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## **Insurance for the future? Potential avian community resilience in cities across Europe**

Morelli, Federico; Benedetti, Yanina; Ibáñez-Álamo, Juan Diego; Tryjanowski, Piotr; Jokimaki, Jukka; Kaisanlahti-Jokimaki, Marja-Liisa; Pérez-Contreras, Tomás; Sprau, Philipp; Jukka, Suhonen; Yosef, Reuven; Diaz, Mario; Møller, Anders Pape

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1 **Insurance for the future? Potential avian community resilience in cities across**  
2 **Europe**

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7 4 *Running head: Bird community potential resilience in European cities*

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29 **Abstract**

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Urbanization is affecting avian biodiversity across the planet, and potentially increasing species vulnerability to climate. Identifying the resilience of urban bird communities to climate change is critical for making conservation decisions. This study explores the pattern in bird communities across nine European cities and examines the projected impact of climate change in order to detect communities facing a higher risk of functional change in the future.

First, generalized linear mixed models were used to explore the potential resilience of urban bird communities in nine European cities, and the effects of land cover, latitude, abundance of potential domesticated predators (dogs and cats), and bird species richness in each trophic guild. Bird community resilience was represented by an index of functional evenness, because it indicates relatively uniform functional space within the species assemblages. Second, bird community resilience in each city was compared with projected changes in temperature and precipitation for the year 2070 to explore potential future threats to conservation.

The results showed that community resilience was not significantly associated with land use or predator abundance. The number of granivorous and granivorous-insectivorous species increases the potential resilience of the community, while the numbers of insectivores, carnivores and omnivores are negatively correlated with resilience. Of the nine cities, Madrid and Toledo (Spain) are projected to experience the largest change in temperature and precipitation, although their bird communities are characterized by relative high resilience.

In contrast, Rovaniemi (Finland) is projected to experience the second highest increase in temperature and the bird community is characterized by low resilience. These findings indicate the importance of future research on the combined effect of urbanization and climate change on urban biodiversity.

**Keywords:** biotic homogenization; bird diversity; community resilience; conservation; functional evenness; urbanization

## 59 INTRODUCTION

### 60 Global urbanization and ecosystem transformation

61 We are living in a new self-induced geological era, called "the Anthropocene",  
62 characterized by the increasing pressure of urban development on ecosystem dynamics  
63 across the entire planet (Crutzen and Stoermer 2000; Lewis and Maslin 2015). The  
64 change in land-use produced by the urbanization process is one of the major drivers of  
65 environmental modification, with strong and deep impacts on both climate and global  
66 biodiversity (Foley et al. 2005; Grimm et al. 2008; Aronson et al. 2014). Expanded  
67 urbanization leads to habitat fragmentation and degradation (Spellerberg 1998;  
68 Schmiegelow and Mönkkönen 2002; Sklenicka 2016), a process which negatively impacts  
69 biodiversity at different levels of organization (Crooks et al. 2004; Wilson et al. 2016).  
70 Thus, the mitigation of the loss of biodiversity is partially dependent on our understanding  
71 of how urbanization structures biological communities and the subsequent development of  
72 wildlife management strategies that incorporate urban ecosystems (Miller and Hobbs  
73 2002).

74 The effects of urbanization on biodiversity are complex and mainly negative (McKinney  
75 2002; Grimm et al. 2008; Secretariat of the Convention on Biological Diversity 2012;  
76 Aronson et al. 2014; Ibáñez-Álamo et al. 2016; Morelli et al. 2016). Especially when trying  
77 to assess the effects of urbanization on overall biodiversity, emphasizing that approaches  
78 focusing only on species richness are limited by their failure to take the ecological role of  
79 species on communities into account (Safi et al. 2013). In fact, estimates of functional  
80 diversity often measure variation in ecosystem functioning better than taxonomic diversity  
81 measures (Petchey et al. 2004). Among the most recognized effects of urbanization on  
82 species assemblage composition stands the 'biotic homogenization' of global communities  
83 (Clergeau et al. 2006; McKinney 2006). Biotic homogenization consists of the range  
84 expansion of cosmopolitan or generalist species and simultaneously the range contraction  
85 of regional and endemic species, a process which is particularly intense in dense urban  
86 settlements (Garcillán et al. 2014). Biotic homogenization was confirmed in several taxa  
87 (McKinney and Lockwood 1999; La Sorte et al. 2007; Knop 2016), but has so far mostly  
88 been studied in urban bird communities (Crooks et al. 2004; Devictor et al. 2007; Godet et  
89 al. 2015; Ibáñez-Álamo et al. 2016; Vázquez-Reyes et al. 2017). Urbanized areas  
90 (farmlands, villages and cities) can provide additional habitat or food sources, attracting

91 specific bird species (Evans et al. 2009a; Evans et al. 2009b; Tryjanowski et al. 2015;  
92 Reynolds et al. 2017). However, urban bird communities are more homogeneous and  
93 often characterized by the absence of specialists when compared with communities from  
94 natural habitats (Jokimäki and Kaisanlahti-Jokimäki 2003; Shochat et al. 2010; Ferenc et  
95 al. 2014). These differences lead to a gradual decline in the overall functional diversity of  
96 species assemblages (Pauw and Louw 2012; Kang et al. 2015; Schütz and Schulze 2015),  
97 potentially also reducing the capacity of such communities to resist drastic changes in land  
98 use or weather. In fact, it has already been demonstrated that urban development patterns  
99 and green areas within cities affect ecosystem dynamics, modifying their abilities to cope  
100 with disturbance and modifying their ecological resilience (Alberti and Marzluff 2004).

