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URBAN STRUCTURE CHANGES AND ENVIRONMENTAL FACTORS OF VALENCIAN COMMUNITY TOWNS AS POTENTIAL DRIVERS OF THE HOUSE SPARROW DECLINE

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PhD Thesis

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Faculty of Biological Sciences

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Urban structure changes and environmental factors of Valencian Community towns as potential drivers of the House Sparrow decline

Cambios en la estructura urbana y factores ambientales en localidades de la Comunidad Valenciana como potenciales impulsores del declive del gorrión común

PhD Thesis

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“Los gorriones son los niños del aire, la chiquillería de los arrabales, plazas y plazuelas del espacio. Son el pueblo pobre, la masa trabajadora que ha de resolver a diario de un modo heroico el problema de la existencia. Su lucha por existir en la luz, por llenar de píos y revuelos el silencio torvo del mundo, es una lucha alegre, decidida, irrenunciable. Ellos llegan, por conquistar la migaja de pan necesaria, a lugares donde ningún otro pájaro llega. Se les ve en los rincones más apartados. Se les oye en todas partes. Corren todos los riesgos y peligros con la gracia y la seguridad que su infancia perpetua les ha dado.”

El gorrión y el prisionero

Cuento inconcluso. Miguel Hernández (1910-1942)

Artwork: Vicent Gisbert Cardona @vigiscar

Aun el gorrión halla casa,
Y la golondrina nido para sí, donde ponga sus polluelos,
Cerca de tus altares, oh Jehová de los ejércitos,
Rey mío, y Dios mío.

Salmos 84:3-4

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No he querido dejar que esta parte fuese la última en ser redactada ya que considero que gran parte de culpa de haber llegado hasta aquí está reflejada en esta pequeña sección.

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Abstract

The House Sparrow *Passer domesticus* (Linnaeus, 1758) is on decline almost all over Europe. In Spain, the case of the Valencian Community, the study area of this thesis, stands out with decreases of over 95% in rural areas and 70% in urban areas. The causes of the decline in urban areas are still unclear. In the towns and cities of the Valencian Community, important processes of transformation of the urban structure, which affect the availability and suitability of the habitat for the species, are taking place. The aim of this thesis was to explore these changes as potential drivers of the decline of the House Sparrow. The thesis is divided into 8 chapters. Chapter 1 contains an overview of the biology and status of the House Sparrow in Europe, Spain, and the Valencian Community. Chapter 2 focused on describing the urban characteristics of contemporary cities that affect urban birds from a holistic perspective. In chapter 3 were studied the main factors that determine the abundance of the house sparrow in urban areas of the Valencian Community, such as parks and surface rubbish containers. In Chapter 4 it was found that the remodelling of urban parks with artificial substrates, such as artificial grass, caused population declines of 60% compared to 15% of traditional parks without remodelling. In Chapter 5 it was reported that replacing surface dumpsters with underground dumpsters could have a negative effect on the species. Chapter 6 concerns a study in captivity that found that the urban diet is capable of increasing oxidative stress in individuals, even in an environment without contamination. Chapter 7 of the thesis showed that the recreational noise of the Moors and Christians festivities in spring significantly reduced the breeding success of the House Sparrow. Chapter 8 summarizes and integrates the information from the entire thesis and also includes a series of urban planning and management measures to alleviate the decline of the House Sparrow in the Valencian Community.

Keywords: decline, diet, House Sparrow, parks, *Passer domesticus*, recreational noise, remodelling, trash containers, urban areas, urbanization.

Resumen

El gorrión común *Passer domesticus* (Linnaeus, 1758) se encuentra en declive en gran parte de Europa. En España, destaca el caso de la Comunidad Valenciana, área de estudio de esta tesis, con descensos superiores al 95% en zonas rurales y al 70% en zonas urbanas. Las causas del declive en zonas urbanas no son claras. En los pueblos y ciudades de la Comunidad Valenciana se están produciendo importantes procesos de transformación de la estructura urbana que afectan a la disponibilidad y adecuación del hábitat para la especie. El objetivo de la tesis fue explorar esos cambios como potenciales impulsores del declive del gorrión común. La tesis se distribuye en 8 capítulos. El capítulo 1 contiene una visión general de la biología y del estatus del gorrión común en Europa, España y la Comunidad Valenciana. El capítulo 2 se centró en describir aquellas características urbanas de las ciudades actuales que afectan a la avifauna urbana desde una perspectiva holística. En el capítulo 3, se estudiaron los factores principales que determinan la abundancia del gorrión común en las zonas urbanas de la Comunidad Valenciana, como son los parques y los contenedores de basura de superficie. En el capítulo 4, se demostró que la remodelación de los parques urbanos con sustratos como el césped artificial provocaba descensos poblacionales del 60% frente al 15% de los parques tradicionales sin remodelar. En el capítulo 5, se detectó que reemplazar los contenedores de basura de superficie por contenedores de basura soterrados podía tener un efecto negativo sobre la especie. En el capítulo 6, se realizó un estudio en cautividad y se comprobó que la dieta urbana es capaz de incrementar el estrés oxidativo de los individuos, aún en un medio sin contaminación. El capítulo 7 de la tesis demostró que el ruido recreativo de las fiestas de Moros y Cristianos en primavera reducía significativamente el éxito reproductor del gorrión común. El capítulo 8 recapitula e integra la información de toda la tesis y, además, incluye una serie de medidas de planificación y gestión urbana para paliar el declive del gorrión común en la Comunidad Valenciana.

Palabras clave: contenedores de basura, declive, dieta, gorrión común, parques, *Passer domesticus*, remodelación, ruido recreativo, urbanización, zonas urbanas.

Manuscripts list

The present PhD thesis has been written in the modality of articles compendium. Manuscripts I, II, II, IV, and V are published or accepted while manuscript VI is under review. List of manuscripts in the thesis:

I. **Bernat-Ponce, E., Gil-Delgado, J.A. & López-Iborra, G.M.** (in press). Efectos de las características de las ciudades occidentales contemporáneas sobre la avifauna urbana. *Ecosistemas.*

II. **Bernat-Ponce, E., Gil-Delgado, J.A. & Guijarro, D.** 2018. Factors affecting the abundance of house sparrows *Passer domesticus* in urban areas of southeast of Spain. *Bird Study* **65:** 404–416. Doi: 10.1080/00063657.2018.1518403.

III. **Bernat-Ponce, E., Gil-Delgado, J.A. & López-Iborra, G.M.** 2020. Replacement of semi-natural cover with artificial substrates in urban parks causes a decline of house sparrows *Passer domesticus* in Mediterranean towns. *Urban Ecosystems* **23:** 471–481. Doi: 10.1007/s11252-020-00940-4

IV. **Bernat-Ponce, E., Ferrer, D., Gil-Delgado, J.A. & López-Iborra, G.M.** 2021. Effect of replacing surface with underground rubbish containers on urban House Sparrows *Passer domesticus*. *Urban Ecosyst*ems. Doi: 10.1007/s11252-021-01138-y

V. **Bernat-Ponce, E., Gil-Delgado, J.A. & López-Iborra, G.M.** 2021. Recreational noise pollution of traditional festivals reduces the juvenile productivity of an avian urban bioindicator. *Environmental Pollution* **286:** 117247. Doi: 10.1016/j.envpol.2021.117247

VI. **Bernat-Ponce, E., Gil-Delgado, J.A., Guardiola, J.V. & López-Iborra, G.M.** (Under review in Science of the Total Environment). Experimental effect of anthropogenic food resources available in cities on the body condition, nutritional status and oxidative stress of an urban bioindicator passerine.

Manuscripts are not listed in a chronologic order but in a logical order for a better understanding of the thesis. Listed manuscripts are the main thesis chapters with the original published or submitted format.

The sections of the project that have not been submitted to a journal (Introduction and General discussion) follow an ornithological format (Bird Study journal).

Chapter 1

General introduction

The House Sparrow, our feathered neighbour

The House Sparrow *Passer domesticus* (Linnaeus, 1758) is a brown small-sized bird species and one of the most recognizable birds all over the world, which has been associated with human settlements since the Bronze Age (Ericson et al. 1997). Originally from the Middle East, the development of agriculture, 10,000 years ago, and the stored grain in human settlements triggered its expansion, by commensalism, to most parts of Eurasia and Northern Africa (Anderson 2006). Natural selection led to genetic changes that shaped their skull and allowed them to digest the abundant rich-starch food grown by humans around their settlements (Ravinet et al. 2018). Furthermore, as a natural cavity nester, humans' first stable settlements with permanent structures also provided new nesting resources for the species from which House Sparrows took advantage (Summers-Smith 1988). For instance, the general current paradigm of the urbanization process of bird species is represented by the House Sparrow: hole-nesting species became urban as soon as they found cavities in buildings and there was food for them in the settlement itself or the surroundings (Summers-Smith 1988).

Nowadays, the House Sparrow presents a wide distribution almost all over the world, always closely linked to human presence, in rural and urban areas (Summers-Smith 1988, Cramp & Perrins 1994, BirdLife International 2021). This wide range of distribution is a consequence of anthropogenic processes, such as deliberate releases of individuals into non-native areas but also to ship-borne dispersal (Summers-Smith 1988, Cramp & Perrins 1994). Furthermore, its adaptability, opportunism, human tolerance, and use of resources provided, usually unintentionally, by people (e.g. food, nesting sites) make this commensalist species a successful exotic invader in non-native areas (Summers-Smith 1980, Sol et al. 2012). The House Sparrow has been traditionally considered as a model species of an urban exploiter, a species that has followed urbanization and is adept to exploit these changes reaching their highest densities in developed sites (Blair 1996, McKinney 2002, Anderson 2006, Kark et al. 2007, Seress & Liker 2015). But, even though the House Sparrow is widely distributed and abundant all around the world; their populations have experienced sharp declines during the last century and the first decades of the present one (BirdLife International 2021). These declines became so important that House Sparrows are practically extinct in important European cities like Glasgow, Edinburgh, London, and Ghent (Summers-Smith 2003) but also in rural areas all over the continent (Gil-Delgado et al. 2002, Hole et al. 2002). This situation worried the scientists but also the general public, so much so that one of the most important British newspapers came to offer a reward of £5,000 to those who solved the mystery of the disappearance of the Londoner sparrows (Crick et al. 2002).

The decline of the House Sparrow

The House Sparrow decline was first noticed in the 1920s in Great Britain, particularly in built-up areas, which coincided with the replacement of the horse by the internal combustion engine (Summers-Smith 2003). During the last decades of XXth century and the beginning of the XXIst, the House Sparrow has experienced sharp declines in rural and urban areas all over its native area, the Western Palaearctic (Gil-Delgado et al. 2002, Summers-Smith 2003, Chamberlain et al. 2005, Anderson 2006, Shaw et al. 2008, Dadam et al. 2019, Mohring et al. 2021). From 1980 until 2017 the species declined 63 % in overall Europe (Figure 1) and in some European countries; the decline has been really significant, reaching even widespread local extinctions (Hole et al. 2002). So much so that in the United Kingdom the species has been included in the Red List for Birds and has the highest conservation priority (Eaton et al. 2015) where a gradual decline since 1990 has ended in a massive decrease and almost a complete extinction in some urban centres and farmland areas (Hole et al. 2002, Summers-Smith 2003).

On the other hand, the species is not considered as globally threatened and is not included in the Red List of BirdLife International (BirdLife International 2021). The reasons that explain this divergent situation are that the worldwide decline is not believed to be sufficiently rapid to approach the thresholds for the “Vulnerable” category under the population trend criterion (does not present >30% decline over ten years or three generations) and the population size is extremely large (does not present <10,000 mature individuals with a continuing decline estimated to be >10% in ten years or three generations, or with a specified population structure) (BirdLife International 2021). It should be highlighted that human-dependent bird species, like the sparrows, are more likely to display negative population trends in the most heavily human-modified environments (Siriwardena et al. 1998). Accordingly, it is appropriate to investigate the declines in House Sparrow numbers in relation to patterns of temporal changes in human activities, especially in urban areas (Anderson 2006).

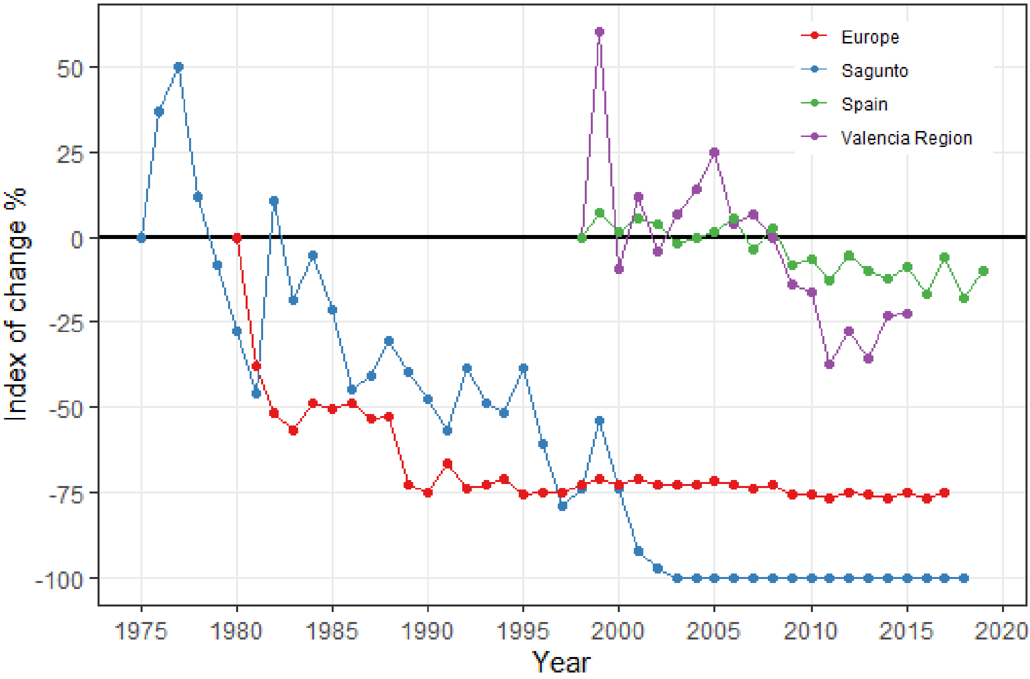

Figure 1. Declining trend of the House Sparrow in Europe (continental scale) between 1980 and 2017 (red), in Spain (country scale) between 1998 and 2019 (green), in Valencia Region (regional scale) between 1998 and 2015 (purple), and Sagunto (local scale) between 1975 and 2020 (blue). Own elaboration with data from Vera and Escandell (2016), EBCC/BirdLife/RSPB/CSO (EBCC 2019), Gil-Delgado (pers. comm. of the number of pairs nesting in orange trees), and SEO/BirdLife (2020).

In Spain, the decline of the species is not so marked when compared to other countries and a 10 % fewer sparrows are found in 2019 than in 1998 (SEO/BirdLife 2020) (Figure 1). However, its decline has worried the Spanish Society of Ornithology (SEO/BirdLife) who in 2016 declared the House Sparrow as “The bird of the year (El Ave del Año)” to show and report the negative trend of the species and to highlight the causes of its decline, especially in urban areas. However, in the Valencian Region (Eastern Spain), the first area where the decline was reported in rural areas (Gil-Delgado et al. 2002), the status of the populations of the House Sparrow is worse than in other Spanish regions. In Sagunto’s orange groves, the breeding pairs of the species declined by 95% between 1975 and 2001 (Gil-Delgado et al. 2002). Nowadays, no House Sparrows are breeding in these trees (Gil-Delgado pers. comm.). Also, in the city of Valencia, Murgui and Macias (2010) detected that urban park populations declined by 70% in ten years (1998-2008). With all, the populations of the Valencian Community have declined more than 20% between 1998 and 2015 (Vera & Escandell 2016) (Figure 1). For these reasons, this region is one of the best areas to study the decline of the House Sparrow in Spain.

Rural decline

House Sparrow declines have occurred in both urban and farmland areas (Summers-Smith 2000, Crick et al. 2002, De Laet & Summers-Smith 2007), even though the decline first started in rural areas (Summers-Smith 2003, Robinson et al. 2005). As an example, in agricultural areas of England, populations of the species showed declining trends since 1970 and in

Spanish rural areas, this trend was also detected since 1975 (Siriwardena et al. 1998, Gil-Delgado et al. 2002, Hole et al. 2002).

Changes in agricultural practices and their intensification seem to be behind the global decline detected in rural and farmland areas (Gil-Delgado et al. 2002, Hole et al. 2002, Anderson 2006). This is an internationally accepted theory to explain the decline of the House Sparrow decline in villages, farms, and small towns surrounded by crops. The main changes in agricultural practices are the switch from spring- to autumn-sown cereals, the increased use of pesticides/herbicides, the switch from haymaking to cutting for silage before the grass has set seed, and the reduced spillage of grain and improved storage to meet EU regulations (Summers-Smith 2003). New methods, such as flood-watering instead of drip irrigation and the use of soil-compacting machinery, reduce the availability of ruderal plants too and hence trophic resources for birds (Gil-Delgado et al. 2002). Also, the abandonment of crops, the urbanization of rural areas, the abandonment of traditional farms with livestock, and even its substitution for industrial farms could have affected the species (Engler & Bauer 2002).

These set of changes have probably reduced trophic resources availability, both of seeds, which sustain birds throughout the year, and of invertebrates, which are required by House Sparrows for feeding nestlings and juveniles (Summers-Smith 1988, Anderson 2006). Studies have suggested that a major impact has been a reduction in the survival rate (Siriwardena et al. 1998), specifically of first-year birds (Crick et al. 2002), rather than a decrease in productivity. Furthermore, this could have lead to local extinctions (Gil-Delgado et al. 2002, Hole et al. 2002). Also, due to food shortage, increased predation pressure, low survival, and reduced movements of the species (Snow & Perrins 1998), the recruitment to sink populations from sources could be restricted (Hole et al. 2002, Anderson et al. 2006). For that reason, Hole et al. (2002) stated for rural areas that local extinction without recolonization may eventually reduce migration between populations as a result of greater separation between remaining populations, a compounding effect that could pervade the rural landscape and which has been predicted by classical metapopulation theory (Hanski 1998).

Urban decline

The causes that explain the decline of the rural populations of House Sparrows seem clear and agreed scientifically. However, there is no clear consensus about a single cause or group of causes that could explain the reported declines in urban areas (Anderson 2006, Figure 2) that even have ended in almost the extinction of the sparrow in some urban centres (Summers-Smith 2003). Since urban declines started in the XXth century, particularly in Great Britain and Western Europe, several hypotheses have been exposed to explain these steep negative trends. However, the most favoured hypothesis to explain the decline of House Sparrows in urban areas is that it is down to a combination of factors rather than to a single one (Anderson 2006). Even though most suggested potential causes for the decline could be encompassed by the increasing urbanization process of our towns and cities, scientists are still struggling to identify which of these are the key urban factors of plummeting populations (Vincent 2005, Anderson 2006). In the following pages can be found a summary of the most important hypotheses suggested to explain the decline of this iconic species:

One of the most famous hypotheses is the increase of predation pressure (Figure 2A) that urban individuals have suffered by Domestic Cats *Felis silvestris catus* or Eurasian Sparrowhawks *Accipiter nisus* (Woods et al. 2003, Chamberlain et al. 2009, Bell et al. 2010). The British House Sparrow populations' long-term release from natural rural predator pressure could have made them especially vulnerable when urban habitats were recently colonized by rural Eurasian Sparrowhawks (Bell et al. 2010). However, this cause seems unlikely to explain the decline in other European countries, such as Spain, where Sparrowhawks tend to avoid urban areas and House Sparrows are still declining there (Zuberogoitia 2012).

Another suggested cause to explain the reduction in the number of indivivuals in Sparrow populations is the increase in competitors in urban areas (Figure 2B). House Sparrows have been human-dependent since the Bronze Age (Ericson et al. 1997, Ravinet et al. 2018). However, the increasing supply of anthropogenic food in cities attracts other bird species that might outcompete the House Sparrow (Kristan et al. 2004). Skórka et al. (2016) found that the abundance of corvids, such as magpies *Pica pica*, were negatively correlated with abundances of House Sparrows and highlighted it as a piece of evidence for competition between these species. Invasive species, such as the Rose-ringed Parakeets *Psittacula krameri*, have also been recorded fatally attacking House Sparrows (Covas et al. 2017). Also, competition for nest sites could be attributed as a potential cause of the decline (Weitzel 1988, Dinetti 2008, Charter et al. 2016). Finally, the competition for food with the Feral Pigeon *Columba livia* and its increasing populations has also been suggested as a potential cause for the decline (Summers-Smith 2003, Dinetti 2008, Murgui & Macias 2010).

Another of the most accepted hypotheses that explain the urban decline of the species is the loss of nesting sites in towns and cities which would be linked with a breeding success reduction (Figure 2C). New urbanization trends, innovative roof materials and high buildings modify the urban structure, usually reducing the number of holes and crevices available for the hole-nesting birds (Wotton et al. 2002, Moudrá et al. 2018). Furthermore, the urban socioeconomic increase of the last decades has allowed house refurbishments and repairments with new materials that improve isolation reducing leaks and crevices with negative effects on building-nesters (Shaw et al. 2008).

The reduction in heterogeneity, number and size of urban green areas linked to new urban planning is another of the main causes suggested for the decline. This increasing urbanization trend of green areas has been commonly linked to a food shortage for urban birds, and especially for the House Sparrow (Figure 2D). These areas are the main suppliers of seeds and invertebrates for urban birds and their habitat suitability reduction will be determinant for the urban birds' future (McKinney 2002, Vincent 2005, Shaw et al. 2008, Peach et al. 2015).

In the last decades, new strains of diseases have been detected in the House Sparrow (Figure 2E), such as avian conjunctivitis and West Nile virus (Anderson 2006). Recently Dadam et al. (2019) reported that on average the 74% of House Sparrows in suburban London were infected by avian malaria *Plasmodium relictum*, which they linked with the decline of the species. However, studies carried out in urban areas of Spain and France showed that the prevalence of the disease in urban areas was more reduced (Jiménez-Peñuela et al. 2019, Bichet et al. 2020, Magallanes et al. 2020). Thus, these divergent results make necessary further studies to assess the real significance of these diseases as a key factor for the species decline.

Cities are a constant and predictable supplier of low-quality anthropogenic food. Intentionally or casually humans offer their food scraps to urban birds in feeders, trash containers, bar terraces, and dumpsters (Bokotey & Gorban 2005, Haemig et al. 2015, Herrera-Dueñas et al. 2015, Reynolds et al. 2008, Spelt et al. 2021). While rural birds mainly feed on crops, poultry food, wild seeds, and invertebrates (Cramp & Perrins 1994, Anderson 2006), these trophic resources are scarcer in urban environments. Thus anthropogenic food might seem to increase their overwinter survival and keep larger populations (Seress & Liker 2015); it does not contain the required amounts of nutritional elements, such as amino acids and/or vitamins (antioxidants). However, it contains an excess of fats, sugars, carbohydrates, or even proteins which can alter some blood plasma biochemical parameters (Ishigame et al. 2006, Jones & Reynolds 2008, Townsend et al. 2019). The alteration of uric acid, glucose, total protein, and cholesterol levels in birds can lead to kidney disease, anorexia, starvation, and heart disease, respectively (Macwhirter 1994, Rosskopf & Woerpel 1996, Fudge 2000, Harr 2002). For that reason, the effects of urban diet on the species survival or breeding success are still unknown and worth studying (Figure 2F).

Besides, urbanization is usually linked to an exponential increase in the emission of pollutant chemicals into the air, especially NO_x, which can have pernicious effects on urban birds (Salmón et al. 2018). Urban populations of House Sparrows exhibit an increased oxidative stress level when compared to their rural conspecifics (Herrera-Dueñas et al. 2017) and pollution has also been proposed as a potential cause for the decline of urban populations (Summers-Smith 2007, Peach et al. 2018, Figure 2G). Air pollution (chemical pollution) is the main suggested cause to explain the alteration of the prooxidant–antioxidant balance in favour of the former leading to potential damage (e.g. lipid, protein, DNA) caused by ROS (Reactive Oxygen Species) (Sies, 1991, Costantini 2019, Herrera-Dueñas et al. 2017). Environmental influences on oxidative stress can be multiple, such as pollution, radiation, disease, and food intake (Isaksson 2018). But another potential explanation for the high oxidative stress of urban individuals is related to their poor-quality diet. Urban diet might also be behind the observed oxidative processes (Isaksson 2015). However, no studies have been carried out to disentangle if higher oxidative stress of urban birds is mainly due to diet, pollution, or both acting synergistically.

Finally, new stressors, such as sensory pollutants (e.g. noise, light, or radiation) are emerging during the last decades and might have also a negative influence on sparrows' populations (Figure 2H). For example, anthropogenic noise reduces their breeding success (Schroeder et al. 2012, Meillère et al. 2015), artificial light at night (ALAN) could be interrupting their sleep, modifying their biological rhythms, and reducing their hatching success (Dominoni et al. 2013, Senzaki et al. 2020), and even electromagnetic radiation from phone masts might reduce their abundance in certain locations of urban areas (Balmori & Hallberg 2007).

Every decade has new contributions to clarify why the House Sparrows have declined so sharply in their native area, the Western Palearctic. However, seems improbable that one of the aforementioned causes on their own could be the only responsible for plummeting populations in that way. Nowadays, the most accepted hypothesis is that the global decline might be due to a combination of factors (Figure 2), some of them with more relevance than others depending on the countries, which synergistically may act against the conservation of this urban exploiter.

