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Strong Decline in Breeding-Bird Community Abundance Throughout Habitats in the Azov Region (Southeastern Ukraine) Linked to Land-Use Intensification and Climate

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Abstract: In recent decades, bird communities associated with agricultural systems have declined in Western and Central Europe and in North America, but how widely these dynamics extend to other regions is poorly known. In this study, we assessed the dynamics and the main drivers of the changes in the abundance, richness, and composition of breeding bird communities over a 30-year period in the various types of habitats of southeastern Ukraine (Azov region), a region that has undergone agricultural intensification for several decades. This study was conducted in the valley of the Molochna River during the 1988 to 2018 nesting seasons. The area studied hosted 81 bird species. Species richness varied from three species in salt marshes to thirty-three in reed beds but did not show a general trend over time. However, we identified a decline in bird communities' total abundance over time in all habitats except shelterbelts and meadows. Similarly, species composition changed over time, reflecting the way in which species varied in their abundance changes. Climatic variables contributed to bird community turnover, but with decreasing importance over time. Overall, our results indicate that the recent declines in bird populations in farmland regions also extend to eastern Europe, with land-use effects interacting with climate to shape temporal dynamics of bird communities.

Keywords: temporal turnover; abundance; birds; communities; climate change; land use intensification

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1. Introduction

Beta diversity is an important concept in ecology, and denotes the heterogeneous distribution of biological objects from taxonomic, phylogenetic, and functional points of view [1,2], and over space or time [3–5]. Spatial turnover accounts for differences between communities due to the relocation of species from one place to another [6–8]. Temporal turnover is defined as a change in species composition observed in a single location over time [9,10]. Growing concern about the continued decline in biodiversity over the past four decades [11] has led to an increased interest in methods used to track the changes in community composition over time [12]. Thus, understanding the influence of internal and external factors on the turnover of natural biotic assemblages is fundamental to the sustainable management of ecosystems [13].

Climate change and habitat destruction are believed to be the main forms of human impact on biodiversity [14,15]. The increasing amount of information indicating that the global climate is becoming more variable heightens the importance of understanding the causes and effects of variability in communities [16–18]. It is well-known that habitat reduction due to land-use change affects population size and decreases genetic diversity within species [19,20]. However, land-use change usually results in more homogeneous landscapes, reducing ecosystem diversity [21,22]. The impact of land-use change on ecological community turnover is usually revealed through changes in abundance, richness, and assemblage composition [23–25]. Birds are a useful group for studying the effects of environmental change, because they are considered excellent indicators of ecosystem health [26]. Bird assemblage variations may indicate changes in both land use [27] and climate conditions [26].

In recent decades there has been evidence of widespread declines in common bird populations in both Western and Central Europe, and North America [28–32]. Burns et al., (2021) [33] estimated a decline of about 20% in the overall breeding bird abundance throughout European Union countries since 1980. In North American biomes, over the last 48 years the abundance of common species has decreased by 29% when compared to the abundance in 1970 abundance, a total loss of 3 billion birds [32]. In particular, there have been major declines in bird abundance in farmland, thought to be caused by agricultural intensification [33–38]. The primary agricultural drivers of reductions in farmland bird abundance include pesticides, habitat loss or fragmentation, mowing and harvesting operations, grazing disturbance, and reduced food availability [39]. Although recent declines in bird communities associated with agricultural systems are well-documented in Western and Central Europe and North America, the extent of these dynamics in other regions, for instance, in Europe east of the European Union, remains poorly understood. For example, although there are studies of the spatial distribution of bird communities in the steppe zone of Ukraine [40,41], temporal trends and potential drivers remain unclear.

To address the above-mentioned knowledge gap, this study is based on a 30-year data set of breeding bird communities in the habitats of the Azov region (Ukraine). The abundant data on birds in the region studied are unique for two reasons: First, in Ukraine there is no unified database of nesting bird populations, and secondly, the data were collected by one experienced observer over 30 years, limiting the scope for census errors. The purpose of this study was to determine principal trends in species richness, abundance, and the composition of breeding bird communities, and to assess drivers throughout habitats in the region studied. Specifically, we examined four hypotheses. (1) The bird communities exhibit monotonic trends over time, notably a decline in overall abundance, as reported from other farmland regions. (2) Temporal species composition change in the bird communities varies among biotopes, reflecting the role of land use in driving temporal community trends. (3) Species composition change is also influenced by temperature and precipitation, with biotope-specific effects, reflecting the interaction of climate and land use.

2. Materials and Methods

2.1. Types of Ecosystems Investigated

This study was carried out in the valley of the Molochna River, near the village of Svitlodolynske (Melitopol district, Zaporizhia region) during the 1988 to 2018 nesting seasons. The Molochna River (47°16″11′ N, 36°22′40″ E) is the largest river in the Azov region (southeastern Ukraine). This part of Ukraine has a homogeneous landscape, and the area studied is representative enough that the results may be extrapolated to a broader territory. Eight habitats for bird nesting were identified in the area investigated: reed beds, meadows, salt marsh (solonchaks), steppe areas, artificial forests (forest plantations), forest shelterbelts, rural areas, and agricultural habitats (Figure 1).

The climate of the Melitopol region is warm-temperate continental with long, hot, sunny, dry summers, frequent droughts and dry winds, relatively cool with little snow, and short, rainy winters, gusty winds and sand-dust storms (a typical coastal semi-desert steppe) [42]. The main natural landscape-forming factor in the investigated area is the Molochna River. The distribution of bird species in the biotopes of the studied region is greatly influenced by the river's hydrological regime and the anthropogenic structures of the habitat.

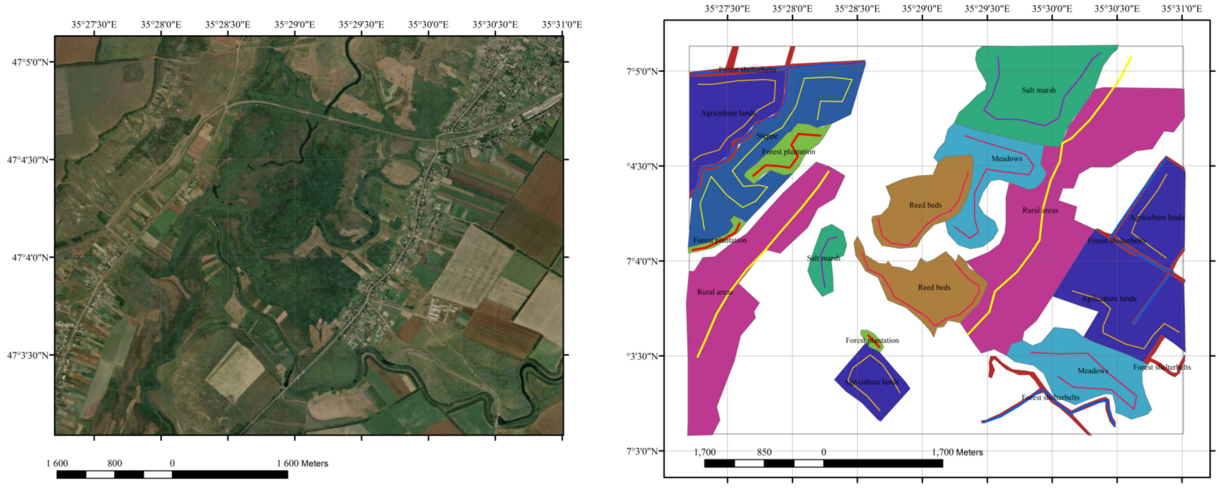


Figure 1. Types of biotopes within the area studied and survey transects.

The flora of reed beds around the Molochna River display significant species diversity, as in the south of Ukraine. The continental climate and the moisture deficit significantly affect the regional features of the flora. The most numerous of the flora's ecological groups was the group of steppe species. Among the biomorphs, herbaceous plants predominate. Meadow vegetation includes *Cynodon dactylon*, *Salicornia*, *Aeluropus*, *Frankenia*, *Juncus acutus*, *Limonium caspium*. The steppe biotope is part of the fescue-grass subzone of the steppe zone. Now, small virgin areas of the steppe are found only on the slopes of gullies and hills. Due to the lack of moisture, plants are drought-resistant, and herbage is sparse [43].

A characteristic element of the seaside landscape of southern Ukraine is solonchaks—the salt marsh along the banks of estuaries, in the valleys of small rivers, and in dried-up ponds. The main abiotic factors that determine the formation of various types of solonchak are humidity and temperature. As saline soils dry out, halophyte meadows of various types form on them, and as they become desalinated, solonchak-meadow associations form. The solonchak habitats are monotonous, low-halophytic or impoverished steppe vegetation dominated by *Suaeda vera*, *Halocnemum strobilaceum*, *Puccinellia fominii*, *Limonium vulgare*, *Artemisia austriaca*, *Tripolium pannonicum*, alternating with open saline areas of soil, small lakes, and mudflats [44].

During the twentieth century, most steppe were transformed into agrocenoses. A network of forest shelterbelts was created to reduce the impact of wind, heavy snowfall, pollution, and erosion. The most common tree species in shelterbelts are *Quercus robur*, *Robinia pseudoacacia*, and *Pinus nigra*. The forest is an intrazonal vegetation type in the area studied. Most of the artificial forest biotopes cover a small area, and the dominant tree species is *Pinus nigra* [45].

The rural area is primarily a subsistence farming region where people who live in suburban homes with gardens grow crops, or raise livestock primarily for their own consumption, but also includes more transformed sections, such as roads. Agricultural land consists of areas used for agricultural production. The main crops grown are cereals, sunflowers, rapeseed, and melons [46].

2.2. Field Data Collection

Bird data were collected using the line-transect method without restricting the width of a transect with a subsequent separate recalculation occurrence per route length [47]. Conditions in various types of biotopes vary greatly, both in terms of biotope size (from several meters wide for forest shelterbelts, to several kilometers for agricultural fields) and in terms of detection probability. Therefore, we counted the number of individual birds along fixed-length transects, obtaining comparable annual data for each biotope, without recalculating the density per unit area [48]. This study focuses on breeding birds because of their strong association with the nesting habitat. We considered only species recorded at least twice during nesting season. Birds were recorded visually and by voice. Flying birds were excluded, except when feeding airborne over the transects. We used 12-X binoculars to identify birds. Each transect was located within a homogeneous habitat (Figure 1). The total length of sampled transects was 36 km (Table A1). The survey design effectively sampled an area of 23 km^2. Bird counts were conducted from 6:00 a.m. to 10:00 a.m., during peak bird activity, and only in good weather (no heavy wind or rain). At least two surveys were conducted during the nesting period, which usually lasts from 20 April to 20 June. The speed of the bird observer along the transect was 2 to 4 km/h. We applied Stegman's (1938) bird taxonomy [49]. Bird encounters were recorded on special cards, scaled to 1:200,000 maps, and then transferred to the ArcMap 10.0 software [50].