### 101 **Functional surrogates of potential community resilience**

102 Ecological resilience is associated with the capacity of ecological systems to resist  
103 invasions, climate or land use changes (Haegeman et al. 2016). This term was introduced  
104 in ecology in the 1970's by Holling (1973) and has been more recently defined as "the  
105 capacity of a given system to change in order to maintain the same identity" (Folke et al.  
106 2010). Thus, resilience is associated with a sort of 'elasticity' of the system. This property  
107 is associated with a relative functional redundancy of components of that system  
108 (Haegeman et al. 2016). Redundant or pseudo-redundant species can be described as  
109 species sharing many functional traits, belonging to the same guild or having a similar role  
110 in the ecosystem (Gitay et al. 1996). Theoretically, the loss or gain of such species should  
111 not strongly affect the overall ecosystem functions (Loreau 2004).

112 Some studies have suggested that indices of functional diversity such as functional  
113 evenness may be useful surrogates for the resilience of communities (Mason et al. 2005;  
114 Villéger et al. 2008; Lee and Martin 2017), even if in recent years some potential  
115 drawbacks were raised (Ricotta et al. 2014; Legras and Gaertner 2018). These indices  
116 could be used to assess the level of utilization of available resources in a given space, by  
117 the species inhabiting it (Mason et al. 2005; Mouchet et al. 2010). In communities with high  
118 functional evenness the resources would be more efficiently used due to a more uniform  
119 distribution of the abundance of species throughout functional space defined by the  
120 species traits (Lee and Martin 2017) (see a schematic exemplification in the Fig. S1,  
121 ESM). In contrast, in communities with low functional evenness available resources could  
122 be underexploited, making the community more susceptible to alterations, as for instance

123 biological invasions (Elton 1958; Shea and Chesson 2002). More resilient ecological  
124 systems should be able to absorb larger shocks, alleviating the effects and reducing the  
125 significant modifications, offering a sort of insurance for the future, facing climate change  
126 scenarios (Folke et al. 2002). In this context, the characteristics of the species composing  
127 each community, as for example the type of diet, acquires significant importance to  
128 determine the overall resilience capacities of such a community. We can expect that  
129 communities composed of many species which are similar in terms of feeding traits (e.g.  
130 several omnivorous species), could be better prepared to face eventual change in land use  
131 or climate, because such communities are able to respond better by adapting to  
132 fluctuations in food resources. On the other hand, communities composed of several  
133 species characterized by a narrow diet (e.g. exclusively frugivorous) could be associated  
134 with a higher extinction risk (Terborgh and Winter 1980).

### 135 **Climate change scenarios and the challenge for conservation of biodiversity**

136 The effects of climate change could be an additional pressure on ecosystems, interacting  
137 with land use change and fragmentation (Eglington and Pearce-Higgins 2012), for  
138 example by promoting biological invasions (Opdam and Wascher 2004; Bellard et al.  
139 2013). Projections suggest that in the next decades human-induced climate change is  
140 expected to continue, and probably accelerate significantly in association with the global  
141 emissions of heat-trapping gases (some sources for projections: for America,  
142 <https://nca2014.globalchange.gov/>; for worldwide, <https://gisclimatechange.ucar.edu/>)  
143 (Hulme et al. 1999; NCAR community 2012; Brown and Caldeira 2017).

144 There exists a vast amount of scientific literature focusing on the main effects of climate in  
145 bird populations, mainly produced by temperature and precipitation (Huntley et al. 2008;  
146 Askeyev et al. 2018; Trautmann 2018). The main effects of climate change on birds could  
147 be associated with changes in distribution range of avian species, phenology and breeding  
148 success, but also genetics and overall population sizes (Trautmann 2018). Briefly, the  
149 main effects of temperature can be associated with variation in the body mass (Andrew et  
150 al. 2018), avian timing of reproduction (Visser et al. 2009) and reproductive performance of  
151 avian species (Conrey et al. 2016). Some studies have shown that extreme temperatures  
152 (e.g., heat waves or drastic fall on temperatures) during the early breeding season can  
153 negatively affect nesting success of grassland birds (Conrey et al. 2016; Zuckerberg et al.  
154 2018). On the other hand, the main effects of precipitation on birds are associated with the



155 direct survival of adults and nesting and hatching success. For example, it was  
156 demonstrated that large amounts of precipitations during the year preceding a breeding  
157 season, can increase the nesting success of some birds (Zuckerberg et al. 2018).  
158 However, excessive precipitations during the breeding season can reduce the nesting  
159 success (Zuckerberg et al. 2018). Additionally, intense rain events are expected to reduce  
160 food availability (e.g. insect resources) or directly foraging efficiency of birds (Siikämaki  
161 1996). Furthermore, many of the effects of weather variables on birds are combined and  
162 associated with different ecological levels (species, populations, communities) (Møller et  
163 al. 2010; Skagen and Adams 2012; Stephens et al. 2016; Trautmann 2018).

