taxa are more predisposed to developing the condition than others by virtue of sharing ensembles of foraging traits. The research project ultimately sets out to determine: which species might be expected to develop the condition in the future, thereby allowing us to target surveillance efforts in the field to detect its occurrence rapidly; at what life stage(s) birds develop the condition so we can target our field surveillance efforts; the chronic impacts of the condition on birds living with the condition; and the causes of the condition so that we can prevent it from occurring in birds in the future.

It is particularly important to increase the number of observations of birds with the condition because, along with other researchers connected to this project, I plan to carry out statistical analysis on data that especially uses phylogenetic comparative methods (where possible) to predict the occurrence of oral fistulas in birds.

Details of my research project can be found at: *https://www.birmingham.ac.uk/research/activity/ornithology/0ral-fistula-project.aspx*. This webpage contains a link to a reporting form that I hope contributors will download and complete before sending it to me via e-mail *(J.Reynolds.2@bham.ac.uk)*, together with accompanying high resolution digital images of birds in the field with oral fistulas. If potential contributors are in any doubt about whether the bird observed has an oral fistula, please send to me an e-mail to which images of the bird are attached so that I can view them and confirm that it has an oral fistula. Following confirmation, I am requesting that contributors complete the reporting form and send it to me for each observation of a bird with an oral fistula, whether it be a novel sighting or a re-sighting.

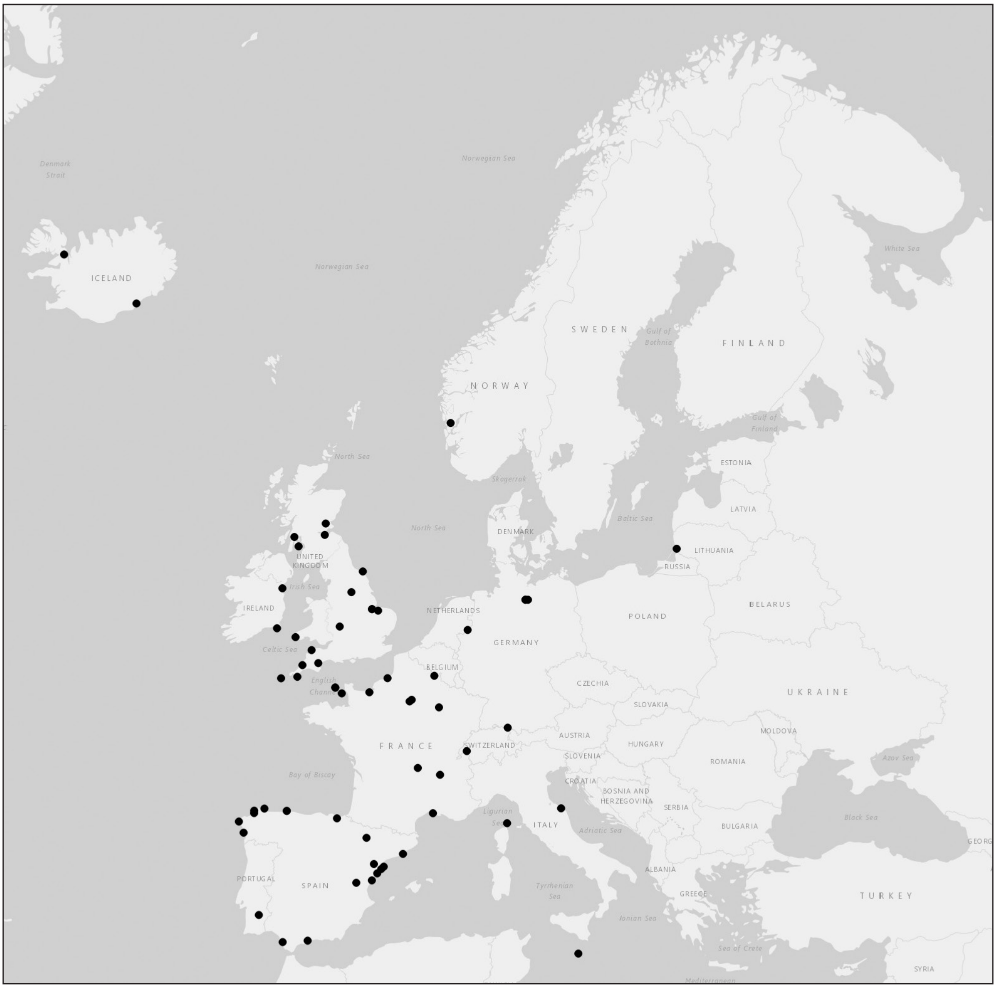

Figure 2. The European distribution of free-living birds with oral fistulas observed between the 3rd of October 2005 and the 27th September 2021

