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Bird communities as bioindicators in urban environments

Supervisor	Coordinator	Ph. D. candidate
Prof. Danilo Russo	Prof. Guido D’Urso	Maurizio Fraissinet

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Chapter I

1. Introduction, the phenomenon of colonization of cities by wild birds

1.1 Introduction

The World Urbanization Prospects (United Nations, 2018) reports the statement of the urban and rural world population up to 2018. The urban and rural population has 4,219,817,000 and 3,413,000,000 human beings respectively. Therefore, in 2018 more than half of the world's population, 55.3%, lived in urban centers and the remaining 44.7% lived in rural areas. It was in 2008 that the urban population exceeded the rural one; in that year, 3.2 billion people lived in the cities all over the world (Dinetti, 2009). A United Nations (UN) projection predicts that in 2050 the percentage of the urban population will reach 68.4% of the world population (United Nations, 2018). The phenomenon started during the 20th century, going from 224 million inhabitants in urban areas in 1900 to 2.9 billion in 1999 (Dinetti, 2009). It should be recalled that the UN defines "urban areas" the built up areas with over 10,000 inhabitants. In Italy, in 2018, 70.5% of the population lived in urban areas and the projection to 2050 expects further growth reaching an estimated 88.1% of urban population (United Nations, 2018).

The growth of the urban population also involves an increase in world’s land areas characterized by urbanized areas, although the latter does not follow the same speed and intensity as that of the human population. In 2005, United Nations Environmental Program (UNEP) estimated 2-4% of lands affected by urban areas. However, a small land area entails a high consumption of energy resources. In fact, UNEP believes that ¾ of the world's energy resources and materials are depleted within cities, with a consequent production of a huge amount of waste (UNEP, 2005; Tortorella and Chiodini, 2008). The increasing phenomenon of the urbanization, even if it can affect small extent areas, implies an environmental impact due to the significant transformations of the land and water use, the emission of pollutants into the atmosphere and hydrosphere, climatic alteration both locally and worldwide, the disruption of biotic communities (Dinetti and Fraissinet, 2001; Mc Kinney, 2006; Gisottti, 2007; Croci et al., 2008; Dinetti, 2009).

The impact of the urbanization of the world population on the biocenosis, both in terms of increased human density on the territory and its upheaval, has become the subject of study all over the world and research activities on this topic have had a remarkable growth during the last two decades (McKinney, 2006; Hedblom & Murgui, 2017). Whilst the considerable concentration of population in urban centers produces a substantial energy inflow and trophic input, which can benefit different animal species, on the other hand the high human density, the intense transformation of the early ecosystems and the introduction of alien species represent limiting factors able to deeply modify the native biocenoses (McKinney, 2006; Luck & Smallbone, 2011; Jokimaki & Suhonen, 2017). Hence the need to have reliable environmental indicators, easy and quick to use, comparable with other realities, understandable and known to the majority of the population, rapid in revealing when environmental changes occur. Such aspects and needs are effectively fitted by the class of Birds, which are also universally considered among the most suitable organisms for developing analysis and bioindication models (Ferenc et al., 2013). In fact, among Vertebrates, Birds is precisely the taxon most used in research on urban fauna because of the high "detectability" that characterizes this class of Vertebrates: they are mainly diurnal; they fly and therefore can be easily observed; they have coloured plumages and emit different vocalizations (Cocker and Tipling, 2013). Hence, the possibility of involving enthusiast and usefully trained volunteers in the field data collection within the context of Citizen Science. There are two particularly exciting research topics about the phenomenon of colonization of the urban environment by birds: the study of the intrinsic ecological and ethological characteristics that a species have to possess in order to colonize and the analyses of the variation over time of the bird communities' composition in urban areas, in consideration of the rapid environmental changes taking place in the cities. In order to study these phenomena it is necessary to have a long-time scale monitoring plan, providing a good coverage of the territory, and a standardized method of field data collection that is constant over time.

The city of Naples is provided with an efficient monitoring plan according to the above mentioned conditions because two ornithological atlases of breeding and wintering species have already been developed and published throughout the municipal area, and a third atlas has been created during our research project. Therefore, three ornithological atlases exists for the municipality of Naples, covering the years 1990 - 1994 (Fraissinet, 1995), 2000 - 2005 (Fraissinet, 2006) and 2014 - 2018 (Fraissinet et al., 2017), and realized using the same survey methods, the coordinator and most of the researchers.

This research project have the following objectives:

- To verify changes occurred in the bird community during the selected time period using community indices;
- To study the causes of the community's variations also as a function of the land use and the environmental change occurred over the years in the city of Naples;

- To analyse population trends for each urban bird species;
- To develop predictive models which measure the probability of presence of a species in the urban area as a function of the extension of a given environmental category;
- To study the biology of a problematic species in the urban environment such as the Yellow-legged Gull (*Larus michahellis*) in order to identify possible management solutions.
- To analyse the impact of urban ecosystem on the potential range expansion of a non-synanthropic species such as the Black Stork (*Ciconia nigra*) by developing a predictive model of the risk posed by the even more widespread electric power lines and wind turbines on the species' potential breeding range in Italy.

Supported by a monitoring plan carried out on a long-time scale, with this theses' project I aim to confirm or not the knowledge acquired so far on the phenomenon of birds' colonization in the urban area. In particular, in some cases, studies about this topic were carried out over short time periods (Fidino & Magle, 2017) and need to be deeply investigated. Moreover, this work will effectively assess how much the environmental changes in the urban areas affect the presence and distribution of species in the city. Hence the possibility of defining more precisely the role, but also the measure, of bioindication in the urban environment.

1.2 The urban ecosystem

The first definitions of urban ecosystem developed in the context of American ecological school in the 1970s, in particular by Detwyler and Marcus (1972) and Stearns and Montag (1974). However, the Japanese Numata (1976, 1982, 1990), as part of the UNESCO's Project 11 MAB - Man and the Biosphere - developed the basic ecological concept of urban ecosystem, including analogies towards the natural ecosystem. His aim was to simulate models of urban ecosystem in order to adopt an efficient integrated management of it. Another researcher who provided a significant contribution to understanding the urban ecosystem was the Belgian Duvigneaud (1974), who in addition to studying urban metabolism, analysed the different ecological factors characterizing the urban ecosystem such as topography, hydrography, climate, edaphic factors and biocenosis. The latter, being an ecosystem with a predominant human component, was defined "anthropocenosis". Ellemberg (1973), within a classification of the mega-ecosystems present in the world, defined "abiotic ecosystems" those ecosystems that depend 100% on human regulation and, in particular, on the provision of fossil, electrical energy, etc. The paradigm of urban ecology is that cities represent an emerging phenomenon of dynamic interactions, at local scale, between socioeconomic and biophysical forces. These interactions emphasize a specific ecology, ruled by particular ecological functions and forces (Dinetti, 2009). It is interesting to remember that the term "urban ecology" was coined by sociologists who used the ecological theory to describe human behaviour in urban settlements (Collins et al., 2000).

The first feature that distinguishes the urban ecosystem from the "natural" ones is obviously the high density of a single animal species - humans - capable of strongly influencing the territory that it has occupied. Hence, the presence of a large amount of buildings which profoundly modify the soil, the water cycle and some microclimatic conditions (Dinetti and Fraissinet, 2001; Alberti et al., 2003; Gisotti, 2007). Another significant element characterizing urban ecosystems is the almost total absence of primary production resulting in a need for continuous energy flows from other territories. These energy flows, metabolized within the urban ecosystem, are unlikely introduced into a cycle and, consequently, once used and transformed, they are expelled outside towards other territories. Therefore, we are facing with a heterotrophic ecosystem (Alberti et al., 1994). Indeed, every day a city transforms into heat a quantity of energy per square meter 70 times more compared to an adjacent area (Dinetti, 2009). Moreover, unlike natural ecosystems, the main source of energy of urban ecosystems is not the sun but fossil fuels externally derived, involving profound alterations of the ecosphere (Dinetti and Fraissinet, 2001). The urban ecosystem can also be defined as a permanently “juvenile ecosystem”, as the continuous anthropic disturbance maintains phases of continuous transition preventing from achieving conditions of balance and stability typical of a mature ecosystem (Vernetti, 1990). In addition, another characteristic of urban ecosystem is a spatial heterogeneity as it consists of a mosaic of urban habitats (Garden et al., 2006: Wilby & Perry, 2006; Murgui, 2009; Werner & Zahner, 2008).

The main characteristics of urban ecosystem are listed below (Detwyler and Marcus, 1972; Duvigneaud, 1975; Sukopp et al., 1980; Sukopp and Werner, 1983; Celecia, 1997; Dinetti and Fraisisnet, 2001; Sukopp, 2002; Gisotti , 2007; Dinetti, 2009):

- high human density;
- heterogeneous and mosaic structure: the city is an assembly of different environmental categories;
- change in the topography with a tendency to levelling;
- concentration of the building;
- soil sealing;

- import and conveyance of large mass of water;
- microclimate variations;
- import and export of materials;
- strong prevalence of man as a consumer;
- scarce primary production;
- biocenoses’ changes;
- strongly growth in population of species table companion of humans;
- colonization of species adapted to new environmental conditions;
- production of anthropogenic waste;
- air, water and soil pollution;
- large-scale energy consumption.

The above mentioned characteristics are common to all urban environments and lead to a homogenization of biocenoses in the different taxa (Jokimaki et al., 2002; Jokimaki et al., 2003; McKinney, 2006; Clergeau et al., 2006; Moller, 2009; Bellocq et al., 2017; Chen et al., 2017; Ferenc et al, 2018). Among the most relevant aspects characterizing these biocenoses, there are the presence of invasive species, the high presence of generalist species at the detriment of specialist ones, a lower weight of the biogeographical aspects related to latitudinal distribution, a higher density (Sukopp et al., 1981; Sukopp & Werner, 1982; Chudzicka & Skibinska, 1994; Celesti Grapow, 1995; Celesti Grapow et al., 1996; Amori e Boitani, 1998; Clergeau et al., 2001; Mc Kinney, 2006; Malher et Lesaffre, 2007; Fraissinet & Fulgione, 2008; Evans et al., 2009; Litteral & Shochat, 2017).

1.2.1 The urban birds

As previously mentioned, Birds represents one of the most used taxa to study the phenomenon of colonization of the urban ecosystem. The high "detectability" characterizing this class of Vertebrates makes it a good model for studies on urban biocenoses. They are mainly diurnal, fly and therefore can be easily observed, have coloured plumages, emit different vocalizations (Cocker and Tipling, 2013). Hence, the possibility of involving enthusiast and usefully trained volunteers in the field data collection within the context of Citizen Science. In addition, the recognized role of Birds as reliable environmental indicators, easy and quick to use, comparable with other realities, make them universally among the most suitable taxa for developing analysis and bioindication models

(Ferenc et al., 2013). As a consequence, the colonization processes of birds in urban environments represents the topic of a big amount of studies all over the world and that have been strongly increased in the last decades, reaching an exponential trend in the last 15 years (Hedblom & Murgui, 2017).

There are several currents of research about this topic. First studies focusing mainly on ecological aspects and related models, were followed by other researches addressing the following arguments: birds' adaptability, both behaviourally and ecologically, to the different environmental categories of the urban ecosystem; the trend of community and demographic indices of urban populations; the turn over within the communities; the different qualitative and quantitative compositions of urban bird community according to urbanization gradients of the area and within the different environmental categories that compose the mosaic of the urban ecosystem; on the "filters" effecting colonization of urban ecosystem by birds (Hedblom & Murgui, 2017). These studies, carried out in these years in several cities distributed in different continents, allow us to clarify some phenomena that seem to be recurrent on a continental scale.

Firstly, it was explored the phenomenon of the homogenization of urban bird community in terms of ecology and taxonomy, resulting in the dominance of some orders and families over others (Ferenc et al., 2018). Indeed, dispersive and generalist species are constantly prevalent compared to specialist ones and, at the same way, resident species compared to migratory ones. The urban bird community it is mostly composed by a prevalence of forest and rock species compared to those of open and rural spaces, and of species nesting on trees or rocks compared to those nesting on the ground (Devictor et al., 2007; Croci et al., 2008; Moller, 2009; Silva dos Santos et al., 2019; Litteral & Shochat, 2017). Another aspect that seems to remain constant in time is the decrease of biodiversity with increasing human density, hence leading to a biodiversity gradient proceeding from the most central areas of the city outward (Batten, 1972; Evans et al., 2009). Moreover, a relative biogeographical "anarchy" is found in the urban ecosystem as, in urban centres, some typically biogeographical phenomena such as the correlation between species richness and latitude, (Ferenc et al., 2013), or peninsular regions (Fraissinet & Fulgione, 2008) do not occur, or are less evident. Such findings can be explained by the presence of several ecotonal conditions in the urban mosaic (Erz, 1966) and by the direct effect of human's actions on the trophic richness of the urban ecosystem (Jokimaki et al., 2002; Jokimaki et al., 2003). From recent taxonomic analyses it is emerged that the species of the *Columbiformes, Apodiformes, Bucerotiformes and*

Caprimulgiformes orders, followed by those of the *Piciformes* and *Passeriformes*, are more frequent in the city. The least frequent are those of the orders *Otidiformes*, *Procellariformes, Galliformes, Suliformes, Accipitriformes* and *Falconiformes* (Ferenc et al., 2018). The high tolerance of taxonomic groups to colonize cities on a global scale, as avian families like *Columbidae, Corvidae* and *Sturnidae*, also suggests a phylogenetic predisposition (Sol et al., 2014).

In the last years, a particular attention has been given to the importance to have scientific data on urban bird collected over a long time scale (Murgui, 2014; Fidino & Magle, 2017). In fact, studies carried out for short periods on the evolution of species richness and composition in the urban environment have led to ambiguous and sometimes contradictory results. Given the wide range of responses we observed in the colonization processes, it is evident that spatial differences observed along a gradient over short time frames do not adequately capture the many temporal forces that may subtly influence species richness and the different bird compositions which may occur as urbanization increases through time (Fidino & Magle, 2017). In the review by Fidino & Magle (2017), which analyses the results achieved over 30-year of monitoring studies, it is emphasized the importance of long-term studies in documenting evolutionary change in urban bird populations. Regardless of reported increases or decreases in species diversity, a high rate of turnover is a common trend in long-term studies (Travis, 2003), together with the introduction of invasive species (Foster et al., 2002), maturation of local native or non-native vegetation (Gleditsch, 2016), land-use legacies (DeGraff and Wentworth 1986; DeGraff, 1991) and habitat fragmentation influencing species persistence, colonization, and extinction rates (Tait et al., 2005; Walk et al., 2010).

Thus, although species richness may not change at a site over time, the composition of species present does. However, while the notion that urbanization may not decrease species diversity through time is encouraging, such a conclusion does not consider the relative values of particular species, the functional diversity of the urban bird community, or the type of habitat that is urbanized. The turnover in species composition often ends to benefit invasive and generalist species with a wide distribution, and to penalize native species and those with more specialized habitat requirements that loose suitable habitats caused by urbanization (Fidino & Magle, 2017).

1.3 Contents

This thesis is divided in the following parts:

- Update on the level of knowledge on birdlife of the city of Naples and analysis of the changes that have taken place in the Neapolitan urban birdlife during 28 years of monitoring. We adopted this large time scale, unusual for researches on urban birdlife (Fidino & Magle), 2017) in order to verify the validity of the information collected so far about the phenomenon of colonization of cities by wild birds (Chapter II).
- A large-scale analysis of the phenomenon of the relative independence of urban bird community from biogeographical factors in order to investigate and provide additional elements of knowledge (Chapter III).
- Analysis of how the size of breeding range can influence birds' colonization of cities on a European scale (Chapter IV).
- Study of a particularly synanthropic species, such as the Yellow-legged Gull (*Larus michahellis*), focusing on the problematic aspects deriving from the coexistence with humans within cities (Chapter V).
- Analysis of the impact of urbanization on the colonization dynamics of non-synanthropic species, such as the Black Stork (Ciconia nigra) (Chapter VI).
- Conclusions (Chapter VII).

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A view of Naples city by San Martino hill.

Chapter II

1 Long-term monitoring reveals responses of avian assemblages to multiple temporal and spatial scales of land use dynamics in urban landscapes

Maurizio Fraissinet[1,2,*], Leonardo Ancillotto[1,*], Antonello Migliozzi[3], Silvia Capasso[2], Luciano Bosso[1], Danilo Russo[1]

1 Wildlife Research Unit, Dipartimento di Agraria, Università Federico II di Napoli, Via Università 100, 80055 Portici (Napoli)

2 Associazione Studi Ornitologici Italia Meridionale, ASOIM, via cavalli di bronzo 95, 80046 San Giorgio a Cremano (Napoli)

3 Dipartimento di Agraria, Università Federico II di Napoli, Via Università 100, 80055 Portici (Napoli).

*= co-first authors

Abstract

The study of birds in urban areas has increased in recent times, yet few studies provide a picture of colonization dynamics over long time windows, at the same time building on extensive and complete datasets on species presence. Within the city of Naples, three bird monitoring campaigns took place, namely in 1990/95, 2000/05 and 2014/18, each time maintaining the same sampling designs, field methods and effort, the campaign coordinator and most of involved personnel. Such continuity provided a solid database on which to analyse occupancy trends of nesting urban birds across 28 years, allowing to test the relationships between species and land use dynamics throughout such extended time. Species richness of breeding birds remained stable over time, yet a significant species turnover occurred between the first and the third monitoring campaigns, with a bias due to the species'favorite habitats; namely, bird species associated with forest and rocky habitats increased their presence, while birds dependent from open and cultivated areas showed a general decrease. Similarly, carnivorous and omnivorous species showed an increase when compared to insectivorous taxa. The quantitative analysis to associate each bird species to a favoured land use class also produced models that may be used to predict the probability of a species' presence in relation to such land use class, and thus provide a simple but powerful tool to inform land use managers and conservationists that work on landscape and bird conservation in urban environments.

Keywords

Urban Ornithological Atlas, Naples, historical dataset, estimated probability of presence

1. Introduction

In comparison to most natural habitats, urban areas represent very dynamic landscapes: renovations, urban expansion, management of green and blue spaces, and urban horticulture all contribute to a continuous change in land use within the perimeter of most cities worldwide, and provide a rich mosaic of different habitats types (McKinney, 2006; Croci et al., 2008; McKinney, 2008). Given the strongly dynamic nature of urban areas in terms of land cover and management, the study of urban wildlife assemblages needs long time-scales to detect genuine trends, and results may significantly vary when different time frames are adopted (Weiserbs & Jacob, 2007; Murgui, 2014; Fidino & Magle, 2017).

Birds are excellent models to test hypotheses on urbanization and its effects on animal communities thanks to well-established monitoring protocols, the generally high detectability of most bird species, as well as the high numbers of the latter that thrive in urban habitats. For such reasons, a wealth of literature exists on the ecology of birds in urban environments (see review Hedblom & Murgui, 2017). A comprehensive understanding of changes in avian assemblages in urban areas requires the usage of historical datasets to allow for comparisons across long time series, but such datasets are difficult to obtain and thus very rare (Fidino & Magle, 2017). From this point of view, the city of Naples (Southern Italy) represents an important exception, since bird presence data in the urban area have been collected for the preparation of three bird atlases, respectively in 1990-95 (Frassinet, 1995), 2000-2005 (Fraissinet, 2006) and 2014-2018 (Fraissinet et al., 2017). Naples represents one of the largest urban areas of Europe and is the third Italian city by number of inhabitants. It has undergone considerable expansion since 1960s, so its territory provides an important opportunity to test hypotheses related to the temporal and spatial effects of urbanization on bird communities. The city's geographic scale is such as to represent an effective model for many large European urban areas.

In our study, set in Naples over the temporal interval of 28 years spanning across the three above-mentioned data collection phases, we formulate the following hypotheses and predictions:

1) *Changes in avian assemblages over 28 years*

   Bird assemblages are known to react to urbanization over time, so we hypothesize that bird communities have changed across the time period considered. Based on studies conducted elsewhere, urbanization progressively leads to reduction or disappearance of open-space specialists, highly specialized species or long-distance migratory birds, while species that dwell in rocky habitats or forests, as well as sedentary species are favoured (Clergeau et al.,2006; Devictor et al., 2007; Malher & Lesaffre, 2007; Croci et al., 2008; Moller, 2009; Murgui, 2014; Litteral & Shochat, 2017). Therefore, we predict that avian assemblages in the study area have undergone detectable changes according to the same patterns.

2) *Time taken to detect changes in avian assemblages*

   Bird species react to land use change, but the latter takes time to occur. Therefore, long-term monitoring is necessary to highlight such reactions (Murgui, 2014; Fidino & Magle, 2017). We hypothesize that in our case too, only a data coverage of a sufficiently long time will make it possible to detect responses in avian assemblages. We therefore predict that most changes will be only evident between the first (1990-1995) and the last (2014-2018) epochs of data collection.

3) *Small-scale response of breeding birds to land use*

   As often found in highly mobile species, birds show ecological responses to multiple spatial scales (McKinney, 2008; Ferenc et al., 2018). While changes in community composition as those we explore in hypothesis 1 are likely detected on a large scale (McKinney, 2008), breeding species may respond finely to a much smaller spatial scale which influences resource availability in terms of nesting microhabitat structure and food (Clergeau et al., 2001; Mc Kinney, 2008; Chiari et al., 2009; Evans et al., 2009; Murgui, 2014). We therefore hypothesize that breeding bird species will show clear responses to land use composition also on a small spatial scale (1km^2) and predict that land use change over such a scale will make it possible to estimate the likelihood of breeding for the species that are present in urban areas.

2. Materials and Methods

2.1 *Study area*

The study was conducted in the city of Naples (40°50′ N, 14°10′E), the largest city in southern Italy, covering ca. 117 km^2 and inhabited by 959,188 (as to December 1t 2019) with population density of 8,179 inhabitants / km^2. It is a city facing the Tyrrhenian Sea with 8 km of coastline. Elevation ranges from 0 to 457 m a.s.l., and the climate is typically Mediterranean, with hot and dry summers and precipitations concentrated in autumn.

The habitats represented within the metropolitan area of Naples are mainly human-modified, with built-up areas strongly dominating the landscape, which is characterised by a mosaic of patches of parkland, including both natural habitats and recreational green spaces, and small agricultural areas. The city is also surrounded by one of the largest conurbations in Italy, extending for > 1,100 km^2 and with a population density of 2,634 inhabitants / km^2.

2.2 *Bird surveys*

Data collection was conducted following the same protocol for all the three atlases. A 142 1-km quadrat grid was overlapped to the city's map to identify standardized sampling points (Fraissinet, 1995; Fraissinet, 2006) (fig.1). The project coordinator, most field assistants (n=57) and sampling protocols were consistent throughout the years of this study, which adopted the guidelines of the European Bird Census Council (Hagemeijer & Blair,1997). All data were recoded and mapped in Q Gis (2018).

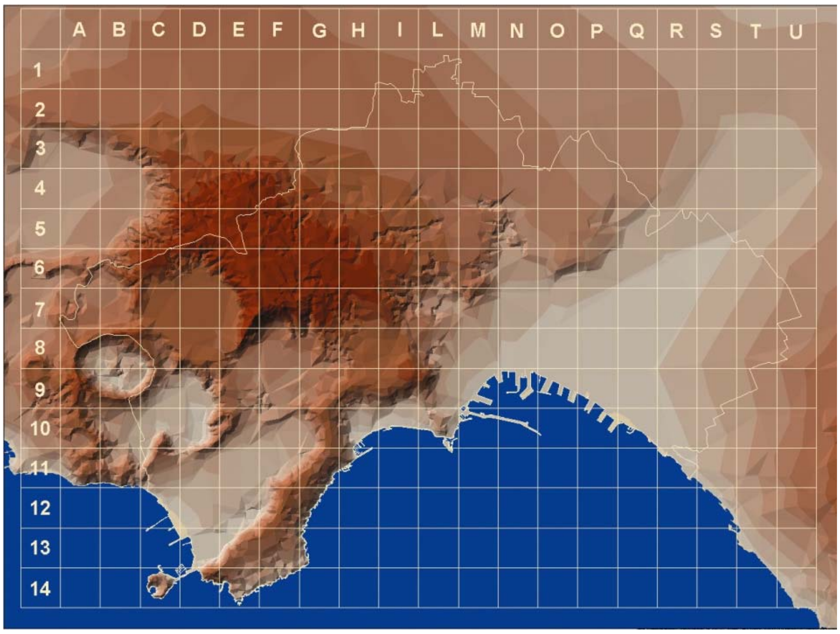

Figure 1 – Map of the urban area of Naples, with the 1 km UTM square grid overlaid. The white line indicates the administrative boundaries of the city, also indicating the study area.

2.3 Land use mapping

Land use of the study area was obtained by photo-interpretation of satellite images (scale 1:1500) and classified into 42 vegetation and land use classes following the Corine Land Cover (*European Environment Agency* - EEA, 1991-1999), by using ArcView 3.2 (with the following extensons: Xtools, Spatial Analyst, ECW v2.0 and ER Mapper Images, CLU Quality Control). Images were obtained separately in 1995, 2006 and 2018, obtaining three different maps corresponding to the final year of each bird sampling session. This classification was then simplified before analyses, with land use classified into the following 5 classes:

1. Urban areas (urb): built-up and residential areas including infrastructures and paved roads, characterized by impervious surfaces, sometimes with small patches of green areas (<0,3 ha) e.g. private gardens;
2. Non-intensive farmland (nif): traditionally managed crops including orchards, vineyards and vegetable gardens, usually separated by hedgerows and small patches of woodland and grassland;
3. Wooded areas (war): areas featuring extensive tree cover, both natural and managed, i.e. including treed recreational areas such as public parkland, historic villas, botanical gardens. These mostly comprised native broadleaved tree species (*Quercus* spp., *Castanea sativa*), conifers (*Pinus pinea, P.halepensis* and *P. pinaster*) and non-native taxa (e.g. *Robinia pseudacacia*, *Ailanthus altissima, Eucaliptus* spp.);
4. Cropland (cro): intensively managed agricultural areas, mostly featuring herbaceous cultivations, also comprising large areas covered by greenhouses;

5. Fallow (fal): pioneer vegetation colonizing newly unmanaged areas, mostly characterized by overgrown herbaceous vegetation and low scrubland (e.g. *Rubus* spp.).

We excluded poorly represented classes such as mining sites, landfills and water bodies, each consisting in <5% of the entire urban territory.

Land use maps were then overlapped with the bird sampling grid, and habitat composition within each cell was quantified as the percent of each of the five land use classes, separately for each sampling session. All procedures were implemented in ArcGis 10.3.1 b

We ran a series of paired Student t-tests to assess whether the amounts of different land use classes within each sampled cell varied between intervals, considering significant all results with $p<0.05$.

2.4 Description of bird assemblages

We calculated commonly used indices to assess temporal changes in the alpha and beta diversities of the bird assemblages, i.e. how the community structure changed among the three sampling sessions between 1990 and 2018. We calculated species richness, the Sorensen Index (S; as the proportion of shared species between samples over the total number of species), Simpson's diversity index (using the numbers of cells occupied by each species as a proxy of abundance), and species evenness. Besides, we calculated the Index of Ornithological Value (IVO) following Massa et al. (2004); the IVO is an index adopted to assess an avian assemblage in terms of conservation value, by evaluating its composition weighted for species' conservation status at global and/or local scale. This index is calculated as follows: IVO = Stot [(SSpec1 x 1) + (SSpec2 x 0,75) + (SSpec3 x 0,50)+ (SNonSpecE x o,25) + (SCR x 1) + (SEN x 0,75) + (SVU x 0,50) + (SNT x 0,25) + S147] x 100-1, where Stot is the number of breeding bird species, SSpec1, SSpec2, SSpec3 e SNonSpecE are the numbers of bird species falling in the SPEC1, SPEC2, SPEC3 and NonSPECE categories as defined by

BirdLife (2017), respectively. SCR, SEN, SVU, SNT are the numbers of bird species assigned to different risk classes (CR, EN, VU, NT) by the Italian IUCN red list (Peronace et al. 2012); and S147 is the number of bird species included in the Annex I of the EU Bird Directive.

In order to evaluate the time scale at which changes in community structure are detectable, for each index we ran a similarity permutation test as implemented in Past 3.0 (Hammer et al.2001), each based on 9999 permutations. We considered different those samples with non-significant similarity, i.e. index values <0.4 and p>0.05. The analysis presented in this chapter only focus on samples showing significant differences.

2.5 Species' relationships with land use classes

We first built single-species generalized linear mixed models (GLMM) for evaluating the effect of land use composition on species' probability of presence in a cell; for each species separately, we modelled the probability of presence as a function of habitat composition inside each cell. Models were built with a binomial error distribution and a logit link function, using presence/absence data as response variable, the amount of each of the five land use classes as the explaining variable, year of sampling and cell identity as random effects. For ease of comparison, we retained full models for all species, considering significant only those habitats with p<0.05. Each species was then assigned to a specific land use class, i.e. the class which explained most variance of probability of presence among those with significant effects. In case more than one land use class contributed equally (i.e. with a difference of <10%), we assigned the species to the land class with the highest effect size. We included in this analysis only those species with >20 cells of presence in at least one sampling session. We excluded the taxa known to be dependent on excluded habitat types (aquatic birds).

Species for which no land class had any significant effect on their probability of presence were excluded from further analyses.

2.6. Effects of traits on urban bird dynamics

In order to test whether eco-behavioural traits affect the bird assemblages occurring in our study area, and how such traits differently influence bird occupancy dynamics in the urban area, we first assessed how species occupancy changed between the first and the third sampling session, grouping species in functional groups. We selected traits related to habitat use by birds during the reproductive season, i.e. main habitat used, foraging habits (diet) and reproductive ecology (classified as the favoured type of nesting substrate). Data on diet and nesting substrate were obtained from literature (Del Hoyo et al., 1992; Anton et al., 2017), while the main used habitat was identified by previous analyses (see section 2.5.1). For each species, we calculated the difference between the numbers of cells occupied in different years, comparing only datasets that resulted significantly different in their community structure (see 2.4). We first assessed whether such difference followed a normal distribution by running Shapiro-Wilk tests; we then used difference values as response in multiple comparison ANOVA models, entering eco-behavioural classes as categorical variables, separately for each trait. Each significant test was followed by a Tukey's post hoc test with Bonferroni correction for multiple comparisons in order to test for the significance of differences between paired categories. All tests were considered significant when $p<0.05$.