164 The use of simulated scenarios to forecast the environmental implications of potential  
165 climate or land use changes is a recognized tool for the development of ecological policies  
166 (Princé et al. 2013). Climate change causes different ecological fingerprints (Parmesan  
167 2006): Bird populations will be affected in many ways, causing ecological adaptations in  
168 response to such changes (Butler and Taylor 2005; Barbet-Massin and Jetz 2015; Howard  
169 et al. 2015). Among the ecological responses, a potential geographic shift and spatial re-  
170 distribution of species assemblages is a critical focus for ecosystem functioning, with deep  
171 implications for conservation (Barbet-Massin and Jetz 2015; Princé and Zuckerberg 2015).  
172 Induced changes could also be species-specific, altering the composition of bird  
173 communities, with potential effects on overall resilience. Additionally, global climate  
174 change is expected to strongly affect even the local climate of cities worldwide (Lauwaet et  
175 al. 2015), which constitutes important core areas for conservation of many bird species  
176 (Ives et al. 2016; Jokimäki et al. 2018). In fact, recent studies have highlighted the  
177 important role of cities for global or regional biodiversity conservation, by providing  
178 opportunities for settlement of species (Zerbe et al. 2003; Aronson et al. 2014; Møller and  
179 Díaz 2017). Cities can be characterized by a mix between global warming and the urban  
180 heat island effect (Oke 1973; Kim 1992). Early identification of specific conservation  
181 targets where climate can accelerate the already negative impact of urbanization should  
182 be a priority in an increasingly urbanizing planet.

183 The aims of this study were (a) to explore the patterns and geographic variation in  
184 resilience of urban bird communities in European cities, (b) to identify urban bird  
185 communities more likely to be threatened by climate change according to future  
186 projections in temperature and rainfall, and (c) to explore the characteristics of such bird



187 assemblages in terms of species traits (e.g. diet), in order to increase our understanding of  
188 potential conservation concerns, and, therefore, attempts to mitigate the negative effects  
189 of such changes.

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191 **METHODS**

192 **Study area, environment and collection of bird data**

193 Fieldwork was performed in nine cities, located along a large latitudinal gradient, in six  
194 European countries (Fig. 1). This approach, involving different urbanized areas, is  
195 particularly suitable for investigating general patterns (Ibáñez-Álamo et al. 2016). In this  
196 study were included only urban areas, with multi-story buildings, single family houses,  
197 roads and parks. Our classification of environments as urban (percentage of built-up area  
198 >50, building density >10/ha and residential human density >10/ha) followed the  
199 description made by Marzluff *et al.* (2001). It has been used in many other studies focusing  
200 on urban avian ecology (Clergeau et al. 2006; Loss et al. 2009; Møller et al. 2015; Morelli  
201 et al. 2016). We collected data on vegetation cover and land use composition within a  
202 distance of 50 m from each survey point (Díaz et al. 2013). Land use / cover categories  
203 were classified in 6 types: building (which includes residential building, built with  
204 infrastructure and processing areas and roads), trees (isolated trees, tree lines and  
205 patches), bushes (which includes plants from gardens), grass, bare soil, and water.

206 Data on bird species were collected during the 2016 breeding season. The surveys were  
207 locally adjusted to the start of the breeding season (e.g. early April in southern Spain or  
208 late May in northern Finland). Data on bird species were collected by expert ornithologists,  
209 following the standardized methodology of point counts randomly selected (Bibby et al.  
210 1992; Voříšek et al. 2010) within each city (ESM, Table S1). All survey points were visited  
211 between 06:00 and 10:00 only during favorable weather conditions (i.e., no rain or strong  
212 wind). A total of 5 minutes in early spring and 5 minutes during a second visit in late spring  
213 allowed for inclusion of both early breeders and late migrants, minimizing issues related to  
214 differences in the detectability of bird species (Kéry et al. 2005). During the point counts all  
215 individual birds seen or heard within a 50m distance from the observer were recorded, with  
216 the only exception being raptors and nocturnal species, because they are subject to a  
217 different strategy of survey. The location of each sampling point was recorded with a GPS  
218 to ensure that counts were made at the same points during the two surveys. For each  
219 sampling point, we also calculated the abundance of mammals as the mean number of  
220 dogs and cats seen within 50m during the two 5-minute point counts.

221 **Surrogate of potential resilience of bird communities**

222 The bird community in each sampling site was defined as the total number of bird species  
223 recorded during the two visits. Thus, species richness was expressed as the largest  
224 number of bird species of the combined data for the two surveys performed during the  
225 breeding season (Magurran 2004).