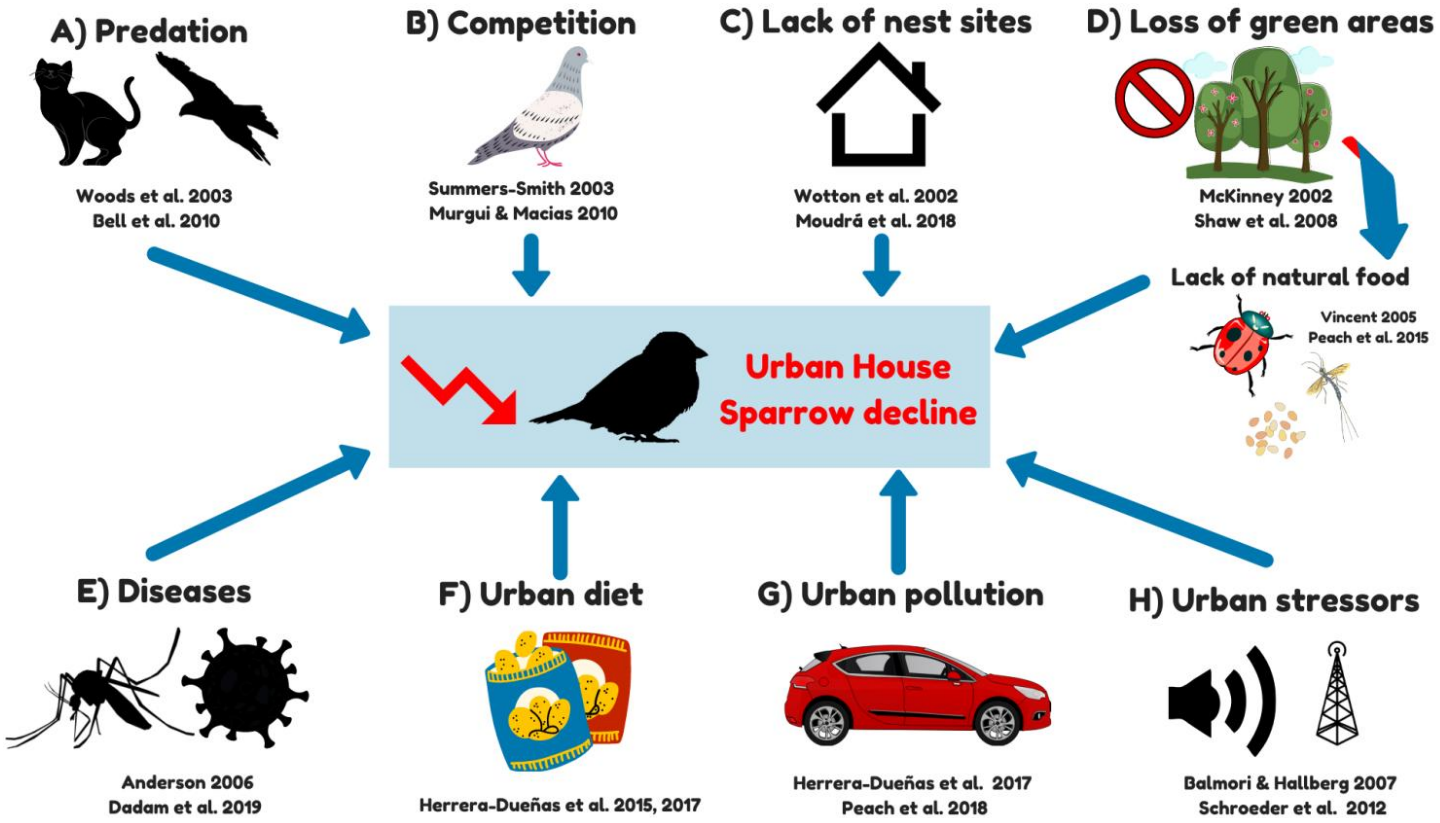

Figure 2. Summary of the main hypotheses suggested explaining the decline of the House Sparrow in urban areas of the Western Palearctic.

The House Sparrow as an urban health bioindicator

The House Sparrow is an extremely sedentary bird that presents a reduced urban home-range with very low-plasticity (Vangestel et al. 2010). When breeding, it usually feeds in a radius of fewer than 100 m around the nest, and out of breeding season its foraging area rarely exceeds two kilometres (Vincent 2005, Anderson 2006, Vangestel et al. 2010). Furthermore, their natal dispersion distances are relatively short and breed usually very close to their natal colonies (Vincent 2005, Anderson 2006). Wild bird species are sensitive to pollutants and rapid changes in their habitats what makes them important structural components of the ecosystem (Swaileh & Sansur 2006, Kekkonen et al. 2011, Berglund et al. 2014). For all the aforementioned, the House Sparrow's status, body condition, abundance, breeding parameters, and movements are a good reflection of the habitat quality where it lives and converts the species into a useful bioindicator for urban areas.

Justification and aims of the thesis

The main objective of this thesis is to shed more light on potential factors that explain that one of the most human-linked and known urban birds, the House Sparrow *Passer domesticus*, is drastically declining in urban areas of Valencia Region (Eastern Spain). The present project also pretends to suggest landscape and urban planning measures useful for the conservation of the species in urban areas of Eastern Spain but also to produce applicable knowledge for other parts of Spain and Europe. This research project covers the study of the focal species from an integrative ecological perspective including habitat selection, nutritional physiology, oxidative stress, conservation ecology, breeding success, and populations dynamics.

This thesis is distributed into 8 chapters. The present Chapter (Chapter 1) is a general introductory glimpse into the biology of the species and its present status in Europe, Spain, and the Valencia Region, the geographical area where the present project was developed. In the introduction section, it is explored its decline, especially focused on urban areas, and its main potential causes found in the literature. Furthermore, the initial chapter includes the justification of the thesis and its main objectives. Chapter 2 is also an introductory section focused on describing the effects of contemporary cities features on urban avifauna from a holistic and general vision. Chapter 3 aims to detect which were the determinant urban factors for the abundance of the House Sparrow in the Valencian Community. In the Chapter 4, there were explored the effects of urban park reurbanizations as they might negatively impact the House Sparrow local populations. In Chapter 5 the potential effects of replacing surface rubbish containers with underground ones which might reduce anthropogenic food resources were linked to the abundance indexes of the species. In the Chapter 6 was carried out a captivity experiment to study if the higher oxidative stress found in urban individuals might be due to urban diet (processed food) or to urban pollutants. Chapter 7 was focused on the exploration of the potential effect of urban recreational noise (traditional festivities with fireworks) on urban birds' breeding performance using the House Sparrow as a model. The last chapter (Chapter 8) collects and discusses the information of the whole thesis and also includes urban planning and management measures to stop the decline of this bioindicator species in the Valencian Community and hence in other parts of Spain and Europe. The final section of the thesis includes the main conclusions of each one of the chapters.

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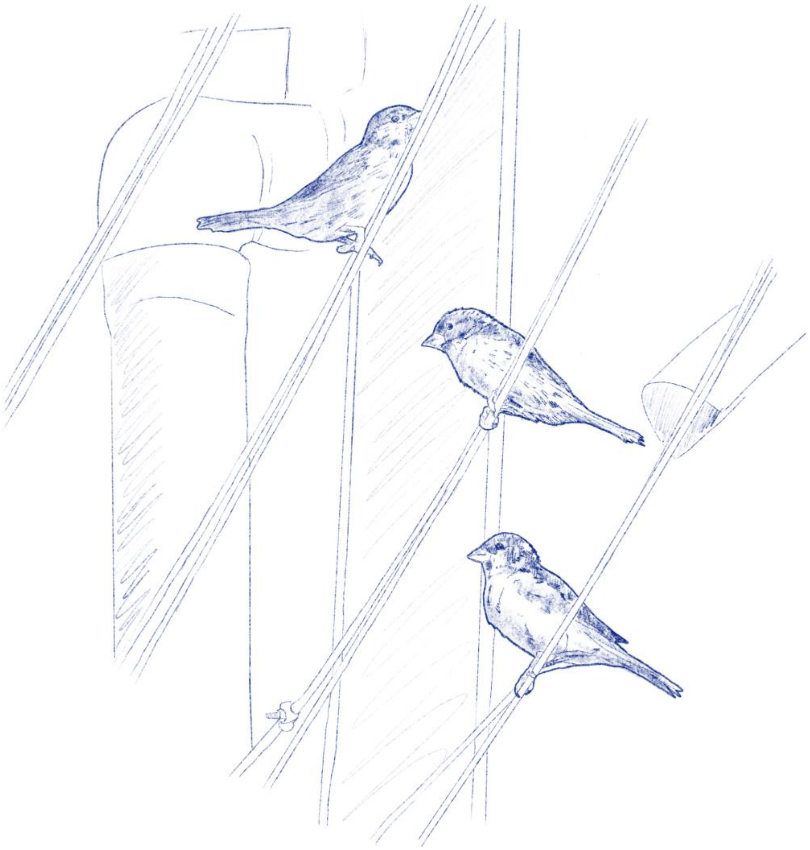
Chapter 2

Efectos de las características de las ciudades occidentales contemporáneas sobre la avifauna urbana

Effects of the features of contemporary occidental cities on urban avifauna

Edgar Bernat-Ponce, José Antonio Gil-Delgado, & Germán Manuel López-Iborra



Ecosistemas

EFECTOS DE LAS CARACTERÍSTICAS DE LAS CIUDADES OCCIDENTALES CONTEMPORÁNEAS SOBRE LA AVIFAUNA URBANA

EFFECTS OF THE FEATURES OF CONTEMPORARY OCCIDENTAL CITIES ON URBAN AVIFAUNA

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Resumen

Desde los primeros asentamientos humanos permanentes del Holoceno, hace 10 000 años, hasta las ciudades contemporáneas, las aves han cohabitado con los seres humanos. En las últimas décadas la urbanización ha crecido exponencialmente en el planeta y, en 2030, más del 60% de la población mundial vivirá en zonas urbanas. En función de su tolerancia a la urbanización las aves se clasifican en tres categorías: evitadoras, adaptadoras o explotadoras urbanas. Las ciudades occidentales contemporáneas atraen a las aves por la presencia de recursos tróficos abundantes y predecibles, la reducción en la diversidad de depredadores o la provisión de estructuras donde ubicar los nidos, entre otras. Sin embargo, la urbanización es uno de los mayores problemas actuales para la biodiversidad y el modelo de ciudad contemporánea puede dejar de ser atractivo para las aves e incluso causar el declive a ciertas especies ligadas a medios urbanos. Algunas razones que explican este proceso son: la gestión urbana y la pérdida de zonas verdes, la contaminación, la comida antropogénica y las nuevas tendencias arquitectónicas. Un cambio en el modelo de ciudad contemporánea que proteja la biodiversidad, aunque es un reto difícil, es posible siguiendo ejemplos como el de la infraestructura verde y sostenibilidad ambiental de la ciudad de Vitoria-Gasteiz (España).

Palabras clave

Área urbana; aves; contaminación; declive; infraestructura verde; urbanización.

Abstract

Since the first permanent human settlements in the Holocene, 10 000 years ago, until contemporary cities birds have cohabited with humans. In the last decades, urbanization has grown exponentially on the planet and in 2050 the 60% of the world population will live in urban areas. Based on their urbanization tolerance birds are classified into three categories: urban avoiders, adapters, or exploiters. Current occidental cities attract birds due to the presence of abundant and predictable trophic resources, reduction of predators' diversity, or the existence of nesting places, among other factors. However, urbanization is nowadays one of the greatest problems for biodiversity and the model of a contemporary city may become unattractive to birds and indeed some urban species are declining in the present. Some reasons that explain this process are loss of green areas, pollution, changes in abundance and composition of trophic resources, and new building trends. Changing the model of the present city that protects is a difficult challenge but it is possible and there exist inspiring experiences like the green infrastructure of the city of Vitoria-Gasteiz (Spain).

Key words

Urban area; birds; pollution; decline; green infrastructure; urbanization.

La urbanización, el modelo actual de ciudad occidental y su relación con las aves

Los primeros asentamientos humanos permanentes surgieron hace más de 10 000 años con la aparición de la agricultura, y fue el inicio de la coexistencia pre-urbana entre las aves y los humanos (Negro et al. 2020). Muchas ciudades contemporáneas son producto de asentamientos antiguos que funcionan como un ecosistema vivo en permanente cambio.

La urbanización es el proceso de conversión de hábitats rurales a urbanos/urbanizados (Clergeau et al. 2006; Elmqvist et al. 2013). Las zonas rurales se pueden definir como paisajes en los que predomina el uso agrícola-ganadero o la cobertura de vegetación natural o espontánea, localizados fuera de pueblos y ciudades (Clergeau et al. 2006). Las zonas urbanizadas se caracterizan por una densidad moderada o elevada de casas unifamiliares con espacios privados seminaturales como jardines o solares (grado medio de urbanización). No obstante, las zonas urbanas (grado alto de urbanización) son áreas con una densidad de habitantes superior a las 1000 personas/Km^2 cubiertas por edificios de varias alturas y más de un 50 % de superficie impermeable al agua (Marzluff et al. 2001; MacGregor-Fors 2011).

La urbanización supone una gran amenaza para la biodiversidad y es uno de los mayores desafíos ambientales de nuestra época (United Nations 2012; Isaksson 2018). Desde la Segunda Guerra Mundial este proceso ocurre a una velocidad sin precedentes, y se estima que en 2030 más del 60% de la población mundial vivirá en zonas urbanas (Fernández Durán 2011; United Nations 2018). Dada la diversidad de características y complejidad del proceso de urbanización a lo largo del mundo, en esta revisión nos centramos en las ciudades occidentales contemporáneas y sus efectos sobre las aves desde el final del siglo XX hasta la actualidad. Este tipo de ciudades comparten grandes similitudes de planificación urbanística y de desarrollo social y urbano debido a la globalización, en muchos casos sin importar su localización geográfica (Gordon y Cox 2012; Clark et al. 2019): desindustrialización en los 70, gentrificación (transformación de un espacio urbano deteriorado mediante la rehabilitación de edificios con mayores alturas que las existentes y desplazamiento de los habitantes de clase media-baja por las clases altas) en los 80, mejora en el transporte en los 90 y gran ampliación de los núcleos urbanos a partir del año 2000 (Clark et al. 2019). Una de las grandes revoluciones en las ciudades de este periodo fue la masificación en el uso del automóvil privado y el desarrollo de estructuras a su servicio (López de Lucio 1993).

En función de su tolerancia a la urbanización y dependencia de los recursos provistos por los humanos, las aves se han clasificado en: evitadoras, adaptadoras o explotadoras urbanas (Blair 1996). La urbanización puede provocar la expulsión de especies de aves de zonas altamente modificadas. Estas son las llamadas evitadoras urbanas (Croci et al. 2008); especies que para su alimentación y nidificación requieren extensiones de hábitats sin urbanizar, o con una urbanización muy reducida (**Fig. 1**; Møller 2009). Por otra parte, las aves adaptadoras urbanas son aquellas que toleran (vía plasticidad o evolución) niveles intermedios del gradiente de urbanización, aunque también se encuentran en otros ambientes (**Fig. 1**; Croci et al. 2008). Finalmente, en los ecosistemas altamente urbanizados existen especies adaptadas y pre-ajustadas para explotar estos cambios ambientales, y que apenas se encuentran en otros ecosistemas (**Fig. 1**; Croci et al. 2008). Estas especies son las explotadoras urbanas (Blair 1996), las cuales suelen ser generalistas, trogloditas (nidifican en oquedades) y con dietas amplias (omnívoras e insectívoras aéreas) que obtienen sobre la superficie del suelo o durante el vuelo (Erz 1966; Kark et al. 2007; Evans et al. 2011). Las explotadoras urbanas muestran una tendencia a desarrollar un mayor tamaño cerebral relativo y una mayor plasticidad comportamental (Evans et al. 2011; Sol et al. 2013; Griffin et al. 2014; Palacio 2020). Asimismo, las dos últimas categorías de especies, las adaptadoras y explotadoras urbanas, se engloban dentro de las denominadas especies sinantrópicas, las cuales prosperan en hábitats modificados por la acción humana (**Fig. 1**; Martin y Boruta 2014). Además, también pueden ser clasificadas como especies sinúrbicas si estas alcanzan mayores densidades en el medio urbano que en zonas rurales (**Fig. 1**; Francis y Chadwick 2012).

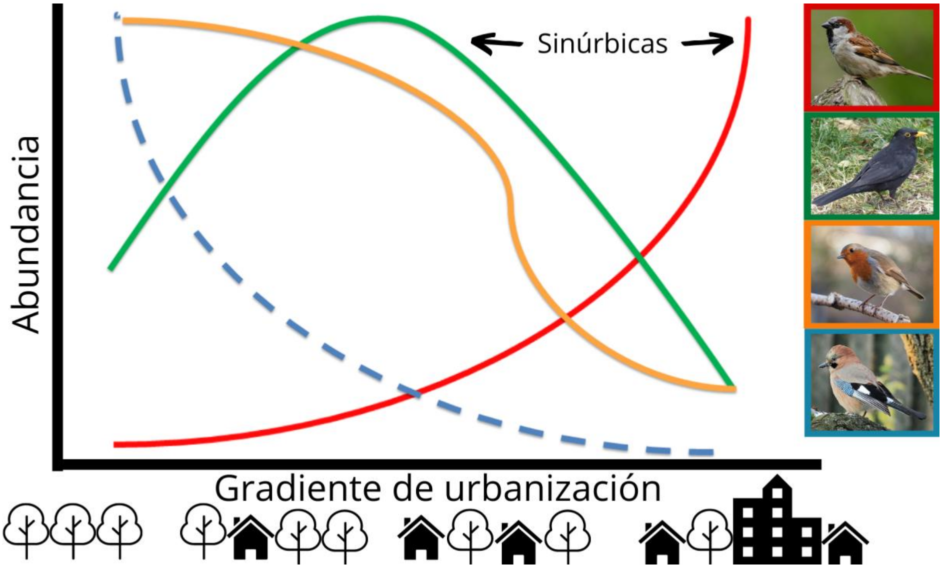

Figura 1. Ejemplos y representación gráfica de la abundancia teórica de las tres categorías de aves en función de su tolerancia a la urbanización. En rojo se muestra una especie explotadora urbana sinúrbica (Gorrión común *Passer domesticus*) y que no se encuentra fuera de entornos antropizados (Anderson 2006). En verde se encuentra un ejemplo de ave adaptadora urbana sinúrbica (Mirlo común *Turdus merula*), especie forestal que en los espacios verdes de la ciudad, como los jardines y parques, presenta densidades muy superiores a las que muestra en los bosques ocupados originalmente (Luniak 1970; Møller et al. 2014). En naranja encontramos otro ejemplo de ave adaptadora urbana (no sinúrbica) que tolera la urbanización, aunque su abundancia en estas zonas no sea mayor que en áreas rurales (Petirrojo europeo *Erithacus rubecula*). En azul se representa la evitadora urbana (Arrendajo euroasiático *Garrulus glandarius*), especie que requiere de grandes masas forestales sin o con una urbanización muy leve para su alimentación y reproducción. Las líneas continuas muestran las tres especies sinantrópicas. Elaboración propia en Canva.

¿Por qué las ciudades actuales son atractivas para algunas especies de aves?

A pesar de los efectos perniciosos de la urbanización sobre la biodiversidad, las ciudades son un ecosistema único utilizado por decenas de especies de aves (Gil y Brumm 2014). En ciudades como Madrid (España) se llegan a catalogar más de medio centenar de especies (SEO/BirdLife 2006). Además, el 20% de las especies del mundo han sido detectadas en áreas urbanizadas e incluso en ciertos ambientes su diversidad es mayor que en zonas naturales próximas (Aronson et al. 2014; Verma y Murmu 2015), por lo que el ecosistema urbano tiene potencial para contribuir a la conservación de determinadas especies si es gestionado de manera favorable para la biodiversidad. Las características más destacadas que atraen a algunas especies de aves a las ciudades occidentales contemporáneas son:

Fuente de alimento constante y predecible

Las ciudades suponen una fuente constante, abundante, variada y predecible de algunos alimentos, generalmente derivados de la actividad humana, aspecto que es especialmente importante para las aves omnívoras (Møller 2009). Estas aves se alimentan, en muchos casos, de los desperdicios de comida que los humanos generamos y que ponemos a su disposición de manera constante en determinados lugares de la matriz urbana (vertederos, contenedores, terrazas de restaurantes), que afectan a su abundancia y presencia (**Fig. 2a**; **Tabla 1a**). Habitualmente, los jardines particulares en muchos países constituyen fuentes fijas y predecibles de alimento gracias a los comederos para aves (**Fig. 2c**; Reynolds et al. 2017) y, a su vez, son un punto de atracción para depredadores como el gavilán común *Accipiter nisus* (Chamberlain et al. 2009b). Asimismo, las zonas verdes urbanas son una importante fuente de recursos tróficos para especies de aves granívoras e insectívoras. En algunos casos, la introducción de especies vegetales exóticas (p. e. palmera datilera *Phoenix dactylifera*), junto a las especies vegetales nativas, permiten un suministro continuo y predecible de frutos e incluso de invertebrados durante diferentes periodos del año (Smith et al. 2006; Murgui 2009; Nieves y Gil-Delgado 2018; Tasker et al. 2020). Esta mayor disponibilidad de alimentos en la ciudad puede incrementar la supervivencia y contribuir a un adelanto de la reproducción (**Tabla 1b**). Además, la iluminación artificial de las zonas urbanas atrae insectos y permite la alimentación nocturna de especies migratorias y vulnerables, como el cernícalo primilla *Falco naumanni* (Negro et al. 2000; Lebbin et al. 2007).

Efecto isla de calor

La isla de calor urbana (ICU) es un fenómeno que se da frecuentemente en las ciudades de climas templados y fríos en el cual las urbes están varios grados por encima de las áreas rurales adyacentes (Bornstein 1968; Landsberg 1981). Este efecto es debido a la generación de calor por la quema de combustibles fósiles y otras actividades residenciales/industriales de origen antrópico, así como a la función colectora/emisora de calor del asfalto y cemento en la ciudad (Botkin y Beveridge 1997). La magnitud de este efecto es variable en función del tamaño y áreas verdes del área urbana (Oke 1973; Shishegar 2014).

Las temperaturas más elevadas (microclima urbano) debidas a la ICU contribuyen, junto con la mayor disponibilidad de alimento para algunas especies, a que las especies migratorias lleguen antes a la ciudad, a que las poblaciones urbanas inicien antes sus puestas que sus congéneres rurales y a que el periodo reproductivo se extienda más en el tiempo (**Tabla 1b**). La ICU se considera universal y sus efectos se han observado en muchas especies, pero no siempre son positivos para su éxito reproductor (Chamberlain et al. 2009a; Deviche y Davies 2014). Asimismo, también tiene importantes consecuencias en el comportamiento, supervivencia y balances energéticos de las aves, como por ejemplo, su asociación en dormideros urbanos (**Fig. 2b**) para protegerse de las bajas temperaturas de las noches invernales (**Tabla 1b**).

Reducción en la diversidad de depredadores

Dado que las aves pequeñas toleran mejor la presencia del hombre que sus depredadores silvestres de mayor tamaño, especialmente aves rapaces (Luniak 2004), las zonas urbanas pueden actuar como refugios frente a la depredación (**Tabla 1c**; pero véase Kettel et al. 2019). Además, en las ciudades, la riqueza de depredadores silvestres suele ser menor que en las zonas rurales (Díaz et al. 2013; Rebolo-Ifrán et al. 2017), por lo que esta liberación frente a la depredación puede mejorar los parámetros demográficos de las especies de presas en las ciudades (**Tabla 1c**).

Sustratos para nidificación

Algunas especies de aves se benefician de estructuras antrópicas urbanas, tanto antiguas como modernas, para la nidificación (**Tabla 1d**). En las zonas urbanas, los campanarios, iglesias y otros edificios son útiles por sus tejas y orificios para la nidificación de especies trogloditas por los elementos salientes como los aleros para la ubicación de los nidos o directamente por la superficie de nidificación que ofrecen los tejados para especies de gran tamaño (**Tabla 1d**). Los edificios altos de las ciudades contemporáneas sirven como oteadero y zona de nidificación de rapaces como el halcón peregrino *Falco peregrinus* (Fraissinet y De Rosa 2012; Kettel et al. 2019). Igualmente, otros elementos antrópicos como los puentes, postes de electricidad o estructuras industriales son importantes sustratos de nidificación para muchas especies de aves (**Fig. 2d***;* Mainwaring 2015).

Hábitats que faltan en el entorno natural

Las zonas urbanizadas pueden albergar hábitats que no se encuentran en sus inmediaciones y que pueden suponer fuentes de recursos esenciales. Por ejemplo, la creación de lagos o estanques artificiales dentro de las urbes incrementa la diversidad ornítica (Shwartz et al. 2008; Chamberlain et al. 2020). En los sectores de mayor estatus económico es más probable que se generen hábitats artificiales que de forma natural no se encuentran en el entorno de las ciudades. Este efecto es especialmente patente en zonas residenciales de áreas áridas o semi-áridas, donde la mayor disponibilidad de recursos para invertir en irrigación de jardines privados, la presencia de cuerpos de agua (p. e. piscinas o estanques), el manejo de la vegetación y la mayor cobertura vegetal son de gran relevancia para las aves (Hope et al. 2003; Leong et al. 2018; Bermúdez-Cavero et al. 2021). Esta situación recibe el nombre de "efecto lujo". No obstante, este efecto también puede deberse a la selección de las zonas más biodiversas para establecer los barrios de población de mayor poder adquisitivo (**Tabla 1e**; Jenerette et al. 2011; Chamberlain et al. 2020; Kuras et al. 2020).

Figura 2. Ejemplos de diferentes elementos de la matriz urbana que atraen a las aves a las ciudades: a) contenedores de basura); b) grandes árboles dormidero; c) comederos para aves; d) estructuras antrópicas útiles para la nidificación.

Tabla 1. Ejemplos de algunas características urbanas que pueden actuar como atractivos para las aves y sus efectos.

Característica urbana	Localización	Especie	Efecto	Referencias
a) Fuente de alimento constante y predecible	Norte de Bélgica	Carbonero común *Parus major*	Individuos urbanos inician las puestas seis días antes que sus conspecíficos rurales pero su éxito reproductor (0.62 volantones/huevo) es menor que en rurales (0.84)	de Satgé et al. 2019
	Bristol (Reino Unido)	3 especies de gaviotas	Ajuste de desplazamientos urbanos en función de la disponibilidad de alimento antropogénico en colegios, vertederos y parques	Spelt et al. 2021
	Suecia, España	Gorrión común *Passer domesticus*	Mayor presencia cerca de las terrazas de restaurantes y mayores abundancias en los alrededores de los contenedores de basura	Haemig et al. 2015, Bernat-Ponce et al. 2018b, 2019
	Pensilvania central (EEUU)	Carbonero cabecinegro *Poecile atricapillus*	Individuos con acceso a comida suplementaria en zonas suburbanas incrementan sus tasas de supervivencia frente a los de zonas control forestales (94% vs. 81%)	Egan y Brittingham 1994
	Londres (Reino Unido)	Gorrión común *Passer domesticus*	Mayor número de volantones (+62%) en zonas suburbanas con alimentación suplementaria frente a zonas no suplementadas	Peach et al. 2015
b) Efecto isla de calor	Łódź (Polonia)	Carbonero común *Parus major*	Puestas en zonas urbanas 3-16 días antes que en zonas rurales gracias al microclima urbano, entre otros aspectos	Wawrzyniak et al. 2015
	Oeste de Polonia	18 especies	15 de 18 especies migratorias tienden a llegar antes a las ciudades que a las zonas rurales	Tryjanowski et al. 2013
	Fort Campbell, Kentucky (EEUU)	Estornino pinto *Sturnus vulgaris*	Dormidero 2ºC por encima de la temperatura exterior	Francis 1976
	Saskatoon (Canadá)	Esmerejón *Falco colombarius*	Colonización de zonas de invernada limitada por la disponibilidad de árboles dormidero urbanos	Warkentin y James 1990
	Leicester, (Reino Unido)	Estornino pinto *Sturnus vulgaris*	500 000 individuos en árboles dormidero de parques urbanos	Peach y Fowler 1989
c) Reducción en la diversidad de depredadores	Toledo (España), Orsay (Francia), Brønderslev (Dinamarca)	Diferentes especies	Correlación positiva en entre la distancia media de inicio de vuelo y la distancia media a la casa más cercana, relacionado con la tolerancia a la presencia humana y el menor riesgo de depredación	Møller y Díaz 2018
	Oeste de Polonia	Mirlo común *Turdus merula*	Poblaciones urbanas 10-20 veces más densas que las rurales gracias, en parte, a protección frente a los depredadores	Gliwicz et al. 1994
	Oslo (Suecia), Orsay (Francia), Brønderslev (Dinamarca)	Diferentes especies	Menor densidad de rapaces (mayor de mamíferos) en zonas urbanas que hace que las aves urbanas canten en posiciones un 25% más elevadas que sus congéneres rurales	Møller 2011

Tabla 1. Continuación.