2.7 Landscape and assemblage variation in time

We examined whether habitat composition and the bird assemblage changed between 1990 and 2018, and whether different land use classes and bird functional groups of species contributed to such changes, respectively. We visualized differences in composition by using non-metric multidimensional scaling (NMDS), an ordination technique that locates samples in a multidimensional space, based on a dissimilarity matrix derived from Bray-Curtis index values. For habitat composition, such a matrix was obtained through the comparison of land use classes within each cell sampled in the two time intervals, while for the bird assemblage it was obtained by comparing species composition. Such analyses were followed by an Analysis of Similarity (ANOSIM), a randomization technique that calculates the probability that a sampling unit belongs to a given group of samples, based on intra- and inter-sample variances; significant test results ($p<0.05$) indicate that samples, i.e. those of 1990/1994 and 2014/2018 in our case, differ significantly in composition, i.e. in land use class and bird species respectively. As a last step, we ran Similarity Percentage analyses (SIMPER) to calculate the relative importance of each group to the detected changes, i.e. which land use classes and bird functional groups changed most in time.

GLMs and GLMMs were run in R 3.2.2 using the package nlme (Pinheiro et al. 2011), while all other tests were run in Past 3.0 (Hammer et al. 2001).

3. Results

3.1 *Landscape mapping*

Across the entire study area, landscape changed in terms of land use types between 1995 and 2018. Urban areas increased (7.8%), and so did cropland (26.4%) and wooded areas (17.4%), while non-intensive farmland and fallow areas decreased by 41.6 and 28.8%, respectively. A similar pattern emerged at the cell scale, as mean urban habitat coverage per cell increased significantly ($t=-3.98$, $p<0.001$), shifting from 53.4 ± 30.7 to 57.6 ± 32.3 ha, and so did wooded area coverage ($t=-1.78$, $p<0.05$), increasing from 7.4 ± 13.0 to 8.7 ± 14.9 ha. Non-intensive farmland decreased significantly ($t=7.16$, $p<0.001$; from 13.2 ± 14.2 to

7.7±13.1), and so did fallow (t=-4.78, p<0.001, from 5.4±9.3 to 1.4±3.9 ha per cell), while cropland showed no significant variation (t=-1.2, p=0.17, from 1.4±3.9 to 1.8±4.9).

Changes in avian assemblages over 28 years

Forty-five bird species were detected as breeding in the study area between 1990 and 2018, for 36 of which we had enough data for quantitative analyses. All considered indexes were relatively stable between each subsequent study period, with similarity index, species richness, Simpson's diversity, evenness and the Bird Value Index showing no significant variation from an atlas to the next one. Conversely, the similarity index between the first and the third atlas differed significantly (Tables 1 and 2), as well as the community Dominance (1990: 0.04; 2018: 0.03; p<0.001), Simpson's diversity (1990: 0.96; 2018: 0.97, p<0.001), and evenness (1990: 0.51; 2018: 0.58; p<0.01), while species richness remained stable (1990: 62; 2018: 64; p>0.05).

Table 1 – list of the bird species breeding in Naples and diversity indexes during three monitoring campaigns (1990/1994, 2001/2005, 2014/2018). R= species richness; S= Simpson's diversity; H= Homogeneity index; I.V.O.= Index of Ornithological Value.

Species	**1990 / 1994**	**2001 / 2005**	**2014 / 2018**
Tahcybaptus ruficollis		X	X
Ixobrichus minutus		X	
Accipiter nisus		X	X
Buteo buteo		X	X
Falco tinnunculus	X	X	X
Falco peregrinus	X	X	X
Coturnix coturnix	X		
Phasianus colchicus			X
Gallinula chloropus	X	X	X
Fulica atra		X	X
Charadrius dubius		X	
Larus michahellis	X	X	X
Columba livia var. *domestica*	X	X	X
Columba palumbus		X	X
Streptopelia decaocto	X	X	X
Psittacula krameri		X	X
Cuculus canorus	X		
Tyto alba	X	X	X

Otus scops	X	X	X
Athene noctua	X	X	X
Strix aluco	X	X	X
Apus apus	X	X	X
Apus pallidus	X	X	X
Tachymarptis melba	X	X	X
Merops apiaster		X	X
Upupa epops		X	X
Jynx torquilla	X	X	X
Picoides major	X	X	X
Picus viridis			X
Calandrella brachydactyla	X	X	
Alauda arvensis	X		
Hirundo rustica	X	X	X
Delichon urbicum	X	X	X
Anthus campestris	X		
Motacilla cinerea	X	X	X
Motacilla alba	X	X	X
Troglodytes troglodytes	X	X	X
Erithacus rubecula	X	X	X
Luscinia megarhynchos	X	X	X
Saxicola torquata	X	X	X
Monticola solitarius	X	X	X
Turdus merula	X	X	X
Cettia cettii	X	X	X
Cisticola juncidis	X	X	X
Acrocephalus scirpaceus	X	X	X
Acrocephalus arundinaceus	X	X	
Sylvia cantillans	X		X
Sylvia melanocephala	X	X	X
Sylvia communis	X	X	X
Sylvia atricapilla	X	X	X
Phylloscopus collybita	X	X	X
Regulus ignicapillus	X	X	X
Muscicapa striata	X	X	X
Aegithalos caudatus	X	X	X
Periparus ater	X		X
Cyanister caeruleus	X	X	X
Parus major	X	X	X
Sitta eruopaea	X		
Certhia brachydactyla	X	X	X
Oriolus oriolus	X		X
Lanius collurio	X	X	
Lanius senator	X	X	
Garrulus glandarius	X	X	X
Pica pica	X	X	X
Corvus monedula	X	X	X
Corvus cornix		X	X
Corvus corax		X	X
Sturnus vulgaris			X
Passer italiae	X	X	X
Passer montanus	X	X	X

Fringilla coelebs	X	X	X
Serinus serinus	X	X	X
Chloris chloris	X	X	X
Carduelis carduelis	X	X	X
Emberiza citrinella	X		
Emberiza cirlus	X	X	X
Emebriza calandra	X		
R	62	64	64
S	0,96	0,96	0,97
H	0,38	0,52	0,46
IVO	14,10	13,60	10,72

Table 2. Similarity indexes among the three ornithological atlases of Naples significance\correlation r *=p<0.05; n.s.=non significant.

	1990 / 1994	2001 / 2005	2014 / 2018
1990 / 1994		0,41	0,30
2001 / 2005	n.s.		0,45*
2014 / 2018	n.s.	*	

Species occupied between 1 and 142 cells per year (i.e. the entire study area). The six most common species did not change in any of the three sampling sessions, and are all well-known urban adapters or exploiters, i.e. the Italian sparrow *Passer italiae* (91.5-99.2% of occupancy), the rock pigeon *Columbia livia* var. *domestica* (79.8-94.3%), the blackbird *Turdus merula* (86.8-93.6%), the European serin *Serinus serinus* (85.4-94.3%), the blackcap *Sylvia atricapilla* (72.2-88.0%), and the green finch *Carduelis chloris* (65.2-80.9%). The bird assemblage in the urban area of Naples was yet not completely stable, with 10 and 12 species respectively going extinct and colonizing the area between the first and second atlas, while 6 extinctions and 6 colonisations (including 3 re-colonisations) occurred in the second interval.

Table 3. The commonest bird species in the urban area of Naples at three time intervals, with an indication of percent occupancy of the study area (as the percent of 1x1 km square cells; n tot=142).

1990/1995	**%**	**2001/2005**	**%**	**2014/2018**	**%**

Passer italiae	96,5	*Passer italiae*	99,2	*Passer italiae*	91,5
Turdus merula	86,8	*Columba livia* var. *domestica*	94,3	*Columba livia* var.*domestica*	93,6
Serinus serinus	85,4	*Serinus serinus*	94,3	*Serinus serinus*	91,5
Columba livia var. *domestica*	79,8	*Turdus merula*	93,6	*Turdus merula*	90,8
Sylvia atricapilla	72,2	*Sylvia atricapilla*	83	*Sylvia atricapilla*	88
Carduelis chloris	65,2	*Carduelis chloris*	80,9	*Carduelis chloris*	73,2
Paris major	59	*Passer montanus*	54,9	*Streptopelia decaocto*	69,7
Passer montanus	54,8	*Parus major*	52,1	*Corvus corone*	67,6
Fringilla coelebs	49,3	*Carduelis carduelis*	48,5	*Pica pica*	61,9
Carduelis carduelis	49,3	*Falco tinnunculus*	45,7	*Columba palumbus*	52,8

3.2 Species relationships with land use classes

Species were grouped into five categories corresponding to land use classes from which their probability of presence was mostly influenced (Figure 2). Most species were closely dependant on the surface of wooded areas (n=18), followed by species favouring cropland (n=6), non-intensive farmland (n=5), fallow (n=4) and urban areas(n=3). Five species were not influenced by land cover (n=5). For all these species, we only detected positive relationships between probability of occurrence and amount of a given land use class (Table A1 in Appendix), i.e. we found no negative relationship. Models for 18 species actually featured >1 land use class as significantly affecting occupancy, namely 2 (n=13) and 3 (n=5).

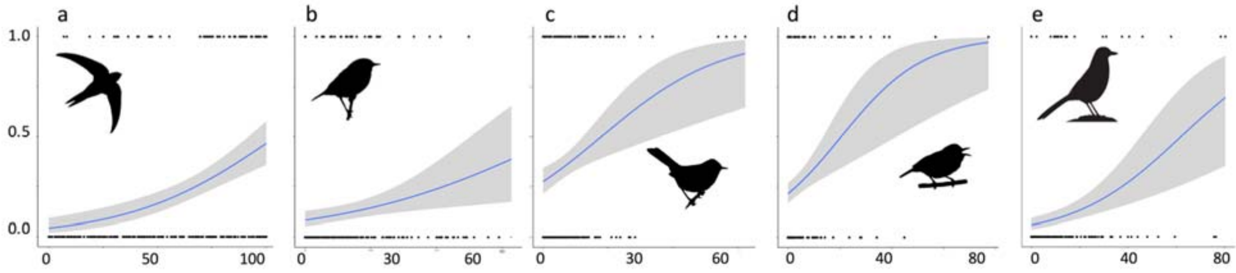

Figure 2. Estimated probability of presence of five bird species breeding in the urban area of Naples in relation to the amount of different favoured land use classes within a 1x1km cell, calculated with binomial generalized linear mixed models. a) Common Swift (*Apus apus*) as a

function of urban land cover; b) Stonechat (*Saxicola torquatus*) as a function of non-intensive farmland; c) Cetti's warbler (*Cettia cetti*) as a function of fallows; d) Fan Tailed warbler (*Cisticola juncidis*) as a function of cropland; e) Eurasian jay (*Garrulus glandarius*) as a function of wooded areas. Grey shaded area: estimate error.

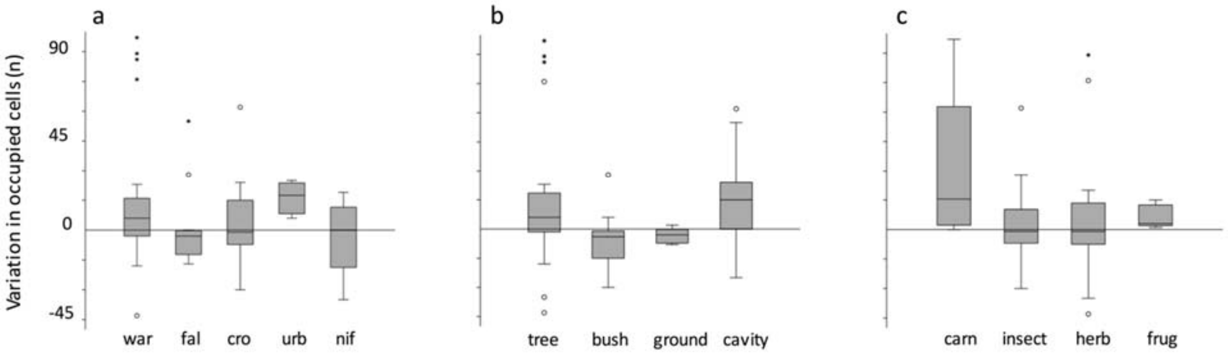

Figure 3. Variation in occupancy (whiskers=standard error) of bird guilds in the urban area of Naples between 1994 and 2018, separately according to favoured habitat type (a), nesting substrate (b) and diet (c). * and ° indicate outliers.

3.3 Landscape and assemblage variations in time

Composition of land use classes in the urban area of Naples did not change significantly between 1994 and 2018 (ANOSIM: R=0.02, p=0.068, based on 9999 permutations). Land-cover class-specific changes were mostly due to urban and wooded areas, and by non-intensive farmland, accounting for 51.0, 15.5 and 18.7% of total variation between the two samples, respectively, and followed by fallow (11.1%) and crops (3.7%).

On the other hand, bird assemblages differed significantly in the two samples (ANOSIM: r=0.05, p<0.001, based on 9999 permutations). The bird species that contributed the most to differences were those inhabiting wooded areas (49.8%), cropland (15.1%) and fallow (14.0%), with smaller differences due to species using non-intensive farmland (12.4%) and urban areas (8.8%). This differential increase in forest species contribution to the bird assemblage mostly involved colonization, while the changes in species associated to cropland and fallows mostly included extinctions (Figures 3 and 4).

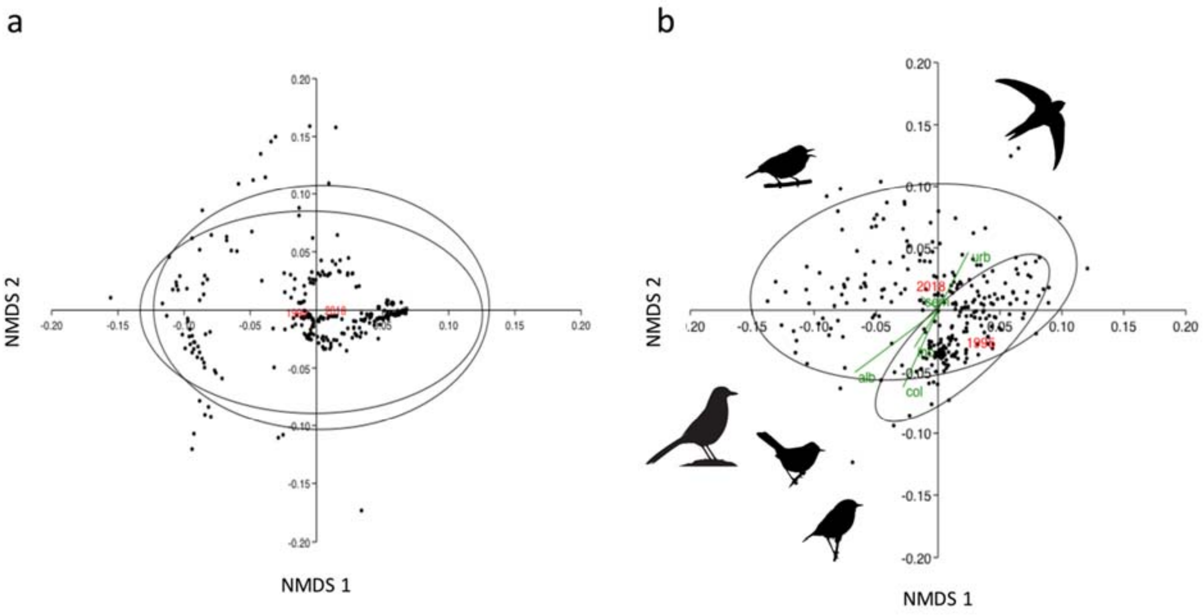

Figure 4. Non-metric multi-dimensional scaling of the land cover (a) and bird assemblage (b) in the urban area of Naples between 1994 and 2018. Ellipses represent 95% confidence intervals around each group's centroid; vectors indicate percent contribution and direction of birds from different habitat types in the difference between samples.

4.Discussion

The long-term perspective we offer on a bird community within one of the largest urban areas of southern Europe provides an unprecedented chance to test hypotheses on the relationship between biological assemblages and the urban environment. By assessing bird species distribution and land cover with consistent methods over a period of 28 years, we provide strong evidence that urban bird communities are more strongly influenced by fine-scale changes in land use rather than large scale ones, at least in highly dynamic and modified landscapes such as cities. The comparison of bird assemblages at different time intervals also allowed us to highlight the importance of long term studies to detect such changes in biological assemblages, as evidenced by the lack of significant differences at a 10-12 years' interval in our study system, and particularly if such studies aim at assessing trends in extinction and colonization rates within an area (Hedblom & Murgui, 2017; Fidino & Magle, 2017; Fercenc et al., 2018). The bird assemblage in Naples proved to be relatively stable only in terms of species richness, whereas diversity and species composition significantly changed over time, as also observed in other urban bird studies (Travis 2003; Fidino and Magle 2017). The urban environment acts as a filter to wildlife, allowing only taxa with specific traits to colonise and eventually thrive in such habitat type. Traits usually related to niche width (e.g. niche generalist) in terms of dietary and ecological niches, as well as behavioural, cognitive (e.g. behavioural plasticity) and reproductive traits (litter/brood size, numbers of breeding events per year, age at first reproduction) are all relevant characters, yet probably not the only ones, determining a species' success in urban environments, in birds as in mammals (Santini et al. 2018). In birds, habitat generalists, large breeding range sizes and naturally high population densities also seem to play a major role in influencing a species' tendency to colonise urban environments (Bonier et al. 2008; Ferenc et al. 2018; Moller 2009). As predictable according to the active filtering imposed to the bird community by urbanisation, the most common bird species in the assemblage remained relatively stable in our study system across 28 years, and mostly comprised taxa featuring traits associated with the ability to colonize and persist in urban areas. Conversely, rarer species were those most subjected to variations in persistence and distribution across the urban landscape. Among these, birds more closely associated to open and semi-open habitats such as agricultural land and fallows were those facing the most evident declines. Such negative trends are probably linked to the rapid replacement of these land use classes with urban areas due to the human demographic expansion, as well as to the development of wooded areas after land abandonment, as also evident at wider scales (Suarez-Seoane et al. 2002; Dyulgerova et al. 2015). Similarly, the range expansion of bird species associated to forests or using cavities as nesting structure seems to be a widespread phenomenon in urban areas (Malher & Lesaffre, 2007; Murgui, 2014; Evans et al. 2009); forest species were also those more likely to colonize the urban area of Naples throughout the study period, and less likely to go locally extinct. Specifically, only two forest species disappeared from the study area, i.e. the European cuckoo (*Cuculus canorus*) that was only considered as probably resident, and the common nuthatch (*Sitta europaea*), a species known to be very sensitive to habitat fragmentation (Battisti, 2004; Lorenzetti & Battisiti, 2006; Fraissinet, 2006; Fidino & Magle, 2017) and that may suffer indirect competition with the introduced rose-ringed parakeet (*Psittacula krameri*), also present in Naples since the '90s but restricted to very few green areas (Strubbe & Matthysen 2009; Fraissinet et al. 2000).

Although we did not include any correction for phylogeny into our analyses, our data suggest a family-biased trend in the colonisation process of the urban environment, with species from the

Columbidae and Corvidae families being among the most successful taxa to appear and rapidly spread in the study area. This is a well-known phenomenon (Bezzel 1985; Ferenc et al., 2018), probably related to a set of traits that characterize birds from these two families and their consequent ability to cope with the complex structure and dynamics of the urban environment. Our results highlight that urban bird assemblages are the product of the complex interactions between intrinsic factors such as species' biological or ecological traits, and the intrinsic structure of the urban environment, with its high rates of land use change at small scale (Hustè & Boulinier 2007). Other factors that may play a role in shaping the urban bird community are the pool of available species in the neighbouring natural areas, as well as landscape configuration within and around the city itself (i.e. habitat-specific connectivity). Our analysis highlighted species-specific relationships between birds and land use classes, also providing a methodological exercise potentially allowing to inform land use managers and urban planners, e.g. taking into account species' probability of presence as a consequence of the amount of a given habitat class. Such tools may only become reliable after long-term and standardized studies on urban bird assemblages, yet they represent a fundamental step towards an informed urban planning that takes into account biodiversity, not only for conserving species, but also for its positive effects on public perception and wellbeing associated to urban green spaces (Nutsford et al. 2013; Aronson et al. 2017).

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Appendix

Table A1. Generalized Linear Mixed Models testing the effect of percent land cover classes upon the probability of occupancy by bird species in the urban area of Naples, S Italy, as assessed in 1x1 km square cell grid (n=142) at three time intervals (1990-95, 2000-05, 2014-18). Abbreviations: Urb=urban areas; Nif=Non-intensive farmland; War=Wooded areas; Cro=Cropland; Fal=Fallow. Significance: *=p<0.05; **=p<0.01; ***=p<0.001. Species assignment to a specific land cover class is indicated by italics.

Species	Land cover class	Estimate ± standard error	p	Explained variance
Apus apus	*Urb*	*0.02±0.00*	**	*0.22*
	Nif	-0.08±0.03	n.s.	0.00
	War	0.02±0.01	n.s.	0.00
	Cro	-0.07±0.06	n.s.	0.01
	Fal	-0.07±0.03	*	0.06
Athene noctua	Urb	0.01±0.01	n.s.	0.00
	Nif	0.04±0.01	***	0.21
	War	0.02±0.01	n.s.	0.02
	Cro	0.04±0.03	n.s.	0.01
	Fal	0.01±0.02	n.s.	0.00
Buteo buteo	Urb	0.00±0.01	n.s.	0.00
	Nif	0.01±0.01	n.s.	0.00
	War	0.02±0.04	n.s.	0.09
	Cro	0.05±0.04	n.s.	0.11
	Fal	0.00±0.00	n.s.	0.00
Carduelis carduelis	Urb	0.00±0.00	n.s.	0.00
	Nif	*0.03±0.00*	**	*0.11*
	War	0.01±0.01	n.s.	0.01
	Cro	0.02±0.03	n.s.	0.02
	Fal	0.03±0.01	*	0.05
Carduelis chloris	Urb	0.01±0.01	n.s.	0.00
	Nif	0.02±0.01	n.s.	0.02
	War	0.04±0.01	*	0.05
	Cro	0.03±0.04	n.s.	0.02
	Fal	*0.05±0.02*	*	*0.11*
Certhia brachydactyla	Urb	-0.01±0.01	n.s.	0.05
	Nif	0.01±0.01	n.s.	0.01

	War	*0.06±0.01*	*****	*0.59*
	Cro	0.02±0.05	n.s.	0.00
	Fal	-0.01±0.02	n.s.	0.00
Cettia cettii	Urb	0.01±0.01	n.s.	0.00
	Nif	0.03±0.01	**	0.13
	War	0.02±0.01	*	0.07
	Cro	0.06±0.03	n.s.	0.05
	Fal	*0.05±0.01*	****	*0.22*
Cisticola juncidis	Urb	0.00±0.01	n.s.	0.00
	Nif	0.01±0.01	n.s.	0.00
	War	-0.04±0.01	*	0.09
	Cro	*0.14±0.04*	*****	*0.20*
	Fal	0.07±0.02	***	0.15
Cyanistes caeruleus	Urb	0.00±0.00	n.s.	0.00
	Nif	0.03±0.01	**	0.04
	War	*0.09±0.02*	*****	*0.31*
	Cro	-0.01±0.03	n.s.	0.02
	Fal	-0.01±0.02	n.s.	0.01
Columba livia var. *domestica*	Urb	0.01±0.01	n.s.	0.05
	Nif	-0.01±0.01	n.s.	0.01
	War	0.00±0.01	n.s.	0.00
	Cro	0.04±0.05	n.s.	0.00
	Fal	0.05±0.03	n.s.	0.00
Columba palumbus	Urb	0.01±0.01	n.s.	0.00
	Nif	-0.00±0.01	n.s.	0.00
	War	*0.03±0.01*	****	*0.39*
	Cro	0.01±0.03	n.s.	0.01
	Fal	-0.01±0.02	n.s.	0.00
Corvus corone	Urb	0.00±0.00	n.s.	0.00
	Nif	-0.02±0.01	n.s.	0.03
	War	*0.02±0.00*	***	*0.10*
	Cro	0.03±0.03	n.s.	0.01
	Fal	-0.01±0.02	n.s.	0.00
Corvus monedula	Urb	0.03±0.01	*	0.03
	Nif	0.01±0.01	n.s.	0.00
	War	*0.04±0.01*	*****	*0.19*
	Cro	0.02±0.03	n.s.	0.02
	Fal	-0.01±0.02	n.s.	0.00
Delichon urbica	*Urb*	*0.02±0.00*	***	*0.16*
	Nif	0.00±0.01	n.s.	0.04
	War	0.02±0.01	n.s.	0.01
	Cro	0.02±0.03	n.s.	0.08
	Fal	-0.04±0.03	n.s.	0.00
Erithacus rubecula	Urb	-0.01±0.00	n.s.	0.01
	Nif	0.03±0.01	**	0.03
	War	*0.05±0.01*	*****	*0.26*
	Cro	-0.10±0.08	n.s.	0.00
	Fal	0.03±0.02	n.s.	0.00
Falco tinnunculus	Urb	0.01±0.01	n.s.	0.00
	Nif	0.00±0.01	n.s.	0.00

	War	0.03±0.00	**	0.11
	Cro	0.03±0.02	n.s.	0.04
	Fal	*0.03±0.01*	*	*0.15*
Fringilla coelebs	Urb	0.00±0.00	n.s.	0.00
	Nif	0.04±0.01	***	0.05
	War	*0.07±0.01*	***	*0.17*
	Cro	-0.04±0.03	n.s.	0.03
	Fal	0.01±0.01	n.s.	0.00
Garrulus glandarius	Urb	-0.01±0.01	n.s.	0.00
	Nif	0.02±0.01	n.s.	0.05
	War	*0.04±0.01*	***	*0.38*
	Cro	0.06±0.03	n.s.	0.02
	Fal	0.02±0.02	n.s.	0.00
Hirundo rustica	Urb	0.02±0.01	*	0.01
	Nif	0.02±0.01	n.s.	0.01
	War	0.00±0.01	n.s.	0.00
	Cro	*0.07±0.02*	*	*0.19*
	Fal	-0.01±0.02	n.s.	0.01
Jynx torquilla	Urb	0.00±0.01	n.s.	0.00
	Nif	0.03±0.02	n.s.	0.08
	War	*0.04±0.01*	**	*0.38*
	Cro	-0.11±0.13	n.s.	0.01
	Fal	0.04±0.02	n.s.	0.00
Luscinia megarhyncos	Urb	-0.01±0.01	n.s.	0.01
	Nif	0.04±0.01	**	0.06
	War	-0.01±0.02	n.s.	0.00
	Cro	-0.04±0.07	n.s.	0.00
	Fal	*0.06±0.01*	***	*0.30*
Monticola solitarius	Urb	0.00±0.01	n.s.	0.00
	Nif	*0.09±0.04*	*	*0.13*
	War	-0.00±0.12	n.s.	0.00
	Cro	-0.14±0.12	n.s.	0.00
	Fal	-0.00±0.02	n.s.	0.00
Motacilla alba	Urb	0.00±0.01	n.s.	0.00
	Nif	0.01±0.01	n.s.	0.02
	War	-0.00±0.01	n.s.	0.02
	Cro	*0.10±0.03*	**	*0.21*
	Fal	0.03±0.02	n.s.	0.09
Muscicapa striata	Urb	0.01±0.01	n.s.	0.00
	Nif	0.01±0.01	n.s.	0.01
	War	*0.04±0.01*	***	*0.23*
	Cro	0.02±0.03	n.s.	0.00
	Fal	0.01±0.02	n.s.	0.00
Otus scops	Urb	0.00±0.00	n.s.	0.08
	Nif	*0.25±0.01*	*	*0.21*
	War	0.00±0.02	n.s.	0.00
	Cro	-0,09±0.08	n.s.	0.02
	Fal	-0.01±0.02	n.s.	0.05
Parus major	Urb	-0.00±0.00	n.s.	0.00
	Nif	0.05±0.01	***	0.07

	War	*0.10±0.02*	*****	*0.17*
	Cro	-0.04±0.03	n.s.	0.02
	Fal	0.02±0.02	n.s.	0.01
Passer italiae	Urb	0.01±0.01	n.s.	0.00
	Nif	*0.10±0.08*	***	*0.21*
	War	0.01±0.03	n.s.	0.00
	Cro	0.10±0.20	n.s.	0.07
	Fal	0.06±0.07	n.s.	0.02
Passer montanus	Urb	0.00±0.00	n.s.	0.00
	Nif	*0.03±0.00*	****	*0.06*
	War	-0.01±0.00	n.s.	0.00
	Cro	0.09±0.04	*	0.02
	Fal	0.02±0.01	n.s.	0.00
Pica pica	Urb	0.00±0.00	n.s.	0.00
	Nif	-0.01±0.01	n.s.	0.02
	War	0.01±0.00	n.s.	0.01
	Cro	0.05±0.03	n.s.	0.06
	Fal	0.00±0.01	n.s.	0.00
Picoides major	Urb	0.01±0.01	n.s.	0.00
	Nif	-0.01±0.01	n.s.	0.00
	War	*0.05±0.01*	*****	*0.61*
	Cro	0.04±0.05	n.s.	0.00
	Fal	0.05±0.03	n.s.	0.00
Regulus ignicapilla	Urb	0.01±0.01	n.s.	0.00
	Nif	-0.02±0.02	n.s.	0.01
	War	*0.05±0.01*	*****	*0.23*
	Cro	-0.25±0.17	n.s.	0.10
	Fal	-0.01±0.03	n.s.	0.02
Saxicola torquata	Urb	0.02±0.01	n.s.	0.00
	Nif	*0.04±0.01*	****	*0.34*
	War	0.02±0.02	n.s.	0.00
	Cro	0.04±0.04	n.s.	0.00
	Fal	0.03±0.02	n.s.	0.00
Serinus serinus	Urb	0.00±0.49	n.s.	0.00
	Nif	0.01±0.02	n.s.	0.04
	War	0.05±0.03	n.s.	0.05
	Cro	0.07±0.06	n.s.	0.00
	Fal	0.03±0.02	n.s.	0.01
Sitta europaea	Urb	-0.00±0.00	n.s.	0.01
	Nif	0.01±0.01	n.s.	0.00
	War	*0.05±0.01*	*****	*0.57*
	Cro	0.02±0.03	n.s.	0.00
	Fal	0.01±0.02	n.s.	0.01
Streptopelia decaocto	Urb	-0.01±0.00	n.s.	0.00
	Nif	-0.01±0.01	n.s.	0.00
	War	*0.02±0.01*	***	*0.19*
	Cro	0.04±0.03	n.s.	0.09
	Fal	-0.01±0.01	n.s.	0.01
Sturnus vulgaris	Urb	0.01±0.01	n.s.	0.01
	Nif	-0.14±0.07	n.s.	0.09

	War	-0.07±0.06	n.s.	0.03
	Cro	*0.11±0.05*	*	*0.20*
	Fal	0.01±0.03	n.s.	0.03
Sylvia atricapilla	Urb	0.01±0.01	n.s.	0.00
	Nif	0.02±0.01	n.s.	0.00
	War	*0.20±0.06*	***	*0.21*
	Cro	0.01±0.05	n.s.	0.00
	Fal	0.04±0.03	n.s.	0.02
Sylvia communis	Urb	0.00±0.01	n.s.	0.00
	Nif	0.03±0.02	n.s.	0.03
	War	0.01±0.02	n.s.	0.01
	Cro	0.01±0.05	n.s.	0.00
	Fal	0.03±0.02	n.s.	0.05
Sylvia melanocephala	Urb	0.01±0.01	n.s.	0.01
	Nif	0.03±0.00	**	0.11
	War	0.04±0.01	**	0.21
	Cro	0.00±0.03	n.s.	0.01
	Fal	*0.04±0.01*	***	*0.25*
Tachymarptis melba	*Urb*	*0.02±0.01*	*	*0.19*
	Nif	-0.16±0.10	n.s.	0.02
	War	-0.00±0.03	n.s.	0.00
	Cro	-0.02±0.08	n.s.	0.00
	Fal	-0.30±0.15	n.s.	0.01
Troglodytes troglodytes	Urb	-0.00±0.00	n.s.	0.00
	Nif	0.03±0.01	**	0.11
	War	*0.08±0.01*	***	*0.51*
	Cro	-0.00±0.03	n.s.	0.00
	Fal	0.02±0.01	n.s.	0.00
Turdus merula	Urb	0.00±0.01	n.s.	0.00
	Nif	0.00±0.01	n.s.	0.00
	War	*0.14±0.05*	**	*0.34*
	Cro	-0.02±0.04	n.s.	0.02
	Fal	0.05±0.03	n.s.	0.05
Tyto alba	Urb	-0.01±0.01	n.s.	0.00
	Nif	0.00±0.02	n.s.	0.04
	War	-0.01±0.02	n.s.	0.00
	Cro	0.03±0.04	n.s.	0.02
	Fal	0.00±0.03	n.s.	0.01
Upupa epops	Urb	0.01±0.01	n.s.	0.00
	Nif	0.03±0.02	n.s.	0.15
	War	-0.00±0.02	n.s.	0.02
	Cro	*0.08±0.03*	*	*0.22*
	Fal	-0.70±0.05	n.s.	0.09

Stonechat (*Saxicola rubicola*) , open environmental typical species in strong decreasing in Naples city

Chapter III

3 Large-scale commonness is the best predictor of bird species presence in European cities[1]

3.1 Abstract

Urban bird communities are homogenized across large spatial scales, suggesting that the urban environment acts as an environmental filter. We hypothesize that large scale commonness is a better predictor of urban affinity of birds than any particular species trait. We estimated the relative importance of taxonomy, reproductive, ecological and morphological traits, and commonness of individual bird species. We compiled data on i) breeding bird communities of 41 European cities from urban bird atlases, and ii) regional bird assemblages defined by nine grid cells of the Atlas of European Breeding Bird around each city, and quantified the urban affinity of each species by comparing its incidence in cities and in randomly drawn communities from respective regional assemblages. Conditional inference tree-based ran- dom forest analysis was utilized to assess the importance of individual predictors. A sign test was used to detect differ- ences between congeneric pairs of species with contrasting affinity to cities. Birds associated with woody habitats and those having altricial chicks had higher affinity for cities. Of the other reproductive traits, only clutch size showed an association with urban affinity. Different bird orders differed sig- nificantly in their urban affinity, exemplifying the homogenizing effect of cities. However, by far the most important factor associated with bird tolerance to the urban environment was species commonness, indicating that either the traits associat- ed with commonness, or population effects driven by commonness, are responsible for their presence in cities.