226 For each sampling site or bird community, we estimated two functional diversity indices  
227 which are traditionally associated with resilience of the community: Functional evenness  
228 ('FEve') (Villéger et al. 2008) and the measure 'Fever' for calculating the functional  
229 evenness of a species' assemblage (Ricotta et al. 2014). Both measures are based on a  
230 species-trait approach, which focuses on functional aspects of biodiversity (de Bello et al.  
231 2010). Both indices used in this study as surrogate for bird community resilience were  
232 calculated using the feeding and breeding avian niche traits provided in Pearman et al.  
233 (2014). The bird traits consists of 73 variables describing the niche of each bird species,  
234 including (i) body mass, (ii) food type (14 variables), (iii) behavior used for food acquisition  
235 (9 variables), (iv) substrate from which food is taken (9 variables), (v) period of day during  
236 which a species are active (3 variables), and (vi) used habitats (38 variables) (Pearman et  
237 al. 2014). All variables, except for body mass, are binary variables (scored as either 0 or 1)  
238 (see a complete description of the used bird traits in ESM, Table S2). The functional traits  
239 are associated with many different characteristics of birds including morphological,  
240 physiological, and phenological attributes of species, which are related to individual fitness  
241 via their effects on growth, reproduction and survival (Violle et al. 2007).

242 The functional evenness (FEve) indicate how regular is the degree to which the biomass of  
243 the species assemblage is distributed in niche space to allow effective utilization of the  
244 entire range of resources available (Villéger et al. 2008) and is calculated as follows:

$$\text{FEve} = \frac{\sum_{i=1}^{S-1} \min\left(\text{PEW}_{i, \frac{1}{S-1}}\right) - \frac{1}{S-1}}{1 - \frac{1}{S-1}}$$

246 where  $i$  is the species (or functional unit),  $S$  is the total species richness and PEW is the  
247 partial weighted evenness.

248 The FeveR index reflects the regularity in the distribution of abundances of species,  
 249 together with the evenness in their pairwise functional dissimilarities (Ricotta et al. 2014),  
 250 and is calculated as follows:

$$R_U = \frac{E_U - 1/N}{1 - 1/N}$$

252 where  $E_U$  is the index of Bulla (1994) and  $N$  the number of species, calculated as follows:

$$E_U = \sum_i^N \min\{\pi_i, 1/N\}$$

254 where  $\pi_i$  is the relative contribution of species  $i$  to  $U$  (average community uniqueness  $U$ ).

255 The average community uniqueness  $U$  is the expected dissimilarity between one individual  
 256 of species  $i$  chosen at random from a given community and all other  $j$ -th species in the  
 257 community, calculated as follows:

$$U = \sum_{j \neq i}^N p_i \times U_i = \sum_i^N p_i \sum_{i \neq j}^N \frac{p_j}{1 - p_i} d_{ij}$$

259 where  $d_{ij}$  is the functional dissimilarity between species  $i$  and  $j$  (with  $d_{ij} = d_{ji}$  and  $d_{ii} = 0$ ),  $p_i$   
 260 and  $p_j$  are the relative abundance of species  $i$  and  $j$  respectively, and  $U_i$  represent the  
 261 expected dissimilarity between species  $i$  and all other species in the community (Ricotta et  
 262 al. 2014).

263 The functional diversity indices used in this study were calculated using the 'FD' package  
 264 for FEve (Laliberté et al. 2015) and the function provided in Ricotta *et al.* (2014) for FeveR.

265 Additionally, we calculated species richness for each trophic guild: Granivorous (diet  
 266 containing primarily seeds and grain), insectivorous (diet containing primarily insects and  
 267 other invertebrates), granivorous-insectivorous (diet containing primarily seeds, grain,  
 268 insects and other invertebrates), and carnivorous and omnivorous birds. The trophic guilds  
 269 were defined by crossing information on bird species from published sources (Cramp and  
 270 Perrins 1994; Storchová and Hořák 2018) and the 14 food types described in Pearman et  
 271 al. (2014). With the species richness per each trophic guild we estimated also the diet  
 272 diversity in each community by applying the Shannon-diversity index (Shannon 1948).