Característica urbana	Localización	Especie	Efecto	Referencias
d) Sustratos para nidificación	Hungría	Lechuza común *Tyto alba*	Nidificación en torres de iglesias y campanarios	Klein et al. 2017
	Reino Unido	Gorrión común *Passer domesticus* Avión común *Delichon urbica*	Nidificación en tejas, grietas y aleros de casas y edificios	Wotton et al. 2002
	Venecia (Italia)	Gaviota patiamarilla *Larus michahellis*	Nidificación en tejados de edificios	Soldatini et al. 2008
	Nápoles (Italia)	Halcón peregrino *Falco peregrinus*	Oteadero y nidificación en rascacielos	Fraissinet y De Rosa 2012
e) Hábitats que faltan en el entorno natural	Phonenix (EEUU)	Comunidad de aves	Correlación positiva entre la riqueza de aves nativas y la vegetación nativa de los jardines privados (número de árboles y arbustos en jardines privados). Mayor riqueza de especies en barrios de elevado nivel socioeconómico	Lerman y Warren 2011
	34 ciudades de todo el mundo	Comunidad de aves	Mayor biodiversidad (incluyendo aves) en las zonas urbanizadas de mayor nivel socioeconómico	Kuras et al. 2020

¿Qué está cambiando en las ciudades contemporáneas para que dejen de ser atractivas para las aves?

Especies de aves urbanas que durante mucho tiempo se han beneficiado de la interacción con el ser humano, se encuentran en serio declive desde finales del siglo XX hasta la actualidad, como el gorrión común *Passer domesticus* (Shaw et al. 2008) o el cernícalo primilla *Falco naumanni* en Europa (Negro et al. 2020). Esto indica que algo está cambiando en las ciudades contemporáneas que está causando una disminución de la calidad del hábitat urbano. Entre las posibles causas se han identificado las siguientes:

Nuevas tendencias arquitectónicas

La urbanización lleva asociada la construcción de nuevos edificios y la reforma de los antiguos con las técnicas y materiales más innovadores (**Fig. 3**). Los edificios de nueva construcción suelen tener diseños mucho más lineales y geométricos que antaño, además de carecer de elementos salientes como los aleros y repisas que ayuden a la nidificación de las aves (**Tabla 2a**; Wotton et al. 2002; Shaw et al. 2008). Los nuevos materiales de construcción y aislamiento reducen la formación de grietas y oquedades en los edificios, y además permiten su remodelación más eficiente, lo que disminuye las oportunidades de nidificación de las aves (Shaw et al. 2008). Igualmente, muchos edificios modernos son diseñados con grandes cristaleras y ventanales, que actúan como una trampa mortal para las aves, debido a las colisiones que generan, especialmente en el periodo de migración (Seewagen y Sheppard 2019).

Gestión urbana y reducción de zonas verdes

En la ciudad contemporánea las zonas verdes son manejadas con pesticidas y herbicidas desde hace décadas. Esta gestión está convirtiendo los parques urbanos en zonas verdes "domesticadas", lo que reduce drásticamente la disponibilidad de recursos tróficos naturales (insectos y vegetales) para las aves urbanas (Summers-Smith 2003; Archibald et al. 2017). Además, la urbanización lleva implícita la pérdida de hábitats naturales que rodean o se encuentran dentro de las ciudades (McKinney 2002). No obstante, una vez que la urbanización se ha llevado a cabo, no es extraño modificar de nuevo el escenario (reurbanización), proceso en el que muchas veces se eliminan total o parcialmente los pocos elementos seminaturales que quedaban en la ciudad (**Fig. 3**), como zonas verdes, parques, jardines y calles arboladas (McKinney 2002; Verbeeck et al. 2011). El objetivo de estas actividades es la construcción de nuevos edificios, la creación de espacios de aparcamiento para vehículos o incluso la remodelación de zonas verdes mediante la colocación de sustratos impermeables, como el cemento o el césped artificial, para reducir costes de mantenimiento (Pauleit et al. 2005; Perry y Nawaz 2008; Bernat-Ponce et al. 2020). Estos reductos seminaturales urbanos, además de ser zonas de alimentación, refugio y descanso para las aves (**Tabla 2b**), son corredores ecológicos claves para sustentar la biodiversidad urbana (Fernández-Juricic 2000; Pena et al. 2017; De Laet y Trappeniers 2019). Este proceso reduce la conectividad, compromete la viabilidad y las posibilidades de dispersión de las poblaciones de aves urbanas, especialmente de las más sedentarias y con menores distancias de dispersión juvenil (Fernández-Juricic y Jokimäki 2001; Isaksson 2018).

Contaminación

Una característica global y compartida por las ciudades es la creciente contaminación ambiental, ya sea química, acústica o lumínica, con efectos muy variados en las aves urbanas (Isaksson 2018). La contaminación urbana más conocida es la química, derivada de la combustión de combustibles fósiles, como gases procedentes de vehículos, industrias y calefacciones (**Fig. 3**). Aunque muchas áreas urbanas de países de industrialización temprana, principalmente europeos, se encuentran mucho más limpias en la actualidad que hace pocos siglos debido a la desindustrialización y nuevas tecnologías, los niveles de polución (p. e. NO_x, metales pesados, hollín) a los que se enfrentan las aves aún son elevados. Esta contaminación afecta a la avifauna urbana, tanto directa como indirectamente (ver **Tabla 2c**).

Las ciudades son lugares ruidosos *per* se, donde el transporte y la construcción, son los contaminantes acústicos más frecuentes (**Fig. 3**; Sordello et al. 2020). Los niveles de ruido urbano se han incrementado durante las últimas décadas y este contaminante cada vez despierta mayor interés en ecólogos y biólogos de la conservación. Estudios recientes demuestran que el ruido antropogénico tiene multitud de efectos sobre la biodiversidad urbana en función de su volumen, frecuencia, consistencia y duración (Francis y Barber 2013; Isaskson 2018; Sordello et al. 2020). Algunos de los efectos perniciosos más destacados del ruido sobre las aves urbanas son las alteraciones en el comportamiento, reducción del éxito reproductor y problemas en la comunicación (**Tabla 2c**).

Durante las últimas décadas se ha producido un incremento exponencial de la superficie nocturna iluminada, así como su intensidad en las zonas urbanizadas (**Fig. 3**; Isaksson 2018). Esta iluminación nocturna altera el comportamiento de las aves urbanas, modificando los periodos normales de actividad y de reproducción e incluso alterando la fisiología de las poblaciones afectadas (**Tabla 2c**). Además, la iluminación nocturna afecta negativamente a la migración de muchas especies de aves, ya que actúa como atrayente e incrementa su mortalidad al entrar en las áreas urbanas (Negro 2016; Horton et al. 2019). Asimismo, la contaminación lumínica también afecta a las poblaciones de insectos y polillas reduciendo su abundancia (Boyes et al. "en prensa") lo que podría tener consecuencias negativas sobre especies de aves insectívoras.

Comida antropogénica

Los recursos tróficos más abundantes para las aves omnívoras en las zonas urbanas suelen ser "comida basura" ultraprocesada (e.g galletas, pan o snacks) procedente de los restos de alimentación humana que se encuentran alrededor de los contenedores de basura, vertederos y terrazas de restaurantes (**Fig. 3**), o incluso de la alimentación de nuestras mascotas (Anderson 2006; Bernat-Ponce et al. 2019; Spelt et al. 2021). Esta "comida basura" es nutricionalmente deficiente debido a un exceso de grasas, azúcares y carbohidratos (Herrera-Dueñas 2018). A pesar de que en muchos casos incrementa la supervivencia invernal o la abundancia de ciertas especies, este tipo de dieta desequilibrada conlleva diversos costes ocultos asociados a su condición corporal, comportamiento, fisiología nutricional, estrés oxidativo, reproducción, patologías y salud general (**Tabla 2d**; Harr 2002; Ishigame et al. 2006; Herrera-Dueñas 2018; Isaksson 2018).

Depredadores y competidores

A pesar de que en la ciudad disminuye la riqueza de depredadores silvestres, otras especies de depredadores son beneficiadas por las actividades humanas y llegan a ser muy abundantes (Sims et al. 2008; Sorace y Gustin 2009). Entre ellos destacan los gatos domésticos *Felis silvestris catus* que salen al exterior. Por ejemplo, se estima que en EEUU los gatos matan anualmente 2400 millones de aves, entre las que destacan las que se alimentan en el suelo (p. e. petirrojo europeo *Erithacus rubecula*) (**Fig. 3**; Baker et al. 2008; Loss et al. 2013). Otros depredadores clásicos de las aves en las áreas urbanas, especialmente de sus nidos, son las ratas como *Rattus norvegicus* y *R. rattus* (Matthews et al. 1999; Donnelly y Marzluff 2004).

Por otro lado, en muchas ciudades contemporáneas se está produciendo un aumento de la abundancia de mesodepredadores, que puede tener que ver tanto con el aumento de estas especies en el medio rural como con los subsidios de alimento antropogénico de este ecosistema (**Fig. 3**; Fischer et al. 2012). Estos mesodepredadores incluyen a los córvidos, como las urracas, que han aumentado en número, expandiéndose y colonizando las ciudades (Gregory y Marchant 1996; Górski 1997). Además, varias especies de aves rapaces, atraídas por la gran abundancia de presas entre otros aspectos, están asentándose recientemente en las zonas urbanas (**Fig. 3**; **Tabla 2e**). Este incremento de mesodepredadores tiene potenciales efectos negativos por la competencia y/o la depredación sobre nidos de otras aves urbanas (**Tabla 2e**; Jokimäki y Huhta 2000). Sin embargo, en muchos casos la tasa de depredación es menor en la ciudad que en el entorno rural, a pesar del aumento de mesodepredadores; lo que ha llevado a la formulación de la paradoja de la depredación en las ciudades (Fischer et al. 2012).

En la actualidad, las aves urbanas se enfrentan a un problema relativamente nuevo, la conversión de las ciudades en puntos calientes para las invasiones biológicas (**Fig. 3**; Gaertner et al. 2017). Tanto es así que, actualmente, 31 de las especies de aves más invasoras del mundo se pueden encontrar en las ciudades (Murgui y Hedblom 2017). Estas especies pueden explotar los mismos recursos que las aves urbanas nativas, pero de una forma agresiva o más eficiente que acaba impactando negativamente sobre ellas, reduciendo su eficacia reproductora, compitiendo por los lugares de nidificación y/o recursos tróficos e incluso llegando a afectar a su supervivencia (**Tabla 2e**).

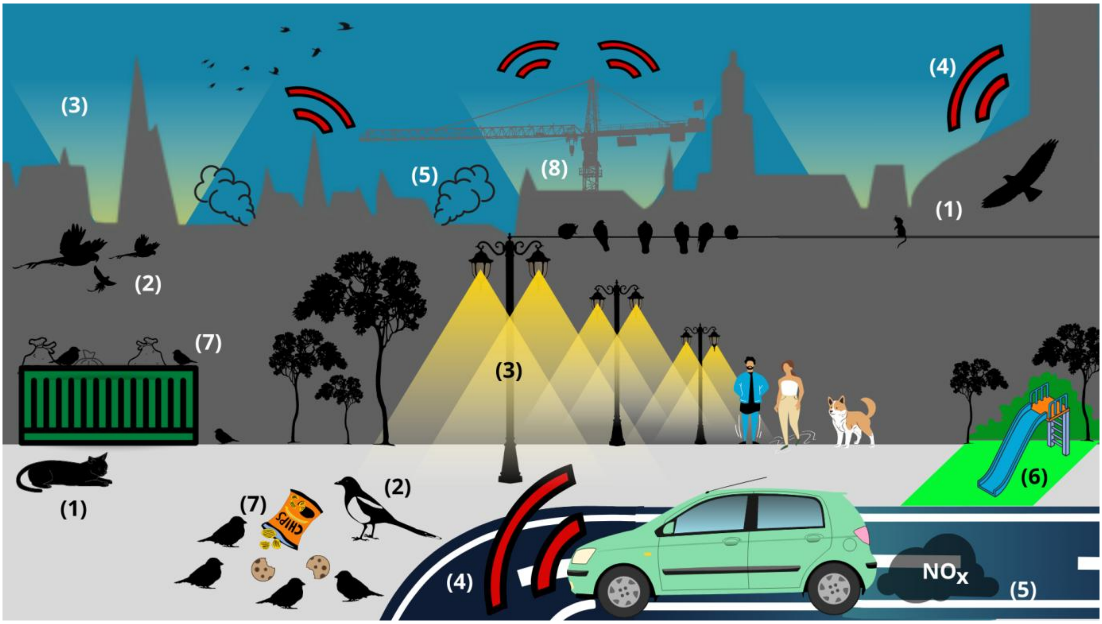

Figura 3. Ilustración que muestra algunos de los aspectos negativos de las ciudades para las aves: 1) depredadores urbanos; 2) competidores (en algunos casos invasores); 3) contaminación lumínica; 4) contaminación acústica; 5) contaminación química; 6) pérdida de hábitat (sustratos artificiales en zonas verdes); 7) dieta urbana de baja calidad y 8) nuevas tendencias arquitectónicas. Elaboración propia en Canva.

Tabla 2. Ejemplos de algunas características urbanas que pueden hacer que las zonas urbanas no sean atractivas para las aves y sus efectos.

Característica urbana	Localización	Especies	Efecto	Referencias
a) Nuevas tendencias arquitectónicas	Praga (República Checa)	Gorrión común *Passer domesticus*	Mayor preferencia por las zonas de los barrios con casas antiguas frente a las de nueva construcción (100% vs.71% de ocupación)	Moudrá et al. 2018
	Comunidad Valenciana (España)	Gorrión común *Passer domesticus*	Alta preferencia por tejas morunas tradicionales frente a nuevos diseños de tajados (75% nidos vs. 29% disponibilidad en la zona)	Bernat-Ponce et al. 2018b
b) Gestión urbana y reducción de zonas verdes	Comunidad Valenciana (España)	Gorrión común *Passer domesticus*	Poblaciones de 10 parques urbanos declinaron un 60% (vs. 15% en parques control) en 4 años por la sustitución del césped natural por césped artificial y otros sustratos impermeables.	Bernat-Ponce et al. 2020
c) Contaminación Química	Hungría	Carbonero común *Parus major*	Reducción de la biomasa de orugas disponibles para las aves de entre 8.5 y 24 veces por la contaminación	Seress et al. 2018
	Scania (Suecia)	4 especies de paseriformes	Correlación positiva entre los niveles estimados de NO_2 y el grado de urbanización con la capacidad antioxidante total en las 4 especies	Salmón et al. 2018
	España	Gorrión común *Passer domesticus*	Individuos urbanos con mayores niveles de daño oxidativo y mayor actividad de enzimas antioxidantes pero una menor capacidad antioxidante en comparación con individuos rurales	Herrera-Dueñas et al. 2017
	Harjavalta (Finlandia)	Carbonero común *Parus major* Papamoscas cerrojillo *Ficedula hypoleuca*	Reducción de la calidad del alimento aportado a los pollos por la contaminación de zonas urbanas. Larvas de mitad de tamaño en territorios contaminados	Eeva et al. 2005
	Londres (Reino Unido)	Gorrión común *Passer domesticus*	Correlación negativa entre la concentración de NO_2 y la tendencia de las poblaciones (abundancia) de machos	Peach et al. 2018
Acústica	EEUU	142 especies	Periodo reproductor alterado y correlación negativa entre del éxito de eclosión de los huevos y el ruido antrópico a lo largo de todo el continente	Senzaki et al. 2020
	San Francisco (EEUU)	Gorrión de corona blanca *Zonotrichia leucophrys*	Mejor comunicación (al doble de distancia) entre individuos debido a la ausencia de ruidos antropogénicos durante el confinamiento domiciliario debido a la COVID-19 en primavera de 2020	Derrrybery et al. 2020
	Países Bajos	Carbonero común *Parus major*	Puestas de menor tamaño en zonas próximas a la autovía a causa del ruido del tráfico	Halfwerk et al. 2011
Lumínica	Alemania	Mirlo común *Turdus merula*	Aves urbanas empezaron antes su actividad (29 minutos) y fue más prolongada en el día (40 minutos) que para sus congéneres rurales	Dominoni et al. 2013a
	Radolfzell (Alemania)	Mirlo común *Turdus merula*	Aves expuestas a la luz artificial se pueden reproducir 13 días antes y adelantan la muda 22 días	Dominoni et al. 2013b
	Viena (Austria)	Herrerillo común *Cyanistes caeruleus*	Inicio de puestas 1.5 días antes (en promedio) en hembras afectadas por la luz artificial de las farolas	Kempenaers et al. 2010
	Poznań (Polonia)	Gorrión común *Passer domesticus* Urraca común *Pica pica* Corneja cenicienta *Corvus cornix*	Correlación negativa entre la abundancia de córvidos y la abundancia de gorrión común. Correlación positiva si la disponibilidad de recursos tróficos antrópicos es elevado. Competencia entre especies nativas dependientes de recursos antrópicos urbanos	Skórka et al. 2016

Tabla 2. Continuación.

Característica urbana	Localización	Especies	Efecto	Referencias
d) Dieta antropogénica	Nueva York (EEUU)	Cuervos americanos *Corvus brachyrhynchos*	Niveles elevados de colesterol en el plasma sanguíneo en individuos salvajes suplementados con hamburguesas con queso de McDonalds’ frente a controles (149 vs. 142.1 mg/dL plasma)	Townsend et al. 2019
	Costa oeste (EEUU)	Gaviota occidental *Larus occidentalis*	Reducción en la eclosión de los huevos y los años de reproducción (<10 años con dieta predominantemente urbana) debido al escaso valor nutricional para los pollos y la formación de los huevos	Annett y Pierotti 1999, Pierotti y Annett 2001
	Arizona (EEUU)	Gorrión común *Passer domesticus*	Individuos urbanos con mayor glucosa en plasma (>400 mg/dL) que los individuos rurales (<300 mg/dL) debido al consumo de restos de comida antrópica (ricos en carbohidratos)	Gadau et al. 2019
	Hungría	Gorrión común *Passer domesticus*	Peor condición física y un 5% menos de masa corporal en individuos urbanos debida posiblemente a la alimentación juvenil con una dieta urbana pobre en artrópodos pero rica en recursos antrópicos	Liker et al. 2008
e) Depredadores y competidores	Reino Unido	Gavilán común *Accipiter nisus* Gorrión común *Passer domesticus* Halcón peregrino *Falco peregrinus*	Colonización urbana por parte del gavilán común como responsable del declive de las poblaciones urbanas de gorrión común Colonización urbana del halcón peregrino y mayor éxito reproductor que en zonas rurales por la mayor disponibilidad de presas	Bell et al. 2010; Kettel et al. 2019
	Finlandia, Italia, España	Diversas especies	Nidos de zonas urbanas con igual o mayor riesgo de depredación que en pueblos o bosques	Jokimäki et al. 2005
	Sevilla (España)	Cotorra de Kramer *Psittacula krameri* Cernícalo primilla *Falco naumanni*	Pérdida de lugares de nidificación para el cernícalo (nativo) por competencia con cotorra (invasora)	Hernández-Brito et al. 2014
	Tel Aviv (Israel)	Miná común *Acridotheres tristis* Carbonero común *Parus major*	78% de las cajas nido de carbonero común (nativo) usurpados por miná común (invasor)	Charter et al. 2016
	Ile-de-France (Francia)	Cotorra de Kramer *Psittacula krameri* Estornino pinto *Sturnus vulgaris*	Estorninos (nativos) afectados por competencia por recursos tróficos con las cotorras (invasoras) en comederos para aves de jardines privados de zonas residenciales	Le Louarn et al. 2016
	Poznań (Polonia)	Gorrión común *Passer domesticus* Urraca común *Pica pica* Corneja cenicienta *Corvus cornix*	Correlación negativa entre la abundancia de córvidos y la abundancia de gorrión común. Correlación positiva si la disponibilidad de recursos tróficos antrópicos es elevado. Competencia entre especies nativas dependientes de recursos antrópicos urbanos	Skórka et al. 2016

¿Es posible un cambio hacia un modelo de ciudad contemporánea que permita conservar mejor la biodiversidad? El caso de Vitoria-Gasteiz en el norte de España

Durante la última década se ha avanzado en muchos aspectos para la protección de la biodiversidad urbana gracias a la planificación verde y la reducción de la emisión de contaminantes. No obstante, el modelo predominante de ciudad contemporánea no es suficiente para preservar la biodiversidad. Las ciudades de tamaño medio acogen al 80% de europeos que viven en zonas urbanas, por lo que son una parte fundamental del problema para conservar la biodiversidad, pero también de la solución. Es necesario iniciar un cambio hacia nuevos modelos de ciudades sostenibles, lo cual supone un reto difícil, pero no imposible.

La ciudad española de Vitoria-Gasteiz es uno de los exitosos ejemplos que demuestran que un nuevo modelo de ciudad es posible. Esta ciudad de tamaño medio (< 250 000 habitantes) ha sido capaz de revolucionar la planificación de la ciudad con el objetivo de conseguir un desarrollo urbano que a su vez mejore la calidad de vida de sus ciudadanos atendiendo a criterios ambientales. En su planificación urbana tuvieron en consideración la creación de una infraestructura verde que incluyese funciones y servicios ecosistémicos naturales dentro de la misma ciudad (Córdoba Hernández et al. 2015). Por estos motivos, y gracias a su diseño verde urbano, en 2012 consiguió el premio de la Capital Verde Europea.

Uno de los pilares básicos de la ciudad ha sido el desarrollo de infraestructura verde urbana. Desde los años 60 se ha incrementado la superficie total de parques urbanos, creando el cinturón verde en 1993 y pasando de 30 a 130 ha de zonas verdes en la actualidad. Este anillo circular distribuye los parques de mayor tamaño en la periferia urbana funcionando como un elemento de transición y conexión entre el sistema verde urbano interior (parques y jardines) y los sistemas naturales circundantes actuando como corredores fundamentales para la biodiversidad urbana (Fernández-Juricic 2000; Pena et al. 2017; De Laet y Trappeniers 2019). En la actualidad el anillo verde alberga cada año entre 63 y 73 especies de aves nidificantes y el 32% de la superficie urbana son jardines públicos, donde las especies autóctonas representan el 91% de los árboles (Aguado et al. 2013; de Juana Aranzana 2015).

Todas estas medidas de planificación verde, unidas a la mejora en las vías ciclo-peatonales y los nuevos planes de movilidad, gestión del aire y gestión del ruido, convierten Vitoria-Gasteiz en una zona de baja contaminación. Estos aspectos benefician a todos los habitantes de la ciudad, incluyendo a las aves urbanas. Asimismo, en su planificación para la protección de la biodiversidad tienen en cuenta la protección de edificios antiguos en los que nidifican especies como el mochuelo europeo *Athene noctua* o la grajilla occidental *Corvus monedula* y, a su vez, los edificios nuevos integran paredes verdes y cajas nido en las fachadas (Ayuntamiento de Vitoria-Gasteiz 2014). Los resultados obtenidos a largo plazo en la ciudad de Vitoria-Gasteiz son una consecuencia de la aplicación de las políticas adecuadas para el desarrollo de una ciudad que integra la calidad de vida de sus habitantes, la sostenibilidad ambiental y la conservación de la biodiversidad urbana (Aguado et al. 2013).

Conclusiones

Las ciudades son ecosistemas antrópicos únicos que han evolucionado desde su origen en el Holoceno hasta la actualidad. Las aves han acompañado a los humanos a lo largo de todo este proceso y, en función de su tolerancia a la urbanización, han colonizado, abandonado y variado en su abundancia en las ciudades hasta el presente. Estos ecosistemas presentan una serie de elementos, como los recursos tróficos predecibles o una gran variedad de sustratos de nidificación, que han atraído a las aves desde sus primeros asentamientos (Seress y Liker 2015; Negro et al. 2020). Tanto es así que las ciudades contemporáneas albergan, además de especies en expansión o cosmopolitas, especies endémicas o en declive (Aronson et al. 2014) y, en algunos casos, contienen más especies amenazadas que su entorno natural (Ives et al. 2016). Cabe destacar que más del 20% de especies de aves reconocidas en el mundo han sido detectadas en las ciudades (Aronson et al. 2014).

Sin embargo, la velocidad actual de urbanización no tiene precedentes en la historia y constituye uno de los mayores problemas para la biodiversidad. En muchos casos, la urbanización actual induce a que las ciudades dejen de ser atractivas para las aves o que éstas no sean capaces de adaptarse a los cambios ambientales y su viabilidad poblacional se vea comprometida. Recientemente se ha planteado que las ciudades pueden estar transformándose en una trampa ecológica para algunas especies de aves (Hale y Swearer 2016; Pollock et al. 2017).

El modelo de ciudad contemporánea y la velocidad de urbanización parecen incompatibles con la conservación de la diversidad de aves urbanas. No obstante, cada vez más ciudades apuntan hacia el necesario desarrollo de nuevos modelos urbanos en los que la infraestructura verde y la sostenibilidad ambiental son pilares fundamentales en su planificación, como es el caso de Vitoria-Gasteiz. No debemos olvidar que una ciudad saludable y sostenible para las aves también implica una buena calidad de vida para sus vecinos, los seres humanos.

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Factors affecting the abundance of House Sparrows *Passer domesticus* in urban areas of southeast of Spain

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FACTORS AFFECTING THE ABUNDANCE OF HOUSE SPARROWS *PASSER DOMESTICUS* IN URBAN AREAS OF SOUTHEAST OF SPAIN

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Abstract

Capsule House Sparrows in the Valencian Community, where the species is declining, are more abundant in parks and urban areas where rubbish bins are available as a source of food. In heavy traffic areas, crowded streets and high rise building zones the species is less abundant.

Aims To explore biotic and abiotic factors affecting the abundance of urban House Sparrows in the Valencian area. To infer the causes that might explain the urban decline and to propose measures that might halt it and elevate population numbers.

Methods The abundance of urban House Sparrows was analysed in relation to 17 landscape variables by carrying out 181 point counts (50 urban parks/38 schools/93 streets) per season across six locations and over five seasons (breeding and non-breeding seasons) in south-eastern Spain. Urban parks were also described by 21 additional variables. Hierarchical Partitioning Analyses were used to identify favoured or avoided urban factors by birds in each season.

Results House Sparrows were most abundant in parks and other urban areas where rubbish bins were available as their foraging sites. The presence of rubbish bins, area and proximity to other parks favoured their abundance in parks. Furthermore, sparrows were less abundant in crowded streets, heavy traffic areas and high rise buildings zones.