3.2 Introduction

Urbanization is a transformation of the environment with far reaching influences on biota, including loss of species and functional diversity, and altered ecological processes (McKinney

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[1] This paper is published as follows: *Ferenc M., Sedlacek O., Fuchs R., Horak D., Storchova L., Fraissinet M. & Storch D. 2018. Large scale commonness is the best predictor of bird species presence in the European cities. Urban Ecosystems 21: 369 – 377.*

2006; Aronson et al. 2014). Considerable evi- dence for the homogenizing effects of urbanization (McKinney 2006; La Sorte et al. 2007; Ferenc et al. 2014; Sol et al. 2014) suggests that urban areas act as environmental filters and probably systematically favour some species based on their traits, as shown, for example, by plants and invertebrates (Chocholoušková and Pyšek 2003; Bates et al. 2011; Duncan et al. 2011). Identification of biological traits associ- ated with persistence in cities has recently received consider- able attention in birds, but the outcomes are ambiguous (e.g. Bonier et al. 2007; Croci et al. 2008; Lepczyk et al. 2008; Hu and Cardoso 2009; Evans et al. 2011; Sol et al. 2014). Although many authors agree that off-ground nesting birds have an advantage in urban areas due to reduced nest predation pressure (Croci et al. 2008; Conole and Kirkpatrick 2011; Evans et al. 2011; Leveau 2013; Cardoso 2014; Jokimäki et al. 2014; Dale et al. 2015), results regarding other traits are often inconsistent. For example, some claim that birds that tolerate urban environments tend to be larger (Croci et al. 2008), while others show that they are of intermediate size (Conole and Kirkpatrick 2011) or that body size is unimportant (Sol et al. 2014). Conflicting results have also been reported in regard to fecundity (Kark et al. 2007; Croci et al. 2008; Evans et al. 2011; Sol et al. 2014). Kark et al. (2007) were unable to find a difference in reproductive mode between urban avoiders and exploiters. Findings by Evans et al. (2011) and Sol et al. (2014) do not support other studies showing that omnivores or granivores are advantaged in comparison to insectivores (Croci et al. 2008; Conole and Kirkpatrick 2011; Leveau 2013), and that migratory species are disadvantaged under urban conditions (Kark et al. 2007; Croci et al. 2008; Leveau 2013). However, some traits, such as bill morphology that reflects the utilized food type (Schoener 1965; Lederer 1975) or tarsus length which is linked to habitat preference, have not been explored in this respect so far.

If some traits determine the urban tolerance of birds and if these traits are phylogenetically conserved, related species should respond to urbanization in a similar manner (Sol et al. 2014). However, contrasting patterns have been revealed at different spatial scales. Phylogenetic signal is recognizable on a global scale, as some bird families (e.g. *Columbidae, Corvidae, Sturnidae*) tend to have urban representatives in most parts of the world (Sol et al. 2014). In contrast, at a smaller scale Evans et al. (2011) and Cardoso (2014) found that closely related species often responded to urbanization in different ways, and that sensitivity to urbanization was rather randomly distributed across phylogeny. This indicates that various urban-adapted birds may not necessarily share any common features that affect their tolerance to urbanization. Different species may benefit from the urban environment for very different reasons, and unique sets of traits characterizing different species may have unique selective advantages in the urban environment. If this is the case, it is reasonable to look for factors other than particular morphological or life- history traits that are potentially responsible for the success of species in cities. For these reasons, we hypothesize that generally wide- spread and common species are better able to colonize and persist in urban environments than less widespread and less common species. Our hypothesis thus accentuates the functional equivalence hypothesis emphasizing random community assembly where common species are advantaged, in contrast to the urbanisation tolerance hypothesis that assumes that specific traits determine urban success of individual species (Sol et al. 2014). The advantage of commonness may be due to, e.g., mass effects supporting colonization events (Shmida and Wilson a1n9d8/5o)r functioning metapopulation dynamics that reduce the extinction probability (Hanski 1998), or just due to a greater ability to adapt to any environment, including the urban one. We have previously shown that bird communities of European cities are more homogeneous than their species pools (Ferenc et al. 2014), i.e. they are more similar to each other than communities of adjacent landscapes. Here we ask whether these homogenized urban bird communities are dominated by species with certain biological traits or, alternatively, by species that are generally widespread and common. Towards this aim we attempted to assess the role of: i) ecological and morphological traits, ii) reproductive traits, iii) taxonomy, and iv) large-scale commonness in determining bird species affinities to European cities.

3.3 Materials and Methods

3.3.1 Study Area and Data

We collated data on the occurrence of breeding bird species in cities from 41 atlases of European cities (see Ferenc et al. 2014, and Supporting Information therein for details on the analysed cities). Since we were interested in the effects of urbanization on native European avifauna, non-native species were omitted from this study. Data on regional species assemblages were obtained from the 50 km × 50 km grid cells of the EBCC (European Bird Census Council) Atlas of European Breeding Birds (Hagemeijer and Blair 1997). Regional species assemblages for each city were defined as the set of bird species breeding in nine (if available) grid cells around each city, which has been previously shown to be a suitable definition of regional assemblages (Ferenc et al. 2014, and Supporting Information therein). The European breeding range size of each species was calculated as the total number of occupied grid cells of the EBCC atlas. Data on European population sizes and global breeding range sizes were retrieved from BirdLife International (2013). To quantify species affinities to urban areas in Europe, we developed a ranking scheme relating the incidence (i.e.

frequency of occurrence) of species in cities to their incidence in communities randomly composed from species of respective regional assemblages. It is not feasible to compute an exact average number of cities that would be occupied by a species if the communities were random subsets of the regional assemblages. Therefore we adopted a simulation approach: bird communities were simulated for each city by random resampling of regional assemblages, where the species richness of an each city community and its simulated counterpart was held equal. After repeating this procedure 1000 times we calculated an approximate average number of cities which would be occupied by each species if the com- munities were randomly assembled. The ratio between the incidence of each species in real cities and its average incidence across simulated city-assemblages gave the urban affinity score. This indicates if a species is present in cities more or less often than would be expected based on random sampling of species from the regional assemblages (score of 1.0 represents completely random occurrence in cities, i.e. a species occurs in cities with the same probability as anywhere else). Species ranks based on this scoring were further used as the response variable in subsequent analyses (see Online resource 1).

We also developed another ranking scheme: the urban affinity scores were multiplied by the frequency of species occurrences in regional assemblages across our study area, and species were ranked according to these new scores. The weighted ranking of urban affinity typically gives lower ranks to species occurring in, e.g., only one species pool and city, which thus have low sample size and the calculated urban affinity rank can be rather imprecise (see Online resource 1). However, because the analyses based on the weighted ranks gave similar results and brought no changes to the interpretations, we present these analyses only in Online resource 2.

We used two sets of traits to explain the affinity of species to urban areas: (i) Ecological and morphological traits: body mass, tarsus and bill length; habitat association (open, wet- land, woodland); migratory strategy (sedentary + short dis- tance migrants, long distance migrants); diet (carnivore, piscivore, granivore, herbivore, omnivore, feeding on insects or invertebrates); (ii) Reproductive traits: number of broods per year, clutch size, reproductive mode (altricial, semialtricial, precocial), nest type (arboreal, ground, hole). The data on ecology, morphology and reproduction were extracted from the interactive version of the Birds of the Western Palearctic handbook (Cramp 2006). The taxonomy (classification to orders) of bird species was based on IOC World Bird List v2.11 (Gill and Donsker 2012). We ran separate analyses (except for the pairwise comparisons of congeneric species) for all birds and for passerines only (order Passeriformes) to verify if the results concerning all birds also hold for a phylogenetically and morphologically homogeneous group representing a substantial part of urban bird communities (Ferenc et al. 2014). The taxonomy and reproductive mode were not included as predictors in the case of passerines.

3.3.2 Analyses

To rank explanatory variables according to their importance in predicting urban affinity of birds, we applied a modified random forest analysis (Breiman 2001; Cutler et al. 2007) utilizing conditional inference trees as base learners (Hothorn et al. 2006). Binary recursive partitioning-based conditional inference trees (hereafter CIT) are suitable for the description of complex datasets (Hothorn et al. 2006; Strobl et al. 2009). The CIT approach is similar to traditional regression trees; both are non-parametric methods insensitive to the frequency distribution of variables resulting in no need for their transformation (Jarošík 2011). Further common advantages are their ability to deal with nonlinear relationships and higher-order interactions, as well as their robustness to the collinearity of predictors (Breiman et al. 1984; De'ath and Fabricius 2000; Jarošík 2011). In the first step, the basic dataset was divided (if possible) into two groups which were as homogeneous as possible, based on a specific value (cut-off point) of a selected predictor (spliting criterion). Next, each new sub-group (node) was recursively split by a cut-off point of any splitting criterion, including the one(s) used in previous step(s). The lower nodes contained increasingly homogeneous groups of response variables, with terminal nodes being the most homogeneous in respect to the explanatory variables (Strobl et al. 2009). However, traditional regression trees tend to be biased in split criterion selection towards variables with a high number of potential cut-off points or many missing values (Hothorn et al. 2006; Strobl et al. 2007), and rely on cross validation and pruning of overgrown trees while selecting the resulting tree (Breiman et al. 1984; Jarošík 2011). In contrast, CITs utilize permutation tests: i) to perform unbiased variable selection at each split and ii) to apply early stopping (instead of pruning) to prevent overfitting (Hothorn et al. 2006). The drawbacks of both methods are their strong dependence on the learning sample and thus their sensitivity to small changes in the data (Strobl et al. 2009). To prevent potentially incorrect conclusions derived from a single tree we utilized the random forest approach (Breiman 2001; Cutler et al. 2007) based on CITs (hereafter CIT-RF). Individual CITs were fitted to 500 different subsamples (without replacement) of the original dataset instead of bootstrapped samples (Hothorn et al. 2006; Strobl et al. 2007). Data not appearing in the subsamples are called out- of-bag (OOB) data and were used for assessing the predictive ability of each tree (Breiman 2001; Cutler et al. 2007). The importance of predictors can be calculated by randomly permuting the values of a predictor for the OOB data and calculating the reduction in model accuracy in comparison to the original OOB data (Cutler et al. 2007; Strobl et al.

2007; Jarošík 2011). We calculated the importance of each variable under the conditional permutation scheme ensuring a lower preference for correlated predictors (Strobl et al. 2008). Finally, the raw predictor importance values were converted into percentages. This analysis was separately run for i) reproductive traits and ii) ecological and morphological traits. Although the commonness of species is a qualitatively different variable, analogical CIT-RF analyses were run with European range size included in the explanatory datasets to directly compare its importance with the importance of other traits. All analyses were conducted in R software (R Core Team 2013) using the `party´ package (Hothorn et al. 2013).

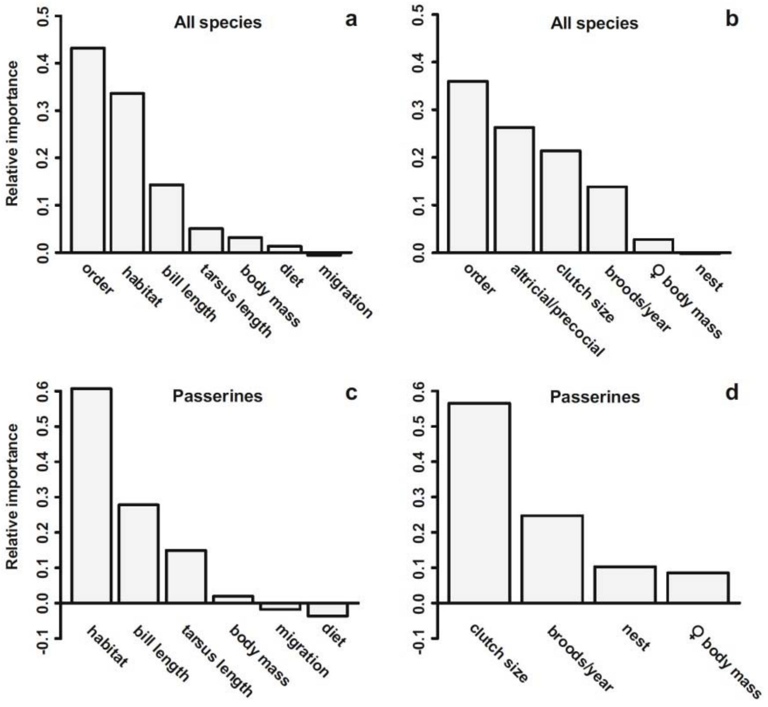


Fig. 3.3.1 - The relative importance of ecological and morphological (a, c) and reproductive (b, d) traits in determining bird species urban affinity ranks based on CIT-RF analyses. Analyses were run for all species (a, b) and passerines (c, d) separately

To test for the differences in continuous traits between bird species with high versus low urban affinity we con- ducted pairwise comparisons of closely related species (using absolute scores instead of ranks). Pairs of conge- ners were selected such that one of them had a higher and the second a lower score than the median urban affinity score. In some cases it was possible to select multiple pairs of congeners; therefore the contrasted pairs were repeatedly randomly chosen (100 randomizations) to ensure a non-arbitrary comparison of species pairs. Subsequently we tested the differences between congeners using the sign test in the R package `BSDA´ (Arnholt 2012), and the proportion of significant tests at the Bonferroni corrected significance level was recorded. The shortcoming of this analysis is that only a restricted number of pairs of bird species could be used (52 pairs), because some species could not be contrasted in respect to the median score (as all had either higher or lower scores than the median) or because some genera were represented by a single species only. The non-parametric Kruskal-Wallis test of analysis of variance was used to test for the differences in ranks of urban affinity among groups of birds characterized by specific categorical variables. Correlations between continuous traits and the ranks of urban affinity of birds were evaluated using non parametric Spearman's rank correlation coefficients. Correlations of global breeding range size, European breeding range size and European population size (all, except for European breeding range size, were ln-transformed) with the rank of urban affinity were analysed using parametric Pearson's correlation coefficients. In all tests, the significance threshold was set to 0.05.

3.4 Results

3.4.1 Ecological and morphological traits

Of the ecological and morphological traits tested, the CIT-RF revealed taxonomic status (order identity) to be the most important predictor of bird urban affinity ranks (Fig. 3.3.1a). Relatively species-poor orders of *Columbiformes, Apodiformes, Bucerotiformes and Caprimulgiformes* had the highest ranks, followed by more numerous *Piciformes* and *Passeriformes* (Fig. 3.4.1). Orders with the lowest urban affinity included *Otidiformes, Procelariiformes, Galliformes, Suliformes* and birds of prey. The habitat association was the second most important predictor of bird urban affinity (Fig. 3.3.1a).

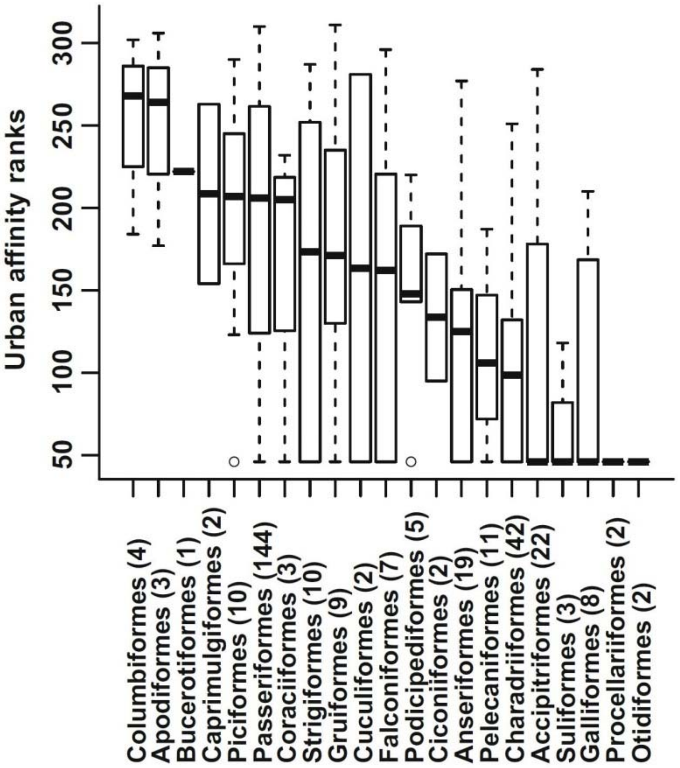


Fig. 3.4.1 - Boxplots of urban affinity ranks of individual bird orders. Boxplots show the median, interquartile ranges, whiskers extending to 1.5 times the interquartile range, and outliers. Numbers in brackets indicate the number of species in each order

Species associated with woodlands showed a higher affinity to cities than species of open habitats and wetlands (Kruskal-Wallis test: H = 38.18, *p* = 0.0000, Fig. 3.4.2). Simple correlations of morphological traits with urban affinity ranks showed that larger birds tend to avoid urban areas (Table 3.4.1). All the other predictors included in the CIT-RF analysis had relatively low importance.

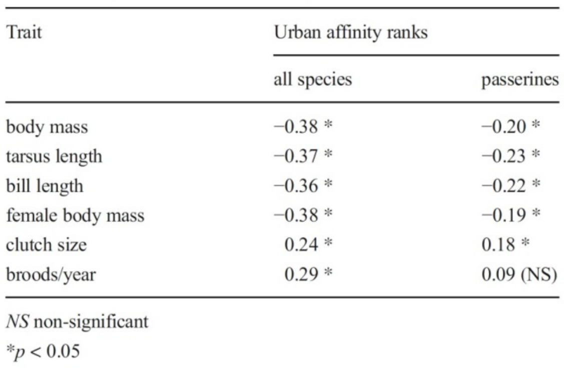
The CIT-RF analysis based on passerines confirmed the importance of habitat associations in explaining bird urban affinities (Fig. 1c). Passerines associated with woodland habitats tended to have the highest urban affinity ranks (Kruskal- Wallis test: H = 10.66, *p* = 0.0049; see Online resource 3). Body size variables had a much smaller effect on urban affinity in passerines, although bill length showed some importance (Fig. 3.3.1c). Simple correlations suggest that passerines with longer bills tend to avoid urban areas. The sign test showed no significant differences between congeners with different urban affinity scores.

Fig. 3.4.2 - The differences in urban affinity ranks between bird species associated with open, wetland, and woodland habitats. Boxplots show the median, interquartile ranges, and whiskers extending to 1.5 times the interquartile range

3.4.2 Reproductive traits

The taxonomic status (order identity) and reproductive mode were among the most important predictors of urban affinity ranks for all bird species (Fig. 1b). Altricial species showed significantly higher affinity to cities compared with the other two strategies (Kruskal-Wallis test: H = 46.71, p = 0.0000; see Online resource 4). Simple correlations suggested that birds with larger clutch sizes and two or more broods per year have higher probability to occur in cities (Table 3.4.1). For passerines, the clutch size and number of broods per year were also identified as relatively important predictors of urban affinity (Fig. 3.4.1 d) A significant positive correlation was detected for clutch size, but not for the number of broods per year (Table 3.4.1). None of the reproductive traits was significantly different among congeneric species with different urban affinity scores.

Table 3.4.1- Spearman's rank correlation coefficients between urban affinity ranks and continuous traits of bird species (all species and passerines only) considered in this study



Trait	Urban affinity ranks	
	all species	passerines
body mass	−0.38 *	−0.20 *
tarsus length	−0.37 *	−0.23 *
bill length	−0.36 *	−0.22 *
female body mass	−0.38 *	−0.19 *
clutch size	0.24 *	0.18 *
broods/year	0.29 *	0.09 (NS)

NS non-significant

$^*p < 0.05$

3.4.3 Geographic distribution and populations size

When European range size was included among the predictors in the CIT-RF analysis, all the other ecological and morphological or reproductive traits had a negligible relative importance (Fig. 3.4.3). We found a strong positive correlation between the urban affinity of birds and their European and global range sizes and population abundance. The European range size was the strongest predictor of urban affinity, followed by population size in Europe and global geographic range (Table 3.4.2; Fig. 3.4.4). Similar results hold for passerines and woodland and non-woodland species separately (Table 3.4.2, Fig. 3.4.4, Online resource 5).

The analyses comparing congeners showed that species with higher affinity to urban areas have significantly larger European and global breeding range sizes, as well as European population sizes. All 100 out of 100 sign tests at the Bonferroni corrected significance level were significant, except for the global geographic range size with only 42 significant tests out of 100.

Table 3.4.2 - Pearson's correlation coefficients between urban affinity ranks and the range size and population abundance of bird species (all species and passerines only)

Variable	Urban affinity ranks	
	all species	passerines
geographic range (World)	0.20 *	0.32 *
range size (Europe)	0.73 *	0.67 *
population size (Europe)	0.62 *	0.60 *

$^*p < 0.05$

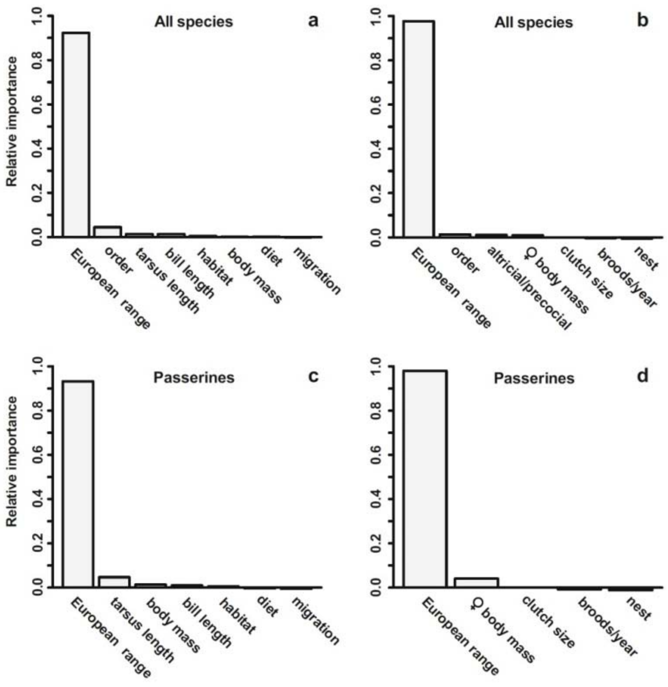


Fig. 3.4.3 - The relative importance of the European range size of species, ecological and morphological (a, c) and reproductive (b, d) traits in determining bird species urban affinity ranks based on CIT-RF analyses. Analyses were run for all species (a, b) and passerines (c, d) separately

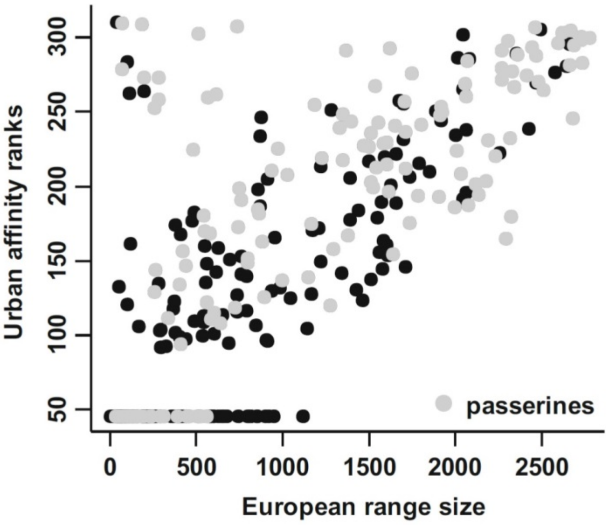


Fig. 3.4.4 - The relationship between European range size and urban affinity ranks of all species (r = 0.73; p = 0.0000) with passerines shaded in grey

3.5 Discussion

Our hypothesis stating that commonness is the major factor determining bird presence in European cities has been largely supported. The worldwide and European range sizes of species and their European population sizes are by far the best predictors of their affinity to cities. Additionally, woodland birds seem to be advantaged in urban areas in contrast to birds of open or wetland habitats, and species with different urbanization tolerance have been shown to be non-randomly distributed among taxonomic groups (orders). Importantly, other ecological, morphological and reproductive traits received relatively weak support throughout our analyses. We therefore suggest that factors determining the overall bird rarity or commonness (either expressed in terms of their geographic distribution or population size) also determine their ability to pass the urban environmental filter and to persist under urban conditions (see also Dale et al. 2015; Aronson et al. 2016). This could be due to the fact species able to survive in large numbers in the human dominated landscape of Europe are preadapted for life in cities (Cardoso 2014). In other words, the traits responsible for widespread distribution and/or high abundances may simultaneously directly and positively influence bird survival in cities. An obvious example is wider environmental tolerance (Bonier et al. 2007; Croci et al. 2007). Alternatively, traits may act indirectly by enabling some species to be widespread and/or abundant, so that their potential for successful colonization of cities is higher (Symonds and Johnson 2006). We propose several non- exclusive mechanisms translating rarity/commonness into the urban affinity of birds: First, widespread species tend to be locally abundant (Gaston et al. 2000), so that common species are more resistant to local extinction due to their high local abundances (Purvis et al. 2000). A favourable metapopulation dynamics can further enhance their persistence in ur- ban areas via the rescue effect (Brown and Kodric-Brown 1977; Hanski 1998). Rare species, on the other hand, may be absent from a city, because their abundances are too low in the surrounding habitats to enable invasion and establishment in the city. Second, the currently expanding urban environment represents an extreme form of environmental alteration, which requires new adaptive responses of species (Partecke and Gwinner 2007; Ibáñez-Álamo and Soler 2010). Initial urban populations of less common species might be too small for the emergence and fixation of beneficial mutations (Kimura 1983), resulting in their lower adaptability and persistence. Third, abundant species may have a higher probability of early colonization of cities. Initial urban populations of less common species might be too small for the emergence and fixation of beneficial mutations (Kimura 1983), resulting in their lower adaptability and persistence. Third, abundant species may have a higher probability of early colonization of cities. Birds inhabiting cities for longer periods of time become gradually adapted to this type of environment and attain higher within-city densities (Møller et al. 2012) resulting in higher urban population stability and persistence.

In our analyses, birds primarily associated with woodlands showed higher urban affinity in comparison to birds of open and aquatic habitats (cf. Dale et al. 2015). The absence of many non-woodland species from cities might be caused by the lower availability and/or inferior quality of non-woodland habitats in urbanized areas (Croci et al. 2008). Alternatively, the dramatic decline in the population sizes of many farmland and wetland birds in Europe, due to intensifying agriculture and aquaculture (Voříšek et al. 2010; Wetlands International 2010), potentially contributes to the low incidence of these species in urbanized areas.

A clear taxonomical pattern in the urban affinity of birds was also revealed, consistently with Sol et al. (2014) who found a non-random phylogenetic pattern in bird sensitivity to urbanization. This indicates that specific combinations of traits represented by individual taxa determine the tolerance of species to urbanized areas (Kark et al. 2007; Croci et al. 2008; Leveau 2013). Our findings therefore explicitly demonstrate the non-random filtering of the assemblages entering cities (Bonier et al. 2007; Croci et al. 2008; Evans et al. 2011; Leveau 2013; Meffert and Dziock 2013), which may ultimately lead to the taxonomically and functionally homogenized avifauna of cities (McKinney

2006; Luck and Smallbone 2011; Ferenc et al. 2014; Sol et al. 2014). In contrast, no phylogenetic signal in urban tolerance was detected by Cardoso (2014) and Evans et al. (2011). However, the former study considered only passerines, and the latter disregarded species completely avoiding the urban environment.