273 **Climate change projection data: differences in temperature and precipitation**

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274 Data on climate change projections were obtained from NCAR GIS Program, through  
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275 Climate Change Scenarios, version 2.0, 2012. URL: <https://gisclimatechange.ucar.edu/>.  
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276 Data Access Date: 25/05/2018 (NCAR community 2012). The climate change scenarios  
7  
277 have been redesigned for the Intergovernmental Panel on Climate Change (IPCC) Fifth  
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278 Assessment Report. The Community Climate System Model (CCSM) as a community-  
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279 wide effort led by the National Center for Atmospheric Research (NCAR, URL:  
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280 <https://ncar.ucar.edu/>), and it is a key component of the National Science Foundation  
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281 program on Climate Modeling, Analysis and Prediction. We used the data provided by  
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282 CCSM, because they are one of the world's leading general circulation climate models,  
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283 and a community wide effort led by NCAR. The data define the Representative  
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284 Concentration Pathways (RCPs), which provide concentrations of atmospheric  
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285 greenhouse gas (GHG) and the trajectory that is taken over time to reach those  
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286 concentrations. The spatial resolution of CCSM-3 climate change projections is  
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287 approximately 1.4 x 1.4 degrees, and represents a plausible alternative scenario for the  
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288 future, not a prediction or forecast (Moss et al. 2008). We downloaded three projected  
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289 scenarios for current and future climate change in the years comprised between 2017 and  
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290 2070: scenarios RCP 4.5, 6.0 and 8.5. In order to visualize the potential climate change  
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291 expected in Europe, we used the differences in annual means (raw CCSM data) between  
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292 2017 and the simulated data for 2070 in a) temperature (in °Celsius) and b) amount of  
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293 precipitation (in mm) in each cell provided in the CCSM data.

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294 The three different scenarios were mapped for the European continent (ESM, Fig. S2) by  
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using the ArcGIS 10.1 (ESRI 2012) tool “inverse distance weighted” (IDW). The IDW is a  
technique that interpolates a raster surface from several point values, taking into account  
the distances among points (Lu and Wong 2008). The distance among points is uniform  
across Europe and corresponds to the distance among the centroids of the cells provided  
in the CCSM data (ESM, Fig. S3). This technique (IDW) is adequate to visualize the areas  
more subject to potential changes in terms of both temperature and precipitation. We used  
the maps produced to overlap with the cities where bird community resilience was  
calculated, and then extracted the relative values of climate change expected for each city.  
For visualization of data, we set the IDW on 6 different classes from the minimum to the  
maximum values of delta temperature and precipitation, by using the natural breaks

305 (Jenks) classification. This classification identifies the best group similar values and  
306 maximizes the differences between classes (De Smith et al. 2007).

307 Finally, in order to compare the congruence/mismatch between potential resilience of  
308 urban bird communities in nine European cities and climate change scenarios, we focused  
309 on the high pathway in which radiative forcing reaches 8.5 Watts per square meter ( $Wm^{-2}$ )  
310 by 2100 (Moss et al. 2008), because the worst scenario is the most suitable to better  
311 highlight potential areas of largest changes in temperature and precipitation in the  
312 continent. The main justification for this choice is supported by a recent study suggesting  
313 that the worst predictions of climate change are probably the most accurate (Brown and  
314 Caldeira 2017).

### 315 **Statistical analyses**

316 In order to investigate the strength and direction of the association between the two  
317 indices of functional diversity (FEve and FeveR) we used the Spearman correlation  
318 coefficient (Triola 2012). Because both indices were significantly positively correlated ( $R^2 =$   
319  $0.55$ ,  $p = 4.6e-12$ ), the subsequent analyses only focused on one of these indices. We  
320 chose FEve because this variable showed a largest range if compared with FeveR in each  
321 city, guarantying a better visualization of data. A linear regression was used to explore the  
322 potential association between the indices of functional diversity and bird diet diversity in  
323 avian communities.

324 To explore differences in avian niche traits among species of different trophic guilds, the  
325 analysis of dissimilarity was used. In this procedure, a dissimilarity matrix was constructed  
326 by the “Gower” dissimilarity measure among bird species, considering the 73 avian traits  
327 described above. We used the function ‘daisy’ from the package ‘cluster’ in R (Maechler et  
328 al. 2018). The average dissimilarity with all the 279 species in the pool was calculated for  
329 each bird species. The Kruskal-Wallis and post hoc Kruskal-Dunn were used to check for  
330 significant differences in species dissimilarities among the five trophic guilds. The post hoc  
331 contrasts were performed using the package ‘PMCMR’ in R (Pohlert 2014).

332 We used Mantel tests to check for spatial autocorrelation of data (SAC) (Mantel 1967),  
333 based on Monte Carlo permutations with 9999 randomizations to test for significance  
334 (Oksanen et al. 2016). Sampling sites were treated as statistically dependent observations  
335 because the values of SAC between geographic distance (km) and FEve distance among

336 sites was very low though statistically significant ( $r_M = 0.048$ ,  $n = 588$ ,  $p = 0.001$ ) (Manly  
337 2006). In order to alleviate SAC issues, the geographical coordinates of sampling sites  
338 were introduced as covariates to incorporate spatial variation during the modeling  
339 procedure (Legendre 1993; Dormann et al. 2007).