Conclusion House Sparrows are present in high abundance where they feed but scarce in polluted, crowded and heavily built-up areas of the city. Changes in the structure of these urban habitats may underlie the documented declines across the Valencian Community and Spain. Green urban planning and management are needed to ensure heterogeneous green areas with enough vegetation coverage to provide high availability of natural food versus anthropogenic food scraps. Sufficient numbers of buildings offering nest sites and reduced air pollution might ameliorate cities adequately to support larger populations of House Sparrows.

Introduction

The House Sparrow *Passer domesticus* has been associated with humans since the Bronze Age (Ericson *et al*. 1997) and presents the widest distribution of any species in the world (Cramp & Perrins 1994, BirdLife International 2016). This is a consequence of an anthropogenic process, namely releases of birds into non-native areas and is, at least initially, partly also a consequence of its dependency and co-evolvement with human agricultural development (Sætre et al. 2012, Liebl et al. 2015). However, this species has declined in the Western Palearctic during the last quarter of the 20th century (Summers-Smith 2003) and at the beginning of the 21st century (Raven *et al*. 2003, Hayhow et al. 2014). Moreover, these declines have occurred in both urban and farmland areas (Summers-Smith 2000, Crick *et al*. 2002, De Laet & Summers-Smith 2007), even though the decline first started in rural ones (Summers-Smith 2003, Robinson *et al*. 2005).

Their decline was first detected in Great Britain (Summers-Smith 2003) where 65% of birds have been lost in 42 years, between 1970 and 2012 (Hayhow et al. 2014). For instance, the species declined in London by 71% between 1994 and 2002 (Raven et al. 2003). On the other hand, it is still abundant in Spain (Carrascal & Palomino 2008), despite showing an annual decline of 0.9% (SEO/BirdLife 2013). However, trends in Valencian populations mirror those in Great Britain. The first signs of decline were detected in orange groves of Sagunto (Valencia) where the population decreased by more than 90% between 1975 and 2001 (Gil-Delgado et al.

2002). Later, in the parks of the city of Valencia, the population declined by 70% between 1998 and 2008, equivalent to an average annual decline of 15% (Murgui & Macias 2010). In addition, large-scale studies within the Valencian Community detected an overall decline of 22% between 1998 and 2015 (Vera & Escandell 2016).

Human-dependent bird species are more likely to display negative population trends in the most heavily human-modified environments (Siriwardena et al. 1998). Accordingly, it is appropriate to investigate declines in House Sparrow numbers in relation to patterns of temporal changes in human activities (Anderson 2006). Several hypotheses to explain their declines in urban areas have been proposed including: predation by Domestic Cats *Felis silvestris catus* or Eurasian Sparrowhawks *Accipiter nisus* (Woods et al. 2003, Bell et al. 2010); competition for food resources with Feral Stock Doves *Columbia livia*; and competition with Common Starlings *Sturnus vulgaris* and Spotless Starlings *Sturnus unicolor* for nest sites (Weitzel 1988). Furthermore, it has also been attributed to other causes such as: structural habitat changes by the improvement of human socioeconomic status in towns and cities which means loss of green space, invertebrate and nest availability sites (McKinney 2002, Shaw et al. 2008, Moudrá et al. 2018); increased exposure to electromagnetic radiation (Balmori & Hallberg 2007); poor-quality trophic resources availability in urban areas (Herrera-Dueñas et al. 2015) and increased pollution from traffic (Summers-Smith 2007, Herrera-Dueñas et al. 2017). The most favoured hypothesis to explain the decline of

House Sparrows in urban areas is that it is down to a combination of factors rather than to a single one. Nevertheless, we are still some way from identifying these key urban factors (Vincent 2005, Anderson 2006).

A very useful approach to identify such factors may be provided by a fine-scale habitat study of areas where the sparrow is declining (Robinson et al. 2005, Chamberlain et al. 2007). Therefore, here, we aimed to explore the key biotic and abiotic factors affecting the abundance of House Sparrows in several locations in Valencia across different seasons. Most previous research studied the species just within its breeding season (e.g. Chamberlain et al. 2007, Shaw et al. 2011), whereas we studied birds in early and midsummer (between late June and July, late breeding season in Spain which is from mid-March to early August (Gil-Delgado et al. 1979, Bernis 1989)) and autumn and winter (their two non-breeding seasons in the study area). Taking into account that summer counts could be biased towards dispersing juvenile House Sparrows present in non-selected areas by adults, it would also be interesting to know these important factors for conservation purposes of urban populations of sparrows. From previous studies in Valencia (Murgui 2009) and Great Britain (Shaw 2009), we predicted some factors determining bird abundance would be related to the availability of food and nest sites. However, the species has declined throughout the whole eastern of Spain (Vera & Escandell 2016). Therefore, we explored habitat factors in coastal and inland areas of the south-eastern Iberian Peninsula as a whole to increase our ability to identify factors that might explain general declines in this species' abundance. We hope that our findings may inform conservation policy to halt and reverse its decline.

Materials and methods

Study area and bird census

The study was carried out in five locations in the Valencia area in south-eastern Spain (four of located in the northern area of Alicante's province and one in the south-east of Valencia's province) (Figure and Table 1). This last location was split across two sites, Gandía and Grao/Playa, separated by orange groves (Figure and Table 1). Further information about human demographics, agriculture and industry of these locations is provided by Jordà (1976), Banyuls (1985), and Tormo (2011).

House Sparrow abundance was determined using point counts (Ralph et al. 1996, Gibbons & Gregory 2006). To avoid double counting of birds, point counts were separated by a minimum distance of 100 m. At each point count, abundance of birds within a space limited by a radius of 50 m was recorded by a single observer (EB-P) for 5 minutes. Birds were recorded by sight and vocalizations avoiding double counting. It was established a settlement period of 1 minute.

Table 1. Geographical, demographical and climatic characteristics of the selected locations for the study of factors affecting the abundance of House Sparrows in urban areas of south-east of Spain between 2014 and 2016. Table devised with data from SigPac (2014), AVAMET (2016) and Instituto Nacional de Estadística (Spanish National Statistics Institute) (2016).

Location	Coordinates	Number of inhabitants 2015	Density (Inhabitants/Km^2)	Temperature 2015 (°C)			Rainfall 2015 (mm)	Altitude (m.a.s.l.)	Point counts[a]	Studied urban surface (ha)
				Max.	Mean	Min.				
Banyeres de Mariola	38°42´57´´ N 0°39´26´´ O	7155	143.65	39.6	15.4	-5.2	489.2	816	14 (3/3/8)	93.33
Muro de Alcoy	38°46´47´´ N 0°26´10´´ O	9167	297.91	42.3	17.0	-4.1	421.4	410	18 (3/3/12)	116.97
Cocentaina	38°44´42´´ N 0°26´26´´ O	11 406	218.93	42.8	16.9	-1.8	533.2	430	20 (5/5/10)	133.56
Alcoy	38°41´54´´ N 0°28´25´´ O	59 567	468.48	41.2	16.0	-3.3	548.4	562	60 (15/15/30)	401.41
Gandía[b]	38°59´39´´ N 0°10´56´´ O	61 319	1258.17	43.3	20.1	3.5	677.1	22	31 (10/3/18)	248.25
Grao/Playa[b]	38°58´00´´ N 0°09´45´´ O	13 895	-	-	-	-	466.6	0	38 (14/3/21)	278.44

[a] Total point counts in each locality and their distribution among the three sampled environments (parks/schools/streets).

[b] Sectors Gandía and Grao/Playa are separated by orange groves.

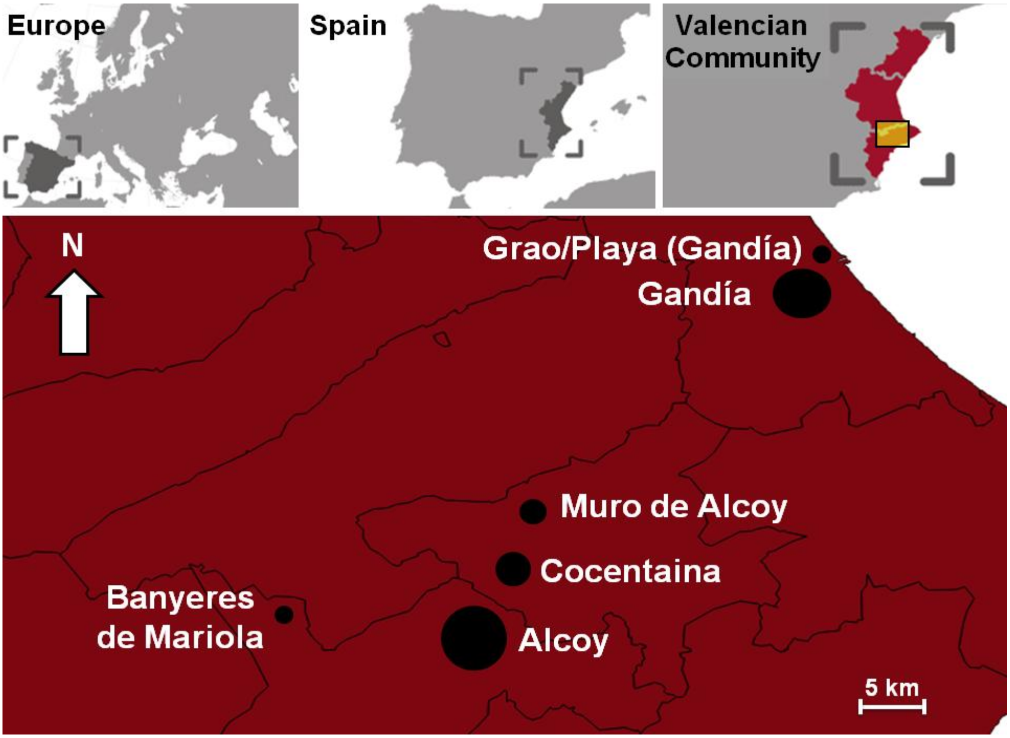

Fig. 1 Maps showing selected Valencian localities (Spain, Europe) for the study of factors affecting the abundance of House Sparrows in urban areas of south-east of Spain between 2014 and 2016. The sizes of the circles show how large the urban areas are in terms of their overall number of inhabitants.

The area of each location was calculated using the cartographic viewer SigPac (SigPac 2014). It was divided into several homogeneous parcels in terms of type and age of the buildings. The number of point counts in each homogeneous parcel and thus in each locality was related to its urban surface (Table 1). At each parcel, point counts were undertaken on streets, in school grounds and in parks, which were distributed randomly taking into account any inaccessible areas (those in parks and schools were randomly distributed within them) via the QGIS platform (QGIS 2014) (Figure 2). Censusing of sparrows was carried out in the autumn of 2014 and 2015 (non-breeding seasons), the winter of 2014-

15 and 2015-16 (non-breeding seasons), and the summer of 2015 (breeding season). During each field season 181 point counts were made totalling 905. Each counting session lasted 2 hours starting at daily official sunrise when House Sparrows are most active (Anderson 2006). Sessions were not conducted on days of rain and/or winds. On average, a point count session comprised 10-14 counts.

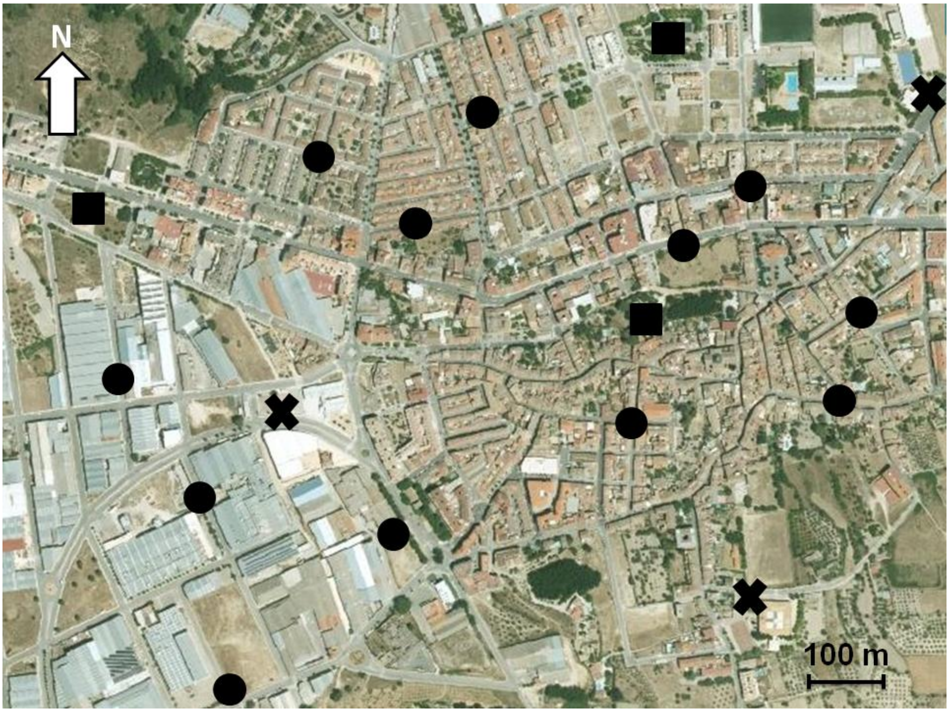

Fig. 2 Distribution of the 18 point counts within the location of Muro de Alcoy (Valencia, Spain) as an example of their distribution in the locations for the study of factors affecting the abundance of House Sparrows in urban areas of south-east of Spain between 2014 and 2016. Symbols represent point count locations in parks (squares), on streets (circles) and in schools (crosses).

Urban landscape and park variables

At each point count in the urban landscape, 17 variables were described (Table 2(A)), some through a qualitative classification of presence/absence (1/0), and others through a qualitative ranking (0, 1, 2, ...) or by quantized values (0 to n). These 17 landscape variables were grouped into three classes: urban structure, human factors and food resources (Table 2A). Variables were selected paying particular attention to human-related urban factors, infrastructure and planning, which could be rather easily addressed and managed according to species' requirements. In the resources group, seven variables related to presence of anthropogenic scraps sources and natural food sources for sparrows were measured (Smith et al. 2006, Murgui 2009, Rega-Brodsky & Nilon 2016, Skórka et al. 2016) (Table 2(A)). In towns and cities finding food could be tough and these variables may be essential for survival. The second group contained six variables related to urban structure (Murgui 2007, Shaw et al. 2008), nest sites, shelter (Savard & Falls 1981, Indykiewicz 1991, Wotton et al. 2002), and urban movements of birds (Bernis 1989, Anderson 2006) (Table 2(A)). The third group included four variables related to the human influences that can promote or depress House Sparrow numbers, such as traffic, pedestrians, socioeconomic status and altitude above sea level (Summers-Smith 2007, Shaw et al. 2008, Menon et al. 2013) (Table 2(A)).

Parks were also described according to 21 variables in four groups (Table 2(B)). The first one contained variables related to vegetation and substrate while the second group described aspects of the location of the park (Table 2(B)). The third one included variables related to resources provided by humans that might influence bird abundance in parks, such as rubbish bins, terraces, children's play areas or water bodies (Table 2(B)), while the last group described variables related to urban factors including the influence of people, socioeconomic status, cats and dogs on bird abundance (Table 2(B)).

Table 2. Summary of the environmental variables used in the Hierarchical Partitioning Analyses for the study of factors affecting the abundance of House Sparrows in urban areas of south-east of Spain between 2014 and 2016. Note that there are two different main groups of studied variables, A) urban landscape and B) parks. The variable geography was included in every group to control overall location effects of the sampling points.

A)	**URBAN LANDSCAPE**	
	Group and variables	
Resources	**Urban structure**	**Human factors**
(1) Presence/absence of park (2) Presence/absence of private gardens (3) Presence/absence of bar/cafeteria terrace (4) Presence/absence of rubbish bins[c] (5)Presence/absence of vacant lot (6) Presence/absence of school building and yard (7) Presence/absence of lawn	(8) Presence/absence of old buildings (built before 1950) (9) Building height (mean number of floors of 8 buildings) (10) Street width (m) (11) Presence/absence of mature trees (12) Distance to the nearest urbanized limit (m) (13) Vegetation cover of the point count (7 854 m^2)[d]	(14) Number of people per minute[e] (15) Number of vehicles per minute[e] (16) Socioeconomic status[f] (17) Altitude above sea level
B)	**PARKS**	
Vegetation and substratum	**Location**	**Urban factors**
(1) Vegetation cover of the park[g] (2) Dominant vegetation[h] (3) Average height of vegetation (4) Presence/absence of shrubs higher than 1 m (5) Presence/absence of exotic vegetation (6) Presence/absence of sand (7) Presence/absence of lawn (8) Dominant substratum[i]	(9) Distance to the nearest urbanized limit (m) (10) Survey zone of the park (edge or centre) (11) Distance to the nearest park (m) (12) Area of the park (m^2) (13) Altitude above sea level (m) **Resources** (14) Presence/absence of rubbish bins (15) Presence/absence of bar/cafeteria terrace (16) Presence/absence of water body (17) Presence/absence of children's play area	(18) Number of people per minute[e] (19) Socioeconomic status[f] (20) Presence/absence of dogs (21) Presence/absence of cats

[c] Rubbish bins are referred to communal above-ground trash containers.

[d] Four categories: 0-5%, 6-30%, 31-60% and >60%.

[e] Both were calculated by counting the number of pedestrians and vehicles entering the point count for 5 minutes and standardising them to obtain the number per minute (Menon et al. 2013).

[f] Five socioeconomic status areas based on type of buildings, height, age, density, presence of private gardens and cleanliness of streets (De Laet et al. 2011). (1) Deprived areas: low-rise old buildings, often with holes for nesting. (2) Average socioeconomic status: the commonest areas in towns and cities, dominated by the highest buildings with a high density of shops and businesses. They present a vast heterogeneity of buildings and very dense traffic. (3) Well-off areas: outside city centres, low-rise apartments or houses, recently built and associated with private gardens. (4) High socioeconomic status areas: away from city centres, with low-density buildings, usually houses of two levels associated with private gardens. (5) Industrial areas (reference level): located on the outskirts of towns and characterized by the presence of industrial buildings with a few green areas.

[g] Three categories: 0-30%, 31-60% and >60%.

[h] Four categories: lawn, shrubbery, trees and trees and lawn.

[i] Four categories: sand, lawn, rubber and pavement.

Statistical analysis

We used Hierarchical Partitioning (HP) to identify the set of predictor variables that best explained the abundance of House Sparrows in each season (Chevan & Sutherland 1991, Mac Nally 1996). HP does not aim to identify a best regression model as such but rather uses all models in a regression hierarchy to distinguish those variables that have high independent correlations with the dependent variable. Such variables are likely to be most influential in controlling variation in the dependent variable (Mac Nally 1996). Their explanatory power is segregated into the independent effect (*I*) and the effects caused jointly with other variables (*J*) (Mac Nally 2002). It also offers a set of negative effects that contains variables that act as suppressors of other variables, suppressing irrelevant variance of independent variables, thus enhancing the relationship between the other independent variables and the dependent one (Chevan & Sutherland 1991). Moreover, model outputs also provide the percentage (*I*%) of the total group *I* counted in each variable.

Hierarchical Partitioning was performed with the 'hier.part' package in RStudio 0.99.903 (Walsh & Mac Nally 2015) using Poisson regression and r-squared (Rsqu) as the goodness-of-fit measure (Jongman et al. 1995). The significance of the contribution of environmental variables was evaluated by 999 tests based on randomizations (Mac Nally 2002), where the random distribution of *I* was generated. For the variable to make a significant contribution, the observed *I* had to be within the 95-percentile with Z-scores $\geq$ 1.65 (Hallstan et al. 2010).

We opted to analyse data from each single season by means of a separate model, to evaluate whether similar or different environmental factors regulated sparrow abundance over the same sites in different periods, according to coherent or varying patterns. This approach, coupled with the use of Hierarchical Partitioning within each season, would help to find stronger and more important effects.

Due to the division of the landscape variables into three groups (Table 2(A)), we avoided performing the HP with more than nine independent variables because higher numbers of variables produce minor rounding errors (Walsh & Mac Nally 2015). Prior to HP analyses of each group of variables in each season, Poisson regression models with the variables of each group were carried out using the 'glm' function with the canonical link 'log' in RStudio 0.99.903 and the variance inflation factor (VIF) among variables was calculated to detect multicollinearity, using the function 'vif' of the package 'car'. When VIF was > 5 in some variable, we considered multicollinearity to be high (Zuur et al. 2010) and in such cases the variable of the highest VIF or the second one, if the first one was considered of key biological relevance, was discarded. This proceeding was repeated to test if the reduced group of variables showed multicollinearity again.

In each of the five seasons, one HP analysis was performed for each urban landscape group: urban structure, human factors and resources (Table 2(A)). The overall location effects of the sampling points were controlled by including a geographical variable in all of the analyses (longitude × latitude) calculated from the UTM coordinates of each point count (López-Iborra et al. 2011). Significant variables from each urban landscape group analysis were then combined in a final HP analysis that assembled selected variables hierarchically for each season. The final HP analysis only retained those key variables for the sparrow abundance. Once again, prior to this, VIF values of selected variables of each group in each season were calculated to avoid multicollinearity. Finally selected categorical variables in the combined HP analysis were analysed to find those significant categories for sparrow abundance. This was conducted in RStudio 0.99.903 through the functions 'kruskal.test' and 'kruskalmc' of the 'pgirmess' package which reduce the probability of making a Type I error (Field et al. 2012). Prior to this, normality and homogeneity of variances of those selected variables were analysed by performing tests of Kolmogorov-Smirnoff and Levene in RStudio 0.99.903 with the functions 'ks.test' and 'leveneTest' from the packages 'stats' and 'car', respectively. Poisson regression models were carried out using the 'glm' function with the canonical link 'log' in RStudio 0.99.903 to determine whether each selected variable was negatively or positively correlated with sparrow abundance in each season. The sign of each category of categorical variables was addressed taking into account their reference level.

The same analytical approaches were adopted to examine the variables of the parks (Table 2(B)).

To test climatic differences between both winters, a paired t-test with the minimum monthly temperatures of the localities in December, January and February of both winter seasons (AVAMET 2018) was conducted in RStudio 0.99.903 with the ‘t.test‘ function. Prior to this, normality and homogeneity of variances were analysed by performing the test of Shapiro-Wilk (‘shapiro.test‘ function) and F test (‘var.test‘ function).

Results

House Sparrows were present in more than 80% of point counts every season (Table 3). Every point count except one (a main street in Alcoy with heavy traffic) yielded House Sparrows in at least one season but there was great variation across them (Table 3).

Table 3. Presence during counts, percentage of occupied points and mean count of House Sparrows in the 181 point counts during five seasons in urban areas of south-east of Spain between 2014 and 2016.

Parameter	**Season of census**				
	Autumn 2014	**Winter 2014-2015**	**Summer 2015**	**Autumn 2015**	**Winter 2015-2016**
Presence	162/181	153/181	170/181	160/181	151/181
Percentage	89.50	84.53	93.92	88.40	83.43
Mean count ± sd	9.39 ± 8.54	6.47 ± 5.06	9.08 ± 7.45	8.43 ± 6.98	7.06 ± 5.78

Urban landscape

Outputs from initial HP analyses are described in Appendix 1. Outputs from VIF analyses revealed that multicollinearity among selected variables was only present in summer counts. Presence of parks and trees had the highest values of VIF (6.53 and 3.01, respectively). Therefore, the second one was not included in the combined HP analyses of that season because the presence of a park is a better representation of an urban habitat. Outputs from combined HP analyses of urban landscape variables revealed that in four seasons of the census, urban parks had a positive significant effect on House Sparrow abundance, which was especially important in summer, with an independent effect higher than 31% (Table 4). Building height had a negative effect on sparrow abundance in every studied season with independent contributions (*I%*) between 7 and 17% (Table 4). The presence of rubbish bins had a positive significant effect on the abundance of birds with a *I%* between 9% and 25% in both winters and summer (Table 4). Geography had an important positive effect on House Sparrow abundance in both winters with *I%* of 25.47% and 35.25%, respectively. Old buildings and vacant lots had a significant positive effect on the abundance of birds in the first autumn with a *I%* of 20.55% and 9.29%, respectively. Old buildings were the second most important variable in that autumn after the 32.07% of *I%* of socioeconomic status (Table 4). In addition, the presence of trees was positively selected in the second winter with a *I%* of 6.91%. However, it had lower explanatory power than urban parks and rubbish bins with *I%* of 21.63 % and 9.89 % respectively (Table 4). Both presence of lawn and street width had, respectively, a positive and negative effect of 6% on sparrow abundance during summer. However, they had less explanatory power than the presence of urban parks and rubbish bins with *I%* of 31.36% and 25.37% respectively (Table 4).

Socioeconomic status had a positive significant effect of 32.07% on the abundance of birds in the first autumn (Table 4), with significant differences between their five categories (χ^2 2014: = 21.53, df = 4, $P <$ 0.001). Post hoc tests revealed that significantly more birds were present in deprived areas than in industrial and medium ones.

Number of people had a negative significant effect on sparrow abundance (Table 4) in both autumns, in the first winter of study and in summer with *I%* of 10.53%, 18.68%, 13.62% and 5.70%, correspondingly. Traffic volume was also significant and negatively related to bird numbers in both winters and in the second autumn with *I%* of 11.85%, 10.11% and 13.93%, respectively (Table 4). These two variables had a medium-low explanatory power in every season but had a negative significant effect in four and three seasons respectively, which could be considered a regular effect for the abundance of the species.

In regard to joint contributions, most of the variables of both autumns and first winter presented lower joint than independent contribution, suggesting a low degree of correlation between them (Table 4). However, summer and the second winter presented the same number of variables with higher independent than joint contributions than variables with higher joint than independent contributions (Table 4). This means that these variables had a slight correlation between them. Nevertheless, multicollinearity was avoided by previous VIF analyses and that is the reason of lower joint than independent contributions in most of groups. Also, there are four variables with negative joint contributions in different seasons that act as suppressors of other variables: vacant lot, number of people, rubbish bin and number of vehicles (Table 4).

Table 4. Variables identified by the combined Hierarchical Partitioning (HP) Analyses as key factors affecting the abundance of House Sparrows in urban areas of south-east of Spain between 2014 and 2016.