Morphological traits as predictors of bird urban affinity received relatively weak support in our CIT-RF analyses, being surpassed by the influence of taxonomy or habitat choice. Moreover, the comparison of congeneric species did not reveal any morphological differences between species with high versus low affinity to cities. Similarly, reproductive traits did not appear to strongly affect the urban affinity of birds. Although our data suggest that birds with altricial nestlings are more frequently found in human settlements, it is not clear whether this strategy genuinely enhances their ability to persist in cities or if it is just a reflection of the taxonomic bias in urban tolerance. Clutch size was moderately correlated with urban affinity, and there was some indication of its importance in passerines, but the results are not clear-cut. Moreover, the lack of differences in reproductive traits between congeners indicated that closely related species with similar reproductive traits can have differing levels of urban tolerance. Higher potential reproductive output (i.e. larger clutches and/or the presence of replacement clutches) has been suggested to enable birds to be better urban exploiters (Chace and Walsh 2006; Croci et al. 2008), but this was not supported by other studies (Kark et al. 2007; Evans et al. 2011) and the role of reproductive traits remains ambiguous.

The strong support for large-scale commonness as a determinant of the response of an individual bird species to urbanization and the much weaker support for other traits (see also Lepczyk et al. 2008) might seem to contradict the conclusions of Sol et al. (2014). These authors suggest that mainly adaptive differences between species cause their differential responses to urbanization. However, our finding that closely related species with differing affinities to cities systematically differed only in their large-scale commonness indicates that non-adaptive effects may be quite important. Unfortunately, it is not possible to tease apart the independent influence of adaptive vs. non-adaptive features based on our analyses. Some unmeasured traits responsible for the large-scale commonness of a species might be similar or identical to the traits providing an advantage in urbanized areas (e.g. wider environmental tolerance; Bonier et al. 2007; Croci et al. 2007) and thus the urbanization tolerance hypothesis (Sol et al. 2014) would still apply in such a case. Our results only show that species commonness is a much stronger predictor of urban affinity than any of the traits we explored.

3.6 Conclusions

Our key finding is that the large-scale homogenization of urban bird communities (Ferenc et al. 2014) is caused by an urban environmental filter preventing rarer birds to invade and/or persist in cities. Traits associated with the ability of bird species to colonize urban environments are thus indistinguishable from traits associated with their geographic distribution and abundance. The relevant traits might act either directly by influencing the urban adaptability and broader environmental tolerance of species, or indirectly by influencing bird population dynamics.

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Peregrine Falcon (*Falco peregrinus*) in the Naples “Centro Direzionale”

Chapter IV

4 Geographic trends in range sizes explain patterns in bird responses to urbanization in Europe[2]

4.1 ABSTRACT

The probability of occurrence of bird species in towns/cities increases with their range sizes and Rapoport´s rule states that range sizes increase with latitude. To test the hypothesis that the increasing number of bird species persisting in cities at higher latitudes of Europe is linked to their larger range sizes we compiled data on bird communities of: a) 41 urban bird atlases; b) 37 city core zones from published sources; c) regions of nine grid cells of the EBCC Atlas of European Breeding Birds around each city. We tested whether the proportion of species from particular regional bird assemblages entering cities (i.e. proportional richness) was related to the geographical position, mean range size of regional avifaunas, proportion of vegetated areas and city habitat heterogeneity. The mean range sizes of observed and randomly selected urban avifaunas were contrasted. The proportional richness of urban avifaunas was positively related to the geographic position and mean range size of birds in regional assemblages. The evidence favoured range sizes if considering European range sizes or latitudinal extents, but was limited for global range sizes. Randomizations tended to show larger range sizes for real avifaunas than in the randomly selected ones. For urban core zones, the results were less clear-cut with some evidence only in favour of European range sizes. No role of vegetation or habitat heterogeneity was found. In conclusion, while vegetation availability or heterogeneity did not show any effects, spatial position and range sizes of birds in regional assemblages seemed to influence the proportional richness of cities and their core zones. Factors correlated with spatial position (e.g. climate) might increase the attractivity of particular cities to birds. However, the effects of range sizes indicated that urbanization possibly has more negative impacts on the avifauna in regions occupied by less widespread species.

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4.2 INTRODUCTION

Urbanization is a process substantially transforming the original environment and its impacts are perceivable at all spatial scales (Seto et al., 2012; Grimm et al., 2015). Urbanized areas act as systematic environmental filters leading to the most significant ecological impact of urbanization, documented especially on avian communities, called biotic homogenization (Jokimäki & Kaisanlahti-Jokimäki, 2003; Clergeau et al., 2006; McKinney, 2006; Luck & Smallbone, 2011; Ferenc et al., 2014a; Sol et al., 2014; Morelli et al., 2016; Leveau et al., 2017). It is exemplified by higher avifaunal similarity among corresponding parts of the urbanization gradient of different cities than among different parts of the urbanization gradient within particular cities (Clergeau et al., 2001, 2006), or by higher similarity among cities than among their species pools (Luck & Smallbone, 2011; Ferenc et al.,2014a). The latter example represents a large scale homogenizing effect and is apparently stronger at higher latitudes of Europe. However, this pattern is paradoxically a consequence of proportionally more species from particular regional assemblages entering European cities at higher latitudes (Ferenc et al., 2014a). The question therefore is: i) what makes these cities of higher latitudes more attractive to birds or ii) what makes birds of higher latitudes more predisposed to persist in cities in comparison to south European species?

External factors might include differences in climate along the latitudinal gradient. Positive effects of urbanization due to greater food availability and predictability throughout the year (Jokimäki & Suhonen, 1993) might be more pronounced at higher latitudes, thereby making urban areas more attractive to birds in this region. Alternatively, differences in history of urbanization at different latitudes of Europe (Jokimäki et al., 2016b) might be reflected in the degree of original habitat alteration in urban areas and thus in their suitability and attractivity to birds (Evans et al., 2009).

On the other hand, particular bird traits also show latitudinal patterns (Cardillo, 2002) and many studies pointed to ecological, behavioural or life-history traits that enable or prevent them to persist in cities (e.g. Bonier et al., 2007; Kark et al., 2007; Croci et al., 2008; Hu & Cardoso, 2009; Conole & Kirkpatrick, 2011; Evans et al., 2011; Díaz et al., 2013; Sol, 2013; Leveau, 2013; Meffert & Dziock, 2013; Cardoso, 2014; Sol et al., 2014; Jokimäki et al., 2016a). Recently it has been shown that commonness (i.e. geographic range or total population size) is a strong predictor of affinity of European bird species to towns and cities (Ferenc et al., 2018). Birds´ range sizes vary in space (Orme et al., 2006) and species of higher latitudes tend to have larger ranges (Rapoport, 1982; Stevens, 1989; Cardillo, 2002). This so called Rapoport’s rule holds quite well at least at the northern hemisphere (Rohde, 1996; Gaston et al., 1998), and it might affect bird responses to urbanization across latitudes in Europe.

The aim of this study was to reveal whether external factors make European cities of higher latitudes more attractive to birds or whether the assumed latitudinal trend in range sizes drives the observed patterns in proportions of species of regional assemblages occurring in cities (proportional richness hereafter). In order to do so, using two dataset on European breeding avifauna of entire cities and city core zones, we tested the following hypotheses: i) Geographical position (especially latitude) has an independent effect on the proportional richness of birds, which might indicate a role of various factors correlated with geographical gradients, such as climate or urbanization history. ii) Patterns of habitat availability and/or habitat heterogeneity within cities of Europe drive the spatially structured differences in proportional richness of their avifaunas. iii) The proportional richness of cities is linked to the Rapoport´s rule predicting a latitudinal increase in range sizes of species of particular regional bird assemblages.

4.3 MATERIALS AND METHODS

4.3.1 Data collection

Data on breeding bird communities in a) 41 European cities (Fig. 4.3.1) were extracted from urban bird atlases (atlas dataset); and b) 37 European city core zones (core zone dataset) were obtained from a dataset published by Jokimäki et al. (2016a, 2016b). St. Petersburg was excluded from this dataset due to incomplete data on its regional bird assemblage (see definition below). All non-natives were removed from the atlas dataset. The core zone dataset contained two non-native species (*Psittacula krameri*, *Alopochen aegyptiaca*), but their inclusion is highly unlikely to substantially influence the results. The atlas dataset contained species with possible, probable and confirmed breeding status to minimize inconsistencies due to different assignment of species into these categories across particular atlases. On the other hand, the core zone dataset contained only species with probable or confirmed breeding status (Jokimäki et al., 2016a). The composition and richness of regional bird assemblages was retrieved from the EBCC Atlas of European Breeding Birds (Hagemeijer & Blair,1997). The regional avifauna of each city was defined as the bird assemblage of nine - if available - atlas grid cells (cell size: 50 by 50km) surrounding each city. The central square included the city or most of the city area and the remaining eight squares surrounded the central square). The proportion of species from each regional assemblage occupying the corresponding city or city core zone was used as the response variable (see also Table S1 in Appendix S1 and Jokimäki et al., 2016a, 2016b).

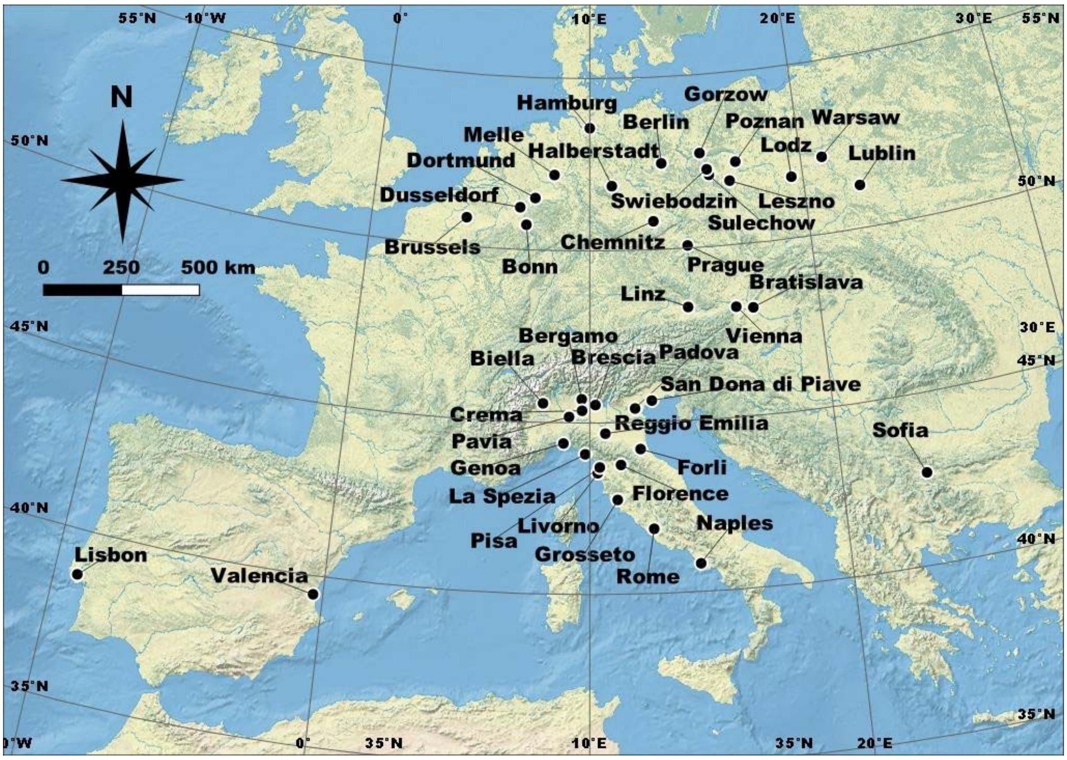

Fig. 4.3.1 - Geographical distribution of the European cities in the "atlas dataset".

In the atlas dataset, the city area was either directly extracted from a particular atlas or calculated as the product of the number of its grid cells and the size of a single cell. While relying on particular authors´ expertise in city border definition, only atlases with comparable methodology were utilized (see e.g. Dinetti *et al.*, 1995). For example the atlas of Paris was excluded as it covered only the inner-city and not the entire city area (Malher et al., 2010). In the core zone dataset only the innermost historical city centers were included and their area recorded (Jokimäki et al., 2016a). Furthermore, the proportion of i) "vegetated", ii) "built-up" and iii) "other" land cover was recorded for each city (Ferenc et al., 2014a; Jokimäki et al., 2016a) and the Shannon index (Shannon´s H) reflecting their habitat heterogeneity was calculated thereof (Shannon, 1948). Finally, we recorded the European breeding range size (i.e. the number of occupied grid cells of the EBCC atlas); latitudinal range extent in Europe (i.e. the number grid cells between the northernmost and southernmost location of occurrence in the EBCC atlas); and the global range sizw (BirdLife International, 2018) for each bird species.

4.3.2 Data analysis

Subsequently, we constructed generalized linear models (with quasibinomial error distribution due to overdispersion and the logit-link function) to relate the proportional richness to the predictor set. In the case of the atlas dataset, both city area and time span of data collection of urban bird atlases influence the recorded species richness and consequently the proportional richness of each city. Therefore, these two variables were included in all models to account for their effects. Similarly, for the core zone dataset all models contained the study plot area. These models were regarded as the baseline models.

A full model containing latitude, longitude, proportion of vegetated area, Shannon index of habitat heterogeneity and either mean European range size (MERS hereafter) or mean latitudinal extent in Europe or mean global range size of birds in particular regional assemblages was created. Thereafter, we constructed the minimal adequate models by backward elimination of predictors while retaining only those causing a significant increase in residual deviance when removed. Finally, we tested for the presence of residual autocorrelation in the resulting models using Moran´s I.

To test whether bird species present in cities are non-randomly selected from regional assemblages according to their range sizes, we performed a randomization test for the atlas dataset. We randomly selected from each regional bird assemblage the same number of species as was actually present in the corresponding city. This procedure was repeated 10,000-times and we recorded the proportion of MERS, mean latitudinal extent and meang global range size of randomized communities that were smaller than those observed in cities.

All analyses were carried out in R (R Core team, 2017) using the following packages: “tidyverse” (Wickham, 2017) for data manipulation and plotting, “fields” (Nynchka et al., 2017) for geographical distance calculation and “ape” (Paradis et al., 2004) for Moran’s calculation.

4.4 RESULTS

The first premise of this study was an increasing proportional richness of urban avifauna with increasing latitude: although such a latitudinal trend of proportional richness was detected in the case of the atlas dataset (R2= 0.34, $p < 0.001$; Fig. 4.4.1a) this relationship did not hold for the birds of city core zones (R2 = 0.00, $p = 0.36$; Fig. 4.4.1b). The second premise of increasing geographical ranges or extent of species with latitude held in dependence on the utilized measure: the MERS of urban birds showed an increasing latitudinal trend in the case of the atlas dataset

(R2= 0.42, $p < 0.001$; Fig. S3.1a in Appendix S3), while this pattern was reversed at higher European latitudes as the core- zone dataset revealed (R2= 0.43, $p < 0.001$; Fig. S3.1b in Appendix S3). The pattern was very similar for the mean latitudinal extent of species in regional assemblages (atlas dataset: R2= 0.88,$_{\mathrm{ADJ}}$ p < 0.001; Fig. S3.2a in Appendix$_{\mathrm{ADJ}}$ S3;$_{\mathrm{ADJ}}$ core zone dataset: R2= 0.83, p < 0.001; Fig. S3.2b in Appendix S3). In contrast, the mean global range sizes of species in regional assemblages increased with latitude consistently in both datasets (atlas dataset: R2= 0.34, p < 0.001; Fig. S3.3a in Appendix S3; core zone dataset: R2= 0.81, p < 0.001; Fig. S3.3b in Appendix S3). Our results lend support to our central hypothesis: the model for all species in the atlas dataset revealed (Table 4.4.1; Table S2.1 in Appendix S2) that spatial position represented by latitude 169(0.033 ± 0.014, t = 2.32, p < 0.05) and longitude (0.02 ± 0.008, t = 2.56, p < 0.05) as well as MERS of species in the regional assemblages (0.0017 ± 0.0007, t = 2.48, p < 0.05; Fig. 4.4.1a) had a significant positive effects on the proportional richness of birds in European cities. On the other hand, only MERS (0.0017 ± 0.0008, t = 2.04, p < 0.05; Fig. 4.1b) had an influence on the proportional richness of birds in core zones of European cities. If the model included the mean latitudinal extent of species in regional assemblages, it was significant (0.18 ± 0.03, t = 5.87, p < 0.001) along with longitude (0.02 ± 0.007, t = 2.43, p < 0.05) in the case of the atlas dataset (Table 4.1; Table S2.1 in Appendix S2; Fig. S3.4a in Appendix S3;). However, no model could be built upon our predictors in the case of the core zone dataset (Table 4.1; Table S2.1 in Appendix S2; Fig. S3.4b in Appendix S3;). Similarly, when including the mean global range sizes of birds in regional assemblages, only latitude had a significant positive effect (0.07 ± 0.01, t = 2.32, p < 0.001) on the proportional richness of birds in the case of the atlas dataset (Table 4.1; Table S2.1 in Appendix S2; Fig. S3.4a in Appendix S3) and no model could be constructed for the core zone dataset (Table 4.1; Table S2.1 in Appendix S2; Fig. S3.4b in Appendix S3). None of the resulting models showed any residual autocorrelation at the 5% significance level. Due to the strong correlation between latitude and mean global range size of birds in regional assemblages, we conducted additional post-hoc analyses. They showed that if the initial full model for the atlas dataset included either of the range size descriptors (MERS, mean latitudinal extent or mean global range size), but not latitude, the range size descriptor was always retained in the final model. The same was true for latitude, when leaving out the range size descriptors from the initial full model. On the other hand, no such effect was revealed in the case of the core zone dataset (results not shown). The role of range sizes of birds in determining their proportional richness in cities was further emphasized by the randomization test using the atlas dataset, as the observed urban assemblages had significantly higher MERS than would be expected by chance at the 5% significance level. Similar results were obtained by analysing latitudinal extents of birds, although there were two cities where the avifauna had mean latitudinal extents as if they were randomly assembled from the regional assemblages. However, with using global range sizes only seven out of

41 cities had avifaunas with higher mean global range sizes than would be expected by chance, while the rest of the cities had mean global range sizes as if the urban avifaunas were assembled randomly.

4.3 DISCUSSION

Urban avifaunas of European cities and city centers seem to be influenced by both their spatial position at the continent per se as well as by the range sizes of species representing the potential species pool in particular regions. Although we detected no influence of vegetation availability or habitat heterogeneity on the proportional richness of urban avifaunas, the effect of geographic location might indicate an influence of unmeasured external factors that are linked to spatial position, such as climatic conditions. On the other hand, features of bird species also influence the richness of urban avifaunas, with relatively more species entering cities in regions occupied by more widespread species.

Cities located towards the north-east of continental Europe (based on the atlas dataset excluding Nordic countries) with relatively continental climate tend to host higher proportions of birds from regional assemblages. Such patterns might be related to the climate moderating effects of urban areas due to the heat-island phenomenon (Erz, 1966; Arnfield, 2003) resulting in greater availability of food resources during the critical periods and their better predictability throughout the year (Jokimäki & Suhonen, 1993). Such effects might enhance the habituation and establishment of populations of particular species within cities (Møller et al., 2014; Tryjanowski et al., 2015). However, the pattern of increasing proportional richness of birds did not hold for the northernmost city core zones (based on the core zone dataset) with the harshest climate, which makes this interpretation dubious. Furthermore, urbanization at higher latitudes of Europe is a relatively recent phenomenon (Jokimäki et al., 2016b) and the responses of bird species might be delayed. That means that avifaunas of different regions have had unequal amount of time to respond to urbanization by adaptation or by going extinct (Essl et al., 2015). The city-age effect can go both ways: i) younger cities can have higher proportional richness as some species (e.g. forest specialists) will still go extinct in the future or ii) younger cities can have lower proportional richness due to less time for adaptation. Again, the fact that the Nordic cities are the youngest, but do not have the highest proportional richness, disfavours the first explanation, but not the second one. Nevertheless, the discrepancy among the two datasets showing an increase in proportional richness with latitude in the case of entire cities, but no such pattern in the case of city core zones, could partly arise because of their different scales of observation. The possibly positive eff effects of urbanization observable at the scale of entire cities (such as the heat-island phenomenon (Erz, 1966; Arnfield, 2003)) might be overridden by its negative impacts in the most urbanized city core zones (Clergeau et al., 2006).

Surprisingly, neither the proportion of vegetation cover nor the habitat heterogeneity played any role in determining the proportional richness of urban avifaunas. This points to the fact that although such factors are of prominent importance in determining bird species richness of urban communities at smaller scales (Evans et al., 2009; Ferenc et al., 2014b, 2016), their importance is not necessarily detectable at large spatial scales (MacGregor-Fors et al., 2010). Instead, spatial patterns of species range sizes seem to be a substantial part of the explanation although our initial hypothesis of the link between Rapoport´s rule (Rapoport, 1982; Stevens, 1989) and bird responses to urbanization does not hold. We found some evidence in favour of the influence of MERS and mean latitudinal extent, for which Rapoport´s rule does not hold, but limited evidence in favour of global range sizes, for which the Rapoport´s rule does hold. On the one hand the link between MERS and proportional richness of cities persisted regardless of whether we focused on entire cities or on city core zones including Nordic cities. On the other hand, the latitudinal extent showed an influence only in the case of entire cities, while not in the case of city core zones. And finally, global range sizes showed no influence in either case. As shown by the randomization tests for MERS or mean latitudinal extent, the urban assemblages systematically filter widespread species, but the evidence is much weaker if considering mean global range sizes. However, even under this scenario, urban avifaunas never showed smaller mean range sizes than the randomly selected assemblages. The discrepant results for MERS or mean latitudinal extent versus mean global range sizes might have two very different reasons: i) the strong correlation between latitude and global range size might lead to a situation when these two variables mask each-others influence on proportional richness. Indeed, the post-hoc analyses revealed such an effect and the mean global range size was always present in the final model if latitude was omitted from the initial full model and vice-versa), at least in the case of the atlas dataset. ii) The shape of ranges of many northern bird species might be elongated in the west-east direction across Eurasia, but they might be relatively narrow in the north-south direction. Such species might in some ways experience less variable environments than species having global ranges more elongated in the north-south direction, but this would require further testing, which is out of scope of this study.

Indeed, the ability of species to cope with variable environmental conditions and its link to their geographic range sizes might be behind the observed relationships (Stevens, 1989). Widespread birds might have high environmental tolerance possibly due to their ecological, behavioural or physiological flexibility, which can be beneficial under urban conditions (Bonier et al., 2007). Traits of widespread species thus might be directly beneficial for survival in cities or indirectly connected to urbanization by influencing the commonness of species (Ferenc et al., 2018). Less common birds might simply be extinction-prone in urban areas or less likely to colonize them (Sol et al., 2014). Alternatively, species with larger ranges tend to be more numerous locally (Brown, 1995, 2013; Gaston et al., 2000; Gaston & Blackburn, 2008), which can enhance their occurrence in cities due to mass effects (Shmida & Wilson, 1985), favourable metapopulation dynamics (Brown & Kodric-Brown, 1977; Hanski, 1995) or by lowering their local extinction risk (Purvis et al., 2000).

Evidence on the influence of species´ commonness on their occurrence in towns and cities is accumulating (Dale et al., 2015; Jokimäki et al., 2016b; Ferenc et al., 2018). Our findings have some important implications for assessing the ecological impacts of urbanization on birds in different geographic locations. Geographical regions showing apparent compositional uniformity of urban bird assemblages are not necessarily the ones most negatively impacted by urbanization (Ferenc et al., 2014a). Based on our results we have to take into account geographic location, the geographically changing patterns of range sizes of birds and the enhanced potential of more widespread species to persist in towns and cities. In a similar vein, the urban filter is not necessarily more permeable in regions showing greater distinctness of urban avifaunas. The growth of cities thus poses a greater threat to the bird assemblages in regions composed of less widespread species having a weaker ability to respond to the challenges of urbanization.

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Table 4.1 - Simplified representation of models constructed by backward elimination showing significant predictors (at the 5% significance level) of proportional richness of urban avifaunas in European cities (atlas dataset) and city core zones (core zones dataset) and their effect ("+" indicates a positive effect). Shaded columns represent different measures of mean geographic range sizes of birds in particular regional assemblages (see Methods for definitions)

Dataset	**mean European range size**	**latitude**	**longitude**	**Shannon´s H**	**% vegetation cover**
atlas	**+**	**+**	**+**		
core zones	**+**				
Dataset	**mean latitudinal extent**	**latitude**	**longitude**	**Shannon´s H**	**% vegetation cover**
atlas	**+**		**+**		
core zones					
Dataset	**mean global range size**	**latitude**	**longitude**	**Shannon´s H**	**% vegetation cover**
atlas	**+**				
core zones					

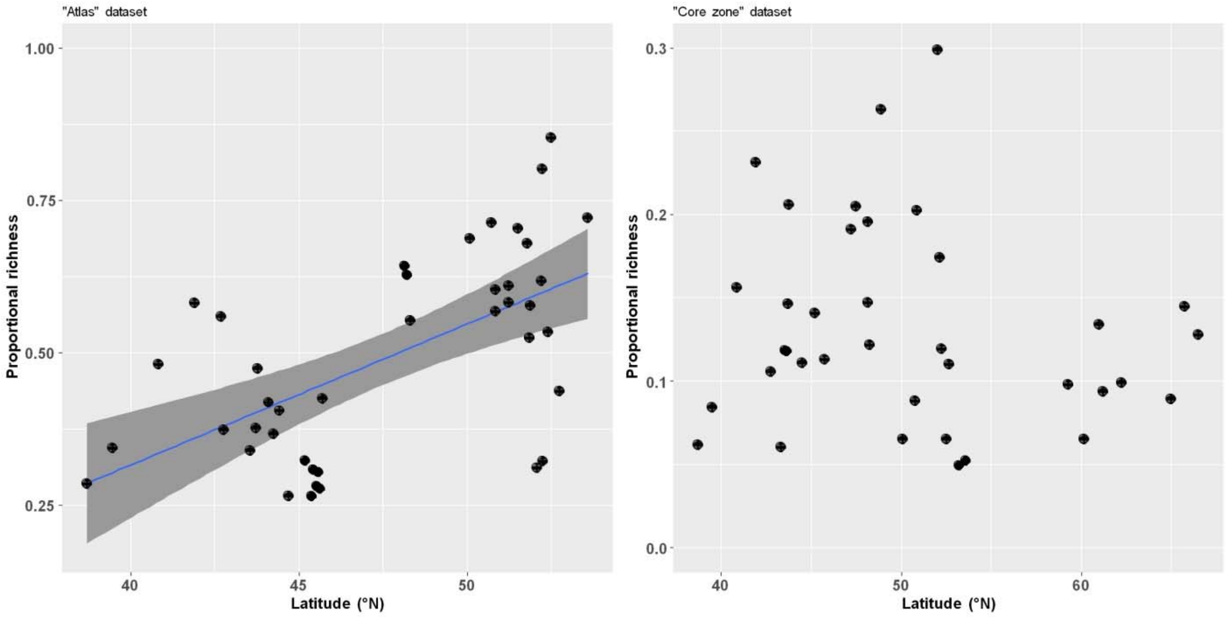

Fig. 4.4.1 - Relationship between the proportional richness of European urban avifaunas (a) atlas dataset (R^2_{ADJ}= 0.34, $p < 0.001$); b) core zone dataset (R^2_{ADJ} = -0.004, $p = 0.36$)) plotted against latitude. The trend was fitted by OLS regression, shaded areas depict 95% confidence intervals.

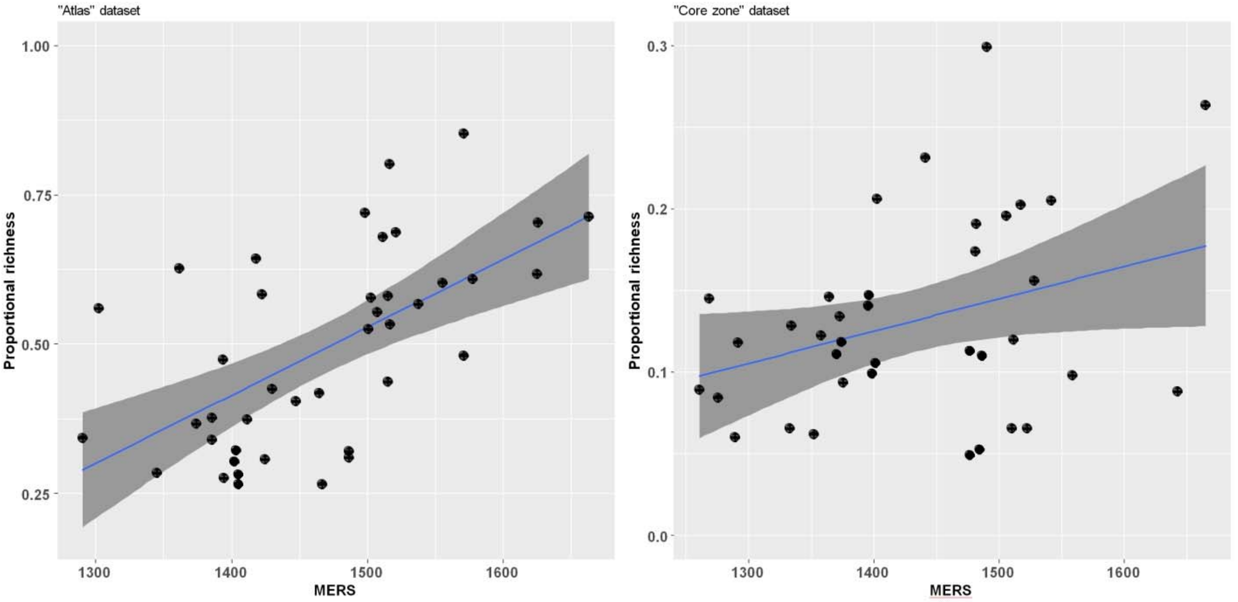

Fig. 4..4.2 - Relationship between the proportional richness of European urban avifaunas (a) atlas dataset (R^2_{ADJ} = 0.34, $p < 0.001$); b) core zone dataset (R^2_{ADJ} =0.08, $p < 0.001$)) plotted against the mean European range size (MERS; based on the number of occupied EBCC atlas grid cells) of species of particular regional assemblages. The trend was fitted by OLS regression, shaded areas depict 95% confidence intervals.