340 Generalized Linear Mixed Models (GLMMs) were used to study the patterns of bird  
341 community resilience (FEve) in relation to geographical coordinates of sampling sites, land  
342 use / cover composition around the point count, abundance of mammals (number of dogs  
343 and cats) and bird species richness for each trophic guild (granivorous, granivorous-  
344 insectivorous, insectivorous, carnivorous and omnivorous), modeled as fixed effects. In  
345 order to avoid any redundancy in the modeling procedure, we checked for the potential  
346 association between the index of FEve (which was estimated considering habitat, breeding  
347 and feeding traits) and the Shannon index estimated based on the species richness for  
348 each trophic guild by using a simple linear regression analysis. The association between  
349 these two variables was not statistically significant (ESM, Fig. S4;  $F = 3.2$ ,  $df = 1$ ,  $586$ ,  $p >$   
350  $0.05$ ), justifying the use of both during the modelling procedure. The following predictor  
351 variables were too strongly correlated and thus were removed, in order to avoid  
352 multicollinearity issues (Graham 2003): building (negatively and significantly correlated  
353 with grass and tree), bare soil (negatively and significantly correlated to grass) and overall  
354 species richness (positively correlated to species richness for each trophic guild). City was  
355 included as a random effect to account for possible consistent differences among cities.  
356 Models were fitted by maximum likelihood, using the package 'lme4' in R (Bates et al.  
357 2014). The model selection was based on Akaike's Information Criterion (AIC) (Burnham  
358 and Anderson 2002) with the package 'AICcmodavg' in R (Mazerolle 2016).

359 All statistical tests were performed with R software version 3.2.4 (R Development Core  
360 Team 2019).



362 **RESULTS**

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363 **Bird community resilience in nine European cities**

364 A total of 107 bird species were recorded at 588 point-counts distributed in nine European  
365 cities (ESM, Table S1). The complete list of species is provided in the Electronic  
366 Supplementary Material (ESM, Table S3). Species richness per point count ranged from a  
367 minimum mean of 7.52 species (min: 3, max: 12 species) in Munich to a maximum mean  
368 of 14.14 species (min: 3, max: 30 species) in Granada.

369 Overall, the two surrogates of bird community resilience varied between a minimum of  
370 0.69 (FEve) and 0.80 (FeveR), and a maximum of 1.00 (FEve) and 0.99 (FeveR). The  
371 differences in FEve among cities were statistically significant (ANOVA:  $F = 51.08$ ,  $df = 8$ ,  
372  $579$ ,  $P < 2e-16$ ). The lower mean value of bird community resilience (FEve) was found in  
373 Munich (0.87, min: 0.69, max: 0.95), while higher mean values were found in the Spanish  
374 cities of Madrid (mean: 0.95, min: 0.90, max: 1.00), Granada (mean: 0.94, min: 0.86, max:  
375 0.99) and Toledo (mean: 0.94, min: 0.88, max: 0.98) as well as in the city of Prague  
376 (mean: 0.94, min: 0.85, max: 0.98) (Fig. 1).

377 The values of bird community resilience (FEve) in nine European cities were not  
378 significantly correlated with the environmental characteristics around the point counts (land  
379 use / cover composition) nor with the abundance of dogs or cats (Table 1). Additionally, we  
380 did not find evidence of latitudinal or longitudinal effects on resilience (Table 1, Fig. 1). Bird  
381 community resilience was higher in cities where the relative number of granivorous or  
382 granivorous-insectivorous species was higher, while it was negatively associated with the  
383 richness of insectivorous, carnivorous and omnivorous-scavenger bird species (Table 1,  
384 Fig. 2).

385 Overall, avian trait dissimilarity was higher in carnivorous and omnivorous species than in  
386 granivorous, granivorous-insectivorous and insectivorous birds. These differences were  
387 statistically significant (Kruskal-Wallis  $\chi^2 = 43.98$ ,  $df = 4$ ,  $P = 2e-09$ , ESM, Fig. S5).

388 **Avian resilience and climate change scenarios**

389 Considering the projection of climate change scenarios for Europe in the next fifty years,  
390 provided by the Intergovernmental Panel on Climate Change (IPCC) in the Fifth  
391 Assessment Report, the cities exposed to a larger change in temperatures (higher than

392 2.4°C) are the Spanish cities of Madrid, Granada and Toledo and the Finnish cities of  
393 Rovaniemi and Turku (Fig. 3). On the other hand, the European cities more exposed to  
394 changes in terms of average precipitation in mm are Madrid, Groningen and Poznan (Fig.  
395 3). Overall, the cities less exposed to a drastic variation in terms of temperature and  
396 precipitation under a high pathway climate change projection are cities from central  
397 Europe such as Prague and Munich (Fig. 3, ESM, Table S4).