2. ábra Európa térképe olyan vadon élő madarak előfordulásának feltüntetésével, amelyeket a szövegben részletezett rendellenességgel (oral fistula) figyeltek meg 2005. október 7. és 2021. szeptember 27. között

I am appealing to readers (irrespective of whether they be birdwatchers, ornithologists, photographers or ringers) for help in collating records of free-living birds with oral fistulas in eastern Europe and also worldwide. To date, I have received 188 novel records of birds of 82 different species *(Table 1)* but the project database contains only one record of a bird with an oral fistula from eastern Europe *(Figure 2)*, a White-tailed Eagle *(Haliaeetus albicilla)* in Lithuania observed in January in 2016. As can be seen in *Figure 2*, observations of birds are concentrated in western Europe with clusters of birds appearing in north-eastern and eastern Spain, and south-western England, including the Western Jackdaw *(Corvus monedula)* photographed in *Figure 1*. The scarcity of observations from eastern Europe is notable; rather than it being indicative of an absence of birds with the condition, however, I conclude that readers who have detected oral fistulas in birds when watching, photographing and/or ringing them are unaware of my research project and that it is ongoing. Therefore, I urge readers please to consider connecting with me in the future if birds with oral fistulas are observed or if notes and images of such birds in some archived field notebooks are available. This would be a great opportunity for me to extend this project to eastern Europe and to other parts of the world.

Acknowledgements

I would like to thank numerous citizen scientists from many countries for contributing sightings of birds with oral fistulas to my research project to date. I thank Jenő Nagy for inviting me to submit this article to the journal.

References

Brown, M. E. 1996. Assessing Body Condition in Birds. – In: Nolan, V. & Ketterson, E. D. (eds.) Current Ornithology, Vol. 13. – Springer, Boston, MA, USA. DOI: 10.1007/978-1-4615-5881-1_3.

Camiña, A. & Guerrero, L. M. 2013. An Eurasian Griffon *Gyps fulvus* disadvantaged for feeding. – Vulture News 64: 66–68.

Castro, I. & Taylor, J. 2001. Survival and reproductive success of Stitchbird (hihi, *Notiomystis cincta*) suffering from a bill abnormality (oral fistula). – Notornis 48: 241–244.

Clayton, D. H., Lee, P. L. M., Tompkins, D. M. & Brodie, E. D. 1999. Reciprocal natural selection on host-parasite phenotypes. – The American Naturalist 154: 261–270. DOI: 10.1086/303237.

del Hoyo, J. (ed.) 2020. All the Birds of the World. – Lynx Edicions, Barcelona

Durell, S. E. A. Le V. dit, Ormerod, S. J. & Dare, P. J. 1996. Differences in population structure between two Oystercatcher *Haematopus ostralegus* roosts on the Burry Inlet, South Wales. – Ardea 84A: 383–388.

Ellington, C. P. 1991. Limitations on animal flight performance. – Journal of Experimental Biology 160: 71–91. DOI: 10.1242/jeb.160.1.71.

Gentle, L. K. & Gosler, A. G. 2001. Fat reserves and perceived predation risk in the Great Tit, *Parus major*. – Proceedings of the Royal Society B: Biological Sciences 268: 487–491. DOI: 10.1098/rspb.2000.1405.

Greño, J. L., Bertolero, A., Bort, J., Howard, R., Luque, E., Oro, D. & Sánchez, A. 2017. Audouin's Gulls *Larus audouinii* affected by sublingual fistulas. – Ardeola 64: 67–73. DOI: 10.13157/arta.64.1.2017.sct.

Houston, A. I. & McNamara, J. M. 1993. A theoretical investigation of the fat reserves and mortality levels of small birds in winter. – Ornis Scandinavica 24: 205–219. DOI: 10.2307/3676736.