Geographic trends in range sizes relate to patterns in bird responses to urbanization in Europe

Michal Ferenc, Ondřej Sedláček, Roman Fuchs, Maurizio Fraissinet, and David Storch

Table S1: Data on 41 European cities and their breeding bird communities

City	Longitude	Latitude	Area (km^2)	Time span of city-atlas data collection (yrs)	% vegetation cover	Shannon index habitat heterogeneity	Species richness - city	Species richness - region	Proportional species richness - city	Mean range size of birds in regional assemblages		
										Europe (# of EBCC squares)	Latitudinal extent (# of cells between southern- & northernmost EBCC squares)	Global ($x 10^6 km^2$)
Bergamo	9,666667	45,7	39	4	0,299	1,036	77	181	0,425	1430,2	28,6	16,188
Berlin	13,4	52,5	900	12	0,168	0,889	145	170	0,853	1571,1	30,8	16,194
Biella	8,066667	45,566667	30,7	1	0,412	0,955	57	188	0,303	1402	28,3	16,046
Bonn	7,099722	50,733889	141	12	0,2	1,039	109	153	0,712	1663,2	31,4	16,743
Bratislava	17,109722	48,143889	376	13	0,336	1,084	126	196	0,643	1418,3	29,3	16,095
Brescia	10,233333	45,533333	15,5	5	0,222	0,869	52	185	0,281	1404,7	28,3	16,224
Brussels	4,35	50,85	162,38	12	0,107	0,527	97	171	0,567	1537,4	30,7	16,375
Crema	9,683333	45,366667	22,75	2	0,079	0,881	49	185	0,265	1404,7	28,3	16,224
Dortmund	7,46666	51,51666	310	6	0,165	1,007	114	162	0,704	1625,4	31,2	16,847
Dusseldorf	6,78333	51,23333	217	6	0,176	0,952	103	169	0,609	1577,3	30,9	16,705
Florence	11,25	43,783333	102,4	2	0,321	1,069	82	173	0,474	1393,9	28,0	15,937
Forli	12,05	44,233333	44,25	3	0,013	0,728	63	172	0,366	1374,3	27,8	16,291
Genoa	8,932778	44,411111	57	5	0,27	1,075	68	168	0,405	1447,4	27,9	16,902
Gorzow	15,25	52,733333	21,1	1	0,236	0,787	79	181	0,436	1515,2	30,5	16,019
Grosseto	11,1	42,766667	17,6	2	0,029	0,788	53	142	0,373	1411,5	27,0	16,511
Halberstadt	11,03333	51,88333	41	5	0,156	0,989	108	187	0,578	1502,2	30,7	16,027
Hamburg	10	53,56666	763	4	0,245	1,007	134	186	0,72	1498,5	30,7	16,008
Chemnitz	12,91666	50,83333	188	4	0,169	1,03	106	176	0,602	1555,1	30,7	16,453
La Spezia	9,816667	44,1	20	2	0,262	1,086	66	158	0,418	1464,4	28,3	15,192
Leszno	16,56666	51,86666	37	4	0,082	0,802	96	183	0,525	1500,9	30,2	15,750
Linz	14,28333	48,3	102,75	2	0,267	1,044	99	179	0,553	1507,4	30,0	16,414
Lisbon	-9,135278	38,706944	84	15	0,073	0,899	41	144	0,285	1345	26,3	17,600
Livorno	10,316667	43,55	38,1	2	0	0,695	56	165	0,339	1385,4	27,3	14,895
Lodz	19,466667	51,783333	236	9	0,225	1,038	123	181	0,68	1511,6	30,3	15,917
Lublin	22,570278	51,248056	147,55	10	0,176	1,024	116	199	0,583	1422,4	29,8	15,422
Melle	8,33333	52,2	311	3	0,242	0,696	100	162	0,617	1624,8	31,3	16,682
Naples	14,25	40,833333	117,2	5	0,148	1,008	62	129	0,481	1571	28,3	16,445
Padova	11,866667	45,416667	35	4	0	0,655	55	179	0,307	1424,8	28,9	16,004
Pavia	9,15	45,183333	33,5	2	0,168	1,029	60	186	0,323	1403,2	28,3	16,408
Pisa	10,4	43,716667	27,4	2	0,019	0,753	62	165	0,376	1385,4	27,3	14,895
Poznan	16,916667	52,4	270	20	0,195	1,027	96	180	0,533	1516,6	30,4	16,051
Prague	14,41666	50,08333	519,4	5	0,171	1,032	125	182	0,687	1520,7	30,5	16,019
Reggio Emilia	10,633333	44,7	21,5	2	0,004	0,536	44	166	0,265	1466,8	28,6	16,627
Rome	12,5	41,9	385	5	0,131	0,956	83	143	0,58	1514,7	28,4	16,123
San Dona di Piave	12,566667	45,633333	12,1	2	0	0,661	53	192	0,276	1394,1	28,6	15,575
Sofia	23,33333	42,7	181	10	0,126	0,854	117	209	0,56	1302,3	27,4	15,806
Sulechow	15,616667	52,083333	4,16	2	0,01	0,583	58	187	0,31	1486,4	30,3	15,802
Swiebodzin	15,533333	52,25	5	2	0,199	0,909	60	187	0,321	1486,4	30,3	15,802
Valencia	-0,36666	39,46666	96,53	16	0,05	0,844	59	172	0,343	1290,3	26,0	16,482
Vienna	16,373056	48,208333	414,89	10	0,207	0,947	131	209	0,627	1361,6	29,0	15,655
Warsaw	21,010833	52,23	545,3	20	0,227	1,036	145	181	0,801	1516,2	30,2	15,612

Geographic trends in range sizes relate to patterns in bird responses to urbanization in Europe

Michal Ferenc, Ondřej Sedláček, Roman Fuchs, Maurizio Fraissinet, and David Storch

Table S2: Minimal adequate models constructed by backward elimination showing significant predictors (at the 5% significance level) of proportional richness of urban avifaunas in European cities (atlas dataset) and city core zones (core zones dataset): parameter values and the associated standard errors and test statistics are shown. Variables shaded in grey were included in all models to account for their effects (data collection time span was not considered for the core zones dataset).

Dataset	mean European range size (MERS)	latitude	longitude	Shannon´s H	% vegetation cover	area	data collection time span
atlas	0.0017 ± 0.0007, t = 2.48, p < 0.05	0.033 ± 0.014, t = 2.32, p < 0.05	0.02 ± 0.008, t = 2.56, p < 0.05			0.82 ± 0.09, t = 9.35, p < 0.001	0.004 ± 0.01, t = 0.41, p = 0.61
core zones	0.0017 ± 0.0008, t = 2.04, p < 0.05					-0.001 ± 0.14, t = -0.09, p = 0.93	

Dataset	mean latitudinal extent	latitude	longitude	Shannon´s H	% vegetation cover	area	data collection time span
atlas	0.18 ± 0.03, t = 5.87, p < 0.001		0.02 ± 0.007, t = 2.43, p < 0.05			0.82 ± 0.09, t = 9.34, p < 0.001	0.004 ± 0.01, t = 0.37, p = 0.71
core zones							

Dataset	mean global range size	latitude	longitude	Shannon´s H	% vegetation cover	area	data collection time span
atlas		0.07 ± 0.01, t = 2.32, p < 0.001				0.90 ± 0.09, t = 9.98, p < 0.001	-0.06 ± 0.01, t = -0.33, p = 0.75
core zones							

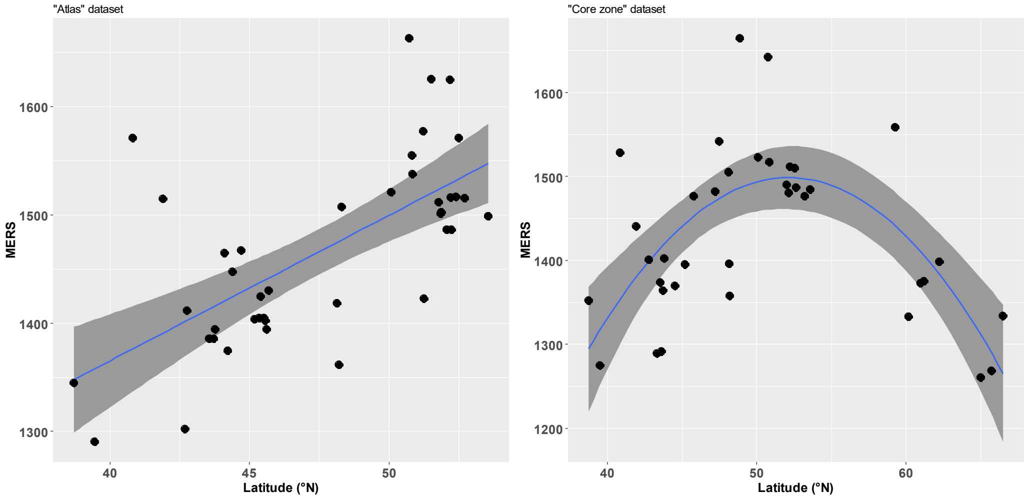

Fig. S3.1 **-** Mean European range size (MERS) of birds in regional assemblages (defined as nine EBCC grid cells around each city) plotted against latitude **a)** atlas dataset ($R^2_{ADJ} = 0.42$, $p < 0.001$); **b)** core zone dataset ($R^2_{ADJ} = 0.43$, $p < 0.001$). Trends fitted by OLS regression, shaded area depicts 95% confidence intervals.

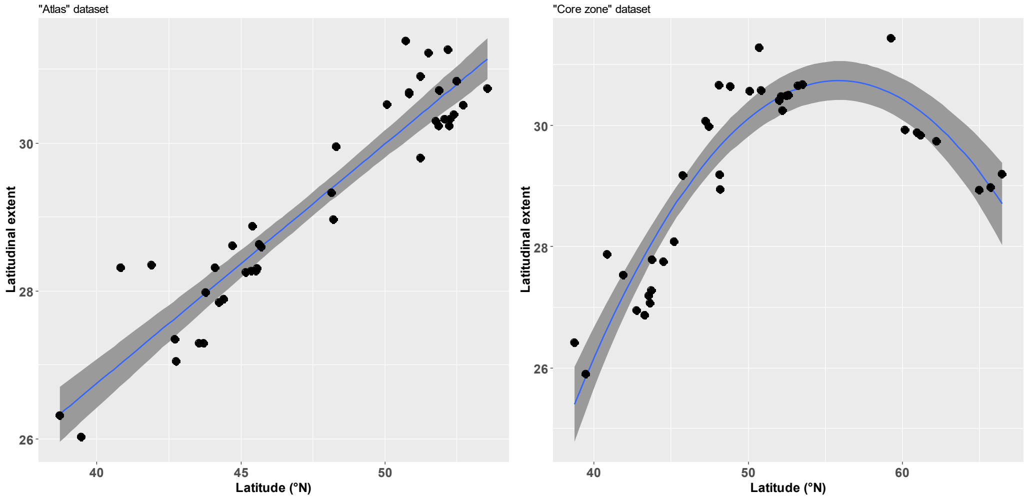

Fig. S3.2 **-** Mean latitudinal extent (defined as the number of cells between the southernmost and northernmost occupied EBCC atlas grid cells) of birds in regional assemblages (defined as nine EBCC grid cells around each city) plotted against latitude: **a)** atlas dataset ($R^2_{ADJ} = 0.88$, $p < 0.001$); **b)** core zone dataset ($R^2_{ADJ} = 0.83$, $p < 0.001$). Trends fitted by OLS regression, shaded area depicts 95% confidence intervals.

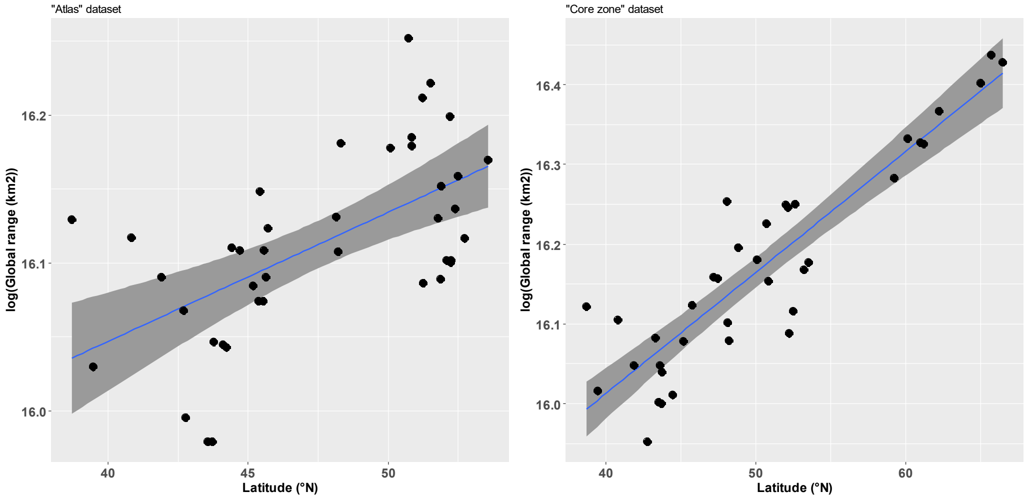

Fig. S3.3 - Mean global range size of birds in regional assemblages (defined as nine EBCC grid cells around each city) plotted against latitude: **a)** atlas dataset ($R^2_{ADJ} = 0.34$, $p < 0.001$); **b)** core zone dataset ($R^2_{ADJ} = 0.81$, $p < 0.001$). Trends fitted by OLS regression, shaded area depicts 95% confidence intervals.

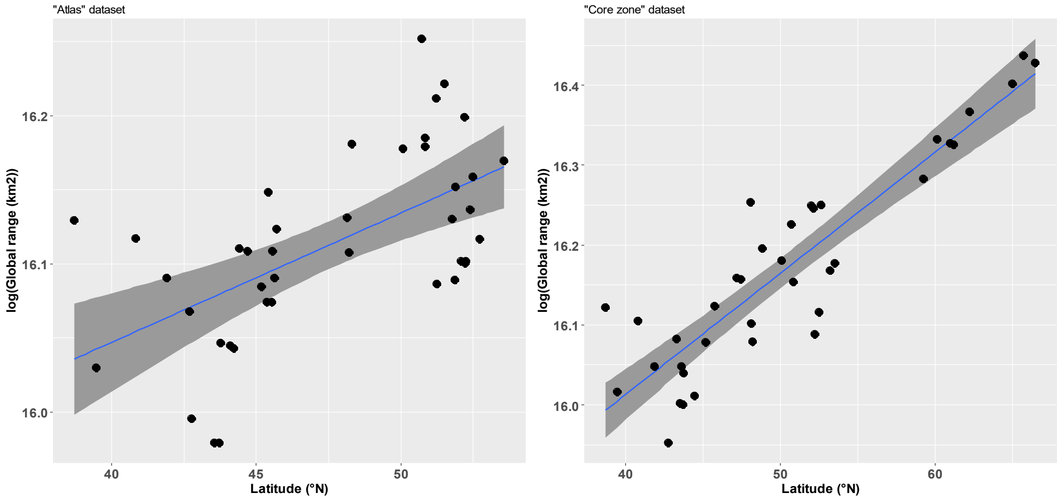

Fig. S3.4 - Relationship between the proportional richness of European urban avifaunas (a) atlas dataset ($R^2_{ADJ} = 0.42$, $p < 0.001$); b) core zone dataset ($R^2_{ADJ} = -0.016$, $p = 0.51$)) plotted against the mean latitudinal extent (defined as the number of cells between the southernmost and northernmost occupied EBCC atlas grid cells) of species of particular regional assemblages. The trend was fitted by OLS regression, shaded area depicts 95% confidence intervals.

Jay (*Garrulus glandarius),* forest species in strong increasing in European cities

Chapter V

5. The colonization of the Western Yellow-legged Gull (*Larus michahellis*) in an Italian city: evolution and management of the phenomenon[3]

5.1 Abstract

The phenomenon of gull urbanization affects several species of *Laridae* and many cities around the world. In European cities, there are several species that have colonized urban areas. The Yellow-legged Gull *Larus michahellis* and the Herring Gull *L.argentatus*, however, there are species that have been more successful in this process. The phenomenon, which started early twentieth century, has grown rapidly and gull urban populations have now reached high numbers and densities. This to the point that it not only represents an interesting ecological phenomenon to study but a management problem for municipal administrations. Gulls represent a discomfort to city dwellers, especially during their breeding season. In Italy, the phenomenon has been monitored for some years, giving way to interesting studies in Trieste (North-East) of population control methods. To inform municipal administrators and the public about the biology of the species, their urban colonization phenomenom, and subsequent strategies for the prevention of this phenomenent a consortium of stakeholders that included municipalities and ornithologists, joined to create a technical- informative document.

Keywords Urban Yellow-legged Gulls in Italy - Monitoring - Management - Population control.

5.2 Introduction

Following the strong numerical increase of the Yellow-legged Gull *Larus michahellis* and Herring Gulls *L.argentatus* in Europe, starting in the early 20th century and mainly due to an increase in food sources of anthropic origin, there has been a progressive saturation of usual breeding sites (Burger & Lesser 1980; Burger and Gochfeld 1983; Vincent 1987). As a result, there has been an expansion

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[3] This paper is in press as follows: *Benussi E. & Fraissinet M. In stampa. The colonization of the Western Yellow-legged Gull (*Larus michahellis*) in an Italian city: evolution and management of the phenomenon. In "Problematic Wildlife Vol. 2. New conservation and management challenges in the human-wildlife interactions" (F.Angeli & L.R.Rossi eds). Springer International ed., Cham (Switzerland).*

of the reproductive area and the colonization of new environments, including urban ones (Cadiou 1997). In this environment the gulls have found very favourable breeding conditions. In fact, man-made constructions offer a wealth of suitable sites, which greatly reduce intraspecific predation and eliminate the problem of terrestrial predators. Moreover, urban waste is a source of additional food, which in contrast to natural food sources, is abundant and constant. In many cities, populations of Yellow-legged Gull have now reached such dimensions as to cause considerable disturbances to the public. Mainly through noise, dirt and aggressiveness of territorial pairs towards people wishing to use terraces and balconies, which the gulls have chosen as a reproductive site. This has prompted various local administrations to study attempts to reduce, or at least contain, gull populations. These interventions are often not completely successful, due in part to the limited involvement of ornithologists. In Italy, where the phenomenon has been monitored for years, there has been a close cooperation between the National Association of Italian Municipalities (ANCI) and ornithological experts that study the phenomena (Fraissinet 2015) to provide information to local administrators and the public.

5.3 The colonization of cities in Europe

The phenomenon of gulls colonizing urban areas is worldwide. In US cities, the Glaucous-winged Gull *Larus glaucescens* and the the Ring-billed Gull *L.delawarensis* are commonly found, while in Argentinian and New Zealand cities, it is the Kelp Gull *L.dominicanus*. In European cities, the Yellow-legged Gull *L.michahellis,* the Herring Gull *L.argentatus,* the Caspian Gull *L.cachinnans* and Lesser Black-backed Gull *L.fuscus* predominate, and, to a lesser extent, the Great Black-backed Gull *L.marinus,* Common Gull *L.canus* and Black-headed Gull *Chroicocephalus ridibundus* do. Lastly, in Barcelona city’s port, in the years 2013, 2016 and 2017, a colony of Audouin’s Gull *Ichthyaetus audouinii* nested. This particular event presumably followed the decline in nest numbers that occurred in a large nesting colony found in the Ebro delta (Anton *et al*. 2017).

In Europe, the colonization of urban centres began in the 20th century with its first instances reported in the United Kingdom as far back as the 1940s. The phenomenon manifested itself in other European countries, in a clear and widespread manner, in the 1970s. In France, urban breeding first began in 1970 and in Spain in 1975 at the Barcelona Zoo. In Italy, the first case of urban nesting dates back to 1971, with a pair that successfully breeded on an artificial rock in the Zoo of Roma. It is interesting to note that both first urban nesting in London, which dates back to

1966, and in Barcelona that dates back to 1975, took place in similar settings (Zoological Gardens) (Garcia Petit *et al*., 1986, Oliver 1997).

France, due to its geographical location, hosts four similar size breeding species of gulls: the Herring, Yellow-legged, Lesser Black-backed and the Great Black-backed Gulls. It would be interesting to examine whether interspecific differences have been observed in the colonization of urban environments by gulls. Cadiou (1997) reports data for the 1980, the first half of the 1990s links the increase of French urban breeding with the increase in nesting pairs along the coasts after the beginning of the twentieth century. At the end of the 1980s, in the French cities, 1,800 pairs of Herring Gull were breeding in 25 colonies and by 1996 the number of urban pairs increased to 7,200 – 7,500 (perhaps 8,000 – 10,000) in about forty colonies, of which seven numbered more than 500 pairs. The average annual growth rate of urban populations was 20%. For a good number of cities, the first pairs nested on the roofs of fish markets. The urban population of the Lesser Black-backed Gull in France in the late 1980s, was about 80 pairs distributed throughout 6 colonies. This represented 0.3% of the French breeding population. In the second half of the 1990s, the population increased to 700 - 800 pairs spread across twenty colonies. This represented about 3% of French nesting pairs, with the species establishing themselves in cities that had pre-existing Herring Gull colonies with spontaneous colonization by the species being unknown. The first hybrid Lesser Black-backed Gull x Herring Gull pair was recorded in Rennes. In France, as in Great Britain, there are no urban colonies where the Lesser Black-backed Gull predominates. At most colonies of the Lesser Black-backed Gull represents micro-colonies within larger colonies of gulls. At the end of the 1980s, there were dozens of urban Great Black-backed Gull breeding pairs in France, this represented 0.4% of the French population and by the mid-1990s, 26 - 29 pairs were nesting in urban areas at a dozen sites representing about 1% of the French breeding population for that species. The Great Black-backed Gulls settle on large roofs already colonized by other gulls, placing their nests in a central position in the colony, as normally happens in non-urban sites. The status of the Yellow-legged Gull in France in the 1990s is interesting. Urban nesting on the Atlantic coast was rare, but it was much more widespread on the Mediterranean coast where even though that by the second half of the 1990s urban pairs numbered only 70 - 100, but this figure was probably underestimated although urban breeding was an exceptional event in Corsica. In addition, it was the the only species to nest in the French mainland cities of Paris, Toulouse and Saint-Girons, in the 1990s (Cadiou 1997). In 2010, Paris had 3 species of nesting gulls: Yellow-legged with 3 pairs and the Lesser Black-backed with 3 pairs, both located within Herring Gull colonies totalling 50 pairs. In 2006 a mixed pair of Yellow-legged Gull x Herring Gull was also observed (Malher 2010).

The Herring Gull, in particular, increased sharply in numbers in the United Kingdom and Ireland from the late 1930s onwards and this led to the need for them to search for new nesting sites which were found on the roofs of abandoned buildings. From the 1970s, the growth of urban populations was quite rapid, passing from 1,250 pairs in 1970 to 3,000 in 1976, and on to 16,900 pairs in 1994. By the end of the 20th century, 8% of Herring Gulls and 4% of Lesser Black-backed Gulls in Great Britain and Ireland were urban breeders. In South Shields, in north-eastern England, the number of urban pairs of Herring Gull increased from 5 in 1963 to 209 in 1976, with an annual increase of more than 20%. Compared to natural breeding populations on the British and Irish coasts it has been observed that urban Herring Gulls suffer less juvenile mortality. The reason for this is attributed to lower rates of cannibalism. It has also been ascertained that urban gulls originate from natural colonies, where the young, once mature and unable to find a suitable site for reproduction, moves to urban centres (Monaghan 1979).

In Bulgaria, the colonization of urban centres on the Black Sea coast is even older, dating back to the late nineteenth century. By 1992, two thirds of the population of Yellow-legged Gull was urban with 95.5% of the breeding pairs found in the coastal towns of the Black Sea and 4.5% nesting in the inland cities (Nankinov 1992). At the time of Nankinov's article, however, there was no separation between the two species - *Larus michahellis* and *L.cachinnans* - so it is not known which species started the colonization process. At the moment both species are present in coastal urban areas of the Black Sea. In the Romanian city of Constance there is a clear prevalence of *L. michahellis.*

In the Spanish coastal cities, the presence of Yellow-legged Gull is particularly frequent and in the city of Barcelona, in 2017, 500 nesting pairs were estimated, 150-200 of which were on roofs and terraces and 15 - 30 at the Zoo (Anton *et al*., 2017).

5.3.1 The colonization of Italian cities

In Italy, as already mentioned, the first urban breeding dates back to 1971 when a pair nested on an artificial rock at the Zoo of Roma (now called 'Bioparco'). The pair nested for several years, but they remained an isolated case for a long time, because it was only from the 1980s onward that there was an increase in the number of urban nests of Yellow-legged Gulls. The first cities to be affected, besides Roma (which by 1984 had 4 pairs), were Sanremo (North-West) with a pair in 1982, Livorno (Central) in 1984 and Trieste (North-East) in 1987. In some cities, the population growth, as observed in other European urban centres, was very fast, with an exponential trend. In Trieste, for example, in the period between 1988 and 2000 there was an annual increase equal to 28.9% with an average annual growth of 4.3% between 2001 and 2018 while in Napoli (South) a figure of 22% per year was measured between 1990 and 2014 (Tab.6.3.1). At present, nesting is known to occur in about fifty urban centres with more than 10,000 inhabitants (Fig.6.3.1). Fig.6.3.2 shows the progress of the colonization process. There is a sharp growth after 2000 and a further increase after 2010. It should be noted that the colonization of the Adriatic cities along the Italian coast with the exception of Venezia and Trieste, has only recently started, in most cases from 2015 onwards. There have been no cases of colonies disappearing so far. An estimate of the breeding population in urban centres with more than 10,000 inhabitants leads to a figure of at least 4,000 pairs, equal to about 8% of the Italian breeding population. Regarding the choice of nest sites in cities, the species mainly uses man-made structures, nesting on terraces, tiled roofs, bell towers and on eaves. Ground nesting has also been observed recently in Roma and Trieste.

Tab.5.3.1 - The evolution of the number of breeding pairs in certain Italian cities

city	no. pairs and period	source
Torino	1 pair in 2007 - 12/15 in 2011	Di Rienzio A. EBN Italia
Genova	1 pair in 1986 – 78 in 2014	Milia L. *unpublished data*
Cremona	10 / 15 pairs in 1987 - 30 / 40 in 1998	Allegri M. - 1999
Venezia	22 pairs in 2003 - 200 / 250 in 2018	Sartori A. *estimate*
Trieste	1 pair in 1987 - 571 in 2018	Benussi E. *unpublished data*
Cesenatico	160 pairs in 2004 – 400 in 2014	Brina S. *unpublished data*
Sesto Fiorentino	45 pairs in 2013 - 72 in 2014	Del Sere M., Malfatti L., Puglisi L. *unpublished data*
Livorno	16 pairs in 1999 – 240 in 2013	Franceschi A. *unpublished data*
Piombino	2 pairs in 1994 - 40 in 2007	Franceschi A. *unpublished data*
Roma	1 pair in 1971 - 1000 / 1500 in 2014	Fraticelli F. *unpublished data*
Napoli	14 pairs in 1990 - 300/350 in 2018	Fraissinet M. *unpublished data*
Portici	2 pairs in 1999 – 25 / 30 in 2014	Fraissinet M. *unpublished data*

Fig.5.3.1 - Italian cities with a population exceeding 10,000 inhabitants with nesting Yellow-legged Gull *Larus michahellis*. Map updated to 2016.

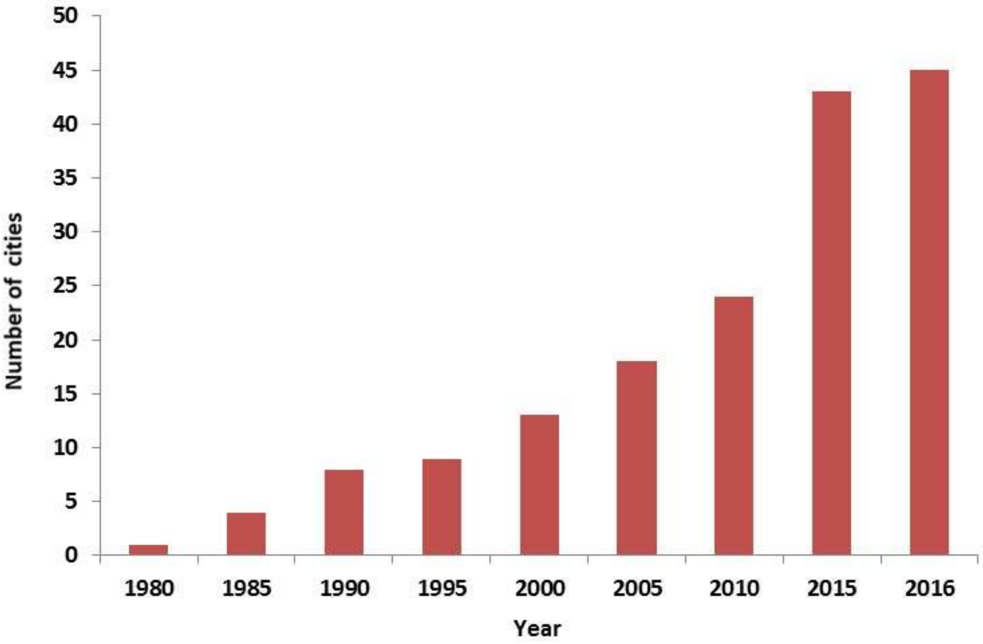

Fig.5.3.2 - Growth in the number of Italian cities with a population of more than 10,000 inhabitants with nesting Yellow-legged Gull *Larus michahellis* updated to 2016.

5.4 The process of colonization and the ecology of the species in urban areas

There may be many causes that lead to the process of colonization of urban centres by Yellow-legged Gulls, a species originally widespread in coastal environments. First of all, the strong numerical recovery of European populations since the mid-twentieth century should be taken into consideration, with the need, therefore, over time, to find new breeding sites. Added to this are more specific reasons: in urban centres, food is abundant and easily available, allowing even the first year birds, still inexperienced, to feed easily. There are also some favourable ecological conditions, such as a higher average temperature than that in areas out of town, and lower rates of predation, although this positive aspect has been decreasing over the years as a result of the entry into the city of species which predate eggs and nestlings such as the Hooded Crow *Corvus cornix*, Carrion Crow *C. corone*, Jackdaw *C. monedula*, Magpie *Pica pica* and even Raven *C. corax*. Many studies highlight the distance between the nests in the city being greater than that of the natural colonies and this might lead to the supposition that urban pairs are components of a single colony widely distributed across an area. There are, however, exceptions and some colonies established on roofs of industrial warehouses or near the coast have distances between nests similar to natural ones (Benussi and Bembich 1998, Fraissinet and De Rosa 2012).