Season	Group	Variable	I^j	J^k	I %	Z score	Sign
Autumn 2014	Resources	Urban park	0.0222	0.0165	8.87	2.19	+
		Vacant lot	0.0233	-0.0009	9.29	2.39	+
	Urban structure	Old buildings	0.0516	0.0010	20.55	5.90	+
		Building height	0.0318	0.0323	12.67	3.09	-
	Human factors	Socioeconomic status[l]	0.0805	0.0089	32.07	3.35	+
		People/minute	0.0264	0.0147	10.53	2.11	-
Winter 2014-15	Resources	Urban park	0.0278	0.0266	13.20	2.80	+
		Rubbish bin	0.0411	0.0163	19.51	4.75	+
	Urban structure	Building height	0.0207	0.0239	9.80	1.80	-
	Human factors	People/minute	0.0287	-0.0022	13.62	3.10	-
		Vehicles/minute	0.0250	0.0315	11.85	2.56	-
		Geography	0.0537	0.0145	25.47	6.04	+
Summer 2015	Resources	Urban park	0.1162	0.1166	31.36	15.19	+
		Rubbish bin	0.0940	-0.0074	25.37	25.36	+
		Lawn	0.0236	0.0546	6.38	2.37	+
	Urban structure	Building height	0.0613	0.0853	16.55	6.71	-
		Street width	0.0209	0.0411	5.63	1.85	-
	Human factors	People/minute	0.0211	0.0049	5.70	2.07	-
Autumn 2015	Urban structure	Building height	0.0257	0.0268	16.37	2.43	-
	Human factors	People/minute	0.0293	0.0171	18.68	2.74	-
		Vehicles/minute	0.0219	0.0157	13.93	1.93	-
Winter 2015-16	Resources	Urban park	0.0804	0.0874	21.63	10.27	+
		Rubbish bin	0.0368	0.0338	9.89	3.76	+
	Urban structure	Building height	0.0264	0.0268	7.09	2.92	-
		Trees	0.0257	0.0480	6.91	2.80	+
	Human factors	Vehicles/minute	0.0376	-0.0118	10.11	4.41	-
		Geography	0.1311	0.0397	35.25	15.64	+

[j] Independent effect of the variable

[k] Joint effects caused with other variables

[l] The reference level of the socioeconomic status was the industrial area, so all other categories were positive for the abundance of the species

Parks

Outputs from initial HP analyses are provided in Appendix 2. Multicollinearity was not found between the initial selected variables of the four groups in each season and, consequently, no variable had to be removed to perform the combined HP analyses. Only three seasons had significant variables selected for the abundance of House Sparrows in parks: summer, autumn 2015 and winter 2015-16. In summer, the surface of the park had a positive significant effect on birds of 56.49% of *I%* while the presence of dogs had a negative significant effect of 43.22% of it (Table 5). Moreover, the distance to the nearest park had a negative significant effect on sparrows during the second autumn and winter (Table 5). Finally, during the winter 2015-16, the presence of rubbish bins had a positive significant effect of 59.94% of *I%* on the abundance of birds (Table 5). In regard to joint contributions, every selected variable had lower joint than independent contributions, suggesting a low degree of correlation between them and none of them had negative joint contributions.

Winter conditions differed between years because means of minimum temperatures of the studied seasons presented significant differences and winter 2015-16 was warmer than winter 2014-15 (Shapiro-Wilk normality text: W = 0.942, P = 0.403 / F test to compare two variances: F = 0.796, n = 14, P = 0.675 / Paired t-test: t = -3.519, df = 14, P = 0.003).

Table 5. Variables identified by the combined Hierarchical Partitioning (HP) Analyses as key factors affecting the abundance of House Sparrows in urban parks of south-east of Spain between 2014 and 2016.

Season	Group	Variable	I^j	J^k	*I %*	*Z.score*	Sign
Summer 2015	Location	Park's area	0.0810	0.0059	56.49	2.23	+
		Dogs	0.0620	0.0058	43.22	1.69	-
Autumn 2015	Location	Distance to nearest park	0.0752	0.0004	99.31	1.97	-
Winter 2015-16	Resources	Rubbish bin	0.1461	0.0354	59.94	4.53	+
	Location	Distance to nearest park	0.0709	0.0177	29.04	1.71	-

[j] Independent effect of the variable.

[k] Joint effects caused with other variables.

Discussion

The House Sparrow is still abundant in urban and suburban habitats of the Valencian area as expected due to their close association with humans (Summers-Smith 1988, Gil-Delgado & Mora 2015), despite their well documented decline across Europe (Summers-Smith 2003, Shaw et al. 2008). However, its abundance varied greatly across different habitats within urban areas (Chamberlain et al. 2007, Murgui 2009) and its presence in almost every studied urban sample was not necessarily a good indicator of healthy populations (Gil-Delgado & Mora 2015).

Key environmental factors in the urban landscape

Eight variables had a positive effect on sparrow abundance. The most significant ones were the presence of urban parks and rubbish bins, which are sources of food and shelter for birds (Mitschke & Mulsow 2003, Bokotey & Gorban 2005, Murgui 2009). Our results, in agreement with Murgui (2009), showed that urban parks are the main feeding location for House Sparrows in cities of south-eastern Spain. Green spaces may offer essential invertebrates for the juvenile sparrows during the breeding season and also plant resources throughout the year (Murgui 2009). However, without their careful management, natural food availability is likely to decline rapidly impacting negatively on the species (Peach et al. 2008, Peach et al. 2015, Weir 2015). We think that modern policies of urban green areas (loss of green spaces, excessive cleaning and use of pesticides) may be displacing urban sparrows to sites where food is easily available and/or of lower nutritional value. Contrary to these arguments, Skórka et al. (2016) found that availability of anthropogenic foods had a negative effect on the breeding abundance of House Sparrows mediated through increased predation risks (Lima 1998, Krauze-Gryz et al. 2013). However, we found that the abundance of House Sparrows was significantly and positively influenced by the presence of rubbish bins, which is a constant and reliable but poor-quality source of human-related food that also represents a sort of proxy for anthropogenic food sources as a whole (Bokotey & Gorban 2005, Herrera-Dueñas et al. 2015). Our findings suggest the existence of seasonality because it was only not selected in autumn. In the coldest seasons, rubbish bins might be important due to the reliability of food scraps that can be found around them (Bokotey & Gorban 2005); while their importance in summer might be due to the easy access to the scraps for the juvenile birds. The replacement of open rubbish bins by closed plastic ones has been suggested as a cause to explain the House Sparrow decline in Ukraine (Bokotey & Gorban 2005) and may have also influence in the studied area where above-ground rubbish bins are being replaced by underground ones (stainless steel chutes on the pavement that feed into underground chambers). Clearly, anthropogenic foods represent a double-edge sword to urban birds because while they may promote survival of larger proportions of populations, they lack many key dietary elements such as vitamins, minerals or essential amino acids (Herrera-Dueñas et al. 2015).

Variables that just had positive effect on bird abundance once or twice such as the presence of old buildings, presence of vacant lots, socioeconomic status, presence of trees, presence of lawn and geography were not considered as key factors for House Sparrows because of their irregular selection. However, geography was positively and consistently selected during both winters suggesting it as an important factor in this season. House Sparrows were more abundant in coastal locations than in inland areas in winter, as suggested Gil-Delgado & Mora (2015), and it implies that inland populations should be the priority for study and conservation purposes of the species in the Valencian area. This result could be explained by three non-exclusive hypotheses: larger winter mortality in inland areas (Senar & Copete 1995), altitudinal movements (Senar & Borras 2004) or a differential predator effect (Chamberlain et al. 2009, Izquierdo & Sánchez 2015); but more research is needed. It is important to note that the positive association of the species to the presence of lawn that we only found in the breeding season reinforces the vision of the need of a green urban planning that boosts the availability of invertebrates and supports the urban breeding populations of the species (Weir 2015).

On the other hand, four variables were negatively associated with sparrow abundance: building height, number of people, traffic volume and street width. Building height emerged as a consistent explanatory variable for reduced sparrow abundance across every season, which suggests that they are negatively linked to heavily built-up areas of the cities, as Murgui (2009) also found in Valencia. Urban House Sparrows feed on the streets, so an increase of building height may increase the vertical distance between feeding areas and roosting sites, probably increasing their cost of foraging. Furthermore, new built-up areas of higher socioeconomic level may provide wholly unsuitable habitats for House Sparrows (Wotton et al. 2002, Shaw et al. 2008, Moudrá et al. 2018). In agreement with Shaw et al. (2008), we found that, at least in one autumn, socioeconomic status could be an important factor for the species because it was more abundant in deprived areas than in industrial ones and more developed parts of urban areas.

The number of people and volume of traffic were negatively related to sparrow abundance in four and three different seasons respectively, findings contrary to Menon et al. (2013). They did not find any correlation between pedestrians or number of vehicles and House Sparrows abundance; probably it was due to urban differences between India and Spain. Our findings suggested that the disturbances to birds caused by many people may outweigh their benefits to birds in terms of food availability and the avoidance of streets with dense traffic, which will be the most polluted. This result suggests that pollution in urban areas may be a major driver of House Sparrow decline (Summers-Smith 2007). Herrera-Dueñas et al. (2017) suggested that air pollution in cities may be increasing oxidative stress in House Sparrows, entailing negative consequences on the body condition of the individuals and on the viability and future fitness of their offspring; while noise pollution may affect negatively to the reproductive fitness of sparrows (Schroeder et al. 2012, Meillère et al. 2015). A possible prediction for this situation would be that the most polluted cities within the House Sparrow's range would see the steepest declines in bird numbers, but more research is needed.

The negative association of street width and House Sparrow abundance in summer suggests their preference by narrow streets than wide ones to breed, features that are being lost in new neighbourhoods to favour motor vehicles (European Commission 2004).

Variables without effect occurred in every season: distance to the nearest urbanized limit, presence of terraces of bars/cafeterias, presence of schools and presence of private gardens. However, these last three variables stood out because they seemed important sources of human-related food scraps and were never selected. It is possible that food scraps of terraces may not have been available for them at the census time because human intense activity has not started yet in those places. In addition, the terraces of the study area are gathered up and cleaned at night, reducing their early morning scraps availability. In line with Magudu

& Downs (2015), we found no association between school presence and House Sparrow abundance. Food scraps of school yards may not have been available for them at the census time because the point counts were carried out before break time. Similar anthropogenic resources for sparrows were found to relate negatively to sparrow abundance in Skórka et al. (2016). Furthermore, we found no association between private gardens and sparrow abundance in any season. In Great Britain private gardens are key habitats for House Sparrows (Chamberlain et al. 2007, Shaw et al. 2011), so a possible explanation for this difference might be a fundamental lack of such land use in Spain compared with Great Britain (Monclús 1998); and they usually have bird feeders, which is not common in Spain (Chamberlain et al. 2005, Reynolds et al. 2017).

Key environmental factors in urban parks

Our results showed that park area is positively related to bird abundance. Contrary to our findings, Murgui (2009) found a negative relationship between park area and abundance of House Sparrows in Valencia. However, the medium-sized parks favoured by birds in Valencia (Murgui 2009) were similar in size to the largest parks supporting the higher bird numbers in our study.

Greater spatial separation between urban parks reduced the abundance of birds, suggesting a positive role for connectedness between neighbouring parks. Again, Murgui (2009) did not find such a relationship, probably due to urban differences between study areas, but clearly more research is required to understand how or if green connectivity promotes abundance of House Sparrows in urban areas. Rubbish bins may also provide a constant and reliable source of food for birds during the winter within the parks increasing their abundance. Contrary to expectations, rubbish bins were selected positively during the second winter, the warmer one. We expected that the species would be more dependent on these reliable food resources in the coldest seasons. However, a possible explanation for this situation could be related to the other selected variable in the second winter, the distance between parks. In this winter, closer parks had more House Sparrows than those far apart, so the movement of individuals between parks could have resulted in increased abundances of the species around key factors, such as rubbish bins.

The presence of dogs reduced the abundance of birds in summer. In this season people walk the dog earlier increasing disturbances and predation risk for House Sparrows (Forrest & St. Clair 2006), especially juvenile ones. Our results suggest that the species avoid parks with dogs, at least in summer. Peris & Montelongo (2014) found no key factors for the presence or absence of House Sparrows in urban parks of Salamanca (Spain) during 2009-10 breeding seasons. Besides, only the abundance and diversity of exotic trees (introduced from the native range of the species) was significant for the abundance of sparrows in a Mexican city during May 2004 (González-Oreja et al. 2012). More fine-scale research is urgently needed within their main feeding habitats to identify precisely the key factors that have led to the House Sparrow decline in urban areas.

Conclusions

We propose that the main causes of urban House Sparrow decline in south-east of Spain may be related to urban land-use and infrastructure. Certainly, the replacement of above-ground rubbish bins by underground ones reduces the availability of food supplies to sparrows (Bokotey & Gorban 2005), but this highlights a major problem: urban areas and parks harbour scarce natural food for this species. In European cities generally, changes in urban parks such as increased paving, use of more pesticides, changes in habitat structure and use of sprinklers instead of hoses for the irrigation of lawns have decreased their suitability for the species and are responsible for its decline (Mitschke & Muslow 2003, Murgui 2009). Consequently, the Green Infrastructure Project of the Valencian Community (Cantó 2014) would be one urban planning option to halt and, in time, reverse the negative trend of the species in the Valencian area because even small green patches provide suitable habitats for it (Rega et al. 2015). Similar projects could create sufficient heterogeneous urban green space that reduces predation of the target species in the long run (Summers-Smith 2003, Shaw et al. 2008) while also ameliorate air quality reducing oxidative stress currently experienced by birds in polluted city areas (Herrera-Dueñas et al. 2017) while also promoting reproductive success of birds by increasing their insects and natural foods availability (Peach et al. 2014). Also, the theory of ‘traffic evaporation’ (European Commission 2004), which proposes to reduce road capacity for cars in city centres, can represent a sustainable and efficient planning solution for

European House Sparrows. Arguably, the development of high-rise buildings using modern materials and designs, increased socioeconomic status and increased urban population density are negatively impacting House Sparrow abundance (Shaw et al. 2008), so careful thought is required to accommodate all of these ‘forces’ to benefit the species through management options (European Commission 2004, Cantó 2014). Clearly, more fine-scale studies have an important part to play if we are to define management options that can be rolled out over large and disparate spatial scales in Spain and elsewhere.

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Appendices

APPENDIX 1- Initial urban landscape results

Multicollinearity was not found among variables of the resources group. The variable vegetal cover of the urban structure group presented high values of VIF (> 5) in four seasons, so it was not included in the initial analyses of that ones. Also, the variable altitude above sea level from human factors group, presented multicollinearity (VIF > 5) in every season and it was not included in the initial analyses of its group.

In the resources group, only the variable presence of park had always a positive significant effect on the abundance of House Sparrows (Table 6A). Furthermore, rubbish bins had also a positive and significant effect on sparrow abundance in three of the five replications but not in autumn (Table 6A). Other variables, such as the presence of private gardens, lawn and vacant lots had a positive significant effect on the House Sparrow abundance in one of the five replications (Table 6A). However in summer, private gardens had a negative significant effect (Table 6B). Also, geography had significant positive effects in both winters (Table 6A).

In the urban structure group, building height always had a negative significant effect on the abundance of birds (Table 6B). Other variables had significant effects once or twice: presence of trees, street width and presence of old buildings. The first one had a positive effect in summer and the second winter. However, the second and third variables had negative and positive effects, respectively, in summer 2015 and autumn of 2014 (Table 6). Besides, geography had significant positive effects in both winters (Table 6A).

In the group of direct effects of humans, the number of vehicles always had negative effects on House Sparrow abundance while the number of people was selected negatively in every season, except in the last winter (Table 6B). The socioeconomic status had positive effects over the species in both autumns and the last winter. The reference level of the socioeconomic status variable was the industrial area, so the other categories were positive for the abundance of the species (Table 6A). Also, geography had significant positive effects in both winters (Table 6A).

In regard to joint effects only one variable, rubbish bins, of the resources group in the second winter had a larger joint than independent contribution, suggesting a slight correlation with other variables (Table 6A). The rest of variables presented larger values of independent than joint contribution, suggesting a low degree of correlation between them (Table 6).

APPENDIX 2- Initial parks results

Dominant vegetation presented high values of VIF (> 5) in every season and in order to avoid multicollinearity was not included in the initial analyses of its group. Also, the variable altitude above sea level of the location group presented high values of VIF (> 5) in every season and was not included in the initial analyses of the group. The rest of the variables were included in their respective analyses because they did not present multicollinearity.

It is noteworthy that 13 variables of the 19 analysed had no significant effects on sparrow abundance in parks: vegetation cover, presence of cats, socioeconomic status, number of people per minute, presence of terrace, presence of water body, survey zone, distance to the nearest urbanized limit, average vegetation height, presence of shrubs higher than 1 m, presence of exotic vegetation, presence of sand and dominant substratum. The variable with positive and significant effect on the abundance of birds in the first autumn was the presence of lawn while the presence of play areas for children in the park was negative for the abundance of the species (Table 7). In the second autumn, the distance to the nearest park had a negative effect on the abundance of the species. In summer, the presence of dogs had a negative effect on sparrows, while the park's area effect was positive for them (Table 7). Finally, the only winter with significant variables showed that rubbish bins and geographical position were positive for the abundance of the species while the distance to the nearest park was negative (Table 7).

In regard to joint effects every variable of each group and season had larger independent than joint contributions in these initial analyses, suggesting a low degree of correlation between them (Table 7).

Table 6. Variables identified by the initial Hierarchical Partitioning (HP) analyses as key factors affecting the abundance of House Sparrows in urban areas of south-east of Spain between 2014 and 2016. A) Variables with a positive effect on the abundance of House Sparrows. B) Variables with a negative effect on the abundance of House Sparrows. The variable geography could be selected in different groups because it was included in every analysis to control overall location effects of the sampling points.

	Season	Group	Variable selected	I^{j}	J^{k}	*I %*	*Z.score*
A)	**Autumn 2014**	Resources	Urban park	0.0392	-0.0004	39.65	4.52
			Vacant lot	0.0256	-0.0033	25.92	2.50
		Urban structure	Old building	0.0607	0.0008	47.65	7.96
		Human factors	Socioeconomic status[l]	0.0016	0.0016	60.69	4.23
	Winter 2014 – 15	Resources	Urban park	0.0493	0.0051	32.00	5.21
			Rubbish bin	0.0309	0.0265	20.06	2.92
			Private gardens	0.0210	-0.0062	13.64	2.07
			Geography	0.0378	0.0303	24.54	4.13
		Urban structure	Geography	0.0694	-0.0012	50.70	8.22
		Human factors	Geography	0.0762	-0.0081	38.99	8.84
	Summer 2015	Resources	Urban park	0.1626	0.0702	50.97	21.38
			Rubbish bin	0.0721	0.0146	22.60	8.61
			Lawn	0.0355	0.0427	11.12	4.06
		Urban structure	Trees	0.0510	-0.0030	24.94	6.12
	Autumn 2015	Resources	Urban park	0.0428	0.0060	50.07	4.81
		Human factors	Socioeconomic status[l]	0.0495	-0.0041	36.43	1.79
	Winter 2015 – 16	Resources	Urban park	0.1208	0.0471	37.73	15.09
			Rubbish bin	0.0348	0.0358	10.86	3.92
			Geography	0.1163	0.0546	36.32	13.78
		Urban structure	Trees	0.0385	0.0353	14.47	4.80
			Geography	0.1561	0.0148	58.68	17.07
		Human factors	Socioeconomic status[l]	0.0508	-0.0159	17.54	1.92
			Geography	0.1867	-0.0159	64.47	23.37
B)	**Autumn 2014**	Urban structure	Building height	0.0524	0.0117	41.09	5.60
		Human factors	People/minute	0.0326	0.0084	22.57	2.98
			Vehicle/minute	0.0225	0.0160	15.55	1.98
	Winter 2014 - 15	Urban structure	Building height	0.0465	-0.0019	33.97	5.16
		Human factors	People/minute	0.0419	-0.0153	21.42	4.91
			Vehicle/minute	0.0384	0.0181	19.63	3.73
	Summer 2015	Resources	Private gardens	0.0175	0.0152	5.48	1.68
		Urban structure	Building height	0.0994	-0.0032	48.66	12.06
			Street width	0.0335	0.0285	16.39	3.28
		Human factors	People/minute	0.0263	-0.0003	26.16	3.03
			Vehicle/minute	0.0193	-0.0015	19.22	1.73
	Autumn 2015	Urban structure	Building height	0.0311	0.0214	27.91	3.30
		Human factors	People/minute	0.0432	0.0033	31.74	5.02
			Vehicle/minute	0.0269	0.0106	19.80	2.67
	Winter 2015 – 16	Urban structure	Building height	0.0390	0.0141	14.67	4.33
		Human factors	Vehicle/minute	0.0404	0.0063	13.94	3.92

[j] Independent effect of the variable

[k] Joint effects caused with other variables

[l] The reference level of the socioeconomic status was the industrial area, so all other categories were positive for the abundance of the species

Table 7. Variables identified by the initial Hierarchical Partitioning (HP) Analyses as key factors affecting the abundance of House Sparrows in urban parks of south-east of Spain between 2014 and 2016.

Season	Group	Variable	I^j	J^k	*I %*	*Z.score*	Sign
Autumn 2014	Vegetation and substratum	Lawn	0.0671	-0.0155	31.38	2.44	+
	Resources	Play area	0.0707	0.0079	62.79	1.69	-
Summer 2015	Location	Park's area	0.0883	-0.0014	56.63	2.52	+
	Urban factors	Dogs	0.0689	-0.0011	32.89	1.78	-
Autumn 2015	Location	Distance to the nearest park	0.0894	-0.0138	57.47	2.45	-
Winter 2015-16	Vegetation and substratum	Geography	0.0734	-0.0160	33.94	2.03	+
	Resources	Rubbish bin	0.1495	0.0320	72.61	4.79	+
	Location	Distance to the nearest park	0.0711	0.0174	46.35	2.15	-

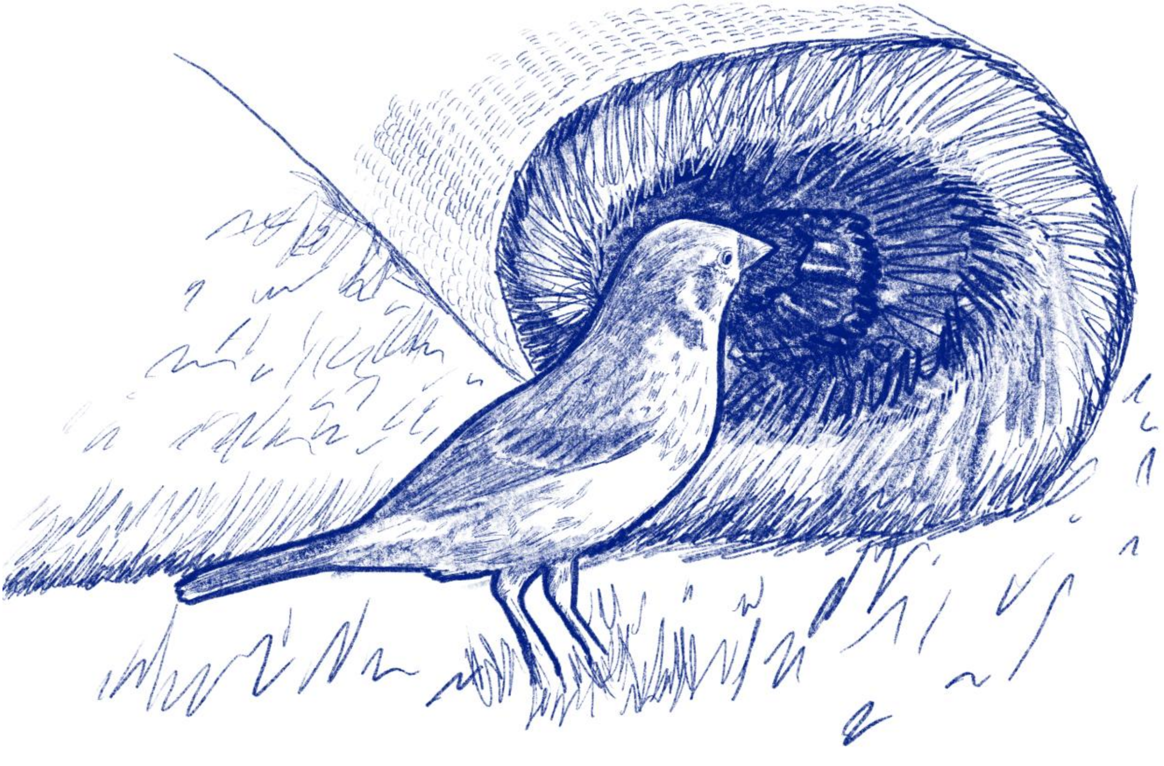
[j] Independent effect of the variable

[k] Joint effects caused with other variables

Chapter 4

Replacement of semi-natural cover with artificial substrates in urban parks causes a decline of House Sparrows *Passer domesticus* in Mediterranean towns

Edgar Bernat-Ponce, José Antonio Gil-Delgado, & Germán Manuel López-Iborra



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REPLACEMENT OF SEMI-NATURAL COVER WITH ARTIFICIAL SUBSTRATES IN URBAN PARKS CAUSES A DECLINE OF HOUSE SPARROWS *PASSER DOMESTICUS* IN MEDITERRANEAN TOWNS

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Abstract

Many European towns and cities have undergone reurbanisation processes in recent decades. However, their effects on urban biodiversity have been poorly studied. Currently urbanisation processes include park remodelling, which involves the replacement of semi-natural substrates (natural grass, bare soil) with artificial ones (pavement, concrete, areas for dogs, artificial grass). Our aim was to explore, for the first time, the effects of park remodelling on the trends and abundance of a declining urban exploiter, the House Sparrow. An abundance index of House Sparrows was obtained in 32 urban parks of four towns in the Valencian Community (Spain) in four summers (2015-2018). Of the studied parks, 10 were remodelled during the study period. Before remodelling, the trends and abundance of House Sparrows in both groups of parks were similar and stable on average, which suggest no bias between both park groups. However, House Sparrow abundance was significantly reduced in parks affected by remodelling works while in those non-remodelled it remained stable. Park remodelling might be linked to a reduction in both habitat suitability and availability of trophic resources, which could harm urban House Sparrows populations and possibly other species as well. Therefore, new park policies and urban planning measures are urgently needed to preserve urban House Sparrows.

Introduction

Urban areas are expanding across the globe and their growth has increased exponentially during recent decades, usually at the expense of rural or suburban areas, and is one of the most important threats to biodiversity in the world (Isaksson 2018). The urbanisation of natural landscapes can affect wildlife through the elimination of some species and the entry of new ones that accept the proximity of humans and the new challenges that this implies, such as chemical pollution, new food resources, new predators or acoustic pollution (Herrera-Dueñas et al. 2017; Isaksson 2018; Shryock et al. 2017). Once urbanisation has been well-established, it is not unusual to change the landscape again through a process that implies new urban organisation, defined as “reurbanisation” or “urban renewal” (Bouzarovski et al. 2010; Buzar et al. 2007a; Van den Berg et al. 1982, Vermeijden 2001). At the end of the 20th century and at the beginning of the present century, most European cities and towns have undergone reurbanisation, which consists in the diversification of household structures, housing functions and the repopulation of inner-city neighbourhoods (Buzar et al. 2007b; Cheshire 2006; Vinuesa 2005), and is also linked to higher human socio-economic levels (Shaw et al. 2008).