2.3. Statistical Analysis

Abundance trends for all bird species were modeled using the *rtrim* package [51], an R package based on Trends and Indices for Monitoring Data (TRIM) software (TRIM v. 3.54. [52]). TRIM is designed to examine count time series and obtain unbiased annual indices and standard errors using log-linear models. The program also estimates the coefficient of variance, correcting for excessive variance, and accounts for consistent correlation between counts at the same site in different years [52]. The general bird population trends were obtained from the Birdlife International Data Zone [53].

The presence of autocorrelation in the time series of the number of species or the number of species in different habitats was assessed using the Durbin–Watson test [54] and the autocorrelation function (ACF). The Durbin–Watson test was calculated using the package *lmtest* [55]. The autocorrelation function was calculated using the built-in function *acf* from the Language and Environment for Statistical Computing R [56]. Generalized linear models to estimate the dependence of the number of species or the abundance of species of bird communities on the year, temperature, and precipitation were calculated using the built-in function *glm* [56] from Project R. Generalized linear models for count time series were calculated using the *tscount* [57]. The lags for the models were chosen based on the ACF. If the ACF indicated no autocorrelation, a lag of 1 was chosen. The selection of the best regression models was based on the Akaike information criterion (AIC) [58]. The links between species number, bird abundance values, and environmental factors for the total metacommunity were determined using the Multiple Generalized Linear Model (MGLM) with Poisson family with log link [59] using the TIBCO Software Statistics v. 12.0 PL software package [60]. MGLM with full factorial design: Biotope, Year, and Biotope × Year was calculated for abundance of species. For the number of species, this design had a larger AIC (1145.7) than the design incorporating Year and Biotope × Year, which had a smaller AIC (1138.8). Therefore, the article discusses the latter design.

This study was based on the framework for measuring temporal turnover developed by Shimadzu et al., (2015), according to which the temporal turnover (D) was decomposed into two components: the first term (D1) focused on the level of change in community composition (relative abundance distribution of the community), and the second term (D2) was determined by the amount of change in community size, in terms of its abundance. This fact emphasizes two critical components for assessing the turnover of a species community: (1) change in community composition and (2) change in total abundance. The turnover measure of the community between times t and u, (u > t) was defined as:

$$\mathrm{D}(\mathrm{t}:\mathrm{u}) = -\sum\nolimits_{s}^{i=1} log\left(\frac{p_i(t)}{p_i(u)}\right)p_i t + log\left(\frac{\lambda(\mathrm{u})}{\lambda(\mathrm{t})}\right) = D_1(p(t)\colon \mathrm{p}(\mathrm{u})) + D_2(\lambda(\mathrm{t}):\lambda(\mathrm{u})), \quad (1)$$

where $\mathrm{p_i(t)}$ is the relative abundance of the i-th species at time t, $\mathrm{p_i(u)}$ is the relative abundance of the i-th species at time u, λ(t) is the expected total abundance of the species in the community at time t, and λ(u) is the expected total abundance of the species in the community at time u.

The expected value of $\lambda_i(t)$ was modelled in the context of the mean annual temperature, total annual precipitation, and time variable (sequence of years). The effect of land cover was evaluated by comparing time series of diversity indicators in the various types of biotopes. To find drivers that influence the turnover measure, D, we determined the contribution ratio of the i-species and the j-th environmental variable. The contribution ratio indicates what proportion each species or factor contributes to the absolute amount of turnover [61].

2.4. Climatic Features

Average daily air temperature data were used to describe the temperature regime, and the data on average annual precipitation were used to describe the precipitation regime [42]. Information about the amount of precipitation and the temperature for the city of Dnipro was obtained from the National Oceanic and Atmospheric Administration (NOAA), and we implemented the library's RNOAA [62] as the language and environment for statistical computing in R [56].

During the period researched, the average annual temperature varied from 8.16 to 12.88 °C (mean ± st.error is 10.36 ± 0.20 °C, CV = 10.9%) (Figure 2). The temperature dynamics followed a linear pattern:

$$Temp = 9.57 + 0.0568 \cdot \mathrm{Y}\ (R^2 = 0.22,\ p = 0.007), \tag{2}$$

where Temp is the average annual temperature, Y is the order of years: 0–1988, 1–1989, . . . , 30–2018.

Annual rainfall varied between 297 and 472 mm (mean ± st.error is 362 ± 8 mm, CV = 12.0%) (Figure 2). The precipitation dynamics function was linear:

$$\mathrm{Prec} = 335 + 1.82 \cdot \mathrm{Y}\ (R^2 = 0.15,\ p = 0.03), \tag{3}$$

where Prec is the total amount of precipitation per year, Y is the order of years: 0–1988, 1–1989, . . . , 30–2018. Temperature and precipitation had a statistically significant relationship (r = 0.44, p = 0.013).

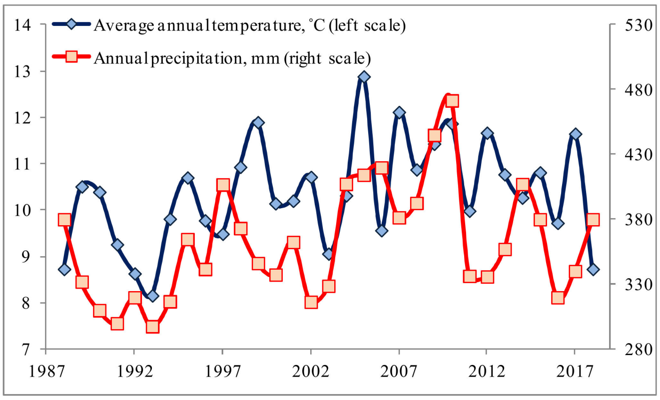


Figure 2. Dynamics of average annual temperature and annual precipitation: *x*-axis is a sequence of years, *y*-axis is average yearly temperature, left scale is average annual temperature in °C and right scale is annual precipitation in mm.

3. Results

The Molochna River valley habitats hosted 81 bird species from 32 families and 14 taxonomic orders (Table A2). The number of breeding bird species varied widely, depending on the biotope, ranging from three species in the solonchaks (salt marsh) to 33 in the reed beds (Table A3).

The analysis of the autocorrelation function (ACF) (Figure S1) and Durbin–Watson test values showed that the time series of the dynamics of the number of bird species in agricultural lands, forest plantations, and steppe had no statistically significant autocorrelation component (Table 1). The time series of the number of bird species in forest shelterbelts (lag 1, 2, 6 years), meadows (lag 1 year), reed beds (lag 1, 2, 7 years), and rural areas (lag 3 years) had statistically significant autocorrelation. The number of species in salt marshes did not change over time and remained low (3 species). The regression models with autocorrelation allow for the fact that the residuals of the time-dependent models of the number of species had no autocorrelation. However, based on the AIC, we can conclude that regression models without autocorrelation are favored. Therefore, multiple generalized linear models were calculated without accounting for autocorrelation (Table 2). Time as a continuous predictor had no statistically significant effect on the overall trend of species richness of bird communities.

Table 1. Generalized linear models of the time and biotope type effect on the bird community richness and Durbin–Watson test (DW) for time series and model residuals. Beta 1, 2, or 3 are autocorrelation regression terms which correspond lag (or lags) given in the "Lag" line.

Predictor	AL	AFB	AF	M	RB	RA	SM	S
DW for initial variable	2.4, $p = 0.80$	1.9, $p = 0.35$	2.0, $p = 0.43$	0.9, $p \leq 0.001$	1.4, $p = 0.03$	2.1, $p = 0.50$	–	1.7, $p = 0.16$
GLM without taking into account the autocorrelation								
DW for model residual	2.3, $p = 0.75$	1.9, $p = 0.38$	2.0, $p = 0.47$	0.9, $p \leq 0.001$	1.2, $p = 0.003$	2.0, $p = 0.45$	–	1.6, $p = 0.08$
Intercept	−4.49 ± 21.37	−30.6 ± 14.3	0.5 ± 18.9	−2.2 ± 15.4	11.1 ± 8.3	10.8 ± 11.6	–	19.1 ± 22.6
Year	0.003 ± 0.011	0.017 ± 0.007	0.001 ± 0.010	0.002 ± 0.008	−0.004 ± 0.004	−0.004 ± 0.006	–	−0.009 ± 0.012
Temp	−0.05 ± 0.09	−0.018 ± 0.065	0.013 ± 0.083	−0.013 ± 0.068	0.008 ± 0.036	−0.021 ± 0.051	–	0.013 ± 0.097
Prec	0.000 ± 0.002	0.000 ± 0.002	0.000 ± 0.002	0.000 ± 0.002	0.000 ± 0.001	0.000 ± 0.001	–	0.000 ± 0.002
AIC	114.3	142.0	121.2	134.7	171.2	151.8	–	113.2
GLM taking into account the autocorrelation								
Lag	4	1, 2, 6	2	1	1, 2, 7	3	–	1
DW for model residual	2.2, $p = 0.65$	2.2, $p = 0.59$	2.2, $p = 0.59$	2.1, $p = 0.49$	2.1, $p = 0.52$	1.9, $p = 0.27$	–	1.9, $p = 0.34$
Intercept	1.65 ± 22.44	−38.5 ± 17.3	1.3 ± 19.0	1.47 ± 15.72	10.8 ± 9.3	2.86 ± 15.95	–	1.8 ± 24.8
Beta1	−0.25 ± 0.62	−0.004 ± 0.002	0.339 ± 1.209	0.47 ± 0.52	0.001 ± 0.037	0.135 ± 0.385	–	
Beta2	–	0.003 ± 0.002	–	–	0.030 ± 0.027	–	–	0.266 ± 0.505
Beta2	–	−0.002 ± 0.001	–	–	−0.021 ± 0.008	–	–	–
Year	0.000 ± 0.012	0.020 ± 0.009	0.000 ± 0.010	0.000 ± 0.008	−0.004 ± 0.005	0.000 ± 0.008	–	0.000 ± 0.012
Temp	−0.03 ± 0.09	−0.014 ± 0.065	0.018 ± 0.083	0.004 ± 0.068	0.007 ± 0.038	−0.027 ± 0.051	–	0.005 ± 0.098
Prec	0.001 ± 0.002	0.000 ± 0.002	0.000 ± 0.002	0.000 ± 0.002	0.000 ± 0.001	0.000 ± 0.001	–	0.000 ± 0.002
AIC	116.2	143.4	123.1	135.1	175.2	154.0	–	115.4

Table 2. Multiple generalized linear model of the effect of time and biotope type on bird community richness (the estimates for factors represent differences from the median value for the metacommunity as a whole).