398 Crossing the mean values of bird community resilience for the nine European cities with  
399 the values of expected delta temperature and precipitation, explored in a 3D plot, highlight  
400 some potential scenarios of urban bird conservation for the next fifty years: Even if Munich  
401 was the city with the lowest avian community resilience, it seems however that bird  
402 communities within that city are not affected by strong climate change events, but only by  
403 a relative moderate change in temperature, with temperature increases of 1.9-2.3 °C, but  
404 less strong changes in precipitation when compared with other European cities (ESM, Fig.  
405 S6). The Spanish cities with higher values of bird community resilience will face more  
406 notorious climate change events: Madrid (with the higher bird community resilience) could  
407 be affected by a relatively large change in the mean amount of monthly precipitation (110-  
408 170 mm) as well as increasing temperatures around 2.4-2.8 °C (ESM, Fig. S6). Granada  
409 and Toledo could face similar scenarios of increasing temperatures, but less strong  
410 changes in terms of mean amount of monthly precipitation (ESM, Fig. S6). The city of  
411 Prague was characterized by bird communities with relative high resilience, and  
412 simultaneously not exposed to so drastic climate change events in the next fifty years  
413 (ESM, Fig. S6). The Finnish city of Rovaniemi could suffer an important increase in  
414 average temperatures, while their bird species assemblages are characterized by lower  
415 resilience than other urban bird communities, suggesting a potential conservation problem.

417 **DISCUSSION**

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418 **Climate change scenarios in cities and bird community resilience**

419 In this study, we suggested potential future conservation concerns in urban bird  
420 communities, by crossing information on a surrogate of species assemblage resilience  
421 (FEve) (Mouchet et al. 2010) and the projection of scenarios facing climate change. Even  
422 if we examined three different scenarios, we used the high pathway climate change  
423 scenario because it could be the most plausible (Brown and Caldeira 2017).

424 Our findings provide important information about the cities that could face stronger climate  
425 change events, and the capacity of their bird communities to reduce any negative impact  
426 associated with them. Briefly, a high probability of climate change and simultaneously a  
427 low value of bird community resilience can be interpreted as the worst scenario,  
428 suggesting a potential conservation concern.

429 A higher pressure of climate change in terms of delta temperatures and delta precipitation  
430 should mainly affect cities from the Southern and Northern regions of Europe. The data  
431 derived from the projections made by NCAR Climate Change Scenarios (NCAR  
432 community 2012) showed that cities like Granada, Madrid and Toledo in Spain and  
433 Rovaniemi and Turku in Finland could suffer variation in temperatures larger than 2.6°C  
434 before the year 2070 (ESM, Table S4). Additionally, Madrid and Toledo could also be  
435 exposed to a significant change in the amount of monthly precipitations. Finally, Madrid  
436 was also the biggest city surveyed in this study, with more than three million people (ESM,  
437 Table S1). This fact is important, if we consider that the number of threatened bird species  
438 could be positively associated with human population size (Pautasso and Dinetti 2009).  
439 However, these two Spanish cities are characterized by species assemblages with higher  
440 resilience, a fact that can offer insurance against the ability of communities to respond to  
441 challenges posed by climate change. On the other hand, from the two Finnish cities,  
442 Rovaniemi could constitute a potential problem for conservation in the future: This northern  
443 city could face an important increase in average temperatures, while their bird  
444 communities are mostly characterized by lower values of resilience within the cities used in  
445 our study. Especially in the case of Rovaniemi, most species are (long)-distance migrants  
446 (Schaefer et al. 2008; Saino et al. 2011) and in most cases insectivores (Fig. 2).

447 Even if all cities could be exposed to potential climate change scenarios, the cities from  
448 the central part of the continent (Groningen, Munich, Poznan and Prague) are less  
449 exposed to drastic changes in temperatures, when compared with the other cities studied  
450 (Fig. 3), suggesting a low conservation concern of urban bird communities. This is  
451 important when planning future conservation, regarding both urbanization and climate  
452 change. For example, Munich was the city with the lowest mean value of resilience of  
453 avian species assemblages, but considering that future scenarios suggest a relative small  
454 impact due to climate change, we consider that specific conservation actions attempting to  
455 promote resilience of urban bird assemblages in this city are not urgently needed. In  
456 contrast, other cities with low/medium levels of resilience, like Rovaniemi, which will be  
457 exposed to important increases in temperature in the future, should be the focus of  
458 conservation practices or at least close monitoring schemes. The bird communities from  
459 Munich were characterized by a high proportion of insectivores and simultaneously a low  
460 proportion of granivorous species, when compared with the other European cities. When  
461 comparing Munich with the nearby city of Prague, Czech Republic, we can highlight how  
462 the risk of increase in the level of threat for bird communities in Prague could be lower,  
463 because the city presents species assemblages with high values of potential resilience,  
464 also being a city not subject to particular intensification of climate change.

465 Another important result to highlight from this study is the weak correlation between  
466 potential resilience of urban bird communities and land use / cover composition where the  
467 communities were assessed. This lack of association in our study may suggest that little  
468 variation in land use / cover composition or heterogeneity in urban green areas do not  
469 have a strong impact on the regularity of bird traits in functional space and how efficiently  
470 the resources are utilized. A study focused on anthropized environments (farmlands)  
471 showed how functional evenness of bird communities is only weakly correlated with the  
472 environmental characteristics of the area (green area size, canopy heterogeneity) (Lee  
473 and Martin 2017).