Fig.5.4.1 - Nesting in the urban centre of Trieste (Photo Enrico Benussi)

The key to the success of the Yellow-legged Gull is also to be found in its high ecological plasticity that allows it, in particular, to modify its diet, adapting it to the most abundant and easily available resources present in urban centres or in the vicinity. This includes eating food waste of anthropic origin, and carrying out an active predatory behaviour on both insects and small mammals but also birds, including especially the Feral Pigeon (*Columba livia* var. *domestica*), abundant in cities. The process of colonization in the reproductive period has also been facilitated by the fact that, in the wild, the Yellow-legged Gull is a species that nests in rocky environments and has therefore found a surrogate of those environments in cities and building a nest on a cliff or on the roof of a building does not make much difference, especially if the roof is rarely accessed and also hosts spontaneous vegetation.

Fig.5.4.2 - The predation of Feral Pigeon is a rather common fact and is carried out by certain “specialist” individuals (Photo Maurizio Fraissinet)

In the city of Napoli, an urban population and a natural population coexist within a short distance of each other. The former breeding on buildings of the city, the latter on the tuff cliffs along the Posillipo and Nisida coast. It was therefore possible to carry out some interesting comparisons between the two populations, to study their ecology and reproductive biology and verify possible differences between them.

In the three years between 2005-2007, the two populations were monitored. In this case as well, as mentioned earlier, we noticed a greater distance between the nests in urban pairs than natural ones, with the sole exception of the nesting colony on the roof of the Royal Palace, a historic building located a short distance from the sea. As far as their reproductive biology was concerned, there were no differences in the average number of eggs nor the average number of fledged chicks. Differences were found however in the fledging dates, where the nesting pairs in the natural context exhibited a certain heterogeneity in fledging dates, with respect to those nesting on man-made structures and in an urban context. This phenomenon has been observed in other Italian urban settings (C. Soldatini, pers.comm.). Further differences are found in diet composition, with the nesting pairs on the Posillipo and Nisida coasts having a more fish-based diet compared to those in urban environments that eat more birds, as shown in Fig.6.4.3 (Fraissinet & De Rosa 2012).

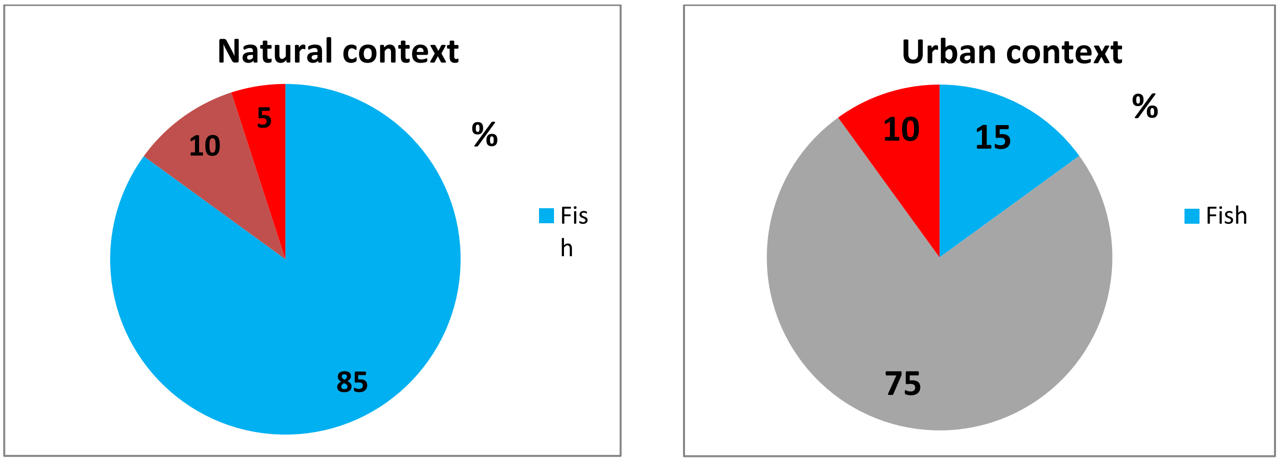

Fig.5.4.3 - Differences in diet composition observed in Napoli's two populations, found in a natural context and in an urban context.

Further monitoring of the breeding pairs carried out in Napoli brought another interesting piece of data on the growth of the urban breeding population compared to that in natural settings. Since 2006, the latter has shown no growth (Fig.6.4.4) (Fraissinet 2016) and it can be noted that the rapid urban population growth coincides with stasis in the natural one which has evidently reached a saturation point in the available sites, in sharp contrast to the urban population which is still a long way from arriving at this point.

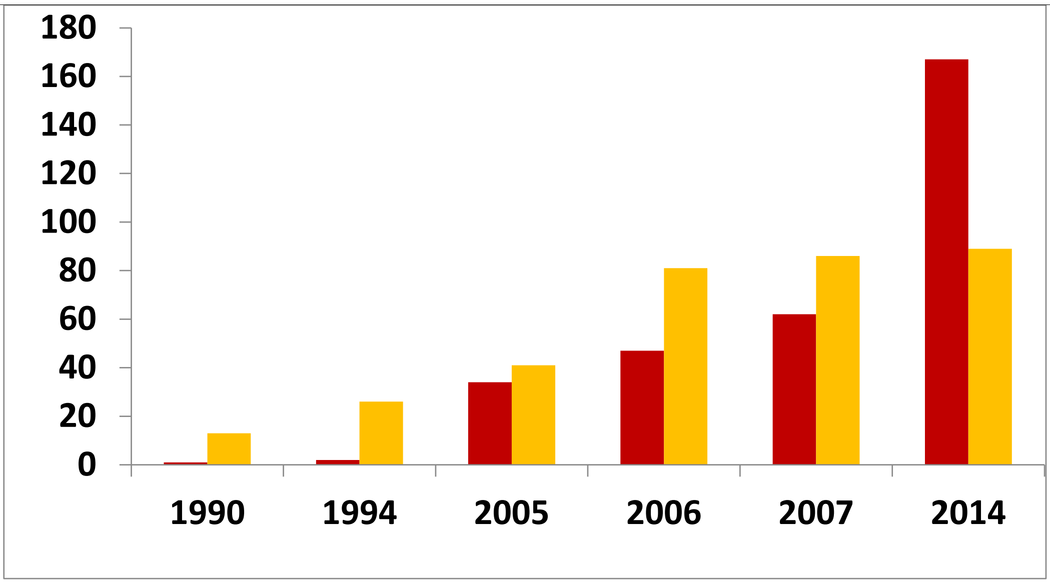

Fig.5.4.4 - Trends in the number of breeding pairs in urban and in natural contexts in Napoli. Red: pairs breeding in urban context. Yellow: pairs breeding in natural context

The Yellow-legged Gull should also be seen as an element of the urban ecosystem, in which it plays the role of secondary and tertiary consumer, depending on the prey, but also that of necrophage, having been observed to feed on carrion of other animals (even rats), as well as human food waste. At the same time, through its broods, it offers trophic resources to other predators such as corvids.

5.5 Problems created by urban gulls to humans

The presence of large numbers of nesting Yellow-legged Gull pairs in a city can cause problems of coexistence with the inhabitants. The difficulties, especially felt during the breeding season and in all European cities where the gulls nest, are manifold. Perhaps the largest impact is the aggressive behaviour of adults towards people - intruders from the gulls' perspective! - who, inadvertently, approach a nest with chicks; such behaviour known as 'dive bombing' is the highest level of aggression that the species can manifest towards humans and although it is 'no more' than flying just over the head of the victim, without any physical contact, it is enough to generate fear, sometimes terror, in the people who experience it. Other problems include the mess from the accumulation of bird droppings that often cover cars, buildings and properties in general and can also block vents, gutters and ventilation ducts as well as the calls emitted especially during the breeding season which take place even at night, with consequent disturbance to peace and quiet, the theft of food – sometimes even from people's hands – a behaviour coming from the kleptoparasitic habits of a bird that normally steals food from other species and conspecifics. Over the last few years these negative effects have multiplied as a result of the numerical growth of populations and the greater consequent contact with humans. Often, however, the episodes are magnified by the press, with the consequent increase in sometimes groundless or even unjustified psychoses. In reality, we are not aware of serious episodes of physical aggression to people outside the reproductive period although there are cases of physical contact attributable exclusively to the defence of nests and chicks at the breeding sites. Even the health risks, deriving from the fact that gulls can be carriers of some pathogenic bacteria to humans, must be traced to particular contexts and circumstances and to date, within the limits of Italian urban contexts, these have never been verified.

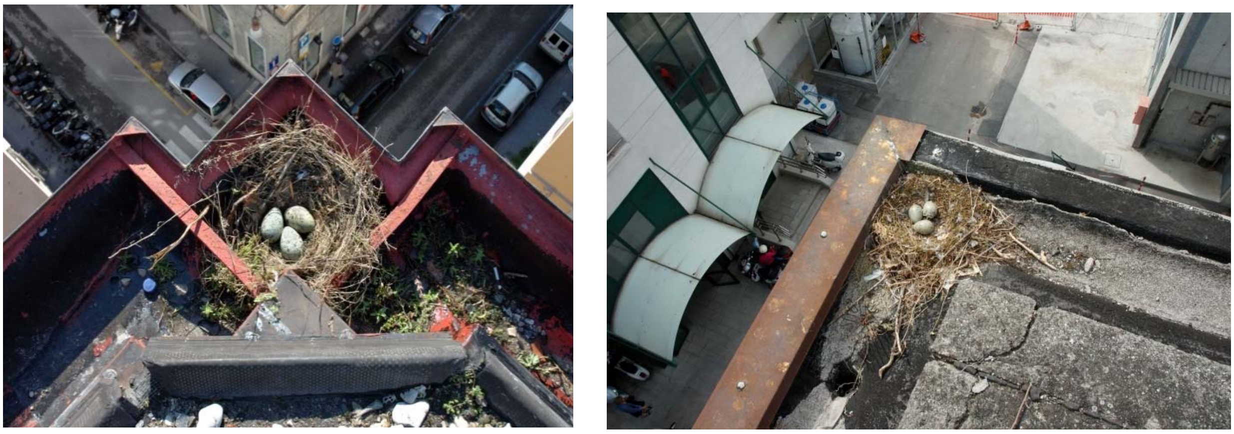

Fig.5.5.1 and 5.5.2 - Nests built in gutters can obstruct the flow of rainwater causing overflows and considerable consequent damage (Photo Enrico Benussi)

Fig.5.5.3 and 5.5.4 - The supply of miscellaneous material for the construction of the nest and an unusual location between the plants of a window-box on a balcony (Photo Enrico Benussi)

It should be remembered that the problems are partly offset by the ecological role that gulls play in the urban ecosystem as predators, as well as the pleasure that may be derived from the observation of their ‘free flight’ and the ‘natural vivacity’ that they bring to the cities. Recently, in some cities, people have increasingly begun to feed the gulls.

5.5.1 Prevention of the phenomenon

Counteracting a natural phenomenon is a difficult, often impossible undertaking. In some cases, it can be curbed by acting on the ecological causes triggering and maintaining it. In the case of the increase in pairs of Yellow-legged Gulls nesting in a city, it is necessary to act, first of all, on prevention. This prevention must be implemented both by tackling the ecological causes and behavioural aspects of the species. Prevention, moreover, must take place both, on a large scale, and be organised and financed by the municipal administrationas. At the level of individual homes, they could be helped and coordinated by the municipal administration but finance it themselves.

In this context, it is worth mentioning the initiative of the National Association of Italian Municipalities - ANCI - and the Municipality of Naples, which, with the involvement of some ornithologists, have produced a brochure that contains information for local authorities of Italian cities affected by the presence of Yellow-legged Gulls on how to prevent and limit the phenomenon, in particular during the breeding season, as well as providing advice to citizens on how to behave in case of forced coexistence with nesting pairs (Fraissinet 2015).

The local authorities are called on to carry out two types of preventive action: the reduction and removal of constant food sources, correct information to the public (civic education) repeated over time. Although the species is very mobile and is able to obtain food from sometimes dozens if not hundreds of kilometres away from a city, it is also appropriate to reduce the opportunities for an urban supply. Therefore, commercial businesses, fishmongers, fruit and vegetable shops, butchers and restaurants of various kinds should be encouraged to avoid disposal of their waste in the area. Many Yellow-legged Gulls are known to have memorized the closing hours of fishmongers or local markets and are ready and waiting to pick the ‘leftovers’ up from the ground. In addition to the commitment from the individual shopkeepers, there must also be that from local authorities whom should be prompted to clean the areas. Another important food source present in the city are rubbish bins for urban waste, although in many cities, these are progressively disappearing to make way for new and more modern disposal techniques based on separated rubbish collection, with organic waste being left out only on certain days, in closed containers, and removed quickly. This is certainly a valid method for reducing food sources available to the gulls. Controlling spontaneous feeding by some people is more difficult to implement and enforcing prohibitions of this activity have regularly failed in the European cities in which it has been attempted, due to the difficulty of its application and the social implications for sometimes problematic human subjects. Another area in which administrations need to take action is the maintenance of roofs and terraces. A roof or a terrace that is not used and is in a state of neglect creates the ideal conditions to entice a pair to nest. It is therefore of great importance that, in the month of February, municipal authorities issue a reminder which states that from the following month (March) Yellow-legged Gulls will resume breeding activity and will also begin the search for suitable nesting sites. This reminder should suggest that from the period between February to March it would be appropriate for homeowners to access roofs and terraces frequently in order to show the gulls that, due to continuous human presence, the site is unsafe. The reminder, moreover, may also provide information regarding the different ways that individuals may adopt to prevent the phenomenon, remembering in any case that this is a species protected by various laws and therefore no cruel or violent action of any kind may be carried out towards these animals. These interventions, on the other hand, might have no effect considering the noteworthy ability of the species to respond to adversity.

For individuals who do not appreciate the presence of breeding pairs in or near their home, as well as the managers of public areas, industrial warehouses, shopping centres or anywhere else that may be negatively affected by the presence of the species during their reproductive period, the information booklet, issued to the town mayors, suggests various types of interventions and reiterates the need to access, if possible, the areas where the gulls have already nested in the past or that could attract them in the future such as roofs, terraces, balconies, chimneys and gutters. In case there are difficulties in carrying this out frequently, it would seem that only two techniques have given positive feedback. The first - an alternative to the 'anti-intrusion net', which is expensive, ugly and not always effective - consists in spreading over the terrace, above head height, a simpler net formed of taut, parallel wires with a mesh of between 50 cm and 3 m, although a distance between 1.50 and 2 m is usually also sufficient. The wires must be strong (with a diameter of at least 3 mm), of a material which is resistant to sunlight and corrosion, such as stainless steel, or even better, fishing line. This method has proven itself effective for flat surfaces, even large ones (such as car parks, squares, etc.), as it acts on the mechanical impossibility of animals to get their wings through the net and the considerable insecurity that it generates in the breeding pair for the raising of chicks. Furthermore, it is not cruel and, above all, does not generate 'familiarity' over time i.e. the birds do not become accustomed to its presence and therefore do not lose their sense of insecurity (Blockpoel and Tessler 1984). The second technique, which is also a mechanical obstruction, can be used to prevent nesting on sloping, tiled roofs, or near chimneys and other roof projections on which the gulls nest. It consists in the use of 'support spikes' similar to the ones used for pigeons, but of a different shape and orientation, i.e. they must be inclined wedges and rather close together, with a height that is not less than 15 cm, but it should be borne in mind that in some cases, longer lengths may have to be used, sometimes as much as 30 cm. Furthermore, it should be underlined that this method requires precise installation to prevent damage to other wild birds that might land on the roof, and continuous maintenance over the years. There are different types of such dissuaders and therefore the identification of those that are best suited to the type and need of each roof is necessary. A wrong choice (elements not curved but straight, not very tall, etc.) may even promote nesting in certain circumstances!

Ultimately, the promotion of scientific research is of fundamental importance. Multi-year projects (at least 5 years) involving the marking of birds with legible rings, for example, are able to provide indispensable information on population dynamics.

5.5.2 Strategies to be adopted in case of nesting

The presence of important urban populations of Yellow-legged Gulls can often lead to real or presumed conflictual situations with people. The most frequently made complaints can be summarized as follows:

- Assaults
- Predation on pets
- Public disturbance
- Dirt and droppings on roofs, monuments and statues
- Health risks
- Damage to structures

Each of these situations present various levels of criticality, shown below, which must be carefully evaluated if specific action is to be contemplated.

- *Assaults*. Even if it is true that, while rearing its chicks, the Yellow-legged Gull applies a behavioural model of defence to its nest that is particularly aggressive towards those, including humans, getting too close, we must consider that, with the exception of isolated incidents, it is mostly just a display of aggression. However, there are more and more frequent cases in which particularly aggressive individuals even engage in physical contact, attacking in flight and striking the head's of intruders with their feet and beak. Regardless of the level of real risk, this behaviour puts a question mark on the everyday use of spaces (balconies or terraces) where a nest is. It can also represent a real element of risk for those who, in the course of their work, must visit the tops of buildings with situations where balance is vital (construction workers, installers of aerials, chimney sweeps). The period of greatest aggressiveness is limited to some extent to incubation but particularly emerges in the weeks after the hatching of the eggs until the young birds fledge. Considering that nesting can last several months in the spring and summer, this type of disturbance is most keenly felt by people.

- *Predation*. In most cases, the attack or active defence (mobbing) behaviour connected with the defence of the nest, or the attempt to steal food put down for dogs and cats, sometimes aggressively, are erroneously interpreted as predation attempts on pets. However, real attempts at predation cannot be ruled out, especially with regard to small animals left free to wander on terraces or in courtyards, such as dwarf rabbits or guinea pigs. Cases of predation have already been extensively documented for newborn kittens.

- *Disturbance.* Yellow-legged Gulls are particularly vocal and annoying at night when they engage in collective flights in large numbers. The meaning of this behaviour is still unclear and probably has a social function. The undoubted disturbance to peace and quiet, especially on the upper floors of buildings near nests, is not however limited to these flights, but also to the constant loud calling of the pairs and their chicks, especially during the weaning period when the latter begin to fly.

- *Dirt and droppings.* Gulls use many types of materials in the construction of the nest, even if in limited quantities. They also carry a range of objects to the tops of buildings, perhaps to play with. These materials, often helped by the slope of the roofs, when driven by rain, tend to accumulate at the mouth of the drainpipes and guttering, causing blockages and leaks. In addition, there are numerous cases of monuments and bronze statues being used as perches that are soiled by the bird droppings which causes corrosion, potentially damaging the structures seriously and even irreversibly. In addition, the remains of predated pigeons and feathers may accumulate on roofs.

- *Health risks*. Gulls' excrement is particularly liquid and therefore does not tend, even if deposited by substantial numbers of individuals, to form layers of guano as happens in the case of urban pigeons. Given that the Yellow-legged Gull is also a predator of other vertebrates, remains of fish, birds or other food waste in a state of putrefaction near the nest may be found. To date, there are no reports of infection directly transmissible from gulls to humans, but in these situations it is advisable to maintain the correct precautions and strive to ensure suitable hygiene conditions in areas frequented by people, as the international scientific community, to date, has no doubts about the possible zoonotic risk linked to the presence of gulls in urbanized areas.

- *Damage*. The Yellow-legged Gull chicks, before starting to fly and moving away from the nesting site, spend several days walking about on the infrastructure near the nest. During these excursions, apparently demonstrating playful behaviour - but probably practising using their beak for their future independence - they tend to fiddle with various objects such as window fittings and dormers, electric or television cables, roof covering materials and so forth. This behaviour can cause serious damage.

Multiple stratagems have been proposed with a view to stopping, or at least containing, the damage and disturbance caused by this species but none has given satisfactory results or is applicable in all situations. The following list refers exclusively to methodologies that have, if nothing else, a rational approach.

- *Distress call.* The alarm or distress call, recorded and amplified, can function as a means of removal for some species. In the management of urban populations of Starling *Sturnus vulgaris*, this has given positive results allowing the removal, or at least, the breaking up of flocks that concentrate in roosts. Using this approach with the Yellow-legged Gull, however, has not produced the same results but has rather generated aggregative responses in a sort of mutualistic behaviour that brings together the individuals present in the vicinity of the site.

- *Ultrasound*. Although there are many ultrasound systems available, which are sold to deter annoying birds, their effectiveness is nil because birds do not have the ability to hear sounds above 20 kHz, in other words ultrasound.

- *Olfactory repellents*. Despite the sense of smell having recently been reconsidered in many species of birds, in the Yellow-legged Gull, the use of products with strong repellent odours did not result in their relocation.

- *Laser lights*. Even this system, which consists in flashing laser light beams at the gulls as they perch, has not provided any result. It is also necessary to consider the real risks to public health that this technique can cause.

- *Protection nets*. Covering the roofs of buildings or terraces with nets can provide positive results especially on small surface areas. Above larger spaces, these structures are not always able to fulfil their purpose. The potential advantages should be weighed against the installation costs and the limitations posed by making the surfaces affected inaccessible to humans. This does not, of course, include the suspended wires mentioned earlier as they can also be placed at heights that allow human access.

- *Spikes.* These consists of plastic or stainless steel points that are designed to prevent gulls from landing or perching on surfaces, or building nests. This system, used in many urban centres to limit the presence of pigeons along the edges of buildings and monuments, does not give appreciable results for gulls, unless the forms and types described above are adopted. Moreover, using the spikes intended for pigeons - besides the obvious difficulty of intervening on all surfaces with often extremely irregular shaped material - may even prove to be counterproductive as the spikes could constitute a support for nest material especially on sloping surfaces where the nest would otherwise slip in the absence of such intervention.

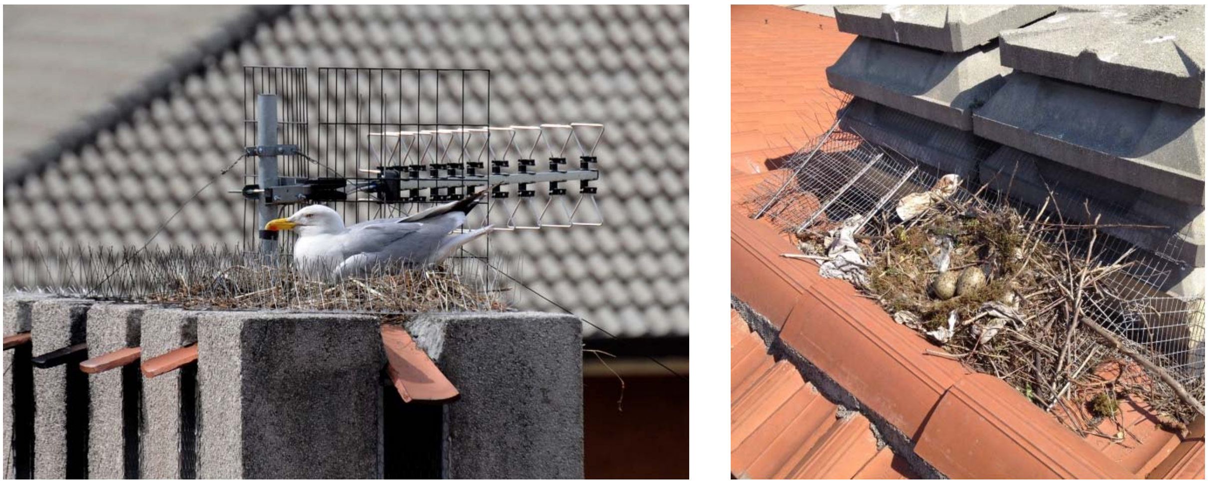

Fig.5.5.5 and 5.5.6 - Spikes and metal nets sometimes serve to better anchor the nest and might not prevent nesting (Photo Enrico Benussi)

Regarding the following techniques, it must be borne in mind that the Yellow-legged Gull, in many countries, is a species protected by law and, consequently, any action involving it or its nests must have a prior permission from the Authorities. The authorization process must be undertaken directly by the individual's public administration, while any planned actions must be carried out by personnel with clear and certified ornithological skills. From a biological point of view, these interventions can be meaningful only in the presence of critical situations related to single individual birds. However, they are totally ineffective as forms of management of the species as a whole, because given that there are so many gulls, local numerical reduction work would be immediately thwarted by the arrival of new birds from neighbouring areas.

- *Culls.* Apart from ethical reservations, the effect of euthanising one of the gull of a breeding pair is nullified in a short time because of the large number of individuals that have failed to pair or which have not found a site suitable for reproduction and thus a new partner is found immediately. Furthermore, the elimination of large numbers of individuals in urban environments would involve techniques that could represent a real risk to the public.

- *Egg removal and nest destruction*. This does not produce results unless it is carried out on a large scale and repeated several times a year on the replacement clutches. This technique can mainly solve the problem on single buildings used as homes, preventing reproduction and therefore removing the source of irritation and the cause of damage to the structures that the presence of breeding pairs may bring. In many cases, the destruction of the nest must be continued for several years because gulls tend to return to the same sites year after year and even when the gulls leave, the problem merely moves elsewhere. In this case laws that protect the species must as well be taken into consideration.

- *Sterilization of eggs*. The technique consists in the drilling of the eggs or in their waterproofing with paraffin, both aimed at interrupting embryonic development. These interventions have proved to be inadequate because within a short time the breeding pair perceives the absence of an embryo in development and lays a replacement clutch. This forces the repetition of the operation several times because there may be multiple replacement clutches. Furthermore, the operational difficulties that this technique entails, the considerable commitment of people involved and the high costs must also be considered. For example, in central Roma alone, fewer than 5% of the nests can be reached, in an area of maximum reproductive density for Yellow-legged Gull, without the use of rock-climbing techniques (F. Fraticelli, pers.comm.).

- *Relocation of nests*. This technique involves physically moving the nest from high impact areas to areas of less impact. Beyond the obvious operational limits similar to the previous intervention, this technique requires that alternative sites for relocating the nest exist. It is therefore understandable that at most this can be applied in a limited number of cases.

- *Administration of baits treated with contraceptive chemicals* This experimentation has thrown up difficulties in the preparation and dosage of baits, such as difficulties in their systematic distribution and controls on consistent and balanced intake as well as high costs which thus makes it inapplicable.

- *Sterilization of adults.* This is not an effective method of control, particularly if performed on sexually immature subjects, as it cannot reduce the size of the population, unless 100% of the territorial males are castrated (or 100% of the females are sterilised), the decrease in the population arriving through the natural mortality of adults. The interventions on juveniles are useless considering the high mortality they undergo in the first two years of life (over 50%) and the marked erraticism that distinguishes all subjects until sexual maturity in their third year, which allows for only a very low percentage, and predominantly males, to return to reproduce at the natal nest site. Castrating/ unnecessarily sterilizing a low percentage of sexually mature subjects without actually altering population dynamics represents an error in planning, but above all is a pointless episode of animal cruelty.

In conclusion, considering the ineffectiveness of the methods described above, the only really effective action is the prevention of the phenomenon. Another fundamental step is the communication to the public of the simple rules of coexistence that may be summarised as follows:

- *Do not feed the gulls*. When checking many situations of presumed risk, it became evident that the same people who had thought themselves threatened had previously offered food to the gulls, thus conditioning their behaviour (Fig.13).
- *Do not make pet food available*. Avoid feeding dogs and cats on terraces and balconies and if necessary remove leftover food.
- *Reduce alternative trophic sources*: in urban centres, avoid leaving rubbish outside bins and limit open landfills outside towns.
- *Protect small pets:* don’t leave them outdoors.

Fig.5.5.7 - “Domesticating” gulls by providing them systematically with food causes the species to become more dangerous towards humans, especially during the breeding season when the most aggressive individuals can physically attack people in defence of the offspring (Enrico Benussi)

Fig.5.5.8 and 5.5.9 - In some cases, particularly tame gulls nest in conditions that allow for them to be approached without problem but greater agressiveness can occur however, following hatching (Photo EnricoBenussi)

Finally, it should be added that in some cities (in Italy: Trieste, Cesenatico and Naples, for example) groups of experts have come forward both from the scientific and the conservation world who, when called on by people, intervene to find solutions that - while guaranteeing the safety of the birds – are also able to meet the expectations of people.

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Yellow-legged Gull (*Larus michahellis*)

Chapter VI

6. Assessing risks posed by wind turbines and electric power lines to soaring birds: a modelling approach[4]

6.1 Abstract

Recent growth of investments in wind energy and electric power industries has increased concerns about their impact on wildlife populations. In particular, the highest impact of these infrastructures affects flying vertebrates, both directly by increasing their mortality rate due to collision with the turbines and indirectly through habitat loss or fragmentation. In this study we proposed a modelling approach that combines the results from species distribution models (SDMs) and geographic information system (GIS) in order to predict and quantify the effects of wind turbines and electric power lines on *Ciconia nigra*'s breeding habitat in Italy. Our results showed high probability of presence of the species in several areas in Italy. However, the mortality rate of *C. nigra* due to collision with turbines and electrocution resulted to be potentially high in Italy but with a different trend between Northern and Southern regions. Our study highlighted the importance of SDMs as tools supporting conservation strategies in order to mitigate the impact of human infrastructures on wildlife and encourage a more sustainable planning.

6.2 Introduction

The increasing anthropic pressure on ecosystems and the global biodiversity loss highlight that further efforts are necessary to improve the currently available tools for the assessment and monitoring of biodiversity conservation at different scales (Tulloch et al. 2016; Di Febbraro et al. 2018). Species distribution models (SDMs) represent an example of effective tool which can be applied to tackle many issues in applied ecology and to support conservation planning in several

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ways. SDMs are the most widely used correlative models which allow to understand the relationships between a species occurrence and environmental variables and to identify areas where a given species is likely to occur, i.e. defining the habitat suitability (Guisan and Thuiller 2005; Elith and Leathwick 2009; McShea 2014). This application may aid conservation decision-making by detecting new areas where the target species presence is unknown and is a useful tool for estimating the potential for species to occur in areas not previously surveyed (Hernandez et al. 2008). Such models are even more relevant for species that are elusive and difficult to observe (Razgour et al. 2016) and, in particular, for rare and threatened species which are usually a conservation priority (Bosso et al. 2016a; McCune 2016; Proosdij et al. 2016).

An important application of SDMs involves predicting the impacts of anthropogenic activities on animal populations and their habitats, and identifying areas of high mortality risks for such species (Santos et al. 2013; Roscioni et al. 2014). In particular, the impacts of wind farms on biodiversity need to be considered because they have been found to affect wildlife, especially flying vertebrates, both directly by increasing their mortality rate due to collision with the turbines and indirectly through habitat loss or fragmentation (Arnett and May 2016; Bastos et al. 2016; Beston et al. 2016).