Effect	Coefficient ± Standard Error	Lower CL	Upper CL	Wald Statistic	*p*-Level
Intercept	3.91 ± 4.23	−4.37	12.20	0.9	0.355
Year	−0.00093 ± 0.00211	−0.00506	0.00321	0.2	0.661
Agricultural lands × Year	−0.00027 ± 0.00004	−0.00035	−0.00020	49.0	<0.001
Forest shelterbelts × Year	0.00011 ± 0.00003	0.00006	0.00016	15.5	<0.001
Meadows × Year	0.00006 ± 0.00003	0.00000	0.00011	3.6	0.05
Reed beds × Year	0.00067 ± 0.00002	0.00064	0.00071	1319.6	<0.001
Rural areas × Year	0.00035 ± 0.00002	0.00030	0.00039	223.4	<0.001
Salt marshes × Year	−0.00048 ± 0.00005	−0.00057	−0.00039	103.7	<0.001
Steppe × Year	−0.00031 ± 0.00004	−0.00039	−0.00023	60.1	<0.001
Forest plantations × Year	−0.00015 ± 0.00003	−0.00022	−0.00008	18.0	<0.001

Stable population dynamics were found for 49 species of birds (60.5% of the total number), and a statistically significant trend of moderate abundance decrease was found for 21 species of birds (25.9%). A strong decrease in abundance was recorded for 3 species (3.7%). Moderate increase in abundance was found for 8 species (9.9%) (Table A4).

The analysis of the autocorrelation function (ACF) (Figure S2) and Durbin–Watson test values showed that the time series of abundance dynamics of species of bird communities in artificial forest belts, artificial forests, meadows, and reed beds had no statistically significant autocorrelation (Table 3). The time series of abundance of species in agricultural lands (lag 1 year), rural areas (lag 1, 2, 3 years), salt marshes (1, 3, 4, 5 years), and steppe (1, 2, 3 years) had statistically reliable autocorrelation. The regression models with autocorrelation allow for the fact that the residuals of the models of time-dependent dynamics of species abundance had no autocorrelation. However, based on the AIC, we can conclude that regression models without autocorrelation are favored. Therefore, the multiple generalized linear model was calculated without accounting for autocorrelation (Table 4). The biotopes differed in the temporal trend of species richness variability. The number of species decreased over time in agricultural land, salt marshes, steppe, and forest plantations. The number of species increased in forest shelterbelts, reed beds, and rural areas. The decrease in bird community abundance over time was a common trend in all habitats.

Table 3. Generalized linear models of the time and biotope type effect on the bird community abundance and Durbin–Watson test (DW) for time series and model residuals. Beta 1, 2, or 3 are autocorrelation regression terms which correspond lag (or lags) given in the "Lag" line.

Predictor	AL	AFB	AF	M	RB	RA	SM	S
DW for initial variable	1.4, $p = 0.02$	1.6, $p = 0.10$	1.5, $p = 0.05$	1.7, $p = 0.15$	2.0, $p = 0.41$	1.1, $p \leq 0.001$	1.9, $p = 0.34$	1.8, $p = 0.28$
GLM without taking into account the autocorrelation								
DW for model residual	1.5, $p = 0.05$	1.5, $p = 0.07$	1.5, $p = 0.06$	1.9, $p = 0.27$	1.9, $p = 0.31$	1.4, $p = 0.03$	2.0, $p = 0.37$	1.8, $p = 0.25$
Intercept	21.27 ± 13.92	−8.11 ± 9.65	19.2 ± 13.2	−4.38 ± 5.66	89.4 ± 1.9	108 ± 2.9	49.2 ± 9.1	34.7 ± 16.2
Year	−0.009 ± 0.007	0.005 ± 0.005	−0.008 ± 0.007	0.004 ± 0.003	−0.042 ± 0.001	−0.052 ± 0.001	−0.022 ± 0.005	−0.016 ± 0.008
Temp	0.043 ± 0.060	−0.024 ± 0.042	0.005 ± 0.057	−0.021 ± 0.024	0.111 ± 0.007	0.053 ± 0.011	−0.073 ± 0.039	−0.020 ± 0.069
Prec	−0.001 ± 0.001	0.002 ± 0.001	0.000 ± 0.001	0.003 ± 0.001	0.002 ± 0.000	0.000 ± 0.001	−0.001 ± 0.001	0.001 ± 0.002
AIC	158.1	194.8	154.7	437.7	4676.6	743.7	218.8	138.6
GLM taking into account the autocorrelation								
Lag	1	1	1	4	1	1, 2, 3	1, 3, 4, 5	1, 2, 3
DW for model residual	2.1, $p = 0.50$	2.0, $p = 0.42$	1.9, $p = 0.27$	2.1, $p = 0.49$	2.0, $p = 0.37$	1.8, $p = 0.21$	1.6, $p = 0.10$	1.9, $p = 0.34$
Intercept	2.5 ± 15.6	1.63 ± 10.2	2.64 ± 13.8	0.82 ± 7.06	95.3 ± 2.0	5.63 ± 4.66	3.28 ± 17.6	2.5 ± 21.9
Beta1	0.284 ± 0.200	0.20 ± 0.15	0.21 ± 0.27	0.033 ± 0.027	−0.004 ± 0.000	0.129 ± 0.025	0.053 ± 0.116	0.090 ± 0.278
Beta2	–	–	–	–	–	0.079 ± 0.022	0.141 ± 0.106	−0.092 ± 0.290
Beta3	–	–	–	–	–	0.131 ± 0.020	0.003 ± 0.117	0.289 ± 0.251
Beta4	–	–	–	–	–	–	0.103 ± 0.109	–
Year	0.000 ± 0.008	0.000 ± 0.005	0.000 ± 0.007	0.001 ± 0.004	−0.045 ± 0.001	−0.001 ± 0.002	0.000 ± 0.009	0.000 ± 0.011
Temp	0.036 ± 0.061	−0.038 ± 0.045	0.003 ± 0.062	−0.021 ± 0.024	0.090 ± 0.008	0.009 ± 0.012	−0.085 ± 0.040	−0.051 ± 0.074
Prec	−0.001 ± 0.001	0.003 ± 0.001	0.000 ± 0.001	0.003 ± 0.001	0.003 ± 0.000	−0.002 ± 0.000	−0.001 ± 0.001	0.001 ± 0.002
AIC	159.1	195.8	157.5	438.1	4701.3	1169.3	236.0	144.4

Increased temperature and precipitation had a positive effect on the abundance of bird communities. A decline in the abundance of bird communities was found in reed beds, rural areas, and salt marshes. An increase in the abundance of bird communities has been found in forest shelterbelts and meadows. No statistically significant temporal trend was found in agricultural land, steppe, and forest plantations.

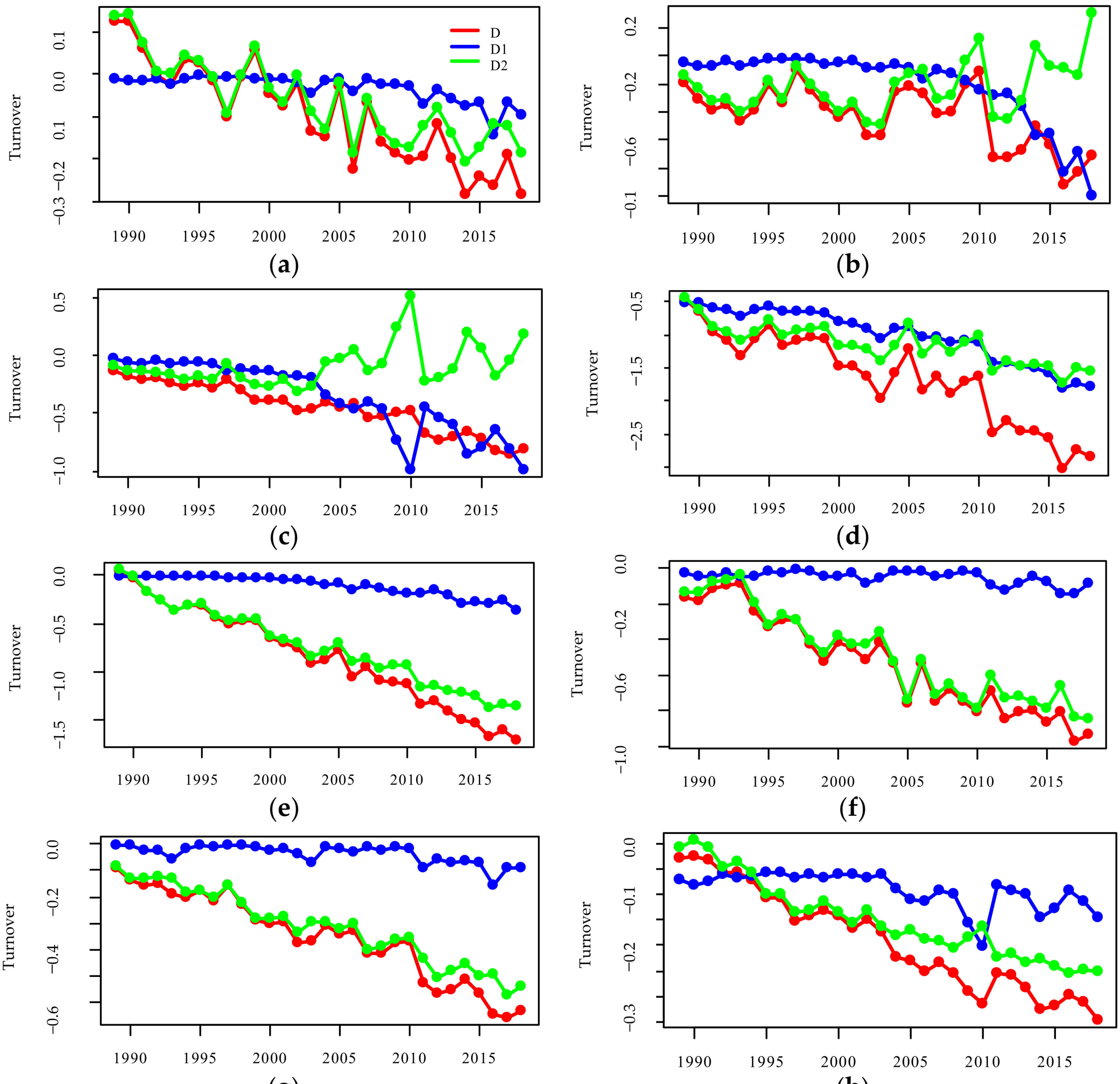
The greatest differences in the number of birds compared to the regional average were observed in forest shelterbelts, meadows, reed beds, and rural areas. Average bird abundance has significantly changed over time, especially in forest plantations, shelterbelts, meadows, and rural areas. Most significantly, precipitation influences the bird abundance in communities of meadows and salt marshes. Therefore, the bird communities in different habitats of the Molochna River valley changed in composition over the years, compared to the initial observations made in 1988 (Figure 3). The turnover measure D is characterized by a descending pattern for bird communities in all types of biotopes, reflecting increasing change over time. This is mainly due to changes in bird community abundance (D2), which decreased in all investigated habitats, except shelterbelts and meadows. Moreover, the dynamics of the turnover of bird assemblages in agricultural land, rural areas, salt, and steppe are completely synchronic with the dynamics of community abundance. Changes in bird community composition in the meadows are accompanied by a decrease in their total abundance (Figure 4). In comparison to the initial period, bird species' relative abundance distribution (D1) remained almost unchanged in agricultural land, rural areas, salt marshes, and steppe. Significant changes in community structure over time were found for birds in shelterbelts, meadows, reed beds, and tree plantations. Typically, strong fluctuations in bird community composition were evident in most biotopes after 2005.