Reurbanisation does not impact natural habitats, but causes degradation, modification or loss of existing urban green spaces, such as urban parks and/or private gardens, which are the main habitat of urban wildlife (Bernat-Ponce et al. 2018; Chamberlain et al. 2009; Cornelis and Hermy 2004; De Coster et al. 2015; Peris and Montelongo 2014; Verbeeck et al. 2011). These green urban spaces are especially important for birds during breeding seasons as they constitute the most important source of invertebrate food, which is essential for their breeding success (Chamberlain et al. 2007, 2009; Peach et al. 2015; Vincent 2005). Current new changes in urban green landscapes, linked to reurbanisation, together with increasing human socio-economic status, tend to remodel parks to meet human purposes which make these habitats less suitable for urban biodiversity. Remodelling activities potentially affecting biodiversity include: the paving of green areas (Mitschke and Mulsow 2003), increasing use of pesticides (Summers-Smith 2003), using sprinklers instead of hoses to water lawns (Murgui and Macias 2010), or building areas for dogs, paving children's play areas with rubber, and replacing natural lawn with artificial grass. These park remodelling actions linked to reurbanisation processes are transforming traditional parks into domestic modified versions of them. For example, replacing natural lawn with artificial grass is seen as a way to save water (Moore 2009) and to reduce management requirements in Mediterranean climates. These changes could lead to a significant reduction of the diversity and number of available invertebrates which could have an important effect limiting the reproductive success and survival of urban bird species (Chamberlain et al. 2009; Peach et al. 2015).

The House Sparrow *Passer domesticus* has been, and still remains, the most closely associated wild bird species with humans since prehistory (Anderson 2006; Ericson et al. 1997). However, throughout the 20th century and at the beginning of the present one, the population of this human-adapted species has drastically declined in urban and rural areas (Hayhow et al. 2014; Summers-Smith 2003). In London, its decline was first noticed in the 1920s when engines replaced the horses that pulled trams (De Laet and Summers-Smith 2007; Summers-Smith 2005). Almost 50 years ago, House Sparrow populations in large cities of Western Europe began showing a new declining trend (Hayhow et al. 2014; Summers-Smith 2005, 2007).

The causes of urban House Sparrow decline still seem unclear and researchers have proposed several potential drivers, including: increased urban pollution (Herrera-Dueñas et al. 2017; Peach et al. 2018; Summers-Smith 2007); lack of nest sites due to the structure of the new buildings (Moudrá et al. 2018); an increased human socio-economic status (Shaw et al. 2008); poor quality of available trophic resources (Herrera-Dueñas et al. 2015); limited access to household waste after replacing communal surface trash containers with underground ones (Bernat-Ponce et al. 2018); increased electromagnetic radiation (Balmori and Hallberg 2007); increased predation by Eurasian Sparrow hawks *Accipiter nisus* or domestic cats *Felis silvestris catus* (Bell et al. 2010; Thomas et al. 2012) and increased exposure to avian malaria (Dadam et al. 2019).

Bird species are likely to present negative population trends in the most modified environments (Siriwardena et al. 1998), and even though urban House Sparrows have adapted to human presence and are considered urban exploiters (Isaksson 2018), they seem less abundant in highly urbanized zones with artificial surfaces (roads, streets, pavement networks or parking spaces), which could be linked to this reurbanisation concept. Furthermore, remodelling their main feeding habitats, such as urban parks and green areas, could have strong effects on their populations (Bernat-Ponce et al. 2018; Murgui 2009; Šálek et al. 2015; Shaw et al. 2008). House Sparrows need green stepstones to maintain an exchange between key urban populations (De Laet and Trappeniers 2019). The reduced suitability of remodelled urban parks could mean the loss of green stepping stones which could, in turn, have negative impacts on populations due to the limited home ranges of this species in urban areas (Heij 1985).

Here, our aim was to explore if park remodelling (including building areas for dogs, paving children's play areas with rubber, paving with concrete and asphalt, and/or replacing lawn with artificial grass), which is linked to the reurbanisation of towns and cities, negatively affects House Sparrow populations. Our main aim was to detect if park remodelling triggers a population decline in these urban green areas. To our knowledge, the effects of these changes to park landscapes on urban biodiversity have never been studied despite being a potential cause of the decline in House Sparrows and other species in urban areas.

Methods

Study area and bird census

The study was carried out in 32 urban parks distributed into four small to mid-sized towns in adjacent areas in the provinces of Alicante and Valencia (SE Spain; Fig. 1) with between 9000-59000 inhabitants (Table 1). Most of the towns in the region (86%) have fewer than 15,000 inhabitants and 12% are medium-sized (15,000-60,000 inhabitants), thus the selected ones are representative of the most frequent type of urban areas in the region (Instituto Nacional de Estadística 2018).

Fig. 1 Location of the four towns selected to study the effects of park remodelling on House Sparrow populations in the Valencian Community (SE Iberian Peninsula) (source: the authors) (QGIS 2018)

Table 1. The geographical, demographical and climate characteristics of the towns selected for the study. This table was produced by the authors using data from AVAMET (2018) and the Instituto Nacional de Estadística (2018).

Locality	Coordinates	Number of inhabitants (2017)	Temperature °C (2017)			Rainfall mm (2017)	Altitude (m.a.s.l.)
			Max.	Mea.	Min.		
Muro de Alcoy	38°46´47´´ N 0°26´10´´ W	9319	40.2	16.3	-5.9	458.6	410
Cocentaina	38°44´42´´ N 0°26´26´´ W	11461	40.0	16.6	-3.6	390.1	430
Alcoy	38°41´54´´ N 0°28´25´´ W	59106	40.4	16.1	-4.4	439.2	562
Onteniente	38°49´20´´ N 0°36´26´´ W	35342	38.4	17.2	-3.4	373.0	353

A House Sparrow abundance index was determined in each park as the number of House Sparrows counted during 5 minutes in one point count of 50 m radius (Gibbons and Gregory 2006) per season in four summers (2015-2018). Counting was done between the end of June and early July, a time that coincides with the end of the breeding season in eastern Spain (Gil-Delgado et al. 1979). The circle around each point count covered an area of 0.79 ha, while the mean size of the 32 parks was 0.90 ha (± 0.01 SE; ranging from 0.01 to 2.80 ha) (Table 2). For the parks whose surface was smaller than 0.79 ha, surrounding streets were included in the surveyed point count. Park areas and distances between them were measured by the SIGPAC cartographic viewer (2018) (Table 2). The mean extension of parks and their shape did not allow us to locate more than one point per park. However given the good detectability of sparrows a single point count per park was considered appropriate to obtain a representative abundance index. Furthermore, for the biggest parks, point counts covered more than the 10% of the park expanse, which is considered a significant part of the study area (Buckland et al. 2015).

Previous studies in the region have shown that although House Sparrows rarely nest in urban parks, these areas are key places for feeding all year round (Bernat-Ponce et al. 2018; Murgui 2009). Thus, by doing counts at the end of the breeding season we expect to include the abundance of the adults breeding nearby and of recently independent young birds. At each point count abundance of birds was recorded by a single observer, to avoid observer bias, for five minutes with a one-minute settlement period. Special attention was paid to avoid double counting and birds were recorded by sight and vocalisations. Counting was done between the daily official sunrise time and two hours later, which comprises the period when House Sparrows are most active and human activity in parks is minimal (Anderson 2006). Counting was not carried out on rainy and/or windy days.

During the study, we detected remodelling works in 10 parks. The exact dates of remodelling works were provided by the Environment Departments of the studied localities. Remodelling works were mainly carried out in autumn or winter (Table 2); hence the abundance indexes do not reflect their immediate impact but the level of the population some months later. We recorded the type of any important remodelling event that could affect the species in each park, and in particular the type of replaced substrate (natural grass or bare soil) and the type of the new substrate, which included artificial grass, concrete or rubber playground surfaces, building areas for dogs and paving. Remodelled surfaces were measured by the SIGPAC cartographic viewer (2018) (Table 2) and the percentage of remodelled area within each park was calculated. In each studied remodelled park the replaced substrate was of just one type (bare soil or natural grass). The average area of the remodelled (1.21 ± 0.34 SE ha; n = 10) and non-remodelled parks (0.76 ± 0.11 SE ha; n = 22) were similar (t = 92.00; *p* = 0.483).

Table 2. Total number of studied parks in each town, mean distance between the studied parks in the same town, mean area (ha) of parks, number of parks remodelled before each census year (number in brackets), remodelling types, the percentage of remodelled surface (in brackets) and remodelling month (in brackets) in these parks. The last row shows the summary of each column, except for the last one.

Location	Studied parks	Mean distance (m) between parks ± SE	Mean area of parks (ha ± SE)	Parks remodelled before census year	Number, type of modifications, % of remodelled surface and remodelling month
Muro de Alcoy	3	507 ± 131	0.56 ± 0.03	-	-
Cocentaina	5	303 ± 16	0.55 ± 0.2	2018 (2)	(1) Building an area for dogs on a bare soil area (7.18) (Jan 2018) (1) Brick paving in a bare soil area (10.55) (Jan 2018)
Alcoy	15	385 ± 41	2.07 ± 0.7	2016 (2) 2017 (2) 2018 (3)	(1) Concrete paving on natural grass and in an area with trees (16.98) (Feb 2018) (1) Building an area for dogs and replacing bare soil with artificial grass (32.03) (Nov 2015) (1) Replacing natural grass with artificial grass in the whole park (72.03) (Feb-Apr 2018) (1) Concrete and rubber paving in a bare soil area (7.47) (Nov 2016) (3) Replacing natural grass with artificial grass in some areas (1.61/1.42/4.49) (Feb 2018 /Sep-Dec 2015 / Sep 2016)
Onteniente	9	338 ± 24	0.62 ± 0.3	2018 (1)	(1) Concrete paving in a bare soil area and installing artificial grass patches (6.20) (Mar 2018)
Total	32	370 ± 24	0.90 ± 0.13	2016 (2) 2017 (2) 2018 (6)	

Statistical analysis

First, we tested if House sparrows' trends and abundance index were similar in remodelled and not remodelled parks before remodelation works. Trends were evaluated by calculating the instantaneous rate of increase (*r*) (Begon et al. 2006) for each park between consecutive seasons (2015-2016, 2016-2017, 2017-2018) as $\ln(N_{t+1}/N_t)$. The zero abundance values in the counts were replaced with one because it was necessary to calculate natural logarithms to estimate the instantaneous growth rates. This implied that if no sparrow was detected, at least one was considered to be present, which seemed a reasonable assumption given the species' ubiquity. Mean growth rates and abundance of remodelled parks (from the mean of the years before changes in each park) and non-remodelled parks (from the mean of the four study years in each park) were compared using two sample *t*-tests. This was useful to check if remodelled parks were a biased sample of the available parks of the study area. These analyses were performed using the Shapiro Wilk normality test ("shapiro.test" function of the "stats" package) (R Core Team 2013) to check for the normality, the Levene test ("leveneTest" of the "car" package) (Fox and Weisberg 2019) to check for homogeneity of variances of both groups of parks (Samuels et al. 2012) and the "t.test" function to perform the two sample *t*-test ("stats" package). We performed all the statistical analyses in RStudio 3.5.1 (RStudio Team 2018).

Given that sparrows' abundance differed between parks, a relative index of abundance was calculated to compare the average trend shape between the park groups defined by remodelling year or lack of remodelling works. To this end, the abundance index of each park in each year was divided by the maximum recorded abundance index of that park in all four years. In this way, it was possible to evaluate if the parks remodelled in different years presented distinct abundance trends.

We used General Linear Mixed Models (GLMM) to evaluate the park variables that could be related to the changes in the abundance index of House Sparrows. Fixed factors tested in the models were: year of study (2015, 2016, 2017, or 2018 as categories), park remodelling (0: absence; 1: presence; i.e. parks were assigned 0 before remodelling works and 1 afterwards), park size category defined according to thresholds of first and last quartiles of park area distribution (small: < 3685 ha; medium: 3686-12400 ha; large: > 12400 ha) and percentage of park area remodelled (0 before remodelling year, constant value after that). Locality and park identity (ID), nested within the former, were considered as random factors. The function “glmmTMB” of the R package “glmmTMB” (Brooks et al. 2017) was used to fit the GLMM.

First of all, we started fitting a GLMM with just the random effects, park identity nested within locality, and using a Poisson distribution. We detected overdispersion after checking this model with the function “check_overdispersion” of the package “performance” (Lüdecke et al. 2019). To avoid this problem, first we fitted to variants of the same model but using the Negative Binomial distribution type 1 (family nbinom1) and type 2 (family nbinom2) (Blasco-Moreno et al. 2019). In both cases we did not detect overdispersion in the fitted models. The model using the “nbinom1” family had lower AIC value, and was selected as a basis to develop subsequent models. Then, we evaluated if random effects could be simplified by examining their variances.

The next step was to include by turn each fixed effect in the GLMM using random intercept and we selected the model that had the lowest AIC. In addition, we tested the significance of fixed effects using the “Anova” function of the “car” package (Fox and Weisberg 2019). Finally, we added the other fixed factors by turn to the best model to check if AIC was reduced by including an additional parameter. For the best model we calculated the intraclass correlation (ICC), to estimate the proportion of the variance in abundance that was accounted for by the random effect, using function “icc” of the package “sjstats” (Lüdecke 2019).

Alteration of lawn or bare soil areas could have different consequences for House Sparrows as the amount of food resources that they provide is likely to be larger in grass-covered areas. Therefore, we checked whether the extension of the remodelling works that affected such types of areas was associated with different declining House Sparrow rates. For this purpose, we used beta regression models, which are appropriate when the dependent variable is a proportion (interval 0, 1) and there is, therefore, neither normally distributed nor homoscedastic (Cribari-Neto and Zeileis 2010). In this analysis, we used the proportion of decline between the year before and the year after remodelling works in the remodelled parks as the dependent variable, and the proportion of remodelled surface and the affected substrate type (lawn n = 5; bare soil n = 5) as the predictor variables. Beta regression was conducted with the ‘betareg’ function of the ‘betareg’ package (Cribari-Neto and Zeileis 2010).

Results

The mean instantaneous growth rate in the non-remodelled parks (r = -0.07 ± 0.06 SE; n = 22) and in the remodelled parks before the remodelling works took place (r = -0.09 ± 0.16 SE; n = 8) were not different (two sample t-test = -0.132; df = 8.76; p = 0.898). Similarly, mean abundance in the non-remodelled parks (13.24 ± 3.05 SE; n = 22) and in the remodelled parks before the remodelling works took place (17.12 ± 1.59 SE; n = 10) were not different (two sample t-test = 1.127; df = 14.104; p = 0.279). Therefore, on average both the non-remodelled and remodelled parks showed similar population trends and abundance before remodelling of the latter.

The House Sparrows in the non-remodelled parks showed a slightly negative trend, while the remodelled parks presented sharp declines in abundance after the remodelling year (Fig. 2). The pattern was less clear in the parks remodelled between 2016 and 2017 because House Sparrow abundance in one park increased in 2018. Overall, the House Sparrows in the non-remodelled parks declined by about 15% between 2015 and 2018, while the populations in the remodelled parks declined by more than 60% during the same period. When considering only the change in the year interval when a park was remodelled, the abundance index lowered by a mean of 62.33% (± 8.86 SE). Thus, in the remodelled parks, most of the decline between 2015 and 2018 occurred in the year when the parks' conditions changed.

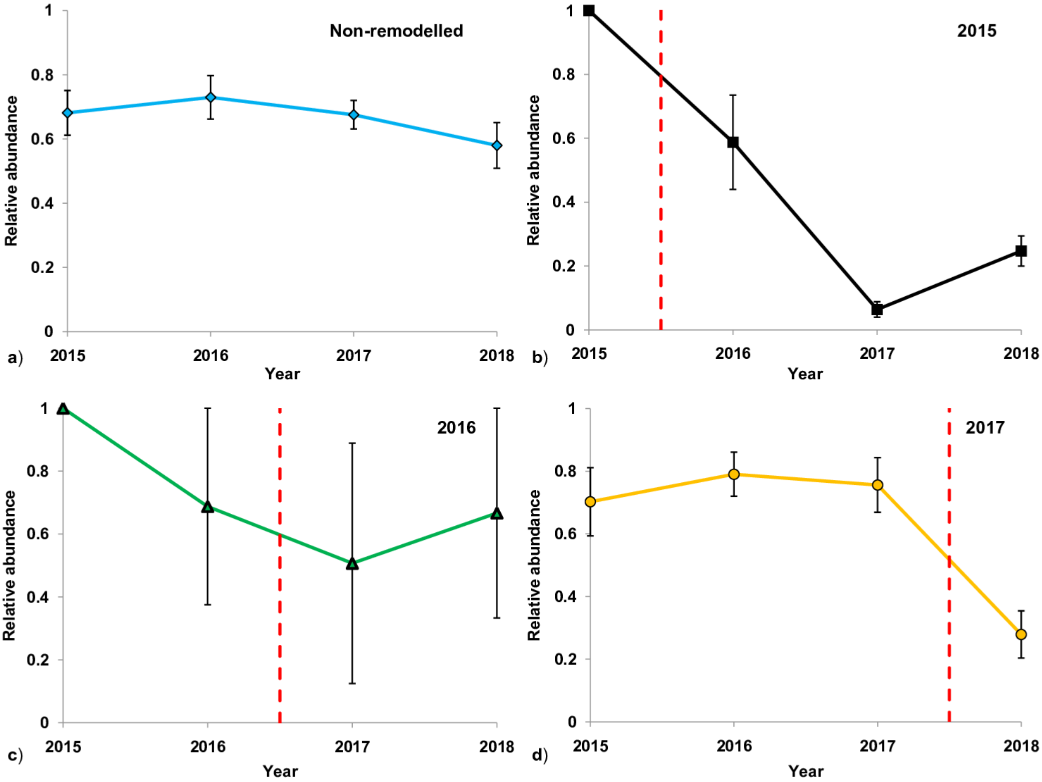

Fig. 2 The House Sparrow relative abundance trends (± SE) in the studied parks classified according to the remodelling year. a) Non-remodelled parks (n = 22). b) The parks remodelled between the summers of 2015 and 2016 (n = 2). c) The parks remodelled between the summers of 2016 and 2017 (n = 2). d) The parks remodelled between the summers of 2017 and 2018 (n = 6). The dashed vertical lines mark the remodelling period

In the first GLMM with random effects the variance for park identity was 0.255 while variance for locality was almost zero (1.19e-09); hence we ran a new model with just the park identity as random effect which had a lower AIC (Table 3). Then we built random intercept models including one fixed effect each. The best model included the presence of remodelling works with a negative coefficient. Models including the other fixed effects were less plausible as presented ΔAIC > 5. The addition of further fixed effects led to the model with lowest AIC when the percentage or remodelled area was added, but ΔAIC for the model including just remodelling was 0.2 (Table 3). Regression coefficients both for remodelling and percentage or remodelled area were negative, although the last one did not reach significance (*p* = 0.17). Adding year to this model did not decrease further the AIC. Thus, we have two equally plausible models that include the presence of remodelling works and one of them the percentage of area remodelled. In these models, ICC for the park identity random effect was 0.495 and 0.503, respectively.

Table 3. Fitted GLMM models to evaluate the park variables that could be related to the changes in the abundance index of House Sparrows. Random effects are in brackets. Significant p-values are marked with (*) = p < 0.05; (**) = p < 0.01; (***) = p < 0.001. ΔAIC is the difference between the selected model and the model with the lowest AIC value.

Model	AIC	ΔAIC
Random effects models		
HS Abund.~ 1 + (1|Locality/Park ID)	868.8	24.8
HS Abund.~ 1 + (1|Park ID)	866.8	22.8
One fixed effects models		
HS Abund.~ Remodelling*** + (1|Park ID)	844.2	0.2
HS Abund.~ % Remodelled area*** + (1|Park ID)	849.4	5.4
HS Abund.~ Year** + (1|Park ID)	859.1	15.1
HS Abund.~ Park area + (1|Park ID)	868.9	24.9
Two fixed effects models		
HS Abund.~ Remodelling*** + % Remodelled area + (1|Park ID)	844.0	0
HS Abund.~ Remodelling*** + Year + (1|Park ID)	845.9	1.9
HS Abund.~ Remodelling** + Park area + (1|Park ID)	846.8	2.8
Three fixed effects models		
HS Abund.~ Remodelling** + Year + Park area + (1|Park ID)	845.6	1.6

In the parks where the replaced substrate was grass, on average the 19.31% ± 13.49 SE (n = 5; range 1.42-72.03%) of the surface was remodelled and the 12.69% ± 4.89 SE (n = 5; range 6.20-32.03%) in the parks where bare soil was affected. Overall, 16.00% of the parks surface (±6.85 SE; n = 10; range 1.42-72.03%) was affected. The beta regression analysis showed a significant interaction between type of substrate affected and the proportion of park area that was transformed (z = -2.35; *p* = 0.019;

Appendix Table 4) (Fig. 3), which indicates that the decline proportion changes depending on whether the substrate affected by remodelling was lawn or bare soil. Replacement of lawn for abiotic substrates affected more negatively House Sparrow abundance than changing bare soil (Fig. 3).

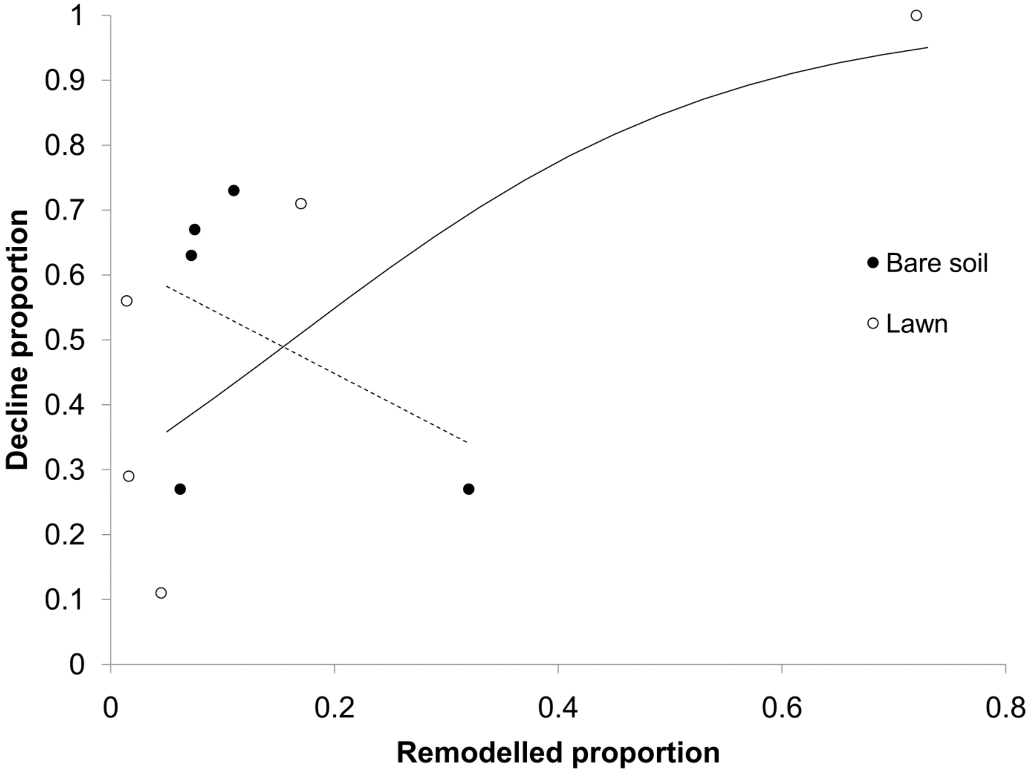

Fig. 3 Beta regression model relating the decline proportion of the House Sparrow abundance index in the first summer after remodelling to the proportion of remodelled area in each park and the affected substrate type (circles: observed data; lines: model prediction: lawn-solid line / bare soil-dashed line)

Discussion

Our results showed that House Sparrow abundance decreased significantly in parks that were remodelled. House Sparrow trends, as measured by the instantaneous growth rate, and abundance index were similar in the parks that were not remodelled and in those that were later remodelled, thus initial differences in the conditions between both groups of parks before works started are unlikely and the most plausible explanation is that changes in park habitat caused by remodelling were responsible for the species decrease. Variability between localities was virtually null, and half of the variance in sparrow's abundance occurred between parks irrespective of their locality, thus the effect of park remodelling was similar in all the studied towns.

Despite 4 years is a short term to precisely determine the effect of park remodelling, abundance lowered immediately after the remodelling was carried out. This scenario indicates a direct link between park remodelling and House Sparrow reductions. This is probably the result of the reduced availability of food resources, which may become even scarcer if suitable foraging areas disappear. In addition, in the few parks modified in 2015 and 2016 abundance tends to remain low several years after remodelling.

It has been suggested that House Sparrow presence and abundance are linked to the available resources offered by green areas (Murgui and Macias 2010; Shaw et al. 2011). Park reforms imply the total or partial replacement of vegetated or soil areas with concrete, asphalt or compact bare ground to build areas for dogs, and even the replacement of natural grass with artificial grass. Even though the park remodelling process is a mixture of diverse changes, they all share the substitution of semi-natural biotic cover (garden plants, weeds and soil) by abiotic artificial substrates (concrete, asphalt, artificial grass), which expectedly reduces food availability for House Sparrows and other urban animal species (Jones and Leather 2012; Smith et al. 2005; Turrini and Knop 2015; Weir 2015). Our results showed that increasing percentage of remodelled area contribute to decrease sparrow abundance, which would be an expected result as it implies a higher reduction in the amount of resources and suitable habitat for the species. In addition, remodelling of the lawn areas of parks is more pernicious for House Sparrow populations than remodelling bare soil areas. Lawn areas are expected to harbour a higher amount of seeds and invertebrates than bare soil, which could explain this result.

To understand the consequences of park remodelling for urban sparrow populations it is necessary to know if the birds that disappeared from remodelled parks died or dispersed to other urban habitats. House sparrows home ranges in urban habitats vary between 0.11 and 0.86 ha (Cheke 1972; Cramp and Perrins 1994; Heij and Moeliker 1990; Vangestel et al. 2010). By assuming a circular home range, the radius of the wider home range reported would be slightly longer than 50 m. The mean distance between parks in the study area was 370 m, thus House Sparrows may have difficulties in finding new parks after their home park has been remodelled. This suggests that most of the birds that disappeared from the remodelled parks either died or switched foraging areas to adjacent urban habitats, such as streets, which are less suitable zones for House Sparrows (Bernat-Ponce et al. 2018). Despite this, it is possible that some of the sparrows that moved from the remodelled parks reached alternative ones or streets with the presence of hedges, green walls and private gardens (Dover 2015; Moudrá et al. 2018), but they would have to compete with resident sparrows there. The population stability that we found in the non-remodelled parks suggests that density-dependent regulation would prevent abundance from increasing in these parks. Consequently, park remodelling could contribute to a general decline of House Sparrow abundance in cities and more research is needed to evaluate its effect on House Sparrow abundance on the whole town scale.