Birds and bats are alarmingly vulnerable to mortality due to wind turbines, because both are volatile taxa (Arnett and May 2016) and the high collision rate of them have been documented worldwide (Penteriani 1998; Barrios and Rodriguez 2004; Rollan et al. 2010; Thaxter et al. 2017; Martin et al. 2018). Actually, another anthropogenic structure may negatively affect wildlife and in particular birds, i.e. the electric power lines. Electrocutions on pylons and collisions with wires seem to be the major cause of many deaths for birds (Bayle 1999; Janss 2000). In the last decades, scientists have attempted to understand population-level effects of wind farms through field studies of species demography (Beston et al. 2016). Moreover, different mitigation efforts aimed to reduce collision rates have been increasing worldwide (Crockford 1992; Drewitt and Langstone 2006; Fox et al. 2006; Kuvlesky et al. 2007; Pearce-Higgins et al. 2009; Busch et al. 2017), especially through the implementation of wire markers (Janss and Ferrer 1998; Barrientos et al. 2012; D’Amico et al. 2019) but, although these mitigation measures can partially reduce species’ mortality risk due to human infrastructures, in most cases they cannot eliminate it. SDMs represent a potentially powerful approach to achieve this goal by generating risk maps, which could provide a crucial tool to locate sites where wind farms or power lines cannot be planned or sites where more survey efforts are needed to assess the actual likelihood of impact (Cathrine and Spray 2009; Roscioni et al. 2013).

Among birds species, soaring birds, including most raptors, storks and other large birds, are the groups of highest concern, as their movement corridors have been populated by wind farms and power lines (Katzner et al. 2012; Cabrera-Cruz and Villegas-Patraca 2016; Martín et al. 2018) leading to high fatality rates through collisions with turbines and wires (e.g., Barrios and Rodriguez 2004; Smallwood and Thelander 2008; Ferrer et al. 2012).

In this study we focused on a soaring and long-distance migratory species, the Black Stork (*Ciconia nigra*), that has the largest breeding range among the *Ciconiidae*, from Portugal to China, with a distinct population in South Africa (Fraissinet et al. 2018). Nevertheless, the species requires special conservation measures in Europe because during the 19th and first half of the 20th century the European population suffered a considerable decrease and disappeared from most of its western European breeding range due to uncontrolled hunting and excessive forest exploitation (Cramp and Simmons 1977; Bordignon et al. 2017). Since the 1970s the European population has been increasing, especially in several countries of Central Europe and the breeding European population is currently estimated to be around 9,800-13,900 couples and classified as Least Concern (BirdLife International 2017). In Italy, where the black stork is a migrating nesting species, we are assisting to a recent recolonization. Since the first breeding attempts established in 1994 in Piedmont and Calabria (Bordignon 1995; Mordente et al. 1998), the colonization has continued with an increase in the number of couples, estimated to be 20 in 2018 (Brunelli et al. 2018). Nevertheless, the increase of both Italian nesting couples and breeding range expansion results to be lower than other Central and Western European countries (Snow and Perrins 1998; Alexandrou et al. 2016; Kalocsa and Tamas 2016; Lorge 2016; Denis and Brossault 2016; BirdLife International 2017). Moreover, monitoring efforts showed that, in Italy, the number of breeding pairs in the North-western population is significantly lower than in Southern and Central regions (Fraissinet et al. 2018).

In our study, we combined SDMs and geographic information system (GIS) to carry out a risk maps' development to analyze the potential distribution of the breeding population of C. nigra in Italy and to evaluate the negative impact posed by anthropic infrastructures on suitable areas.
We hypothesize that: a) in Italy, a species with a wide global distribution like C. nigra will have a high availability of suitable areas to colonize; b) and that, given the widespread presence of electrocution and wind farms in Italy (Marcantonini and Valero 2017; Terna 2017), the black stork's population will be affected by a high risk of collision with power lines and turbines especially in areas close to foraging and nesting sites.

6.3 Materials and methods

6.3.1 Study area and presence records

The study area included the whole Italian territory covering ca. 301,000 km2 between latitudes 45° N – 36° N and longitudes 6°E – 18° E. It has a very heterogeneous topography with an elevation ranging from 0 up to 4,810 m above level sea (a.s.l.).

We obtained occurrence records for *C. nigra* in Italy from experts' personal database updated to 2017 (Maurizio Fraissinet unpublished data) and from the online platform Global Biodiversity Information Facility (GBIF 2019). We downloaded the presence records of the black stork both for Italy and the other European countries including species' breeding distribution in order to include the entire environmental and climatic niche of the species (Barbet–Massin et al. 2010; Raes 2012; Guisan et al. 2013; Smeraldo et al. 2017). Each record corresponded to coordinates of a black stork's nest. The accuracy of species occurrences gathered from GBIF database was assessed by including only those collected after 1990, during the main nesting period (June and July, Maurizio Fraissinet pers. comm.) and whose positional values featured at least two decimal digits (0.01 decimal degrees, corresponding to 1.11 km at the equator; Strubbe et al. 2015). In addition, we filtered these data by removing duplicated records and those with unrealistic coordinates.

Prior to model calculations, records were screened in ArcGis (version 10.2.2) for spatial autocorrelation using average nearest neighbour analyses to remove spatially correlated data points and guarantee independence (Russo et al. 2015; Kwon et al. 2016; Bosso et al. 2018b). Therefore, form the initial occurrence dataset including 280 records, after a filtering procedure, we obtained a dataset of 136 records for *C. nigra* which were used to generate SDMs.

6.3.2. Ecogeographical variables

We selected a set of variables potentially useful to predict summer range of *C. nigra* in Europe.. Climatic variables were obtained from the Worldclim database version 2.0 (Hijmans et al., 2005; http://www.worldclim.com/current). Land cover categories were selected according to the species' preferences for foraging or nesting sites and were calculated as the Euclidean distance from the Corine Land Cover 2012 categories (European Environmental Agency: www.eea.europa.eu/data-and-maps/data/clc-2012-raster). The hydrographic network was taken from the Digital Chart of the World (DCW; http://www.diva-gis.org/gdata). All predictors were rasterized at a resolution of ca. 1 km with ArcGis (version 10.2.2). To take into account the pairwise correlation between the predictors, the final set of variables was subselected considering a Pearson's correlation coefficient

|r|<0.75 (Hernandez et al., 2006; Lobo et al., 2010; Russo et al., 2015; Bosso et al. 2016b) and a variance inflation factor ≤ 5 (Zuur et al. 2010). These procedures were carried out in the R environment (R Development Core Team, 2012). Therefore, from the initial set of predictors, we retained only the eight most relevant variables according to the species' ecological requirements during the breeding season: mean temperature and precipitation of May, Euclidean distance from hydrographic network, Euclidean distance from urban areas, Euclidean distance from inland wetlands, Euclidean distance from mixed forest, Euclidean distance from non-irrigated arable land and Euclidean distance from permanently irrigated land and rice field. We added further details on the most relevant variables used to model potential distribution of *C. nigra* in Europe in the Table S1.

6.3.3 Species distribution models

To build the models, we used presence records of C. nigra selected as described above and localized in all the European species' home-range in order to avoid truncated niche estimations (Barbet-Massin et al., 2010; Raes, 2012; Guisan et al., 2013). SDMs were built through an ensemble forecasting approach, as implemented in the R package "biomod2" (https://cran.r-project.org/bin/windows/base/; Thuiller et al. 2009). We considered the following seven modelling techniques (Thuiller et al. 2009; Jiguet et al. 2010; Ducci et al. 2015): 1) generalized linear models (GLM); 2) generalized additive models (GAM); 3) generalized boosted models (GBM); 4) random forests (RF); 5) multivariate adaptive regression spline (MARS), 6) maximum entropy models (MAXENT), and 7) artificial neural network (ANN; for further details, see Thuiller et al., 2009). Following Pio et al. (2014), the modelling settings were tuned as follows. GLMs and GAMs were calibrated using a binomial distribution and a logistic link function. GBMs were calibrated with a maximum number of trees set to 5000, threefold cross-validation procedures to select the optimal numbers of trees to be kept and a value of seven as maximum depth of variable interactions. RF models were fitted by growing 750 trees with half the numbers of available predictors sampled for splitting at each node. MARS models were fitted with a maximum interaction degree equal to 2, while MAXENT models were fitted with the default settings apart with a maximum value of 1000 iterations. In order to avoid model overfitting, we developed MAXENT models applying species-specific settings selected using the "ENMeval" (Muscarella et al. 2014; Fourcade et al. 2018) R package. The approach implemented in ENMeval runs successively several MAXENT models using different combinations of parameters to select the settings that optimize the trade-off between goodness-of-fit and overfitting.

Here, we set ENMeval to test regularization values between 0.5 and 4, with 0.5 steps, as well as the following feature classes: linear, linear + quadratic, hinge, linear + quadratic + hinge, linear + quadratic + hinge + product and linear + quadratic + hinge + product + threshold, which corresponds to the default ENMeval settings. We then selected the parameters that scored lower AIC value. Default parameters were used to fit ANN models. Each occurrence dataset was randomly split into a 70% sample, used for the calibration of the model, and the remaining 30%, used to evaluate model performance. Because our dataset contained only presence data, a set of 10,000 background points were randomly placed over a region identified by all the WWF terrestrial ecoregions (Olson et al., 2001) were species records occurred (Hirzel et al. 2002; Barve et al. 2011; Barbet-Massin et al. 2012; Smeraldo et al. 2018). Predictive performances of SDMs were assessed by measuring the area under the receiver operating characteristic curve (AUC; Hanley and McNeil, 1982) and the true skill statistic (TSS; Allouche et al. 2006). These validation methods have been widely used (Bosso et al. 2016b; Feuda et al. 2015; Breiner et al. 2015; Balestrieri et al. 2016; Smeraldo et al., 2018) and offer excellent performances. The data splitting procedure was repeated 10 times and the evaluation values averaged. For species' dataset, we ran a total of 80 SDMs (eight algorithms × 10 splitting replicates for model evaluation). After excluding models with $AUC<0.7$, model averaging was performed by weighting the individual model projections by their AUC scores, a method shown to be particularly robust (Marmion et al. 2009). Models were then projected over the study area. The relative importance of variables was also calculated from the ensemble model using the specifically devoted functionality available in the biomod2 package (Jiguet et al. 2010). Final potential distribution was obtained by averaging the projections from the 10 replicated ensemble models coming from the subsampling procedure (see above). Final map was binarized into presence–absence values using a threshold maximizing sensitivity (the percentage of correctly predicted presence) and specificity (the percentage of correctly predicted absence; Fielding and Bell, 1997). Such threshold has been widely used (Algar et al. 2009; Dubuis et al. 2011; Di Febbraro et al. 2015; Bosso et al., 2018a; Smeraldo et al. 2018) and constitutes one of the most accurate (Liu et al. 2005).

6.3.4 Risk maps

We generated risk maps to assess the impact of wind turbines and electric power lines on the potential distribution of *C. nigra* in Italy. To do this, we used the binary map of *C. nigra*, clipped for

Italy, and the shapefiles of Italian regions, wind turbines and electric power lines. We downloaded the administrative boundaries of the Italian regions by Italian national statistical institute (ISTAT) (http://www.istat.it/ambiente/cartografia). We obtained the distribution of the wind turbines by consulting several online databases as: ATLAEOLICO (http://atlanteeolico.rse-web.it/), The Wind Power (https://www.thewindpower.net/country_maps_en_7_italy.php), EMODnet (http://www.emodnet-humanactivities.eu/view-data.php), Harvard University World Map (https://worldmap.harvard.edu/data/geonode:osm_worldwide_wind_turbines_qb8), ATLAIMPIANTI (https://atla.gse.it/atlaimpianti/project/Atlaimpianti_Internet.html), and Overpass Turbo (https://overpass-turbo.eu/). We then critically reviewed all these data in ArcGis (version 10.2.2) by deleting unreliable, ambiguous or duplicate wind turbine records and excluding those whose geographical location was not precisely defined or wrong, checking every single wind turbine in Google Earth Pro (version 7.3.2). Furthermore, when we found a new wind turbine not reported in one of the previous cited databases, we added manually it in our new database in order to carried out the first complete map of the wind turbines in Italy (Fig S1). We acquired geo-referenced linear electric power line by Overpass Turbo (https://overpass-turbo.eu/) (Fig S2).

To assess the surface of *C. nigra*'s habitat suitability that fell inside or close to the wind turbine and the electric power line, we created different buffer areas around these infrastructures as follows: 1) a circular buffer with radius of 100 m, 500m, 1,000m, 3,000m, 5,000m and 10,000m around the wind turbine (Figs S3 and S4); and 2) a linear buffer with distance of 100m, 500, and 1,000m from the electric power line (Figs S5 and S6).

Risk maps for *C. nigra* in Italy were obtained by weighted overlay using spatial analyst tools in ArcGis (version 10.2.2). Weighted Overlay is a technique used to apply a common measurement scale of values to diverse and dissimilar inputs in order to create an integrated analysis (further details on how weighted overlay works are available at the following website: http://webhelp.esri.com/arcgisdesktop/9.3/index.cfm?TopicName=How%20Weighted%20Overlay%20works).

Because weighted overlay use only the raster data, all shapefiles employed in this study were converted to raster format. The input raster data for weighted overlay must contain discrete integer or continuous values and these values must be on a common scale. The weighted overlay tool reclassifies values in input raster onto a common evaluation scale of suitability or preference i.e. on the basis of their relative contribution to the central theme (Iqbal and Khan 2014). In this study all input raster data were reclassified to assign equal intervals of discrete values and then the final maps were reclassified into five categories representing different risk classes, respectively low, medium-low, medium, medium-high and high (e.g. Bosso et al. 2017a).

6.4 Results

7.4.1 SDMs

SDMs showed an excellent level of predictive performance as indicated by the AUC and TSS value which had a mean value and a standard deviation respectively of 0.904 ± 0.017 and 0.640 ± 0.030.

The mean temperature of May provided the greatest contribution among variables, followed by distance from urban areas, inland wetlands and hydrographic network. In particular, for *C. nigra* occurrence was more likely for mean temperature of May comprised between 15 and 25 °C (Figure S7) and habitat suitability decreased for increasing distances from inland wetlands, hydrographic network, mixed forests and non-irrigated arable areas (Figure S7). Regarding the distance from urban areas, the variable's response curve reflected the black stork's preference for areas quite distant from anthropic disturbance (Figure S7).

We found that the potential distribution of *C. nigra* showed high logistic values in several areas in Italy, in particular in central and southern lowlands (Fig. 7.1) while low probability of presence occurred in Alp and Apennine regions (Fig. 7.1). Suitable habitat for the black stork in Italy amounted to ca. 121,380 km^2, which corresponds to ca. the 40% of the Italian territory. Sardinia (14,118 km^2), Sicily (12,268 km^2), Lombardy (11,513 km^2), Piedmont (10,743 km^2) and Apulia (9,140 km^2) are the regions encompassing the largest potentially suitable surface for the black stork while Liguria (2,411 km^2), Marche (2,348 km^2), Molise (1,730 km^2), Trentino-Alto Adige (1,302 km^2) and Valle d'Aosta (49 km^2) were those including the smallest amount of it (Table 1).

6.4.2 Risk map

We found that ca. 34%, 9% and 0.1% of the *C. nigra*'s habitat suitability in Italy fell in the circular buffer areas of the wind turbine to a radius of 10,000 m, 3,000 m and 500 m, respectively (Tab 2). We also detected that ca. 38% of suitable surfaces of black stork was fallen in linear buffer with distance of 1,000m from the electric power line (Tab 3).

Valle d'Aosta, Basilicata, Apulia, Sicily, Sardinia and Molise were the regions at highest risk of impact with the wind turbine for *C. nigra* while Piedmont, Lombardy, Trentino-Alto Adige, Veneto Friuli-Venetia Julia and Emilia-Romagna were those at lowest risk (Fig. 7.2; Tab S2).

Valle d'Aosta, Lombardy, Trentino-Alto Adige, Piedmont and Liguria were the regions at highest risk of impact and electrocution with the electric power line for the Black stork while Sardinia, Basilicata, Apulia, Molise, Sicily and Calabria were those at lowest risk (Fig 7.2; Tab S3).

6.5 Discussion

6.5.1 Model performance and habitat preferences

Our SDMs provided a confident delineation of the potential breeding range of *C. nigra* in Italy, showed by a very high predictive performance with a AUC value > 0.9 and a TSS value > 0.6 (Domíguez-Vega et al. 2012; Di Febbraro et al. 2016; Kabir et al., 2017; Ancillotto et al., 2019). It is important to remark that the high reliability of our models derived from a thorough knowledge of species' year-round physiological requirements which was essential to select suitable occurrences and variables representative of a particular phenological pattern (Feng and Papes 2017; Smeraldo et al. 2018). This procedure allowed to generate a more faithful representation of the potential breeding distribution for the black stork. Using the same approach, in a previous study on *C. nigra*, Jiguet et al. (2011) combined winter tracking data and niche-based modelling techniques in order to predict the species' wintering distribution in Africa. Other studies on different migratory birds used seasonal occurrences with the aim to forecast species' potential breeding/wintering distribution also under future climate change scenarios (Barbet- Massin et al. 2012; Morganti et al. 2017).

Our results confirmed the hypothesis that a wide area of Italy is potentially suitable for the species and might be colonized, in particular the North-western regions, Apulia, Sicily and Sardinia. Nevertheless, the current breeding distribution of the black stork in Italy is only partially in agreement with our prediction, as the spread of the species since its recolonization in 1994 has remained confined to only few Italian regions and the growth process seemed to be slower than other European countries as Hungary, Poland, France, Germany and Czech Republic where a significant increase was registered (Dzyubenko and Bokotey 2011; Kalocsa and Tamas 2016; Lorge 2016; Denis and Brossault 2016; Pojr and Vo Itechovska 2016; Fraissinet et al. 2018). In Denmark, for example, where the recolonization of *C. nigra* was almost contemporary to the Italian one and where the forest area is lower than in Italy, today there are about 10 couples and even Luxembourg hosts between 4 and 7 couples (BirdLife International 2015). Instead, in Italy, the increase of the black stork's population is essentially due to the breeding population in South-central Italy, with the highest number of couples in Basilicata, while the North-western population, 20 years after recolonization, has not showed any increase and has not colonized the Central and Eastern part of the subalpine area and Po Valley (Fraissinet et al. 2018). Other Italian Central regions, like Lazio and Tuscany, which from our study resulted to have many suitable areas for the species, actually showed a very low or null colonization rate. A particular case regarded the Sardinia which currently is not included within the black stork's breeding range but our SDMs predicted a wide suitable area. The regular presence of the species in this Italian region during the winter has been considered by ornithologists as a potential establishment of wintering site for *C. nigra* in Italy (Grussu and Floris, 2005; Fraissinet pers. comm.).

The reasons for the particular colonization trend of the black stork in Italy, which have been under thorough investigation by researchers in the last decade (Bordignon et al. 2006; 2009; Fraissinet et al. 2018), might depend on different biotic and abiotic factors affecting and limiting the species' presence. If we focus on the environmental requirements of *C. nigra* during the breeding season, from our study emerged that mean temperature of May represents the main climatic variable limiting the species' distribution. This is explicable considering that birds which build exposed nests are vulnerable to low temperature that can negatively affect hatching success (Tobolka et al. 2015). Among the habitat preferences of the species for the nesting and foraging activities there were the proximity to mixed and deciduous forests and to streams and inland wetlands. These findings are in agreement with studies carried out on the breeding distribution of the black stork in other European countries. Jiguet et al. (2004) investigated the species' habitat use within core ranges and found that it was predominantly composed by mixture of woodlands and open areas, the former mainly deciduous forests, and that the location of nest-sites of black storks depended on the availability of mature trees (Bakaloudis et al. 2005).This environmental preferences are reflected by species' habitat selection especially in Northern Italy, while studies carried out in Central-southern Italian regions suggested that the breeding couples chose wide woody areas in hills with the presence of small areas with high slope, like cliffs and rock jumps (Fontaneto et al. 2006).

The black storks forage in correspondence of natural wetlands and shallow artificial pools in grasslands (like in rice fields) or along streams (Jiguet and Villarubias 2004). Alexandrou et al. (2016), in Greece, found that, during the breeding season, black storks aggregate in isolated pools which constitute the main feeding habitats in summer. In fact, in Mediterranean countries most streams dry out during the summer, then sufficient water remains in streams and marshy ponds only in lowland areas which, from our study, resulted to have high suitability for the species' presence. It is also important to remark that the habitat of black storks seemed to be linked with water quality of the rivers, as this quality could be expected to be higher closer to species' feeding areas (Jiguet and Villarubias 2004). For this reason, the storks are often considered as flagship species for wetland conservation (Olsson and Rogers 2009).

In consideration of species' habitat requirements, the main factors hindering the black stork's new colonization of suitable areas are the habitat fragmentation and deforestation because of species' large spatial needs during the breeding period (Jiguet and Villarubias 2011). In addition, the degradation of wetland habitats and, especially, the drainage of permanent water courses contributes to the population decline (Lõhmus and Sellis 2001; Czech and Parsons 2002) as well as the excessive use of pesticides because the species frequently feeds in ponds in intensively cultivated agricultural lands (Tucker and Heath 1994). Regarding the risk of nest predation, only few cases were observed. For example, in Latvia, the main predators affecting the breeding success of black storks are pine marten *Martes martes* as well as white-tailed eagle (Ķuze et al. 2008; Strazds 2011).

6.5.2 Risk map

We found that the black stork's probability of presence decreased in proximity of urban areas. Our results are confirmed by the well-known behavior of the species which, unlike white stork (*Ciconia ciconia*), shuns contact with humans and breeds preferentially in undisturbed woodland (Del Hoyo et al. 1992). Nevertheless, the impact of human infrastructures such as wind farms and electrocution resulted to be very high in correspondence of the potential breeding habitat of *C. nigra* as demonstrated by our study. To date, the resulting increase in the number of wind farms has raised concern about their potentially negative and cumulative effects on bird populations (Bellebaum et al. 2013; Wang et al. 2015). Wind turbines and electric power lines may affect birds mainly through direct collision (Barrios and Rodriguez 2004). Moreover, most accidents seem to occur on leading lines defined by macro elements (e.g. mountain ranges, coastlines or plains) that are important for long-distance bird movements, mostly for migrators like *C. nigra* which run the highest risk of collision in correspondence of migratory bottlenecks because they often correspond to areas suitable for wind-power production (Rollan et al. 2010; Mulero-Pázmány et al. 2013). In fact, wind turbines are frequently installed along the top of mountain ranges, in order to maximize exposure to horizontal winds, and these areas also tend to have high orographic uplift potential for soaring birds (Katzner et al. 2012). Similarly, the black stork might be affected by a disturb in the foraging and breeding sites (Barrios and Rodriguez 2004). Studies which compared the pre- and post-construction phases showed that soaring birds reduce their use of the areas where turbines are installed and their population decrease in proximity of wind farms and power lines due to a functional habitat loss (Barrios and Rodriguez, 2004; Pearce-Higgins et al. 2009; Marques et al. 2019).

In our study, the development of a risk map represented the first attempt to quantify the negative effects of wind farms and electrocution separately on potential suitable habitat for *C. nigra* in Italy. A similar approach was applied by Busch et al. (2017) in Germany. They used the current distribution of different birds' species, including *C. nigra*, provided by the Atlas of German breeding birds to carried out an overlap between the habitat occupied by the species and areas of wind farm related risk. Compared to our study, they didn't used an ecological niche model to forecast the potential distribution of a species in areas where it didn't seem to occur, so their approach wouldn't be effective for species in a recolonization phase as in our study case.

Our analyses, carried out on each Italian region, suggested a different pattern of interference of these infrastructures on species' probability of colonization between Northern and Southern Italy. In particular, black stork's population which could be not much threatened by wind farms in Northern regions might be highly hampered by the presence of power lines and vice versa in Southern regions. These findings are in agreement with the increasing number of wind farms in areas where 2/3 of the Italian breeding population of black stork is currently concentrated, i.e. Basilicata and Apulia regions (Fraissinet et al. 2018). Moreover, the only known case of mortality due to electrocution were registered Northern Italy, in particular in 1996 for Piedmont, while 5 cases are known between 1995 and 2003 in Lombardy (Bordignon and Mastrorilli 2004).

6.5.3. Implications for conservation

To date, many countries have been adopting measures and guidelines to mitigate the cause of birds' collision and mortality with wind turbines and power lines (Bright et al. 2008). However, such infrastructures are still increasing and implanted in areas where the impact on the wildlife is still unknown (Janss 2001; Marques et al. 2014). Identifying and understanding all the factors which can be related to mortality rate of birds can help reduce and mitigate the adverse impacts on birds (Bevanger 1998; Wang et al. 2015).

The model-based methodology proposed in our study represents a valuable tool to support strategic actions for impact mitigation and management (Roscioni et al. 2014; Santos et al. 2013). The risk maps obtained allow researchers to identify more vulnerable areas to address conservation efforts, i.e. those particularly interested by a high risk of collision of the species with turbines or power lines. Such method might also forecast, with scientific credibility, future ecological consequences associated with infrastructural impacts on wildlife (Bastos et al. 2016; Guisan and Thuiller 2005). For example, for *C. nigra* we were able to identify many suitable areas not yet colonized by the species but where the presence of power lines or turbines might dramatically increase the likelihood of mortality in the near future. Such areas correspond to riparian habitats or wetlands where birds spend a large part of their day flying between breeding/nesting and foraging areas, often during crepuscular periods with low light levels increasing the risk of collision or electrocution (Garrido and Fernandez-Cruz 2003; Bernardino et al. 2018). Then, conservation actions could be addressed to mitigate the impact of power line routing in proximity of wetlands, nesting and foraging sites and other sensitive bird habitats which are of conservation concern.

Another important issue regards the presence of wind farm routes along the main flight paths of migratory species (Martin et al. 2018; Marques et al. 2019). Prominent landscape features such as important rivers and mountain ridge lines which are important reference elements for birds during the migration (Harness and Carlton 2001; SNH 2016). From our analyses, resulted that the areas with the highest risk of collision for *C. nigra* with turbines occur in regions interested by species' passage during migration (Bordignon et al. 2006). In this perspective, our modelling approach might guide wind farm industry' decisions by suggesting strategies to mitigate the current risk of collision and addressing a more careful route planning for the future.

6.6 References

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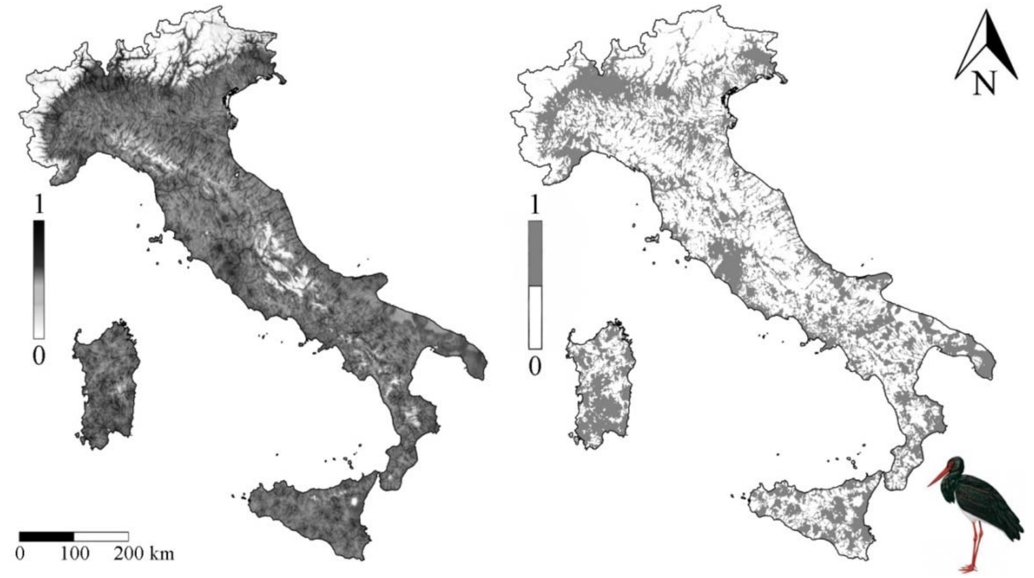

Fig. 6.1 - Species Distribution Models of *C. nigra* in Italy. Left: logistic map; right: binary map. Scales show the probability of presence ranging from 0 to 1.

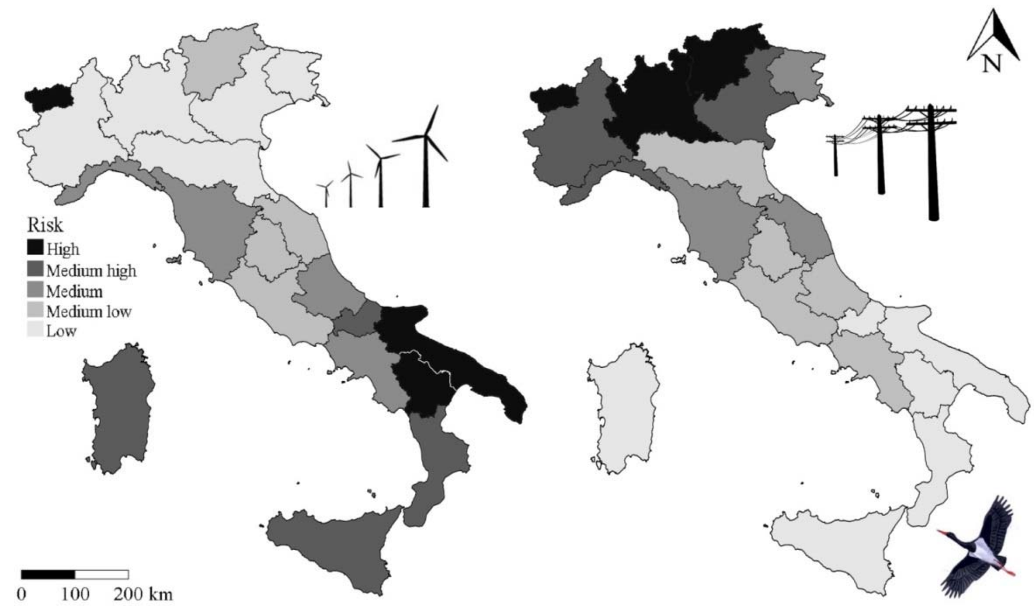

Fig. 6.2 - Risk map of exposure of Italian regions to *C. nigra*. Left: wind turbine; right: electric power line.