Table 4. Multiple generalized linear model of the effect of precipitation, temperature, time, and biotope type on bird community abundance (the estimates for factors represent differences from the median value for the metacommunity as a whole).

Effect	Coefficient ± Standard Error	Lower CL	Upper CL	Wald Statistic	*p*-Level
Intercept	52.82 ± 2.97	47.00	58.63	316.8	<0.001
Year	−0.025 ± 0.001	−0.028	−0.022	284.6	<0.001
Temp	0.079 ± 0.005	0.069	0.090	219.9	<0.001
Prec	0.002 ± 0.000	0.001	0.002	133.0	<0.001
Agricultural lands	−18.19 ± 11.31	−40.35	3.97	2.6	0.11
Forest shelterbelts	−53.56 ± 8.12	−69.49	−37.64	43.5	<0.001
Meadows	−53.24 ± 5.28	−63.59	−42.89	101.6	<0.001
Reed beds	30.70 ± 3.20	24.44	36.97	92.3	<0.001
Rural areas	64.40 ± 3.59	57.36	71.44	321.2	<0.001
Salt marshes	22.75 ± 7.61	7.84	37.67	8.9	<0.001
Steppe	−3.95 ± 12.83	−29.09	21.19	0.1	0.76
Forests plantations	−19.33 ± 10.75	−40.39	1.73	3.2	0.07
Agricultural lands × Year	0.008 ± 0.006	−0.003	0.019	2.2	0.14
Forest shelterbelts × Year	0.026 ± 0.004	0.018	0.034	42.3	<0.001
Meadows × Year	0.027 ± 0.003	0.022	0.032	103.0	<0.001
Reed beds × Year	−0.014 ± 0.002	−0.017	−0.011	76.8	<0.001
Rural areas × Year	−0.031 ± 0.002	−0.035	−0.028	303.0	<0.001
Salt marshes × Year	−0.012 ± 0.004	−0.019	−0.004	9.4	<0.001
Steppe × Year	0.001 ± 0.006	−0.011	0.014	0.03	0.86
Forest Plantations × Year	0.009 ± 0.005	−0.002	0.019	2.8	0.09

The species whose abundance changed most significantly during the period studied may be considered species with the highest contribution ratio to the total temporal β-diversity of community. In some communities, the species whose abundance changed most dramatically are clearly apparent, such as *Alauda arvensis* in agricultural land, *Sturnus vulgaris* in rural areas, or *Vanellus vanellus* in salt marshes (Figure 4). In other biotopes, such as meadows, steppes, and tree plantations, species displayed more equal changes in abundance.

Environmental factors affect the turnover of bird communities in different ways, depending on the habitat (Figure 5). Nevertheless, the significance of the temporal factor in species turnover was most important for communities in every studied biotope. The contribution ratio of temperature and precipitation in turnover decreased during the research period (Figure 5). The ratio of temperature contribution to turnover was greater than precipitation in agricultural land, reed beds, and rural areas, whereas precipitation was the more influential climatic factor in other habitats.



(**a**) (**b**) (**c**) (**d**) (**e**) (**f**) (**g**) (**h**)

Figure 3. Bird species composition turnover (D: the red line) and its components, composition change (D1: the blue line) and community size (species richness) change (D2: the green line) obtained after the turnover analysis of the bird communities at Molochna River valley over the period (1988–2018): the abscissa is the order of years and the ordinate is the turnover. (**a**)—Agricultural lands, (**b**)—Forest shelterbelts, (**c**)—Meadows, (**d**)—Reed beds, (**e**)—Rural areas, (**f**)—Salt marshes, (**g**)—Steppe, (**h**)—Forest plantations.

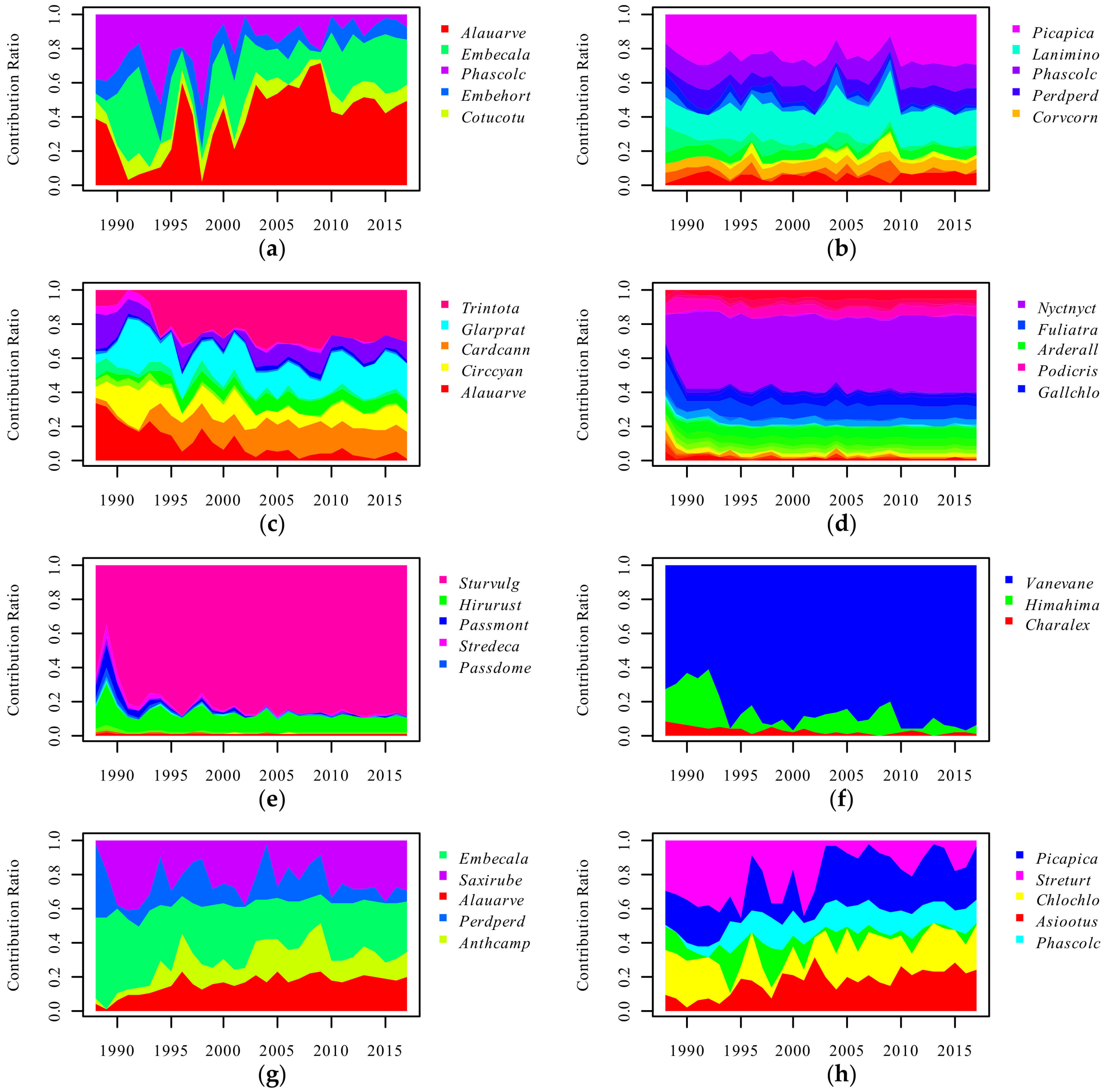


Figure 4. Contribution ratios of each species obtained after analyzing the turnover of bird communities in the Molochna River valley for the period (1988–2018). The top five species with the highest contribution ratio are stated in the legend: the abscissa is the order of years, the ordinate is the contribution ratio; *Alauarve—Alauda arvensis*, *Cotucotu—Coturnix coturnix*, *Embecala—Emberiza calandra*, *Embehort—Emberiza hortulana*, *Phascolc—Phasianus colchicus; Forest shelterbelt*: *Corvcorn—Corvus cornix*, *Lanimino—Lanius minor*, *Perdperd—Perdix perdix*, *Phascolc—Phasianus colchicus*, *Picapica—Pica pica; Meadows*: *Alauarve—Alauda arvensis*, *Cardcann—Carduelis cannabina*, *Circcyan—Circus cyaneus*, *Glarprat—Glareola pratincola*, *Trintota—Tringa totanus; Reed beds*: *Arderall—Ardeola ralloides*, *Fuliatra—Fulica atra*, *Gallchlo—Gallinula chloropus*, *Nyctnyct—Nycticorax nycticorax*, *Podicris—Podiceps cristatus; Rural areas*: *Hirurust—Hirundo rustica*, *Passdome—Passer domesticus*, *Passmont—Passer montanus*, *Stredeca—Streptopelia decaocto*, *Sturvulg—Sturnus vulgaris; Salt marshes*: *Charalex—Charadrius alexandrinus*, *Himahima—Himantopus himantopus*, *Vanevane—Vanellus vanellus; Steppe*: *Alauarve—Alauda arvensis*, *Anthcamp—Anthus campestris*, *Embecala—Emberiza calandra*, *Perdperd—Perdix*, *Saxirube—Saxicola rubetra; Forest plantations*: *Asiootus—Asio otus*, *Chlochlo—Chloris chloris*, *Phascolc—Phasianus colchicus*, *Picapica—Pica pica*, *Streturt—Streptopelia turtur*. (**a**)—Agricultural lands, (**b**)—Forest shelterbelts, (**c**)—Meadows, (**d**)—Reed beds, (**e**)—Rural areas, (**f**)—Salt marshes, (**g**)—Steppe, (**h**)—Forest plantations.