Valencian towns and cities, like many other European ones, are being reurbanized (Boira 1995; López-Gay 2011, 2014) and their parks are being modified by the process that we name “park remodelling”. The substitution of semi-natural biotic cover by abiotic substrates that we describe in this paper contributes to save water for plant irrigation, and reduces long-term maintenance costs. Thus, it is likely that this process will continue to expand, especially in dry or Mediterranean areas where water availability for irrigation is frequently limited (Yaghoobian et al. 2010). Initially, artificial turf was used to replace grass in sport fields in the 1960s, but nowadays it is spreading to schools, child-care facilities, parks, streets, play areas, home lawns and commercial areas (New Yorkers for Parks 2006; Toronto Public Health 2015). Environmental and health concern about artificial turf areas is growing (Cheng et al. 2014; Claudio 2008); especially in relation to its effects on the urban heat island, carbon and water cycle and environmental impacts of its rubber components that derive from scrap tyres (Claudio 2008; Toronto Public Health 2015; Yaghoobian et al. 2010). To our knowledge, no studies have been conducted about its effect on biodiversity, despite the potential ecological consequences that artificial turf has for urban wildlife (Francis 2018). Our results show, for the first time, that replacing semi-natural substrates such as grass or bare soil with artificial grass and other abiotic substrates leads to House Sparrow declines in parks, which could also happen with other species.

Other important feature of the new substrates in the studied urban parks is that they increase their impervious surface. This seems to be part of a general trend for increasing impervious substrates in urban green areas, which has been shown to have an effect on urban hydrological cycle (Perry and Narwaz 2008; Verbeeck et al. 2011; Warhurst et al. 2014). Given this situation, new projects for urban green areas remodelling, either public or private, should consider measures to help maintain them as a suitable habitat for both urban biodiversity and House Sparrows. We suggest prioritizing the traditional design and management of urban green areas and their heterogeneity (Mexia et al. 2018). Furthermore, due to the restricted usual dispersal movements of species, increasing urban green cover, such as tree-lined streets or green stepstones, would favour population connections between parks (De Laet and Trappeniers 2019; Murgui 2007). These urban planning measures, along with the use of native trees, bushes and grass species with fewer watering requirements, might prove beneficial to biodiversity, compensate the water saving produced by artificial grass and reduce its heat island effect (Domene et al. 2005; Moore 2009; Yaghoobian et al. 2010). It would also reduce the amount of management requirements needed by municipalities and would increase the number of harboured invertebrates in urban parks, which would directly benefit urban biodiversity and House Sparrows (Narango et al. 2018; Peach et al. 2008; Turrini and Knop 2015; Vincent 2005). Similar urban planning measures could be taken in already remodelled parks, such as increasing native bush density (Wilkinson 2006), using less insecticides in these green urban areas (Summers-Smith 2003) and appropriate management (Heyman et al. 2017). However, going back to traditional park models would probably be a better option to preserve biodiversity.

Our findings could be useful for suggesting conservation and urban planning measures to be applied on large disparate spatial scales where parks are being, or will be remodelled, and where species are sharply declining (Shaw et al. 2008). More research is urgently needed to precisely identify the short-, mid- and long-term effects of park remodelling and the use of artificial grass on urban biodiversity.

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Appendices

Appendix Table 4. Coefficients of the beta regression model relating the decline proportion of the House Sparrow abundance index in the first summer after remodelling to the proportion of remodelled area in each park and the affected substrate type. Phi is the precision parameter of the model.

	Estimate	**SE**	**z. value**	**p-value**
Intercept	-0.740	0.410	-1.804	0.071
Proportion of remodelled park	5.201	1.620	3.210	0.001
Substrate type	1.259	0.675	1.866	0.062
Interaction (proportion:type)	-8.857	3.768	-2.351	0.019
Phi	6.826	2.942	2.320	0.020

Chapter 5

Effect of replacing surface with underground rubbish containers on urban House Sparrows *Passer domesticus*

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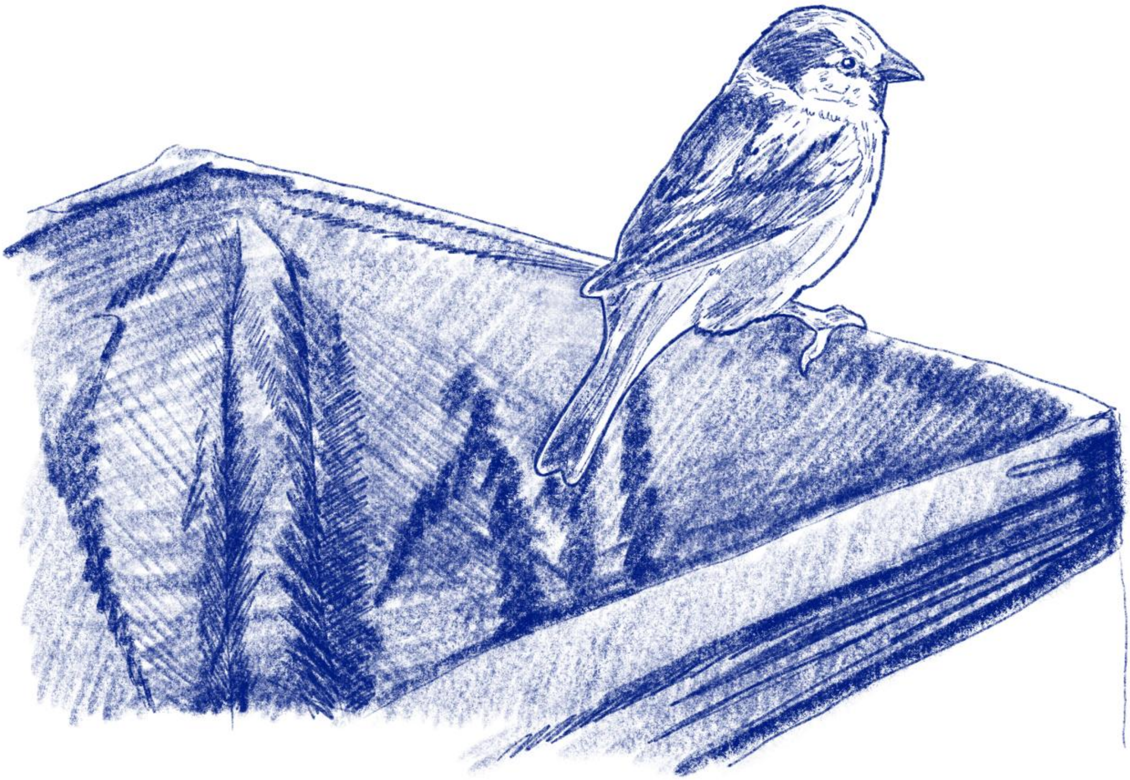

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EFFECT OF REPLACING SURFACE WITH UNDERGROUND RUBBISH CONTAINERS ON URBAN HOUSE SPARROWS *PASSER DOMESTICUS*

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Abstract

Urbanisation processes are increasing worldwide at surprising rates affecting wildlife in many ways: changing habitat structure, reducing resources, and modifying the distribution, composition and abundance of local biota. In different countries, urban waste collection techniques are evolving and surface rubbish containers (neighbourhood receptacles for temporarily storing anthropogenic household waste located above-ground on the streets) are being replaced with underground ones (metal boxes with steel chutes that fed into large underground containers) to improve sanitation measures, to avoid bad smells and waste scattering by animals. We aimed to detect if House Sparrows were more abundant close to surface rubbish containers than close to the underground ones. We recorded an abundance index of House Sparrows during two visits in winter 2018-2019 to point counts located in groups of both container types (80 and 85 groups of underground and surface containers, respectively) in eight towns of Eastern Spain. We modelled the abundance index according to rubbish container type, and 14 other environmental variables at four scales: container, nearest buildings, near urban features, and general locality features using GLMMs. House Sparrows were more abundant close to surface than to underground rubbish containers, which may be linked with higher food debris availability. The presence of other urban features (bar terraces, private gardens, mature trees) interacting with the rubbish containers also influenced the abundance of House Sparrows. The replacement of above-ground rubbish containers with underground ones may deprive House Sparrows resources, which could lead to the decline of this species, especially in urban areas with little green cover.

Keywords

Anthropogenic food; decline; socio-economic level; urban ecology; urbanisation

Introduction

Urbanisation is increasing all over the world at surprisingly quick rates (Murgui and Hedblom 2017; United Nations 2018). This process affects wildlife in many ways, for example, modifying habitat structure which renders some landscapes unsuitable for several species (Bernat-Ponce et al. 2020; Isaksson 2018; Murgui and Hedblom 2017; Verbeeck et al. 2011). These modified environments pose a new challenge for urban avifauna through exposure to novel stressors, such as pollutants, noise, new predators, exotic competitors or processed food (Bernat-Ponce et al. 2018; Francis and Barber 2013; Herrera-Dueñas et al. 2017; Murgui and Hedblom 2017; Schroeder et al. 2012). This significant landscape transformation usually modifies local biota by bringing about changes in its distribution, composition, abundance and structure, or even leads some species to extinction (McKinney 2002; Shochat et al. 2010).

Despite the negative effects of urbanisation on individuals, populations and communities, some species manage to take advantage of several features they find in towns and cities. One of the main attractors for this group of animals to urban areas is constant human-induced food resources (Chace and Walsh 2006), provided intentionally by people such as feeders, or indirectly by rubbish containers or dumpsters (Bernat-Ponce et al. 2018; Reynolds et al. 2017; Tortosa et al. 2002). Thus, urban areas buffer seasonality effects concerning food availability (Murgui and Hedblom 2017).

The House Sparrow (*Passer domesticus*), native to most of Europe, Mediterranean basin and large parts of Asia (Anderson 2006), is a synanthropic species that over the last 50 years has sharply declined in many urban areas (Shaw et al. 2008; Summers-Smith 2003). However, the main cause of the species urban decline remains unclear (Anderson 2006; Summers-Smith 2005). A consensus is starting to emerge that considers the increase in human socio-economic status and urban renewal as key factors for the species' urban decline in Europe. First, changes in urban habitats and loss of green urban areas may negatively impact upon natural food availability, such as seeds and invertebrates (Bernat-Ponce et al. 2020; Pauleit et al. 2005; Vincent 2005). Second, these habitat changes have indirect effects on House Sparrows, such as a rising predation risk by Sparrowhawks (*Accipiter nisus*) and domestic cats (*Felis catus*) due to reduced shelter from predators (Bell et al. 2010; Shaw et al. 2008; Thomas et al. 2012). Finally, as House Sparrows are cavity nesters, nest site availability can diminish due to: changes in design, new materials in modern buildings and the refurbishment of older ones, especially in high income areas which decreases the number of crevices (Moudrá et al. 2018; Shaw et al. 2008). In addition, the competition for nest-sites between House Sparrows and larger invasive species could harm their populations (Charter et al. 2016).

House Sparrows have been human-commensals for more than 10,000 years (Sætre et al. 2012) and this worldwide success has been directly linked with their adaptation to feed on agricultural crops, poultry feed, and human subsidies (Anderson 2006; Bernis 1989; Whelan et al.

2015), and the use of human structures for nesting (Anderson 2006; Moudrá et al. 2018). The link of the House Sparrow with waste food is still important in modern urbanised areas. A previous study conducted in Spain showed that neighbourhood surface rubbish containers (large receptacles, dumpsters or skips, usually of 1000-3500 litres of capacity for temporarily storing anthropogenic household waste located above-ground on the streets; see González-Torre et al. 2003 and Pires et al. 2019) were positively related to House Sparrow abundance, especially in winter (Bernat-Ponce et al. 2018). Food debris is usually found around surface rubbish containers as they are sometimes overfilled, and rubbish bags are deposited outside or sometimes they fall from the containers. Furthermore, surface rubbish containers are accessible to urban animals, such as cats, who break open bags and make the contents available to House Sparrows. In Lvov (Ukraine), a drastic reduction in the number of open surface rubbish containers in the city centre was followed by a rapid decline of House Sparrow populations (Bokotey and Gorban 2005). During the coldest season, natural trophic resources in urban landscapes are scarce and surface rubbish containers reliably provide anthropic high-calorific scraps, such as biscuits or snacks, around them (Bernat-Ponce et al. 2018; Bokotey and Gorban 2005; Herrera-Dueñas et al. 2015).

The effect of rubbish containers on House sparrow abundance is not independent of the influence of other urban landscape characteristics. For example, urban outdoor restaurants can attract House Sparrows to their surroundings as they are a reliable supply of food debris (Haemig et al. 2015). Wooded streets, private gardens and parks may provide shelter and food, especially by native plants (Bernat-Ponce et al. 2018; Chamberlain et al. 2007; Murgui 2007, 2009). The height of the buildings and socio-economic level of the neighbourhood influence House Sparrow abundance (Bernat-Ponce et al. 2018; Shaw et al. 2008). Therefore, it is necessary to take into account the possible effect of urban features at several scales on House Sparrow abundance around rubbish containers.

Recently, neighbourhood underground rubbish containers system (large underground containers where rubbish enters through surface metal boxes with stainless steel chutes on the pavements; see "stand alone underground containers" in ISWA 2013) has spread in towns replacing neighbourhood surface rubbish containers. Even though underground systems are costly and technically more complex than surface containers they are installed to prevent bad odours, increase the cleanliness of urban areas, reduce the presence of mice, rats, cats, and invertebrate animals, save space, and avoid the negative visual effect of traditional neighbourhood above-ground rubbish containers (ISWA 2013; Nilsson 2011). For underground rubbish containers, rubbish bags disappear underground, and thus become inaccessible to House Sparrows. Several European urban areas in different countries, such as the United Kingdom, the Netherlands or Spain, among others (Eroski Consumer 2008; Interesting Engineering 2017; The Hague 2017), have replaced neighbourhood above-ground rubbish containers with underground rubbish containers in the last two decades (ISWA 2013; Nilsson 2011), especially in the centres of towns with higher socio-economic levels (García-Hernández et al. 2017; INTHERWASTE 2019). However, this underground system is quickly spreading to other areas of the towns (ISWA 2013).

In the present study, we aimed to explore if sparrows were more abundant around above-ground rubbish containers than around underground rubbish containers, as well as the potential effects of other urban features surrounding the containers that could affect their use. We hypothesise that the replacement of neighbourhood above-ground rubbish containers with underground ones would have negative effects on House Sparrow populations. If this happens, the change in type of rubbish containers would be an additional mechanism contributing to the negative relation between House sparrow abundance and socioeconomic status proposed by Shaw et al. (2008).

Methods

Study area and bird census

The study was carried out in the Valencian Community (east Spain) (Fig. 1). This region presents a high availability of towns distributed along an altitudinal gradient where there has been a replacement of some neighbourhood above-ground rubbish containers by underground containers. We selected five coastal and three inland towns of several sizes (Table 1). They are representative localities in the study area, where 12% of towns are medium-sized (15,000-60,000 inhabitants) and 86% are small towns (<15,000 inhabitants) (Table 1).

Both neighbourhood rubbish containers types, above-ground and underground (Fig. 2), are usually arranged in groups of several of the same type. The geographical location of rubbish container groups in each urban area was obtained from technical report maps owned by the Town Councils of the studied localities. We randomly selected a representative variable number of groups of underground rubbish containers at the eight localities, according to town size and containers availability (n=80 in all; Table 1). When two underground groups of containers fell within a 75m radius, only one was selected for the study to avoid double counting. A similar number of surface rubbish containers per locality was also randomly selected (n=85 in all; Table 1) by following the same protocol to avoid double counting. Some groups of underground rubbish containers had an additional surface organic rubbish container, in which case this particular group of underground rubbish containers was discarded. Number of organic rubbish containers in each group was counted. At each selected group of containers we conducted 5-minute 25m fixed-radius point counts, centred at the rubbish container, and counted every House Sparrow seen and heard (Gibbons and Gregory 2006). Counting was done in December 2018, January 2019 and February 2019 (winter of 2018-2019) given the special relevance of human-related sources of food in winter (Bernat-Ponce et al. 2018; Bokotey and Gorban 2005). Each group of rubbish containers was sampled twice on two different days separated by a minimum of two-three weeks (First visit = early winter; Second visit = late winter). Counts were not done on windy and/or rainy days. Daily sampling sessions lasted approximately two hours, and started 15 minutes after the official sunrise time when this species is most active (Anderson 2006). On average, a sampling session comprised 14-18 point counts depending on the distance between consecutive sampling points.

For each group of containers we obtained 15 habitat variables potentially important for influencing House Sparrow presence and abundance (Table 2), including some of those proposed by Bernat-Ponce et al. (2018). These variables were classified into four sets that referred to urban landscape characteristics on different scales. The first set of variables defined the features of the rubbish containers: type of container (surface or underground) and number of containers intended to receive rubbish containing organic matter. The second set of variables described the features of the closest buildings at the sampling point: mean number of floors of the eight nearest buildings; the socio-economic level of the area where rubbish containers are located, classified into deprived area, average area, and well-off/high income area (Bernat-Ponce et al. 2018; Shaw et al. 2008). The third set described the urban features in the sampling point and its adjacent habitat (50m radius): presence/absence of private gardens; presence/absence of bar terraces; presence/absence of schools; presence/absence of parks; presence/absence of mature trees on streets; if the sampling point was in the centre of the urban area (defined as the oldest sector of the urban area, where commercial and business activities tend to concentrate) or not. We used a 50 m radius because this represents the average home range of urban House Sparrows (0.86 ha; Vangestel et al. 2010). The last set included variables associated with the locality's general features (town/city scale): geographical location (coastal or inland); socio-economic level of the sampled locality defined by the average budget per inhabitant (ARGOS 2019); number of inhabitants (Instituto Nacional de Estadística 2019); ratio of surface rubbish containers per inhabitant in the locality; ratio of underground rubbish containers per inhabitant in the locality.

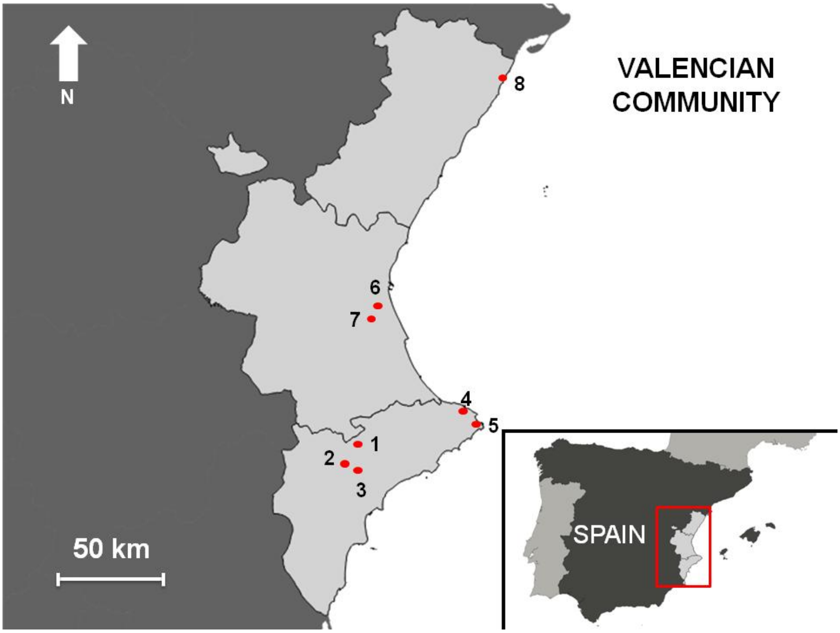

Fig. 1 Map showing the eight selected urban areas to study the rubbish containers in east Spain: 1) Alcoy; 2) Onil; 3) Castalla; 4) Denia; 5) Jávea; 6) Alboraya; 7) Burjasot; 8) Vinaroz

Fig. 2 Types of rubbish containers in this study. a) Neighbourhood surface rubbish containers. b) Neighbourhood underground rubbish containers

Table 1 Geographical, demographic, urban and climatic characteristics of the selected urban areas and information about the groups of rubbish containers studied in them. The number of studied groups of rubbish containers of each type is shown to the left of the slash; the available number is shown to the right of the slash. The number of total containers of each locality is shown in parentheses (when the numbers of groups is not reported, it is shown with -). Source: the authors with data from AVAMET (2019), Instituto Nacional de Estadística (2019), SIGPAC (2019) and technical reports from the localities.

Locality	**Coordinates**	**Number of inhabitants (2018)**	**Urban area (ha)**	**Temperature °C (2018)**			**Rainfall mm (2018)**	**Altitude m.a.s.l.**	**Underground groups**	**Surface groups**
				Max.	**Mean**	**Min.**				
Alboraya	39°30´04´´ N 0°20´58´´ W	24,222	67.02	35.8	18.3	3.2	642.6	5	8/ 14 (28)	8/ - (212)
Alcoy	38°41´54´´ N 0°28´25´´ W	58,977	349.03	40.4	16.1	-4.4	439.2	562	8/ 12 (25)	10/141 (294)
Burjasot	39°30´33´´ N 0°24´39´´ W	37,584	194.74	38.5[a]	17.6[a]	2.8	658.6	59	14/ - (50)	14/ - (242)
Castalla	38°35´48´´ N 0°40´15´´ W	9880	85.74	-	-	-1.7	418.8	675	6/ 52(52)	6/27 (27)
Denia	38°50´25´´ N 0°06´31´´ E	41,733	99.80	37.8	19.5	5.3	625.8	12	17/ 60 (112)	17/65 (520)
Jávea	38°47´21´´ N 0°09´47´´ E	27,224	171.1	36.9	19.1	4.4	675.6	12	13/ - (30)	16/ - (325)
Onil	38°37´46´´ N 0°40´26´´ W	7475	77.76	34.8	14.5	-4.2	360.6	715	4/ 5 (11)	4/- (98)
Vinaroz	40°28´00´´ N 0°28´00´´ E	28,438	97.50	35.1	18.2	1.6	785.9	7	10/ 12 (25)	10/ - (260)

[a] Corresponds to the adjacent town (Moncada) due to the incomplete series in Burjasot.

Table 2 Environmental variables tested for their effect on House Sparrow abundance around groups of rubbish containers in urban areas of east Spain in winter.

Set and sampling scale	Variable	Abbreviation	Type	Sampled/ Obtained	Description
1. Container features					
(sampling point)	Rubbish container	CONT	Categorical	In situ	Underground /Surface
	Number of containers	NCONT	Numerical	In situ	Number of containers at the group
2. Building features (sampling point)	Mean building height	BUILD	Numerical	In situ	Mean number of floors (eight nearest buildings)
	Socio-economic level point	SOCLEVP	Categorical	In situ	Deprived/Average/Well-off
3. Urban features (50m radius)	Private garden	PG	Categorical	SIGPAC (2019)	Absence/Presence
	Terrace	TERR	Categorical	SIGPAC (2019)	Absence/Presence
	School	SCH	Categorical	SIGPAC (2019)	Absence/Presence
	Park	PARK	Categorical	SIGPAC (2019)	Absence/Presence
	Tree	TREE	Categorical	In situ	Absence/Presence
	Centre	CENT	Categorical	SIGPAC (2019)	No/Yes
4. Locality features (town scale)	Altitude	ALT	Categorical		Coastal/Inland
	No. inhabitants	INHAB	Numerical (scaled)	INE[a](2019)	
	Socio-economic level locality	SOCLEVL	Numerical (scaled)	ARGOS (2019)	Average budget per inhabitant
	Ratio of underground containers	RATUND	Numerical (scaled)	SIGPAC (2019)	Number of underground containers/inhabitant
	Ratio of surface containers	RATSURF	Numerical (scaled)	SIGPAC (2019)	Number of surface containers/inhabitant

[a] Instituto Nacional de Estadística 2019

Statistical analysis

Statistical analyses were carried out in RStudio 3.6.1. We used Generalized Linear Mixed Models (GLMMs) to identify the variables in each of the four aforementioned sets that were related most to the House Sparrow number counted at each point (dependent variable). The R package "glmmTMB" (Brooks et al. 2017) was used to fit the GLMMs. The container group identity code was included as a random factor because repeated measures were done at each group of containers (O'Hara 2009). The environmental variables were treated as fixed effects. Number of inhabitants, socio-economic level of the locality and both the ratios of underground/surface rubbish containers per inhabitant in the locality that formed part of the locality's features group were scaled (centred and divided by standard deviations) using the "scale" function in R. Moreover, in the models we included three variables as fixed effects to consider the space or time effect. The spatial autocorrelation effect was controlled by including a spatial term (SPAT) with the coordinates of the form '$x + y + x^2 + xy + y^2 + x^3 + x^2y + xy^2 + y^3$' in all analyses (Legendre and Legendre 1998; López-Pomares et al. 2015). The name of the locality (LOC) was included to control for the locality effect on each group, except for the group describing the locality's general features. The visit (VISIT, first or second) was also included to take temporal variability into account and to check if there were any differences between them (early winter/late winter).

To avoid including a high number of variables and interactions in the analyses the following statistical proceeding was followed for each set of variables. First of all, we fitted a complete GLMM with all the environmental variables in the set, the spatial-time variables and only the interactions that we considered to be biologically meaningful. We started fitting the complete model with Poisson distribution. We tested this model for overdispersion with the "check_overdispersion" function of the "performance" package (Lüdecke et al. 2019). We also fitted two variants of the same model using Negative Binomial distribution type 1 (family nbinom1) and type 2 (family nbinom2) (Blasco-Moreno et al. 2019). Then we selected the model variant with the family that yielded the lowest Second-Order Akaike Information Criterion (AICc) calculated with the appropriate function of the "MuMIn" package (Bartoń 2019).

The multicollinearity of the variables included in each model was checked with the "check_collinearity" function of the "performance" package (Lüdecke et al. 2019). The significance of the fixed effects was tested by the "Anova" function of the "car" package (Fox and Weisberg 2019). We considered multicollinearity to be high when VIF was > 5 (Zuur et al. 2010). In these cases we first attempted to reduce multicollinearity by deleting the interaction between variables with VIF > 5. If several interactions presented similar high VIF values, that with the lowest significance in GLMMs was eliminated. When no multicollinear interactions remained, the same procedure was repeated for the multicollinear main effects. Then the GLMMs in each group of variables were simplified by deleting the least significant interactions in turn and by checking if this deletion was linked to a reduction in the AICc. If this proceeding did not reduce the AICc, the least significant main effect was deleted instead, provided it was not included in the remaining interactions, and AICc reduction was checked. The final model was that with the lowest possible AICc after deleting the non-significant variables/interactions.

The variables included in the final model and the equally plausible models (ΔAICc < 2) of each set were selected for checking in a final combined analysis with the variables selected from all the four sets, along with their biologically interesting interactions and the spatial-time variables (SPAT, LOC, VISIT). This combined GLMM was simplified by following the aforementioned proceeding to obtain a final set of equally plausible models (ΔAICc < 2). Given the large number of variables in the combined analysis, the process of eliminating multicollinear terms was done in two steps to avoid convergence problems in the models. First of all, the variables selected from the first three sets were checked together to eliminate multicollinear terms. Then the remaining terms were joined to the selected variables of the fourth set (locality features) to detect and eliminate additional collinear terms. For the best models, we calculated the conditional intraclass correlation (cICC) to estimate the proportion of variance in abundance that was accounted for by the random effect using the “icc” function of the package “sjstats” (Lüdecke 2019) and the conditional R^2 to obtain the variance explained by the entire model, including both fixed and random effects with the “r.squaredGLMM” function of the “MuMIn” package (Bartoń 2019).