Table 6.1 - Potential surface (km^2) and percent surface of Italian territory potentially suitable to *C. nigra* divided by region according to the biomod2 binarized map

Region	Potential surface by binarized map (km^2)	Regional potential surface by binarized map (%)
Piedmont	10,743	42
Valle d’Aosta	49	2
Lombardy	11,513	48
Trentino-Alto Adige	1,302	10
Veneto	6,652	36
Friuli-Venetia Julia	3,945	50
Liguria	2,411	44
Emilia-Romagna	7,559	34
Tuscany	6,822	30
Umbria	2,882	34
Marche	2,348	24
Latium	8,431	49
Abruzzi	2,668	25
Molise	1,730	39
Campania	5,859	43
Apulia	9,140	47
Basilicata	4,077	41
Calabria	6,867	46
Sicily	12,268	48
Sardinia	14,118	59

Table 6.2 Wind turbine impact

	Buffer (m)					
	100	500	1,000	3,000	5,000	10,000
Surface (km2)	75	935	2,362	10,878	20,491	41,934
Percentage (%)	0.06	0.77	1.95	8.96	16.88	34.55

Table 6.2 Electric power line impact

	Buffer (m)		
	100	500	1,000
Surface (km2)	6,512	27,330	46,530
Percentage (%)	5	22	38

Supplementary materials

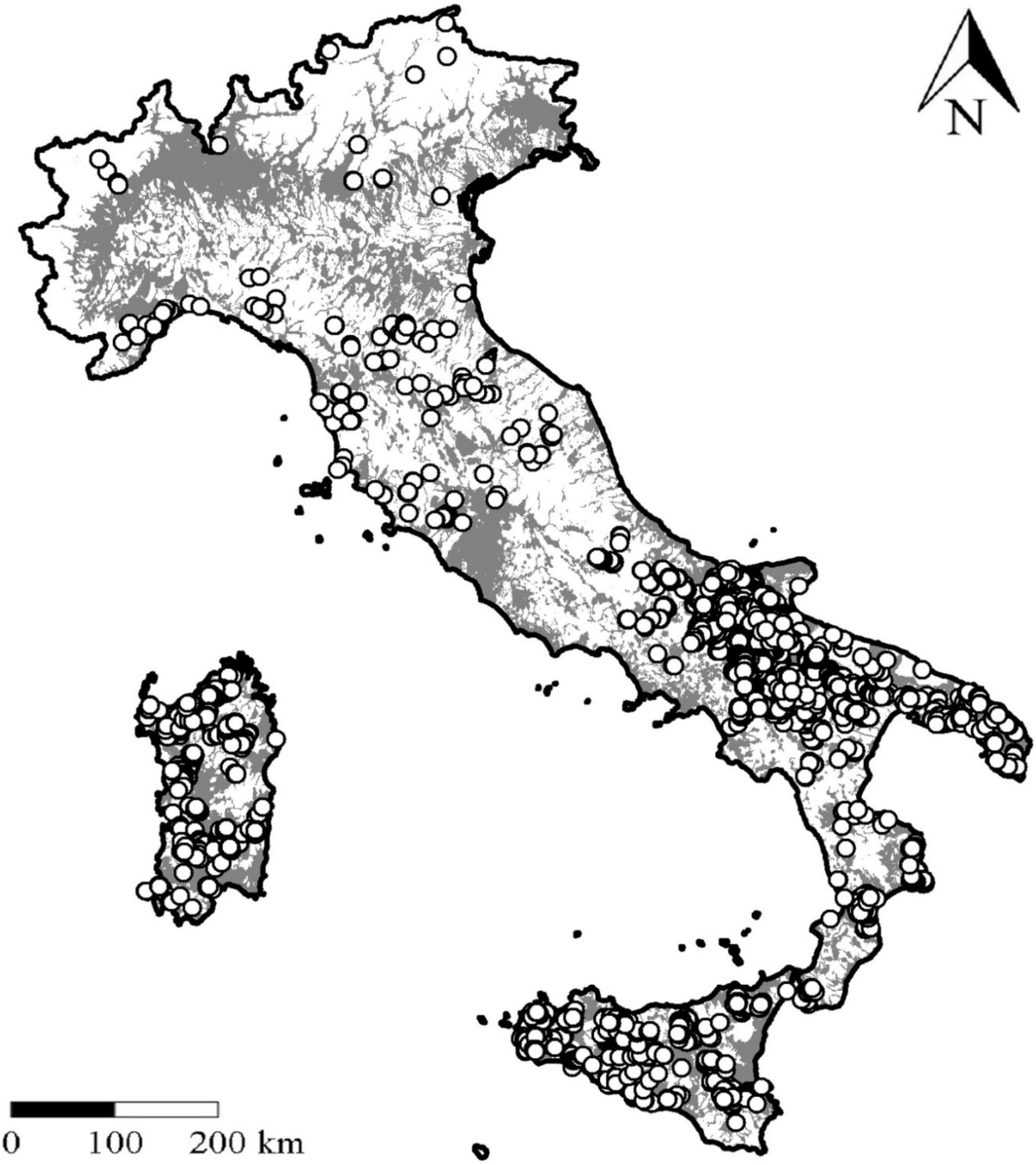

Fig. S1 - Binary map (presence = grey; absence = white) of *C. nigra* and wind turbine distribution (white circle) in Italy.

Fig S2 - Binary map (presence = grey; absence = white) of *C. nigra* and electric power line distribution (black line) in Italy.

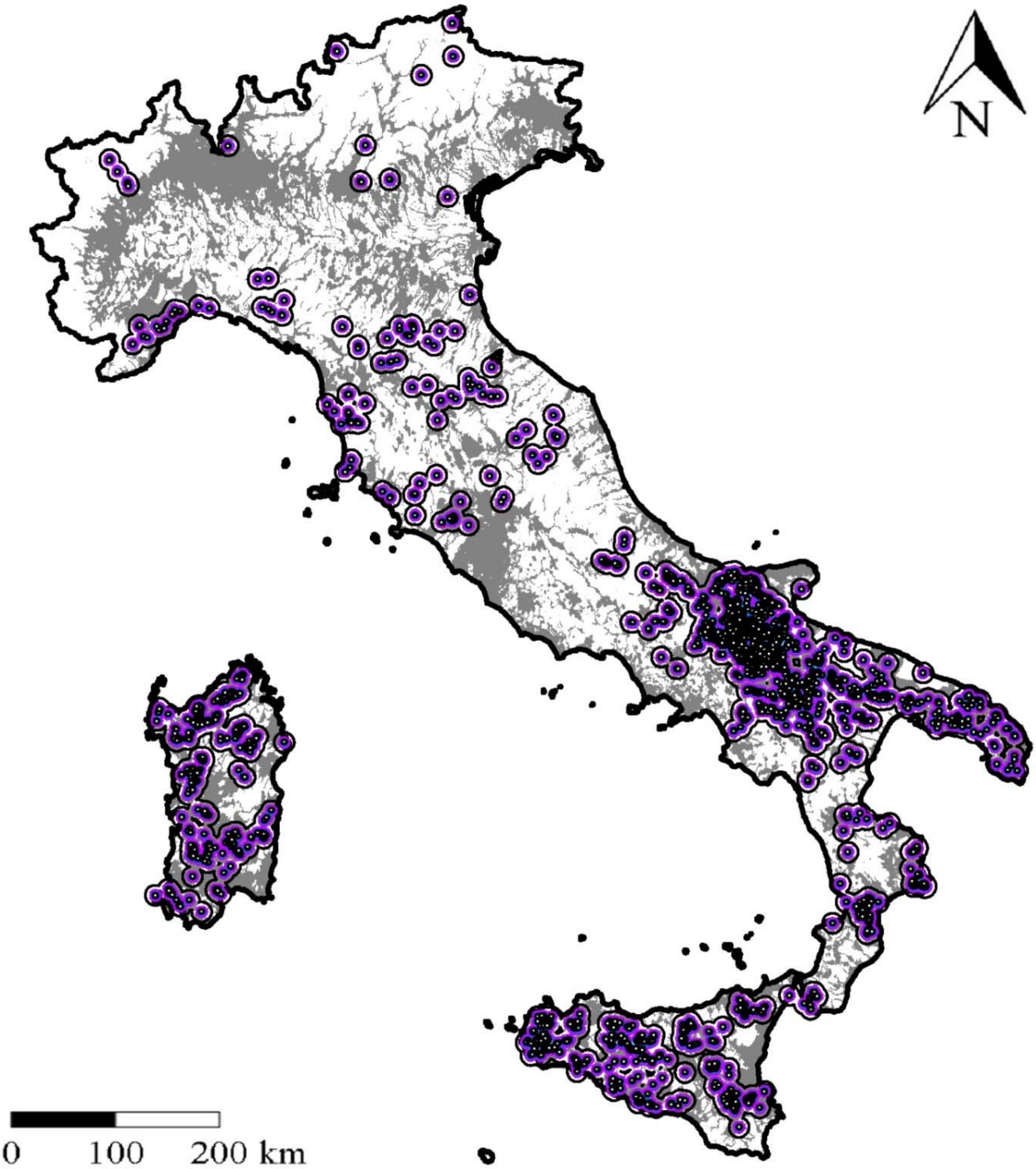

Fig S3 - Binary map (presence = grey; absence = white) of *C. nigra* and wind turbine distribution (white circle) in Italy. Circular buffers around the wind turbine were made with a radius of: 100 m (red), 500 m (orange), 1,000 m (yellow), 3,000 m (blue), 5,000 m (violet) and 10,000 m (black).

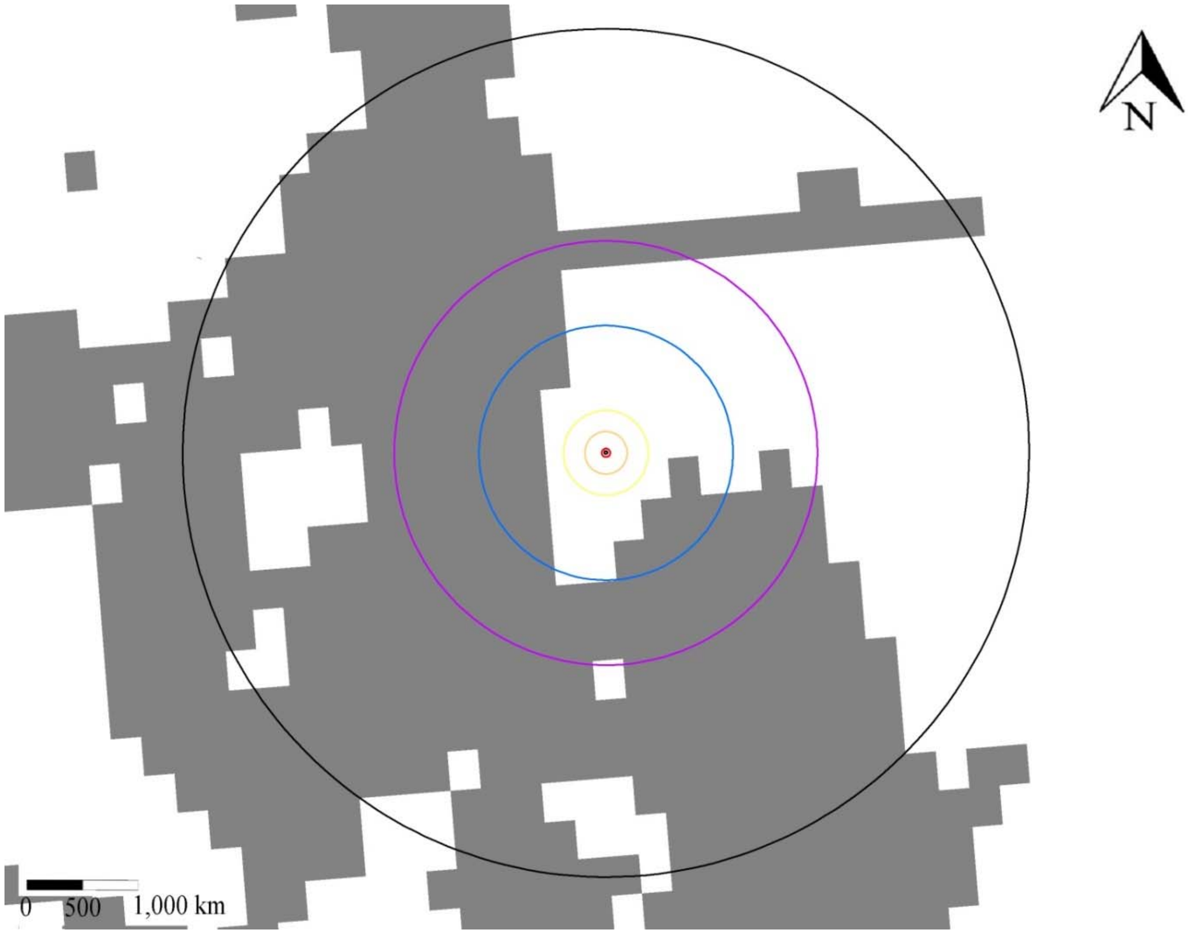

Fig S4 - Zoom on the binary map (presence = grey; absence = white) of *C. nigra* and wind turbine (with circle) in Italy. Circular buffers around the wind turbine were made with a radius of: 100 m (red), 500 m (orange), 1,000 m (yellow), 3,000 m (blue), 5,000 m (violet) and 10,000 m (black).

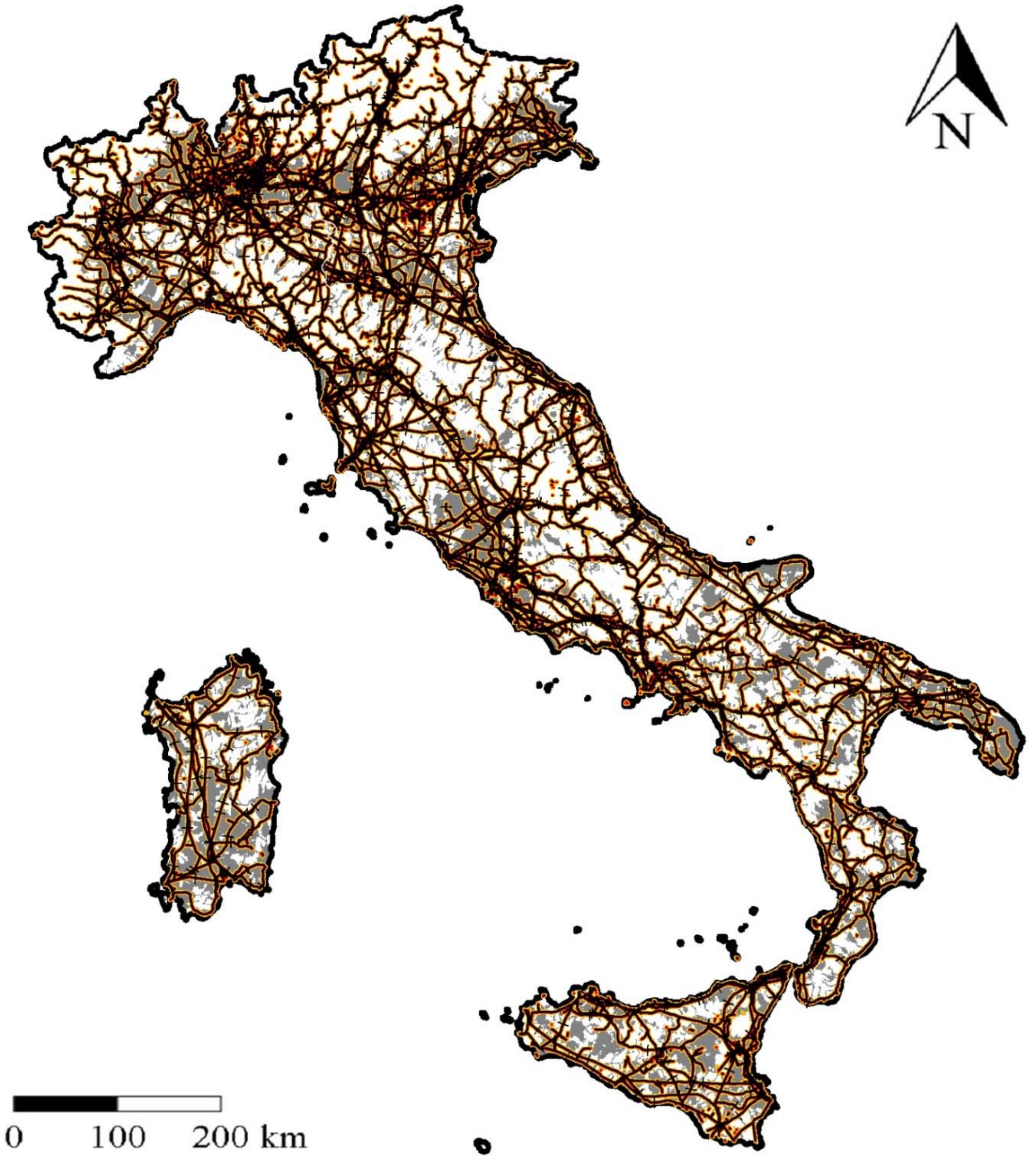

Fig S5 - Binary map (presence = grey; absence = white) of *C. nigra* and electric power line distribution (black line) in Italy. Linear buffers around the electric power line were made with a distance of: 100 m (red), 500 m (orange) and 1,000 m (yellow).

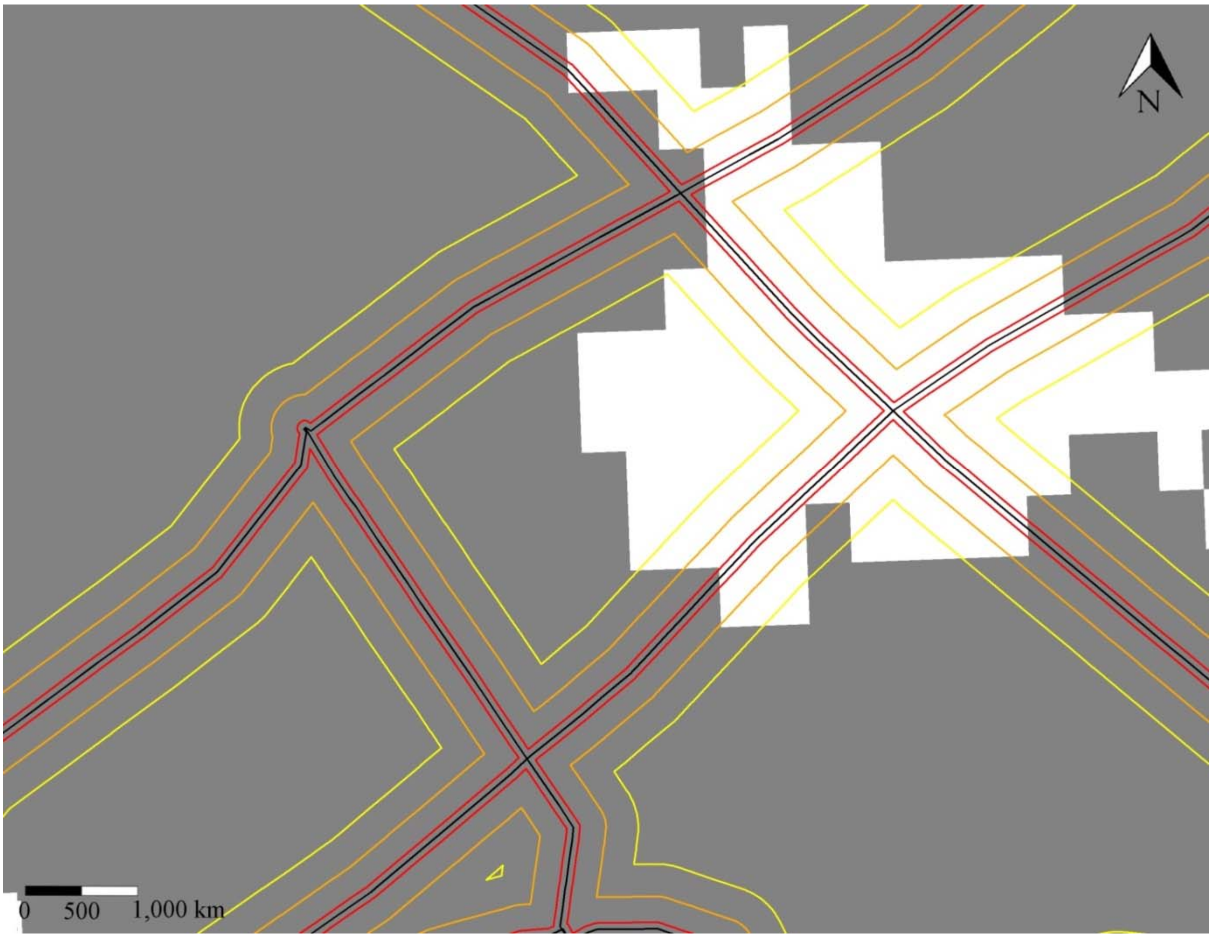

Fig S6 - Zoom on the binary map (presence = grey; absence = white) of *C. nigra* and electric power line distribution (black line) in Italy. Linear buffers around the electric power line were made with a distance of: 100 m (red), 500 m (orange) and 1,000 m (yellow).

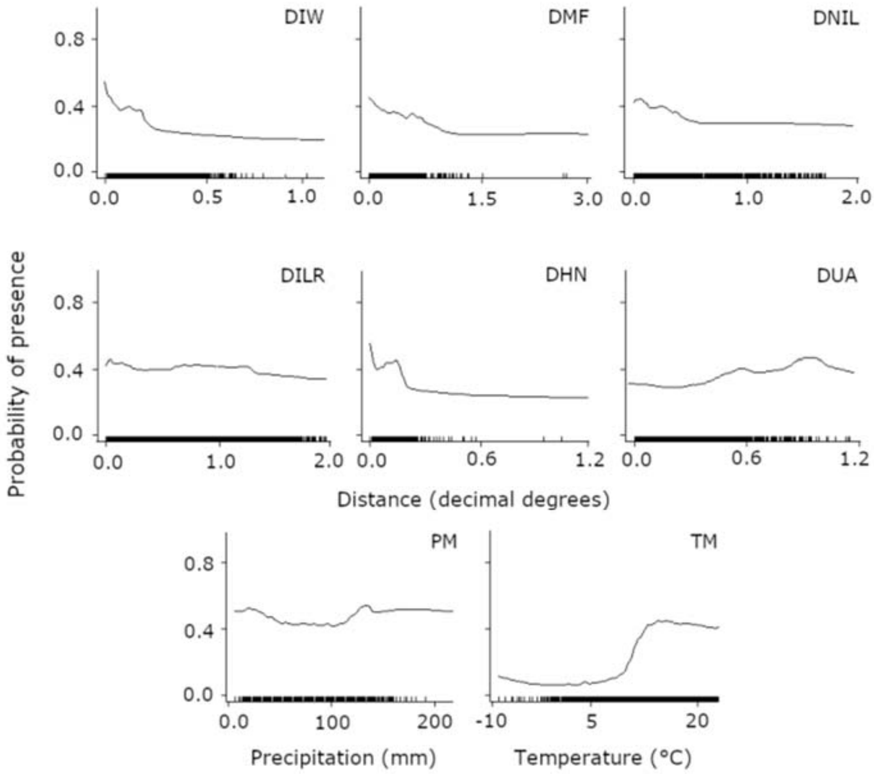

Fig S7 - Response curves for the eight variables used to model *Ciconia nigra* potential distribution in Europe with the "biomod2" computer platform. Response curves correlate probability of occurrence (y axis) with values of the explanatory variables (x axis). Each curve represents one variable (DIW = distance from inland wetlands; DMF = distance from mixed forests; DNIL = distance from non-irrigated agricultural lands; DILR, = distance from irrigated agricultural lands and rice fileds; DHN = distance from hydrographical network; DUA = distance from urban areas; PM: mean precipitation of May; TM: mean temperature of May). Distances are expressed in decimal degrees, precipitation in mm and temperature in °C.

Table S1 - List of Ecogeographical Variables (type and measurement unit) used to predict current potential breeding distribution of *C. nigra* in Europe.

Type	Bioclimatic variable	Unit
Climatic	Mean Temperature of May	°C
	Mean Precipitation of May	mm

Habitat	Distance from urban areas	Decimal degrees
	Distance from mixed forests	Decimal degrees
	Distance from inland wetlands	Decimal degrees
	Distance from hydrographic network	Decimal degrees
	Distance from non-irrigated arable fields	Decimal degrees
	Distance from rice fields	Decimal degrees

Table S2 Percent surface of *C. nigra*'s suitable habitat, according to the biomod2 binarized map, fallen in the buffers made around the wind turbine divided by region

Region	Buffer					
	100	500	1,000	3,000	5,000	10,000
Piedmont	0	0	0	1	1	3
Valle d'Aosta	0	2	6	31	59	94
Lombardy	0	0	0	0	0	2
Trentino-Alto Adige	0	0	0	2	4	9
Veneto	0	0	0	1	3	9
Friuli-Venetia Julia	0	0	0	0	0	0
Liguria	0	1	2	10	21	49
Emilia-Romagna	0	0	0	1	3	9
Tuscany	0	0	1	4	11	31
Umbria	0	0	0	2	5	20
Marche	0	0	0	1	4	20
Latium	0	0	0	2	4	12
Abruzzi	0	0	0	3	6	18
Molise	0	0	1	11	27	63
Campania	0	1	3	11	19	36
Apulia	0	3	7	28	46	79
Basilicata	0	2	7	31	53	85
Calabria	0	1	3	13	25	49
Sicily	0	1	3	16	31	65
Sardinia	0	1	3	14	31	63

Table S3 - Percent surface of *C. nigra*'s suitable habitat, according to the biomod2 binarized map, fallen in the buffers made around the electric power line divided by region

Region	Buffer		
	100	500	1,000
Piedmont	6	26	45
Valle d'Aosta	16	61	90
Lombardy	9	35	56
Trentino-Alto Adige	15	52	74
Veneto	7	29	49
Friuli-Venetia Julia	6	26	44
Liguria	8	30	48
Emilia-Romagna	5	23	40
Tuscany	6	26	43
Umbria	6	23	39
Marche	6	25	43
Latium	5	23	40
Abruzzi	6	24	42
Molise	4	18	31
Campania	5	20	36
Apulia	3	15	27
Basilicata	3	14	25
Calabria	4	18	32
Sicily	4	18	31
Sardinia	3	13	24

Black Stork (*Ciconia nigra*)

Chapter VII

7. Conclusions

The aim of this thesis was to verify the role of birdlife communities as bioindicators in urban environments. To do that, we studied the evolution over time of the urban birdlife of the city of Naples using the method of the ornithological atlas, considering that Naples represented the only city in the world provided with three urban atlases of breeding and wintering birds created over a period of 28 years and developed using the same methodology, cartographic grid, research coordinator and most of the detectors. The wide time span, together with the significant amount of data collected and the possibility of correlating the evolution of the avifauna with the environmental changes occurred in the city during the period under consideration, constitute a good research base for studying the evolution of bird community and to verify the existing knowledge about the phenomenon of urban colonization by birds. The presence of wild bird species in the city also leads to management problems. Hence, the choice of studying the case of the Yellow-legged gull (*Larus michahellis*), while the Black Stork (*Ciconia nigra*) was selected as a non-synanthropic species to investigate and highlight the negative effects of urban ecosystems on the areas outside them. Regarding knowledge acquisition, we obtained the following results:

1. It was confirmed that the breeding urban birdlife undergo a rapid turnover in time. Considering the three urban ornithological atlases created for the city of Naples and the distance of about ten years between each of them, we were able to establish that the turnover is substantial and has a statistical validity in a period of about 30 years. In fact, Sorensen's index, which reveals the similarity existing among the checklists of the breeding species of the three atlases, was statistically significant in the comparison between the first and third atlas. The number of breeding species remains unchanged and in Naples 64 breeding species occur. However, the quality of the population is changed and it is highlighted by: a decrease in Ornithological Value Index; the loss of native species typical of environments pre-existing to urbanization and intolerant to fragmentation habitat; the colonization of cities by common species with a widespread distribution and good ability to adapt to new environments. This also entails a low homogeneity index for the birdlife community characterized by the presence of a small number of widely distributed species and many other species gathered in few areas of the city.

2. Species that tend to decrease and in some cases even become extinct due to the urbanization processes are those typical of open and shrubby environments resulting disadvantaged from both the environmental typology and the substrate of the nesting sites. Those species that manage to colonize urban environments or take advantage of urbanization processes are typical of forest and rock environments, which, in the city, are substituted by the building. These findings represent two phenomena that are reported in the literature and that find further confirmation in this long-term research.
3. Species strongly dependent from wooded environments are both the most represented category in the context of urban environment and those with the highest probability of colonization and the tendency to expand their home range in the city. The urban environment often includes quite extensive wooded areas, mainly represented by public or private gardens, historic houses and portions of natural woods, thus providing suitable habitats for many species typical of the pervious mentioned type of environment. On the contrary, open or cultivated green areas are generally lacking or small in the cities, and often their management is not compatible with the biology of many bird species that depend on these environments. Similarly, areas offering suitable habitat for the species which depend from scrub and uncultivated lands are often those more quickly replaced by newly established urban fabric.
4. Moreover, species characterized by a widespread distribution and mainly resident, are favoured in the colonization of the city, while migratory species show difficulty in surviving for a long time. This finding was also reported in the literature and is confirmed in a long-term study such as this thesis.
5. Our results do not agree with the data relating to North American cities where granivorous species increase with the intensification of urbanization. Nevertheless, the increase in frugivorous and carnivorous species is in line with what is known from the literature.
6. The correlation between the percentage of land use categories identified in the city of Naples within a grid of 1 km^2, as adopted for the urban ornithological atlases of Naples, and the presence / absence data of the species, allowed the development of a predictive model which provide the probability of occurrence of a breeding species in the city. This modelling approach has never been applied before for the study of breeding urban birdlife.
7. This study showed that the Yellow-legged Gull (*Larus michahellis*) has successfully colonised the Italian cities due to its ability to quickly adapt its eating habits, assuming a mainly

ornithophagous and necrophagous diet in the city, compared to the predominantly ichthyophagous diet of the populations living in suburban areas. Moreover, it is emerged the need for a more responsible and effective management of organic waste in order to drastically reduce the winter trophic sources for this species which showed a high reproductive rate in urban areas.

8. We demonstrated that a species that live far from urban areas and with absolutely non-synanthropic characteristics, such as the black stork (*Ciconia nigra*), are affected by the energy needs of urban ecosystems also in areas that are geographically distant. The high percentage of overlap between the predicted areas potentially suitable for the reproduction of the Black Stork in Italy and the records of power lines and wind turbines highlighted the significant anthropic impact on the potential expansion of the species, which, in Italy, counts for little more of 20 breeding pairs.

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Detail of the murals of the metro station of Piscinola