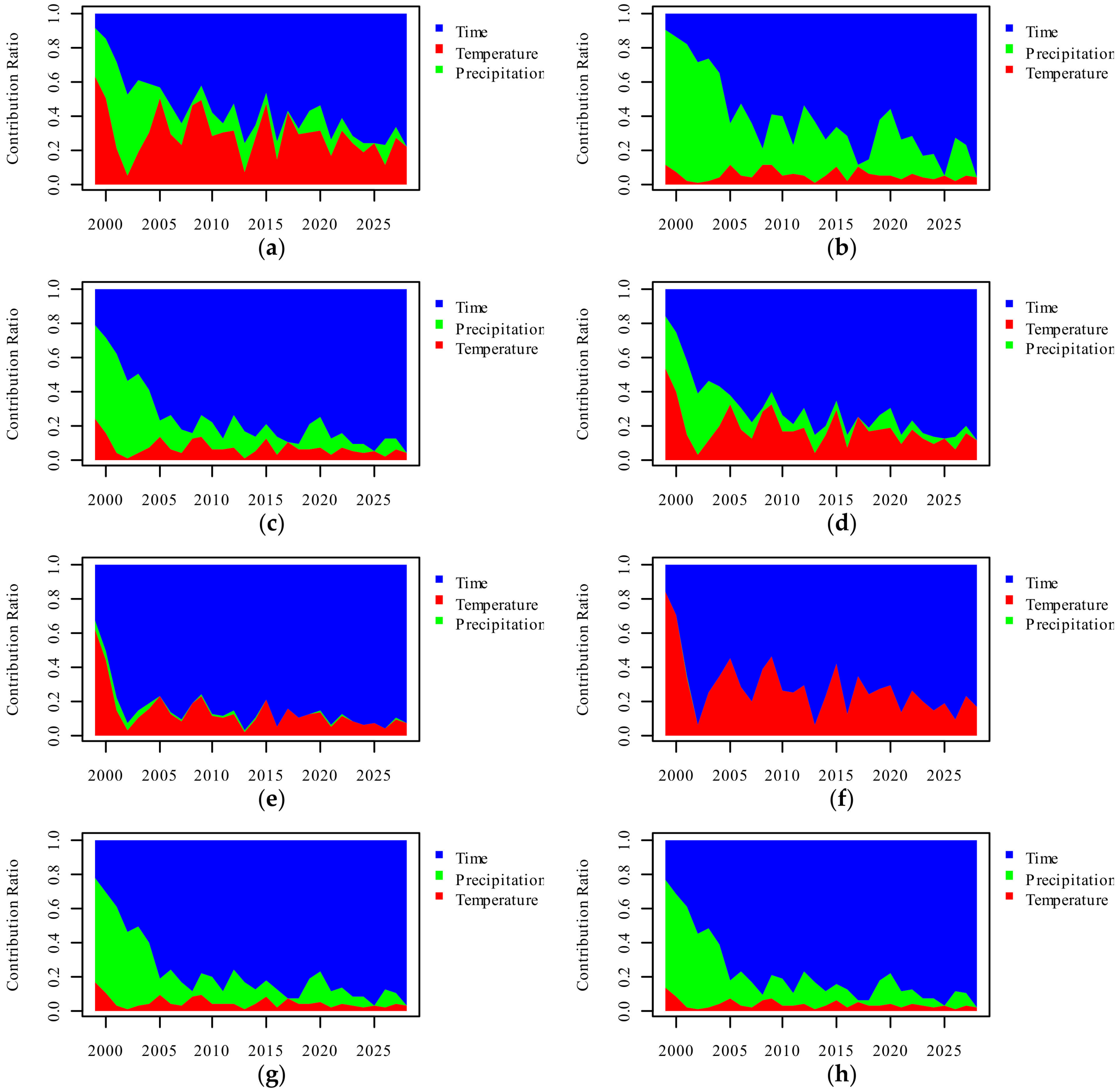


Figure 5. Contribution ratios of each environmental factors estimated after the turnover analysis on bird communities at the Molochna River valley over the period (1988–2018): the abscissa is the order of years, the ordinate axis is the contribution ratio; year (blue), precipitation (green) and temperature (red). (**a**)—Agricultural lands, (**b**)—Forest shelterbelts, (**c**)—Meadows, (**d**)—Reed beds, (**e**)—Rural areas, (**f**)—Salt marshes, (**g**)—Steppe, (**h**)—Forest plantations.

4. Discussion

4.1. General Trends in the Diversity and Abundance of Bird Species in the Area Studied

Although species abundance varies considerably among biotopes, there is a general trend that reveals a decrease of species abundance in most biotopes. Recently, many countries around the world have seen a staggering decline in bird populations, especially in communities associated with agricultural habitats [28,33,36,37,39]. Many birds in agricultural regions have suffered from habitat loss and degradation as a result of agricultural intensification [34,38].

Species richness also depends significantly on biotope type. A feature of the studied area is its highly mosaic habitat system, within which there are many different ecological gradients. The habitat continuum as a function of moisture levels—from steppes to meadows beside rivers, to reedbeds in river floodplains—reflects the most significant ecological gradient. Since this gradient affects the species composition of plants, it also reflects the ecosystem's nutrient regime [63,64]. Consistent with this trend, bird species richness increases from steppe (five species) to meadows (14 species) to reed beds (33 species) (Table A2).

Another type of ecological gradient reflects the level of anthropogenic load. Rural areas were the most transformed biotopes of the area studied and were characterized by comparatively high species diversity (18 species). This may be explained by the fact that some human settlements have displayed some of the greatest potential for increases in species richness over the last 100 years [65]. However, we are currently observing a significant decrease in bird richness and abundance in rural areas over time (Table 2).

Bird communities in agricultural areas, which are derivatives of steppe ecosystems, are less diverse than those in rural areas. Although the total number of bird species is similar in steppe and agricultural areas (five species), the composition of the communities differs (Table A2), though there are some common species. This confirms the fact that the anthropogenic transformation of habitats completely changes the functioning of an ecosystem, and restructures biological communities [66,67]. The total abundance of bird species in both communities strongly declined when compared to the initial period of research (Figures 3 and 4). The driver that limits species composition and the numbers of birds in these types of habitats is the continued intensification of agriculture [68].

Despite their small area, forest shelterbelts have become important centers of bird diversity. Today, thanks to the developed network of forest belts, small forested areas of southern Ukraine are connected to large forests of the Forest–Steppe and Polissia regions [69]. The forest shelterbelts are insular in nature, as they are usually surrounded by agro-habitats. Crows and small passerines predominate among the nesting birds in shelterbelts. The diversity of birds in forest shelterbelts tended to increase over the period studied (Table 2), indicating the importance of this habitat type in maintaining the ecosystem services of adjacent agricultural habitats [70].

Salt marshes belong to the azonal habitat type, and they are scattered in small areas throughout the region. The basis of nesting bird assemblages of solonchaks are mainly the *Charadriidae* and *Recurvirostridae* families and the species composition is poor. Nevertheless, salt marshes are the nesting places of protected bird species. All the breeding bird species we encountered in the salt marshes have a high conservation status. Thus, the Kentish plover (*Charadrius alexandrinus*) and black-winged stilt (*Himantopus himantopus*) are listed in the Red Data Book of Ukraine, and the northern lapwing (*Vanellus vanellus*) is classified as "Near Threatened," according to the IUCN Red List [71]. Therefore, despite its small area and low total species richness, this type of biotope is important for the conservation of the region's biodiversity.

4.2. Temporal Turnover of Bird Communities in Various Biotopes

Our results reveal that the total temporal turnover of bird communities displays a downward trend in all types of biotopes in the area studied, which is consistent with the pan-European trend [29,32,33]. During 1988–2018, the main changes have been to the abundance of bird communities, whereas the composition of the communities (relative abundance of species) in most habitats remained relatively stable. This indicates that, for heterogeneous steppe regions, relative species abundance is a less sensitive indicator of community dynamics influenced by external factors than their total abundance. The only biotopes in which the total abundance of bird communities increased were forest belts and meadows. Since 1988, the species composition of these communities has been gradually changing in forest shelterbelts, meadows, and reed beds (Figure 3).

Shelterbelts are known to provide many ecological and social benefits, including climate change mitigation and biodiversity conservation [72]. Our research shows the importance of forest shelterbelts in maintaining a region's biodiversity, as this is the only biotope in which both species diversity and the total abundance of birds in a community increased over time. The community composition changed substantially during the period studied. A noticeable increase in abundance was observed in species for which human development of an area (an increase the area of arable land and the appearance of settlements) is favorable, for example, *Corvidae*. Nevertheless, the nesting species of shelterbelts are capable of exterminating pests and rodents outside forest belts in adjacent fields and orchards, so it is important to protect them and maintain their abundance.

The reed bed biotope was another that underwent significant changes in terms of species richness and relative abundance. The most probable cause of the decline in bird species diversity and total community abundance in this type of biotope is the gradual decrease in the water level in the Molochna River [72]. Changes in the river's hydrological regime led to the plant succession of reed bed habitat [73]. *Ciconiiformes* birds react most strongly to changes in the river's hydrological regime and the state of the reeds. In high-water years, many isolated feeding bays are formed, which are inaccessible to four-legged predators, and the rapid development of vegetation creates favorable protective conditions for nests with clutches and broods. The area suitable for nesting increases significantly, leading to the dispersal of birds throughout the biotope. In dry years, the opposite pattern is observed: most birds concentrate in a few suitable areas, which reduces breeding success.

Meadow ecosystems in the Azov region are among the intensively transformed habitats; therefore, research into the direction and degree of the transformational processes occurring in them is important for the preservation of biotic diversity in general, and avian fauna in particular [73]. Significant anthropogenic transformation has led to an increase in the total abundance of birds, and to a significant change in the composition of the bird community, compared to the initial period (Figure 3). The formation of avian communities in meadows as a result of natural and anthropogenic changes continues to this day. Nesting populations of several bird species are degrading, but at the same time, new, previously absent bird species are appearing [74].

The decline in bird abundance worldwide is thought to be attributable to the more common species [29,33,75], whereas the less common species have demonstrated an increase in abundance in some areas [29]. In the area we studied, both common and rare species became less abundant. Thus, for example, the relative abundance of species in the salt marsh bird communities changed only slightly during the research period, though the total abundance decreased dramatically (Figure 3). Since the community consists of species that require protection, this situation is unfavorable. Typically, only a few species of birds that permanently nest in farmland—primarily corvids—benefit from increased agricultural intensification [75,76].