474 We found in the nine European cities that urban bird communities with high richness of  
475 granivorous or granivorous-insectivorous species were the communities with higher  
476 potential resilience. In contrast, urban bird communities with an overrated number of  
477 insectivorous, carnivorous and omnivorous species were assemblages with lower potential  
478 resilience. The main reason why carnivorous and omnivorous species richness was

479 negatively associated with potential community resilience could be explained by the fact  
480 that those bird species were characterized by higher trait dissimilarity, when compared  
481 with the entire pool of species (ESM, Fig. S5). The granivorous birds recorded in this  
482 study, on the other hand, were more similar in terms of breeding and feeding traits (ESM,  
483 Fig. S5). By increasing the number of species which are functionally closer, the functional  
484 redundancy will rise, achieving a higher functional evenness and potential bird community  
485 resilience.

486 Regarding the bird's foraging characteristics in relation to the potential resilience of  
487 species assemblages, however, we consider it relevant to highlight the limitations of  
488 simple traits describing the diet of a species. Further studies should also consider the fact  
489 that many species are relatively plastic in terms of foraging ecology. Some species, as  
490 shrikes or sparrows, can exploit different types of diet depending on the geographical  
491 context and environmental conditions, being also scavengers at roadsides (Tryjanowski et  
492 al. 2003; Morelli et al. 2015). The diet plasticity of bird species, also in cities, can shift their  
493 main type of food between seasons. E.g. in the northern part of Europe, while most  
494 species need/use protein-rich arthropods during breeding season for their nestlings, they  
495 must change their diet for berries during winter, because of a lack of insects (e.g.  
496 Turdidae, Sylviidae). Geographic variation in trophic plasticity of bird species could play an  
497 important role in the overall potential resilience of the communities. Since specialist  
498 species are considered more prone to the processes that lead to extinction than generalist  
499 species (McKinney 1997; Colles et al. 2009), we can expect that a given community  
500 composed of many species characterized by high ecological plasticity in terms of foraging  
501 (but also behaviour or breeding) should be better adapted to eventual environmental or  
502 climatic changes, because such species could respond better by adapting to different  
503 types of food or other resources. Further studies could also focus on the overall level of  
504 specialization of bird communities inhabiting cities (Morelli et al. 2019).

505 Finally, the fact that we did not find any evidence of latitudinal or longitudinal effects on  
506 avian resilience among the nine cities could be interpreted as a confirmation of the level of  
507 homogenization of urban settlements in Europe (Devictor et al. 2007).

## 508 **Final considerations and some remarks on the use of a surrogate of community** 509 **resilience**

510 Recent studies have provided evidence of drawbacks for the use of this functional diversity  
511 index (e.g. Ricotta et al. 2014; Legras and Gaertner 2018), which is the evenness  
512 component of functional diversity. Briefly, the main concerns are related to the fact that an  
513 increase in FEve index values is not always synonymous with an increase in functional  
514 evenness, mainly when comparing communities which differ in terms of abundance  
515 distribution (Legras and Gaertner 2018). However, in the present study, we estimated the  
516 FEve crossing a trait matrix with a matrix of solely presence/absence of species, therefore  
517 overcoming (or at least alleviating) this potential problem. Additionally, we explicitly  
518 compared the behavior of the index FEve with the new index proposed by Ricotta et al.  
519 (2014), which measures the regularity in the distribution and abundance of species in  
520 functional space, together with the evenness in their pairwise functional dissimilarities, to  
521 ensure that in our study both could be used similarly. Moreover, any index or metric used  
522 to quantify the hypothetical “resilience” of a species assemblage must be handled  
523 cautiously. An index is not a direct "measure", especially because the resilience or  
524 capacity to respond to an alteration depends on many (and complex) factors: period of  
525 disturbance, intensity, relative plasticity of species, biotic interactions (explicit and hidden),  
526 and cascade effects (Spears et al. 2015; Morelli and Tryjanowski 2016). Ecological  
527 resilience was defined as a multifaceted concept (Cumming et al. 2005). Thus, we  
528 consider that any index should be more efficiently used when applied mainly to  
529 comparisons among sites or areas.

530 Importantly, climate change effects can interact with the distribution of bird species,  
531 facilitating the expansion of invasive birds that could compete, displace and / or prey on  
532 native species (Bellard et al. 2013). Therefore, this could potentially amplify the effects of  
533 changes in land use, precipitation or temperature. Maintenance of the level of functional  
534 redundancy in bird communities can increase the overall tolerance of such species  
535 assemblages to potential scenarios of land use and climate change (Elmqvist et al. 2003).  
536 Thus, the capacity of species assemblages to face and recover from extreme events (such  
537 as climate or land use change) will determine their persistence.

538 In conclusion, our findings highlighted how an approach combining projections of climate  
539 change scenarios and potential resilience of species assemblages (using species trait-  
540 based methods), could be useful to identify in advance conservation concerns. We  
541 hypothesize that the approach used in this study could also be applied to other taxa such

542 as insects and mammals. This may help establish adequate urban planning strategies for  
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543 the promotion of a high diversity of communities in urban exploiter and tolerant species,  
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544 increasing the level of protection of urban ecosystem functioning.  
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