Hughes, B. J., Martin, G. R., Wearn, C. P. & Reynolds, S. J. 2013. Sublingual fistula in a Masked Booby *(Sula dactylatra)* and possible role of ectoparasites in its etiology. – Journal of Wildlife Diseases 49: 455–457. DOI: 10.7589/2012-03-085.

Low, M., Alley, M. R. & Minot, E. 2007. Sub-lingual oral fistulas in free-living Stitchbirds *(Notiomystis cincta)*. – Avian Pathology 36: 101–107. DOI: 10.1080/03079450601142570.

Melville, D. S., Davaasuren, B. & Erdenechimeg, T. 2019. First sub-lingual oral fistula reported in a Swinhoe's Snipe *Gallinago megala*. – Wader Study 126: 64–66. DOI: 10.18194/ws.00133.

Murza, G. L., Bortolotti, G. R. & Dawson, R. D. 2000. Handicapped American Kestrels: Needy or prudent foragers? – Journal of Raptor Research 34: 137–142.

Reynolds, S. J., Martin, G. R., Wearn, C. P. & Hughes, B. J. 2009. Sub-lingual oral fistulas in Sooty Terns *(Onychoprion fuscatus)*. – Journal of Ornithology 150: 691–696. DOI: 10.1007/s10336-009-0377-3.

Rintoul, D. A. & Reynolds, S. J. 2019. Sublingual oral fistula in a Franklin's Gull *(Leucophaeus pipixcan)*. – Kansas Ornithological Society Bulletin 70: 53–56.

Rising, J. D. & Somers, K. M. 1989. The measurement of overall body size in birds. – The Auk 106: 666–674. DOI: 10.1093/auk/106.4.666.

Sharp, M. S. & Neill, R. L. 1979. Physical deformities in a population of wintering Blackbirds. – The Condor 81: 427–430. DOI: 10.2307/1366976.

Sutton, R. R. 1973. Harrier without feet. – Notornis 20: 74.

Temple, S. A. 1987. Do predators always capture substandard individuals disproportionately from prey populations? – Ecology 68: 669–674. DOI: 10.2307/1938472.

Van Hemert, C., Handel, C. M. & O'Brien, D. M. 2012. Stable isotopes identify dietary changes associated with beak deformities in Black-capped Chickadees *(Poecile atricapillus)*. – The Auk 129: 460–466. DOI: 10.1525/auk.2010.10111.

Witter, M. S., Cuthill, I. C. & Bonser, R. H. C. 1994. Experimental investigations of mass-dependent predation risk in the European Starling, *Sturnus vulgaris*. – Animal Behaviour 48: 201–222. DOI: 10.1006/anbe.1994.1227.

FORM FOR REPORTING ORAL FISTULAS IN BIRDS

(NOTE: Please complete a separate form for each bird observed with condition)

Record # (for use by SJR) (please ignore)	
Date	
Time	
Species' name (common, scientific or both)	
Was the bird alive (**A**) or dead (**D**)?	
Age of bird ([**A**] adult or [**C**] chick)	
Location (GPS coordinates, county or region, country etc.)	
Where was the bird seen (e.g. woodland, farmland, urban, landfill, harbour, river, lake, canal, on the coast, at-sea etc.)?	
Behaviour of bird (e.g. feeding, resting, incubating or brooding, fighting with another etc.)	
Was the bird ringed? (**Y**/**N**) (If '**Y**' please also supply the ring number if you have it)	
Is this the only time that you have seen this bird? (**Y**/**N**) (If '**N**' please provide further details including how many times in total you have seen it and at which sighting you first noted the oral fistula)	
Was the protruding tongue extended and coloured OR constricted, dry and discoloured?	
Did you photograph the bird? (**Y**/**N**) (If '**Y**' please send me the photograph(s) via e-mail. You will retain copyright and I will seek permission from you if I would like to use for an output)	
Were you ringing/banding, birdwatching, on a walk, photographing birds or something else when you made the observation?	
Your name and contact details	
Any other relevant information?	

NOTE: Please return to Dr Jim Reynolds via e-mail: J.Reynolds.2@bham.ac.uk