4.3. Identification of Bird Species with the Highest Rates of Contribution to Community Turnover

Each biotope has its own set of bird species that contributes most to turnover within it (Figure 4). This again confirms the distinctiveness of the temporal turnover of communities in each habitat type [65,77]. In some biotopes, the cohort of birds that contribute most to the turnover of the community is unique, that is, there is no overlap with other types of biotopes. Such biotopes include rural areas, reed beds, and salt marshes. Rural areas are characterized by higher anthropogenic pressure, but also better nesting and foraging opportunities for some species. Other researchers have also noted specific patterns of bird-species composition in urban and rural habitats [78,79].

The species composition of vegetation in artificial forest belts is usually limited. The forest litter is poorly developed and poor in invertebrates, and the herbivorous insect fauna is not numerous. Such areas are excellent habitats for migratory birds but are not very suitable for nesting species that forage among trees and shrubs. As a result, species such as *Pica* and *Lanius minor*, which breed in trees and shrubs, but feed on steppe, fields, and meadows, predominate (Figure 4). The bird fauna is supplemented by species capable of nesting in open areas, but that concentrate near areas with bushes for shelter and food (*Perdix*, *Phasianus colchicus*), or that more often nest on the ground but are able to build nests in bushes and small trees (*Sylvia communis*, *Emberiza hortulana*). However, in forest shelterbelts, the composition of core species—birds that have been recorded from year to year, and have adapted to living in this habitat [80,81]—remains unchanged (Table A4). Shifts in species composition and their abundance in forest belts occur because of the appearance of satellite or occasional species, such as *Luscinia megarhynchos*, *Upupa epops*, *Sylvia communis*, and *Streptopelia turtur*, that are less common to the habitat [82]. The contribution ratios of each core species in the turnover process of the forest belt varied within a narrow range (Figure 4), indicating that the increase in the abundance measure of the bird community turnover was primarily caused by a relatively proportional increase in the numbers of both core and occasional species.

The similarity in species composition between the two habitats is believed to decrease as the distance between the two habitats increases [83–86]. This pattern may be traced between communities of birds from steppe and agricultural habitats (probably because one biotope arose from the other), and between communities of forest biotopes, which have the largest number of common species that contribute to temporal turnover. However, this pattern is not confirmed for biotopes with vegetation unique to the steppe zone, such as salt marshes and reed beds [87].

During the last 18 years of this study, the greatest contribution to the increase of the community abundance in meadow habitats was made by the *Tringa totanus* population, whose contribution rate on community turnover increased. However, the contribution of *Alauda arvensis* decreased (Figure 4). We believe that species turnover reflects the transition between the core species and the occasional species [81], so if there is a successional dynamic, then it reflects the fact that the species change place in their roles in community turnover. Such replacement of some species by others may be observed in meadow biotopes, probably because of the successional dynamics of vegetation. *Glareola pratincola* contributes significantly to the turnover of bird communities, the abundance of which has monotonously decreased since 2013, which is consistent with the general global trend of this species' population, which has tended to decline in recent years (Table A4) [53].

The populations of all species in salt marshes have sharply decreased. *Vanellus* is the most numerous species. Although its population is declining, its significance in the community structure is increasing, as the abundance of the other two species has decreased even faster. *Vanellus* populations are declining worldwide [53]. *Charadrius alexandrinus* is the least abundant species in solonchak biotopes; its abundance remained low throughout the entire period studied, so its contribution rate in the community is stable. The abundance of *Himantopus* fluctuated throughout the period studied, resulting in its reduced contribution to the temporal dynamics of the community.

Phasianus colchicus is a species that contributes to the turnover of three types of biotopes simultaneously: agricultural land, forest plantations, and forest belts (Figure 4). In Ukraine, the common pheasant is an introduced, naturalized species. The species is common in forest shelterbelts along riverbanks but rare in forest belt agro-habitats far from fresh water sources. However, its role in agro-habitats has decreased over time, while it is relatively stable in forest shelterbelts, and is increasing in forest plantations. The decline of the common pheasant in agricultural areas worldwide (Table A4) is associated with the intensification of agricultural production, which is accompanied by damage to the habitat (loss of field-edge habitat (fewer fencerows), the removal of bushes, the trend of monoculture, suburban sprawl, etc.) [88]. The increase in the role and abundance of *Phasianus colchicus* in forest habitats is associated with the targeted breeding of the species in forestry enterprises, for hunting.

The steppe avian communities are considered among the most vulnerable in south-eastern Ukraine. Over the past 100 years, more than 10 species of nesting birds have disappeared from the region's steppe habitats, including *Aquila rapax*, *Anthropoides virgo*, *Otis tarda* [89]. The degradation of steppe biotopes has led to a decrease in bird diversity and their total abundance (Table 2, Figure 3). This is largely due to the very limited area of natural steppe areas, which are preserved as spaces in the modern agro-landscape. Small passerines (*Emberiza calandra*, *Alauda arvensis*, *Anthus campestris*, and *Saxicola rubetra*) form the basis of the bird community in the steppe (Figure 4). The predominance of ground-nesting passerines is characteristic of the remaining virgin steppe areas throughout the Eurasian steppe belt [90].

4.4. Main Drivers of Bird Community Turnover in the Molochna River Valley

A sharp reduction in the total abundance of bird species in communities in all biotopes indicates the presence of some external factor whose intense influence is increasing over time. The greatest contribution to turnover in each biotope has a time factor. The time factor (t) may change in magnitude as a result of climate change, biotic influx, or human disturbance [91]. When climatic factors are considered separately, the most probable constant factor whose intensity is increasing over time is an anthropogenic influence.

According to the results of this study, changes in the biotope over time significantly affect species richness (Table 1) and are therefore reflected in the process of community turnover. The most probable reason for temporal fluctuations in bird community turnover is the anthropogenic transformation of habitats. As the largest area in the studied region is characterized by agricultural habitats, we may conclude that the factor intensity which most increased throughout the period studied is the intensification of agriculture, which in turn led to a decrease in the total abundance of birds in ecosystems. From 1988 to 2018, agricultural production in Ukraine underwent many changes: from extensive land use in 1988 through 1991, to its decline in the 1990s (in connection with the collapse of the Soviet Union and restructuring of all economic spheres) and the subsequent intensification of production in recent years [92]. Between 1990 and 2010, Ukraine's ploughed area decreased by 21% [93], although most abandoned land lay outside the steppe zone [94]. Thus, the steppe bird communities in Ukraine did not benefit significantly from post-Soviet land-abandonment, as they did in Kazakhstan and Russia [95,96]. Despite a 6.6% decrease in the arable area of the Zaporizhzhia region in 2010, when compared to 1990, between 2011 and 2018 the area of arable land increased by almost 5% [97]. The area planted for cereals, sunflowers, and rapeseed has continuously expanded since 1990 [93,97]. At the same time, this territory is characterized by the intensification of agriculture, which is defined as an increase in agricultural commodities per unit area [98]. Crop yield in Ukraine, and also in the Melitopol region, has increased over the past 10 to 15 years [99,100]. The intensification of agriculture was accompanied not only by the expansion of arable land, but also by an increase in the number of pesticides and chemical fertilizers used. The influence of agricultural intensification on the decline in farmland bird populations has been proved for many European countries [33,35–39]. Our studies are consistent with the assertion that increases in the intensity of agricultural production inevitably led to a decrease in farmland biodiversity, namely the abundance of bird communities. Moreover, we found that at local (regional) scale, the influence of anthropogenic factors (including land-use change) has a greater impact on a bird community's turnover than climate, whereas at the global scale, climate variables are supposed to have a greater impact [101].

We have found that the direct influence of climatic factors on the turnover of bird communities decreased over time (Figure 5). Also, temperature fluctuations are most strongly reflected in the dynamics of communities in the most anthropogenically transformed habitats (rural areas, agricultural land), and in the most humid habitats (reed beds, salt marshes). Temperature is considered a key climatic factor in wetland ecosystems [45,101–103] This is consistent with our findings: for the bird communities of salt marshes, the temperature is a decisive climatic factor. This may be due to the fact that higher temperatures lead to seasonal drying, which results in increased nest mortality, owing to their greater accessibility to predators and humans, redistributing a large proportion of nesting birds both in a body of water and within the region, and mass non-nesting in unfavorable seasons [42]. Therefore, the relationships between community turnover and environmental factors have a more complex pattern, as time is the most influential factor. Climate change includes not only changes in average temperature and precipitation, but also their impact on land cover [104–106], which were apparent in the area studied.

Consequently, the temporal turnover of avian communities reflects more complex climate dynamics, such as the cumulative effects [107] of changes in climate and land use.

5. Conclusions

In all biotopes, except forest shelterbelts and meadows, there was a sharp decrease in the abundance of avian communities during the research period. For over 30 years, the temporal turnover of bird metacommunities in the diverse habitats of the Molochna River valley has been influenced by temperature and precipitation changes and has shown a monotonous, declining trend during the entire research period, similar to what has been reported from much of Central and Western Europe, as well as North America. The biotope type has an influence on species richness as well as its trend, especially in forest shelterbelts, rural areas, and reed beds, where the number of species has changed significantly over time, compared to the general trend. Although this situation is unfavorable in general, of greatest concern is the decline in bird populations in the salt marsh biotopes, where many of the bird species are protected, and whose conservation status is of concern, such as *Charadrius alexandrinus*, *Himantopus*, and *Vanellus vanellus*. The relative abundance of species in bird communities was most altered when compared to the initial period (1988) in forest shelterbelts, reedbeds, and meadows, which indicates a restructuring of the community structure in these biotopes. The cause of the monotonic temporal trend is probably the anthropogenic transformation of habitats, owing to the constant intensification of agriculture throughout the period studied. The influence of climatic factors (annual temperature and precipitation) on the turnover of bird communities decreased over time. Temperature changes were the second most important factor for bird communities in agricultural land, reed beds, rural areas, and salt marshes. In other habitats, changes in precipitation intensity were the second most important factor. This confirms the negative influence of agricultural intensification on bird abundance in Ukrainian farmland, and this effect is likely to increase over time.

Supplementary Materials: The following supporting information can be downloaded at: https://www.mdpi.com/article/10.3390/d14121028/s1, Figure S1: Autocorrelation functions of time series of the number of bird community species. Figure S2: Autocorrelation functions of time series of the abundance of bird community species.

Author Contributions: Conceptualization, A.Z., O.Z. and J.-C.S.; methodology, O.Z.; software, O.Z.; validation, A.Z., J.-C.S. and T.F.; formal analysis, O.Z.; investigation, A.Z.; data curation, O.Z.; writing—original draft preparation—A.Z.; writing—review and editing—J.-C.S.; visualization, O.Z.; supervision, J.-C.S.; project administration, T.F.; funding acquisition, J.-C.S. All authors have read and agreed to the published version of the manuscript.

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Conflicts of Interest: The authors declare no conflict of interest.

Appendix A

Table A1. Length of sampled transects and surveyed area.

Biotope	Length, m	Area, km^2
Agricultural lands	6000	4.95
Forest plantation	2000	0.58
Forest shelterbelts	7000	0.90
Meadows	4000	2.67
Reed beds	4000	2.30
Rural areas	6000	6.87
Salt marsh	3000	2.56
Steppe	4000	2.06
Total	36,000	23

Table A2. Taxonomic diversity and presence/absence (+/−) of the aviafauna species (Class Aves).

Taxons	Biotope *							
	1	2	3	4	5	6	7	8
Parvclass Galloanserae								
Ordo Anseriformes								
Family Anatidae								
Anas clypeata Linnaeus, 1758	–	–	–	+	–	–	–	–
Anas platyrhynchos Linnaeus, 1758	–	–	–	+	–	–	–	–
Anas querquedula Linnaeus, 1758	–	–	–	+	–	–	–	–
Anas strepera Linnaeus, 1758	–	–	–	+	–	–	–	–
Anser anser (Linnaeus, 1758)	–	–	–	+	–	–	–	–
Aythya ferina (Linnaeus, 1758)	–	–	–	+	–	–	–	–
Cygnus olor (Gmelin, 1803)	–	–	–	+	–	–	–	–
Ordo Galliformes								
Family Phasianidae								
Coturnix coturnix (Linnaeus, 1758)	+	–	+	–	–	–	–	–
Perdix perdix (Linnaeus, 1758)	–	+	–	–	–	–	+	–
Phasianus colchicus Linnaeus, 1758	+	+	–	–	+	–	–	+
Parvclass Passerae								
Ordo Apodiformes								
Family Apodidae								
Apus apus (Linnaeus, 1758)	–	–	–	–	+	–	–	–
Charadriiformes								
Family Scolopacidae								
Tringa totanus (Linnaeus, 1758)	–	–	+	–	–	–	–	–
Family Charadriidae								
Charadrius alexandrinus Linnaeus, 1758	–	–	–	–	–	+	–	–
Vanellus vanellus (Linnaeus, 1758)	–	–	–	–	–	+	–	–
Family Recurvirostridae								
Himantopus himantopus (Linnaeus, 1758)	–	–	–	–	–	+	–	–
Family Scolopacidae								
Limosa limosa (Linnaeus, 1758)	–	–	+	–	–	–	–	–
Ordo Ciconiiformes								
Family Podicipitidae								
Podiceps cristatus (Linnaeus, 1758)	–	–	–	+	–	–	–	–
Podiceps grisegena (Boddaert, 1783)	–	–	–	+	–	–	–	–
Tachybaptus ruficollis (Pallas, 1764)	–	–	–	+	–	–	–	–
Family Ardeidae								
Ardea cinerea Linnaeus, 1758	–	–	–	+	–	–	–	–
Ardea purpurea Linnaeus, 1766	–	–	–	+	–	–	–	–
Ardeola ralloides (Scopoli, 1769)	–	–	–	+	–	–	–	–
Botaurus stellaris (Linnaeus, 1758)	–	–	–	+	–	–	–	–

Table A2. *Cont.*

Taxons	Biotope *							
	1	**2**	**3**	**4**	**5**	**6**	**7**	**8**
Casmerodius albus (Linnaeus, 1758)	–	–	–	+	–	–	–	–
Egretta garzetta (Linnaeus, 1758)	–	–	–	+	–	–	–	–
Ixobrychus minutus (Linnaeus, 1766)	–	–	–	+	–	–	–	–
Nycticorax nycticorax (Linnaeus, 1758)	–	–	–	+	–	–	–	–
Family Ciconiidae								
Ciconia ciconia (Linnaeus, 1758)	–	–	–	–	+	–	–	–
Ordo Columbiformes								
Family Columbidae								
Columba oenas Linnaeus, 1758	–	+	–	–	–	–	–	–
Streptopelia decaocto (Frivaldszky, 1838)	–	–	–	–	+	–	–	–
Streptopelia turtur (Linnaeus, 1758)	–	+	–	–	–	–	–	+
Ordo Cuculiformes								
Family Cuculidae								
Cuculus canorus Linnaeus, 1758	–	+	–	+	–	–	–	–
Ordo Falconiformes								
Family Accipitridae								
Circus aeruginosus (Linnaeus, 1758)	–	–	–	+	–	–	–	–
Circus cyaneus (Linnaeus, 1758)	–	–	+	–	–	–	–	–
Family Falconidae								
Falco tinnunculus Linnaeus, 1758	–	+	–	–	–	–	–	–
Falco vespertinus Linnaeus, 1766	–	+	–	–	–	–	–	–
Family Motacillidae								
Motacilla alba Linnaeus, 1758	–	–	–	–	+	–	–	–
Motacilla citreola Pallas, 1776	–	–	+	–	–	–	–	–
Motacilla feldegg Michahelles, 1830	–	–	+	–	–	–	–	–
Motacilla flava Linnaeus, 1758	–	–	+	–	–	–	–	–
Ordo Gruiformes								
Family Rallidae								
Crex crex (Linnaeus, 1758)	–	–	+	–	–	–	–	–
Fulica atra Linnaeus, 1758	–	–	–	+	–	–	–	–
Gallinula chloropus (Linnaeus, 1758)	–	–	–	+	–	–	–	–
Porzana parva (Scopoli, 1769)	–	–	–	+	–	–	–	–
Rallus aquaticus Linnaeus, 1758	–	–	–	+	–	–	–	–
Ordo Passeriformes								
Family Saxicolidae								
Luscinia megarhynchos C. L. Brehm, 1831	–	+	–	–	–	–	–	–
Luscinia svecica (Linnaeus, 1758)	–	–	–	+	–	–	–	–
Phoenicurus ochruros (S. G. Gmelin, 1774)	–	–	–	–	+	–	–	–
Saxicola rubetra (Linnaeus, 1758)	–	–	+	–	–	–	+	–
Saxicola torquatus (Linnaeus, 1766)	–	–	+	–	–	–	–	–
Family Alaudidae								
Alauda arvensis Linnaeus, 1758	+	–	+	–	–	–	+	–
Family Corvidae								
Corvus corax Linnaeus, 1758	–	+	–	–	–	–	–	–
Corvus cornix Linnaeus, 1758	–	+	–	–	–	–	–	–
Garrulus glandarius (Linnaeus, 1758)	–	–	+	–	–	–	–	–
Pica pica (Linnaeus, 1758)	–	+	–	+	–	–	–	+
Family Emberizidae								
Emberiza calandra Linnaeus, 1758	+	–	+	–	–	–	+	–
Emberiza hortulana Linnaeus, 1758	+	+	–	–	–	–	–	–
Emberiza schoeniclus (Linnaeus, 1758)	–	–	–	+	–	–	–	–
Family Fringillidae								
Carduelis cannabina (Linnaeus, 1758)	–	–	+	–	+	–	–	–
Chloris chloris (Linnaeus, 1758)	–	–	–	–	+	–	–	+
Family Hirundinidae								
Hirundo rustica Linnaeus, 1758	–	–	–	–	+	–	–	–
Family Laniidae								

Table A2. *Cont.*

Taxons	Biotope *							
	1	2	3	4	5	6	7	8
Lanius collurio Linnaeus, 1758	–	+	–	–	–	–	–	–
Lanius minor Gmelin, 1788	–	+	–	–	+	–	–	+
Family Motacillidae								
Anthus campestris (Linnaeus, 1758)	–	–	–	–	–	–	+	–
Family Oriolidae								
Oriolus oriolus (Linnaeus, 1758)	–	+	–	–	+	–	–	–
Family Paridae								
Parus major Linnaeus, 1758	–	+	–	–	+	–	–	–
Family Passeridae								
Passer domesticus (Linnaeus, 1758)	–	–	–	–	+	–	–	–
Passer montanus (Linnaeus, 1758)	–	–	–	–	+	–	–	–
Family Sturnidae								
Sturnus vulgaris Linnaeus, 1758	–	–	–	–	+	–	–	–
Family Sylviidae								
Acrocephalus agricola (Jerdon, 1845)	–	–	–	+	–	–	–	–
Acrocephalus arundinaceus (Linnaeus, 1758)	–	–	–	+	–	–	–	–
Acrocephalus schoenobaenus (Linnaeus, 1758)	–	–	–	+	–	–	–	–
Acrocephalus scirpaceus (Hermann, 1804)	–	–	–	+	–	–	–	–
Locustella luscinioides (Savi, 1824)	–	–	–	+	–	–	–	–
Panurus biarmicus (Linnaeus, 1758)	–	–	–	+	–	–	–	–
Sylvia communis Latham, 1787	–	+	–	–	–	–	–	–
Ordo Picimorphes								
Family Picidae								
Dendrocopos syriacus (Hemprich & Ehrenberg, 1833)	–	+	–	–	+	–	–	–
Ordo Strigiformes								
Family Strigidae								
Asio otus (Linnaeus, 1758)	–	+	–	–	–	–	–	+
Athene noctua (Scopoli, 1769)	–	–	–	–	+	–	–	–
Otus scops (Linnaeus, 1758)	–	+	–	–	–	–	–	–
Ordo Upupiformes								
Family Upupidae								
Upupa epops Linnaeus, 1758	–	+	–	–	+	–	–	–

* 1—Agricultural lands; 2—Forest shelterbelts; 3—Meadows; 4—Reed beds; 5—Rural areas; 6—Salt marshes; 7—Steppe; 8—Forest plantations.

Table A3. Species richness of bird communities over the period (1988–2018, N = 31).

Biotope	Total for All Periods	Mean ± St.Error	Minimum	Maximum	Median
Agricultural lands	5	4.55 ± 0.09	4	5	5
Forest shelterbelts	21	9.77 ± 0.37	7	17	9
Meadows	14	8.77 ± 0.14	7	11	9
Reed beds	33	30.32 ± 0.20	28	32	31
Rural areas	18	15.71 ± 0.19	14	18	16
Solonchaks (salt marsh)	3	3.00 ± 0.00	3	3	3
Steppe	5	4.19 ± 0.13	3	5	4
Artificial forests (plantations)	6	5.84 ± 0.07	4	6	6
The entire landscape system	81	80.45 ± 3.88	39	79	59

Table A4. Trends in the abundance of bird species (1988–2018). Trends in bird populations were classified, according to Pannekoek and van Strien [52], into one of the following categories depending on the overall slope and its 95% confidence interval: strong growth/steep decline is classified if growth/decline is significantly greater than 5% per year; moderate growth/decline is classified if growth/decline is significant, but not more than 5% per year; stable is classified if there is no significant increase or decrease and there is confidence that trends do not exceed 5% per year; and uncertain is classified if there is no significant growth or decline, but there is no certainty that trends are less than 5% per year.

Species	Slope ± St.Error	*p*-Value	Trend Interpretation (Meaning)	World Population Trend *
Acrocephalus agricola (Jerdon, 1845)	−0.014 ± 0.005	0.018	Moderate decrease ($p < 0.05$)	Decreasing
Acrocephalus arundinaceus (Linnaeus, 1758)	−0.016 ± 0.004	<0.001	Moderate decrease ($p < 0.01$)	Decreasing
Acrocephalus schoenobaenus (Linnaeus, 1758)	0.005 ± 0.008	0.510	Stable	Stable
Acrocephalus scirpaceus (Hermann, 1804)	0.016 ± 0.006	0.022	Moderate increase ($p < 0.05$)	Stable
Alauda arvensis Linnaeus, 1758	−0.001 ± 0.003	0.635	Stable	Decreasing
Anas clypeata Linnaeus, 1758	0.000 ± 0.006	0.965	Stable	Decreasing
Anas platyrhynchos Linnaeus, 1758	−0.046 ± 0.004	<0.001	Moderate decrease	Increasing
Anas querquedula Linnaeus, 1758	−0.018 ± 0.006	0.005	Moderate decrease ($p < 0.05$)	Decreasing
Anas strepera Linnaeus, 1758	−0.003 ± 0.007	0.678	Stable	Increasing
Anser anser (Linnaeus, 1758)	−0.045 ± 0.006	<0.001	Moderate decrease	Increasing
Anthus campestris (Linnaeus, 1758)	−0.005 ± 0.006	0.407	Stable	Stable
Apus apus (Linnaeus, 1758)	−0.011 ± 0.006	0.090	Stable	Stable
Ardea cinerea Linnaeus, 1758	−0.039 ± 0.005	<0.001	Moderate decrease ($p < 0.01$)	Unknown
Ardea purpurea Linnaeus, 1766	−0.055 ± 0.005	<0.001	Moderate decrease	Decreasing
Ardeola ralloides (Scopoli, 1769)	−0.048 ± 0.006	<0.001	Moderate decrease ($p < 0.01$)	Unknown
Asio otus (Linnaeus, 1758)	−0.006 ± 0.006	0.339	Stable	Decreasing
Athene noctua (Scopoli, 1769)	−0.007 ± 0.006	0.262	Stable	Stable
Aythya ferina (Linnaeus, 1758)	−0.035 ± 0.005	<0.001	Moderate decrease ($p < 0.01$)	Decreasing
Botaurus stellaris (Linnaeus, 1758)	−0.001 ± 0.005	0.836	Stable	Decreasing
Carduelis cannabina (Linnaeus, 1758)	−0.004 ± 0.005	0.428	Stable	Decreasing
Casmerodius albus (Linnaeus, 1758)	−0.014 ± 0.006	0.026	Moderate decrease ($p < 0.05$)	Unknown
Charadrius alexandrinus Linnaeus, 1758	0.001 ± 0.006	0.843	Stable	Decreasing
Chloris chloris (Linnaeus, 1758)	−0.005 ± 0.005	0.352	Stable	Stable
Ciconia ciconia (Linnaeus, 1758)	0.000 ± 0.005	0.943	Stable	Increasing
Circus aeruginosus (Linnaeus, 1758)	−0.004 ± 0.005	0.456	Stable	Stable
Circus cyaneus (Linnaeus, 1758)	−0.002 ± 0.007	0.809	Stable	Decreasing
Columba oenas Linnaeus, 1758	−0.002 ± 0.007	0.809	Stable	Decreasing
Corvus corax Linnaeus, 1758	−0.006 ± 0.006	0.321	Stable	Increasing
Corvus cornix Linnaeus, 1758	−0.003 ± 0.006	0.564	Stable	Stable
Coturnix coturnix (Linnaeus, 1758)	0.005 ± 0.006	0.404	Stable	Decreasing
Crex crex (Linnaeus, 1758)	−0.011 ± 0.006	0.093	Stable	Stable
Cuculus canorus Linnaeus, 1758	0.008 ± 0.006	0.181	Stable	Decreasing
Cygnus olor (Gmelin, 1803)	−0.008 ± 0.006	0.161	Stable	Increasing
Dendrocopos syriacus (Hemprich & Ehrenberg, 1833)	−0.005 ± 0.006	0.420	Stable	Stable
Egretta garzetta (Linnaeus, 1758)	−0.040 ± 0.005	<0.001	Moderate decrease ($p < 0.01$)	Increasing
Emberiza calandra Linnaeus, 1758	−0.006 ± 0.004	0.172	Stable	Decreasing
Emberiza hortulana Linnaeus, 1758	0.004 ± 0.006	0.507	Stable	Decreasing
Emberiza schoeniclus (Linnaeus, 1758)	0.038 ± 0.008	<0.001	Moderate increase ($p < 0.01$)	Decreasing
Falco tinnunculus Linnaeus, 1758	−0.002 ± 0.006	0.713	Stable	Decreasing
Falco vespertinus Linnaeus, 1766	0.001 ± 0.007	0.843	Stable	Decreasing
Fulica atra Linnaeus, 1758	−0.034 ± 0.001	<0.001	Moderate decrease	Increasing
Gallinula chloropus (Linnaeus, 1758)	−0.034 ± 0.001	<0.001	Moderate decrease ($p < 0.01$)	Stable
Glareola pratincola (Linnaeus, 1766)	−0.041 ± 0.002	<0.001	Moderate decrease ($p < 0.01$)	Decreasing
Garrulus glandarius (Linnaeus, 1758)	−0.010 ± 0.005	0.052	Stable	Stable
Himantopus himantopus (Linnaeus, 1758)	−0.010 ± 0.005	0.052	Stable	Increasing
Hirundo rustica Linnaeus, 1758	−0.035 ± 0.003	<0.001	Moderate decrease ($p < 0.01$)	Decreasing
Ixobrychus minutus (Linnaeus, 1766)	0.002 ± 0.007	0.770	Stable	Decreasing
Lanius collurio Linnaeus, 1758	−0.008 ± 0.005	0.123	Stable	Stable
Lanius minor Gmelin, 1788	−0.001 ± 0.007	0.930	Stable	Decreasing
Limosa limosa (Linnaeus, 1758)	−0.007 ± 0.004	0.125	Stable	Decreasing
Locustella luscinioides (Savi, 1824)	0.013 ± 0.006	0.047	Moderate increase ($p < 0.05$)	Stable
Luscinia megarhynchos C. L. Brehm, 1831	0.024 ± 0.006	0.001	Moderate increase ($p < 0.01$)	Stable
Luscinia svecica (Linnaeus, 1758)	0.002 ± 0.006	0.728	Stable	Stable
Motacilla alba Linnaeus, 1758	0.024 ± 0.006	0.001	Moderate increase ($p < 0.01$)	Stable
Motacilla citreola Pallas, 1776	0.052 ± 0.008	<0.001	Moderate increase ($p < 0.01$)	Increasing
Motacilla feldegg Michahelles, 1830	0.034 ± 0.005	<0.001	Moderate increase ($p < 0.01$)	Decreasing
Motacilla flava Linnaeus, 1758	−0.119 ± 0.004	<0.001	Strong decrease ($p < 0.01$)	Decreasing
Nycticorax nycticorax (Linnaeus, 1758)	−0.001 ± 0.005	0.780	Stable	Decreasing
Oriolus oriolus (Linnaeus, 1758)	0.001 ± 0.007	0.924	Stable	Stable
Otus scops (Linnaeus, 1758)	−0.003 ± 0.008	0.723	Stable	Decreasing
Panurus biarmicus (Linnaeus, 1758)	−0.006 ± 0.006	0.285	Stable	Unknown
Parus major Linnaeus, 1758	0.001 ± 0.003	0.718	Stable	Increasing
Passer domesticus (Linnaeus, 1758)	0.005 ± 0.003	0.130	Stable	Decreasing
Passer montanus (Linnaeus, 1758)	−0.006 ± 0.005	0.264	Stable	Decreasing
Perdix perdix (Linnaeus, 1758)	−0.003 ± 0.005	0.506	Stable	Decreasing
Phasianus colchicus Linnaeus, 1758	0.000 ± 0.007	1.000	Stable	Decreasing
Phoenicurus ochruros (S. G. Gmelin, 1774)	−0.017 ± 0.004	<0.001	Moderate decrease ($p < 0.01$)	Increasing
Pica pica (Linnaeus, 1758)	−0.054 ± 0.005	<0.001	Moderate decrease ($p < 0.01$)	Stable

Table A4. *Cont.*

Species	Slope ± St.Error	p-Value	Trend Interpretation (Meaning)	World Population Trend *
Podiceps cristatus (Linnaeus, 1758)	−0.039 ± 0.005	<0.001	Moderate decrease (p < 0.01)	Unknown
Podiceps grisegena (Boddaert, 1783)	−0.043 ± 0.003	<0.001	Moderate decrease (p < 0.01)	Decreasing
Porzana parva (Scopoli, 1769)	−0.070 ± 0.005	<0.001	Strong decrease (p < 0.01)	Stable
Rallus aquaticus Linnaeus, 1758	0.003 ± 0.005	0.648	Stable	Decreasing
Saxicola rubetra (Linnaeus, 1758)	0.001 ± 0.007	0.856	Stable	Decreasing
Saxicola torquatus (Linnaeus, 1766)	−0.005 ± 0.003	0.160	Stable	Stable
Streptopelia decaocto (Frivaldszky, 1838)	0.003 ± 0.006	0.657	Stable	Increasing
Streptopelia turtur (Linnaeus, 1758)	−0.078 ± 0.003	<0.001	Strong decrease (p < 0.01)	Decreasing
Sturnus vulgaris Linnaeus, 1758	0.032 ± 0.006	<0.001	Moderate increase (p < 0.01)	Decreasing
Sylvia communis Latham, 1787	0.002 ± 0.007	0.836	Stable	Increasing
Tachybaptus ruficollis (Pallas, 1764)	−0.038 ± 0.005	<0.001	Moderate decrease (p < 0.01)	Decreasing
Tringa totanus (Linnaeus, 1758)	−0.001 ± 0.006	0.842	Stable	Unknown
Upupa epops Linnaeus, 1758	−0.027 ± 0.003	<0.001	Moderate decrease (p < 0.01)	Decreasing
Vanellus vanellus (Linnaeus, 1758)	−0.014 ± 0.005	0.018	Moderate decrease (p < 0.05)	Decreasing

* Data obtained from the Birdlife International Data Zone [53].

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