Comparison of the socioeconomic level and building and urban features around groups of containers located in the centre or the outskirts of urban areas can be found in Appendix A (Online Resource). We tested if these variables differed between urban sectors (centre or outskirts) and container type (underground or surface) using log-linear models for categorical variables (SOCLEVP, PG, TERR, SCHO, PARK, TREE) and ANOVA test for the continuous variable (height of buildings, BUILD). These analyses were carried out with the "lm" and "aov" functions from the "stats" package (R Core Team 2020), and the "anova" function from the "car" package (Fox and Weisberg 2019). Mean number of containers per group were compared between container types using the function "wilcox.test" of the package "stats" (R Core Team 2020).

Results

Mean number of containers in underground groups of rubbish containers was significantly higher than in surface groups (Underground: 1.93 ± 0.07 SE; Surface: 1.59 ± 0.09 SE; Wilcoxon W = 2290.5; $p < 0.001$). House sparrows were frequently found around rubbish containers as they were detected in 90-97% of surface containers and 65-81% of underground containers. Even though no overdispersion was detected in the Poisson family models, GLMMs were fitted using Negative Binomial distribution type 1 (nbinom1 family) because they had lower AICc values (Appendix B-Online Resource). The simplification process of GLMMs for each set of variables and the combined model are found in Appendix C (Online Resource). The final model for the variables related to container features included the container type, spatial term and visit, which were all significant. The final model for the building feature variables included the mean number of floors, the visit and the spatial term, all significant. The variables associated with urban features included in the final model were: private gardens; mature trees; bar terraces; location of the point count in the centre; the centre-visit interaction; the visit and the spatial term. In the group of variables related to locality features, the following variables were selected: socio-economic level; ratio of surface containers per inhabitant and its interaction with visit; altitude and its interaction with visit; the spatial term and visit.

The combined GLMMs showed that three final models were equally plausible (ΔAICc < 2) (Table 3). The model with the lowest AICc (Model 1; AICc = 1,750.6) was built with the variables container type (CONT), private garden (PG), mature trees (TREE), bar terraces (TERR), centre (CENT), spatial term (SPAT) and visit (VIS); with four interactions: centre (CENT) with visit (VIS); container type (CONT) with private garden (PG); centre (CENT) with container type (CONT); mature trees (TREE) with centre (CENT). Model coefficients were all significant ($p < 0.05$), except for the variable TREE ($p = 0.06$), the variable CENT ($p = 0.168$) and the interaction CENT with CONT ($p = 0.104$) (Table 3). There were two other equally plausible models. The first was like Model 1 but without the CENT with CONT interaction (Model 2; AICc = 1,751.0); the second resulted from adding the variable BUILD to Model 1 (Model 3; AICc = 1,751.9) (Table 3).

All the final models showed that the rubbish container type is related to the abundance of the House Sparrows around them. This species was more abundant in those areas with surface rubbish containers than in the areas where rubbish containers were located underground. Presence of bar terraces was also related to the abundance of House Sparrows, regardless of the container type in their vicinity. On the contrary, presence of private gardens was positively related to House Sparrow abundance only around underground rubbish containers (Fig. 3a). These models also showed that mature trees on streets were negatively correlated to the abundance of the House Sparrows around containers. However, the interaction between the presence of mature trees and location at the centre of the urban area showed that the absence of trees contributed to lower House Sparrow numbers in outskirt areas while had not effect in the urban centre (Fig. 3b). Furthermore, spatial term and visit were significant. The significant interaction between visit and centre revealed that House Sparrow abundance index significantly increased during the second visit only around containers in central areas of towns (Fig. 3c). The cICC of the three final models ranged between 0.180 and 0.191, thus around 20% of variance in the House Sparrow abundance index was due to the rubbish containers group’s identity. The variance explained by the three models was similar and around 90%.

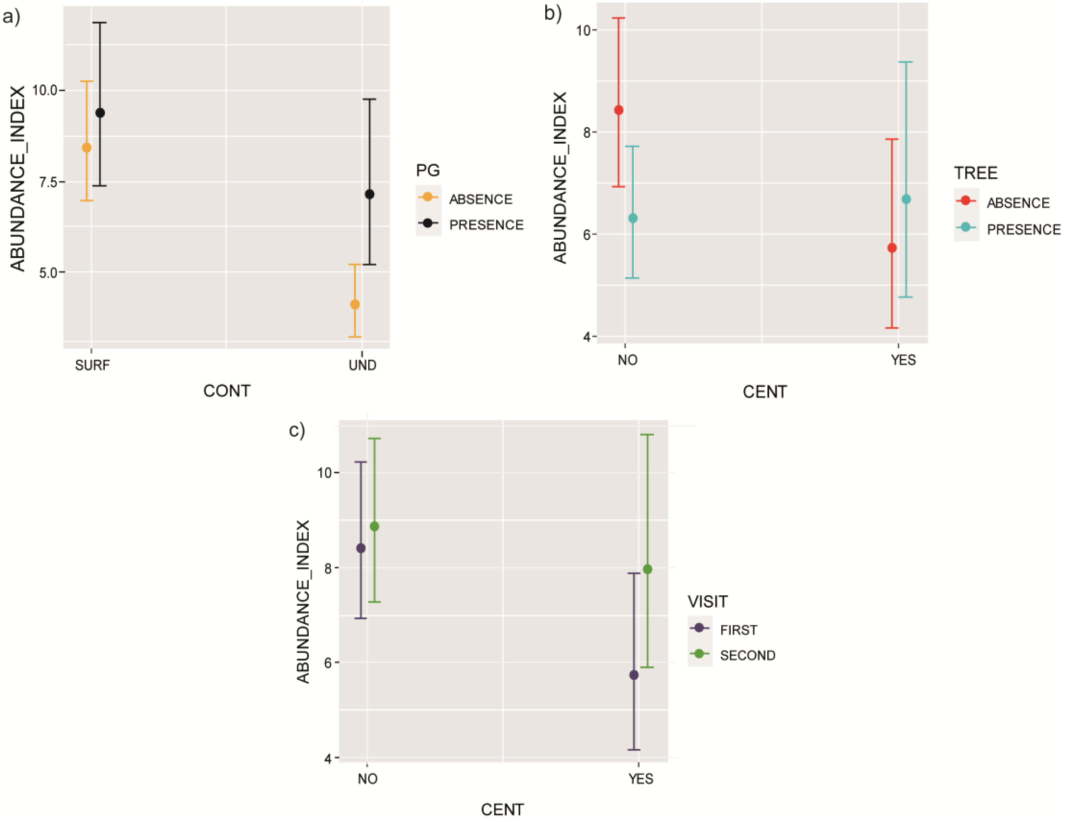

Fig. 3 Graphs of the significant interactions detected in GLMMs (Table 3). Predictions of the abundance indices of House Sparrows were calculated using Model 1. a) Interaction between rubbish container type and presence/absence of private gardens. b) Interaction between location at the centre of the urban area and presence of mature trees in streets. c) Interaction between location at the centre of the urban area and visit. CONT: Rubbish container; SURF: Surface rubbish containers; UND: underground rubbish containers; PG: Private gardens; CENT: Centre of urban area; TREE: Presence of mature trees

Table 3 Best final combined GLMM models (nbinom1 family) built with the variables selected from the four sets. Models are ordered by AICc value (Second-order Akaike Information Criterion). cICC: conditional intraclass correlation coefficient. R^2 interpreted as a % of variance explained by the entire model, including both fixed and random effects. Significant p values are marked with (*) = p < 0.05; (**) = p < 0.01; (***) = p < 0.001, and marginally significant p values with (˙) = p < 0.1. Brackets identify the level of the factor at which the model coefficients were calculated.

Model	**AICc**	**R^2**	**Set**	**Variables**	**Estimate**	***p***
1	1750.6	*91.0*		Intercept	0.834	
				Fixed		
			1	CONT	-0.725 (UNDERGROUND)	<0.001***
			3	CENT	-0.385 (YES)	0.168
				PG	0.106 (PRESENCE)	0.002**
				TERR	0.238 (PRESENCE)	0.017*
				TREE	-0.288 (PRESENCE)	0.060˙
			Control	SPAT	0.182	<0.001***
				VIS	0.050 (SECOND)	0.042*
			Interactions	CENT*CONT	-0.332 (CENT YES * CONT UNDERGROUND)	0.104
				CENT*VIS	0.282 (CENT YES * SECOND)	0.030*
				PG * CONT	0.451 (PG PRESENCE * CONT UNDERGROUND)	0.015*
				TREE*CENT	0.442 (TREE PRESENCE * CENT YES)	0.030*
				Random		
				CONT-ID	Variance (0.148) cICC (0.182)	
2	1751.0	*90.9*		Intercept	0.874	
				Fixed		
			1	CONT	-0.823 (UNDERGROUND)	<0.001***
			3	CENT	-0.522 (YES)	0.165
				PG	0.071 (PRESENCE)	0.005**
				TERR	0.260 (PRESENCE)	0.009*
				TREE	-0.279 (PRESENCE)	0.062˙
			Control	SPAT	0.181	<0.001***
				VIS	0.050 (SECOND)	0.044*
			Interactions	CENT*SUR	0.278 (CENT YES * SECOND)	0.033*
				PG * CONT	0.483 (PG PRESENCE * CONT UNDERGROUND)	0.010**
				TREE*CENT	0.399 (TREE PRESENCE * CENT YES)	0.050*
				Random		
				CONT-ID	Variance (0.154) cICC (0.191)	
3	1751.9	*91.0*		Intercept	0.978	
				Fixed		
			1	CONT	-0.702 (UNDERGROUND)	<0.001***
			2	BUILD	-0.035	0.334
			3	CENT	-0.420 (YES)	0.121
				PG	0.087 (PRESENCE)	0.005**
				TERR	0.242 (PRESENCE)	0.015*
				TREE	-0.282 (PRESENCE)	0.073˙
			Control	SPAT	0.181	<0.001***
				VIS	0.050 (SECOND)	0.042*
			Interactions	CENT*CONT	-0.331(CENT YES * CONT UNDERGROUND)	0.105
				CENT*VIS	0.282 (CENT YES * SECOND)	0.030*
				PG * CONT	0.444 (PG PRESENCE * CONT UNDERGROUND)	0.017*
				TREE*CENT	0.457 (TREE PRESENCE * CENT YES)	0.025*
				Random		
				CONT-ID	Variance (0.146) cICC (0.180)	

Discussion

Our results showed that House Sparrows were less abundant around underground rubbish containers than around surface containers. However, the House Sparrow abundance around rubbish containers was also influenced by other urban habitat characteristics. On one hand, the presence of bar terraces around rubbish containers was in general associated with higher House Sparrow abundance while presence of private gardens was related to higher House Sparrow abundance only around underground containers. On the other hand presence of mature trees around containers located at outskirts was negatively related to House Sparrow abundance.

It is assumed that the main reason for House Sparrows association to rubbish containers is that birds easily find scraps of anthropogenic food, usually of high calorific value, in their surroundings and in fact sparrows are frequently seen pecking on the ground around these places (pers. obs.). This situation seems to be especially important in winter, when natural food in urban areas can be scarcer or harder to obtain as Bokotey and Gorban (2005), and Bernat-Ponce et al. (2018) found in Ukraine and Spain, respectively. A limitation of the present study is that we assumed that trophic resources were more abundant around surface than around underground containers but we did not actually quantify their abundance. The design and functioning of each container type are so distinct that we expected some difference in abundance of food scraps around them, and this should be an intended consequence of the design of underground containers. Therefore, it seems reasonable that the reduced abundance of House Sparrows recorded around underground rubbish containers reflects the increase in cleanliness associated to underground containers. Replacing surface rubbish containers by underground ones is a growing urban trend in European cities to improve sanitation measures (Eroski Consumer 2008; Interesting Engineering 2017; The Hague 2017). This trend could lead to a potential limitation of the number of sparrows that these areas can maintain. However, future studies should check the assumption of different abundance of food scraps around container types or other urban features as well as their temporal variation along the day, which could help to explain some of the patterns of urban habitats use by House Sparrows.

Bar terraces around rubbish containers were associated with higher House Sparrow abundance because they are an important supply of anthropogenic debris (Haemig et al. 2015). No matter what type of rubbish container group was studied, House Sparrows were always more abundant when bar terraces were present. However, this link between House Sparrows and bar terraces was not found in east Spain by Bernat-Ponce et al. (2018). This difference could be due to differences in the sampling design between the studies because sampling point counts of the present research were located, exclusively, in the vicinity of rubbish containers, while the other study considered the entire urban matrix. The presence of bar terraces and containers could have a synergic effect on House Sparrows' abundance, since both provide food resources in complementary times, and therefore the effect of bar terraces could be easier to detect in the proximity of rubbish containers. Bernat-Ponce et al. (2018) found that urban parks were positive for the abundance of House Sparrows while in the present study we did not find this effect around rubbish containers of any type. In the aforementioned study, most point counts in parks were located inside them and not on their surrounding streets, where rubbish containers were located. Parks might offer alternative abundant natural food resources that could reduce the link between House Sparrows and urban rubbish containers located on their edge, or even could reduce their detectability due to the presence of abundant vegetation. Conversely, private gardens had an effect only on sparrow abundance around underground containers, where they partially mitigate reduced House Sparrow abundance compared to surface containers. Thus the effect of private gardens seemed weaker than that of bar terraces. This is likely explained by the variability in the quality of these gardens as a habitat for sparrows. Even though some Spanish gardens could provide shelter and food from native vegetation, they are usually very small, mostly planted with exotic species that do not produce berries, and food provision for birds by owners (e.g. bird feeders) is not as frequent as in other European countries. Consequently, we may expect food availability in private gardens to be lower and more unpredictable than on bar terraces. However, Chamberlain et al. (2007), Murgui (2009), and Shaw et al. (2008) found that these gardens were a key factor for the House Sparrow abundance in urban environments of the UK and Spain. Therefore, it would appear that the importance of some urban landscape components for House Sparrows may differ between localities, depending on their specific characteristics, as Murgui (2009) suggested to explain the discrepancies among several studies.

Presence of trees in the vicinity of rubbish container groups studied in the centre of urban areas had no effect on the abundance of House Sparrows, but trees had a negative effect on sparrow abundance around containers located at outskirts. This was an unexpected result as we thought that trees could be used as a shelter and a food supply, and would always have a positive effect. Several explanations are possible for this effect of trees. As the presence of mature trees around rubbish containers was similar in the centre and outskirts (Appendix A-Online Resource) this result might be due to the different features of wooded areas. When present in outskirts, trees tend to be located on avenues which cover larger areas, where House Sparrows would be less linked with rubbish containers and would, therefore, group less around them than in the centre. In addition, House Sparrows also tend to be more abundant in outskirts (Fig. 3c this paper; Murgui 2009), thus when trees are absent, the House Sparrow grouping around rubbish containers would be more evident than in the town centre.

Visit had no effect on the House Sparrow abundance around the containers located in outskirts, while more birds were counted during the second visit (late winter) in the centre. Variation in the urban landscape between centre and outskirts could help to explain this interaction. Schools, bar terraces, and private gardens, which are associated with more abundance of food (Gaston et al. 2005; Haemig et al. 2015; Spelt et al.

2021), tend to be more frequent in outskirts (see Appendix A, Online Resource) and therefore House Sparrows should concentrate more around rubbish containers in town centres during the harshest winter period than in outskirts. Finally, no variable related to the locality's general characteristics was selected in the final models. As we have seen, all the significant variables explaining the abundance of House Sparrows around rubbish containers were related to the presence of some particular urban features around them. This suggests that the pattern found herein was the same in all the studied towns and supports the view that the container type effect we detected in this work is likely generalizable to other cities in different geographical locations.

Evidence reveals that replacing surface rubbish containers with underground rubbish containers is associated with a reduction of House Sparrows abundance around them. As urban cleanliness is a social demand and is correlated with positive outcomes (high hygienic standards, avoiding odours, etc.) (ISWA 2013), we do not advocate removing underground containers and return to surface waste collecting systems. However, we believe that it is important to be aware that an increase in underground rubbish containers would add another negative impact on House Sparrows in modern cities and, thus, compensatory measures that increase food supply in other urban landscape component should be taken to mitigate its impact. First, design and management of green areas should promote weeds, bushes and native plants that harbour important food resources for House sparrows such as invertebrates and seeds (Bernat-Ponce et al. 2018; Narango et al. 2018). The substitution of grass and soil with impervious substrates such as artificial grass and concrete has been shown to affect negatively to House sparrows (Bernat-Ponce et al. 2020; Verbeeck et al. 2011), likely due to the reduction of food resources and thus should be avoided. Second, it is important to promote new urban green areas that work as green stepstones, including parks and private gardens, especially in areas with underground rubbish containers (De Laet and Trappeniers 2019; Shaw et al. 2011). More research is urgently needed to precisely identify the short-, mid- and long-term effects of urban diet on urban wildlife and to boost alternative trophic resources through green urban planning.

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Availability of data and material Data available on request from the authors

Code availability Code (R software) available on request from the authors
Authors' contributions E.B-P. and J.A.G-D. conceived of the study idea. E.B-P., J.A.G-D. and G.M.L-I. designed the fieldwork. All authors collected field data. E.B-P., J.A.G-D. and G.M.L-I. contributed to the analysis of the results and to the writing of the manuscript. All authors read and approved the final manuscript.

Online Appendices
Appendix A

Table 1 of electronic supplementary material Comparison of the urban habitat around rubbish containers of each type (underground, surface or both pooled) located in the centre or the outskirts of towns. Variables related to closest buildings, to the containers and urban features around them are shown. Mean building height is the mean number of floors of buildings (±SD). The socio-economic categories show the percentage of the point counts included in each category. Urban features variables are the percentage of point counts with the presence of each urban feature. The significant terms of the ANOVA test and Log-linear models are shown in the second column. CONT: effect of the container type. CENT: effect of the container being in the centre or outskirts. The final row denotes the total number of studied groups of rubbish containers in each category

		Centre			Outskirts		
Urban variable	**Significant terms**	**Underground**	**Surface**	**Both**	**Underground**	**Surface**	**Both**
Building features							
Mean building height (BUILD)	CONT CENT	4.04 ± 1.22	3.15 ± 0.86	3.70 ± 1.17	4.60 ± 1.24	3.90 ± 1.27	4.20 ± 1.30
Socioeconomic level point (SOCLEVP)	SOCLEVP*CONT SOCLEVP*CENT						
Deprived		25.0	72.2	43.5	9.6	16.4	13.4
Average		64.3	27.8	50.0	80.8	64.2	71.4
Well-off		10.7	0	6.5	9.6	19.4	15.1
Urban features							
Private garden (PG)	PG*CONT*CENT	21.4	5.6	15.2	23.1	37.3	31.1
Terrace (TERR)	TERR*CONT	28.6	22.2	26.1	53.3	22.4	36.1
School (SCHO)	SCHO*CENT	5.6	0	2.2	9.6	13.4	11.8
Park		21.4	11.1	17.4	26.9	19.4	22.7
Tree		64.3	44.4	56.5	75.0	61.2	67.6
Number of container groups		28	18	46	52	67	119

Appendix B

Table 2 of electronic supplementary material GLMMs family selection for each set of variables used to study the effects of the rubbish container type on House Sparrow abundance in urban areas of E. Spain in winter 2018/2019. The best model (lowest AICc) is marked by (*). ∆AICc is the difference between the selected model and the model of that set with the lowest AICc value.

Set	Family	AICc	∆AICc
Rubbish containers features	Poisson	1799.02	7.92
	Negative Binomial 1 *	1791.10	0
	Negative Binomial 2	1800.36	9.26
Building features			
	Poisson	1844.31	7.26
	Negative Binomial 1 *	1837.05	0
	Negative Binomial 2	1845.97	8.92
Urban features			
	Poisson	1850.50	9.41
	Negative Binomial 1 *	1841.09	0
	Negative Binomial 2	1852.88	11.79
Locality features			
	Poisson	1834.20	9.19
	Negative Binomial 1 *	1825.01	0
	Negative Binomial 2	1835.58	10.57
Combined set	Poisson	1786.14	11.74
	Negative Binomial 1 *	1774.40	0
	Negative Binomial 2	1786.14	11.74

Appendix C

GLMMs variable selection process for each set of variables and the final set of selected variables. Multicollinear variables or interactions (VIF >5) were excluded from the process and are identified at the end of each table. The variables or interactions included in the models are identified by **X** if they are significant or by x if they are not. Variable codes are found in Table 2 in the main text.

Table 3 of electronic supplementary material Steps in the selection of variables related to rubbish containers.

	Step		
Variable	**1**	**2**	**3**
CONT	**X**	**X**	**X**
NCONT	x	x	
SPAT	**X**	**X**	**X**
VIS	**X**	**X**	**X**
CONT*VIS	x		
AICc	1770.62	1768.53	1766.49
Variables or interactions excluded because of collinearity			
LOC			
CONT*NCONT			
NCONT*VIS			
VIS*LOC			

Table 4 of electronic supplementary material Steps in the selection of variables related to building features.

	Step	
Variable	**1**	**2**
BUILD	**X**	**X**
SOCLEV	x	
SPAT	**X**	**X**
VIS	**X**	**X**
AICc	1813.6	1809.6
Variables or interactions excluded because of collinearity		
LOC		
VIS*BUILD		
VIS*LOC		
VIS*SOCLEV		

Table 5 of electronic supplementary material Steps in the selection of variables related to urban features.

	Step													
Variable	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**
CENT	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**
PARK	x	x	x	x	x	x	x	x	x	x				
PG	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**
SCH	x	x	x	x	x	x	x	x	x	x	x			
SPAT	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**
TERR	x	x	x	x	x	x	x	x	x	x	x	x		
TREE	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**
VIS	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**
CENT*PG	x	x	x											
CENT*TERR	x	x	x	x										
CENT*SCH	x	x	x	x	x	x								
CENT*PARK	x	x	x	x	x									
CENT*TREE	x	x	x	x	x	x	x	x	x	x	x	x	x	
VIS*CENT	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**
VIS *PG	x	x	x	x	x	x	x	x	x					
VIS*TERR	x	x												
VIS*SCH	x	x	x	x	x	x	x	x						
VIS*PARK	x	x	x	x	x	x	x							
VIS*TREE	x													
AICc	1825.0	1822.7	1820.4	1818.4	1816.5	1814.7	1812.9	1811.2	1809.5	1807.8	1805.8	1803.8	1802.4	1802.7

Variables or interactions excluded because of collinearity

LOC
CENT*LOC VIS*LOC

Table 6 of electronic supplementary material Steps in the selection of variables related to locality features.

							Step	
Variable	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**
ALT	x	x	x	x	x	x	x	x
INHABIT	x	x	x	x	x			
RATUND	x	x	x	x				
RATSURF	x	x	x	x	x	x		x
SOCLEVL	x	x	x	x	x	x		
SPAT	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**
VIS	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**
ALT* VISIT	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**
INHABIT* VISIT	x							
RATUND* VISIT	x	x						
RATSURF* VISIT	**X**	**X**	**X**	x	x	x	x	x
SOCLEVL*VISIT	x	x	x					
AICc	1823.5	1821.7	1819.9	1818.9	1816.8	1814.8	1814.0	1812.9

Variables or interactions excluded because of collinearity
INHAB*SOCLEVL

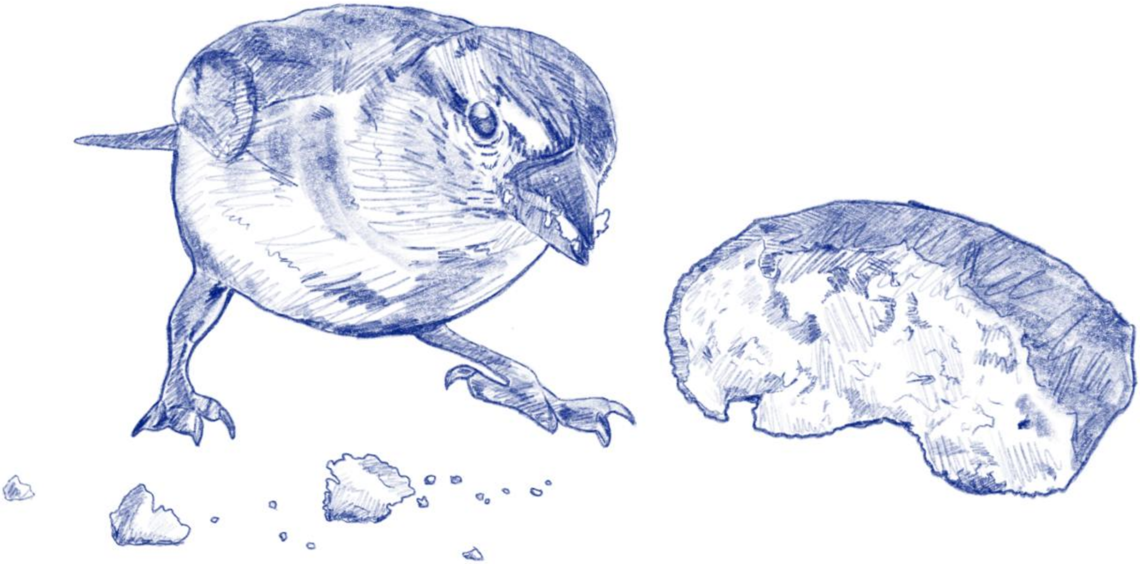
Table 7 of electronic supplementary material Steps in the selection of variables of the combined set of

Set	Variable	Step 1	2	3	4	5	6	7	8
Container features	CONT	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**
Building features	BUILD	x	x	x	x	x	x		
Urban features	PG	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**
	TREE	x	x	x	x	x	x	x	x
	CENT	x	x	x	x	x	x	x	x
	TERR	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**
Locality features	RATSUR	x	x	x	x	x			
	SOCECOLEVP	x	x	x	x				
Control variables	SPAT	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**
	VIS	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**
Interactions	CENT*CONT	x	x	x	x	x	x	x	
	CENT*VIS	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**
	PG*CONT	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**
	RATSUR*VIS	x							
	SOCECOLEVP*CONT	x	x	x					
	TERR*CONT	x	x						
	TREE*CENT	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**
	AICc	1760.3	1758.1	1756.1	1756.1	1753.8	1751.9	1750.6	1751.0
Variables or interactions excluded because of collinearity									
	ALT LOC ALT*CONT ALT*VIS BUILD*ALT BUILD*CENT BUILD*CONT							BUILD*PG BUILD*TERR BUILD*TREE BUILD*RATSURF BUILD*SOCLEVL CONT*RATSURF CONT*TREE	

Chapter 6

Experimental effect of anthropogenic food resources available in cities on the body condition, nutritional status and oxidative stress of an urban bioindicator passerine

Edgar Bernat-Ponce, José Antonio Gil-Delgado, José Vicente Guardiola & Germán Manuel López-Iborra



Under review in Science of the Total Environment

EXPERIMENTAL EFFECT OF ANTHROPOGENIC FOOD RESOURCES AVAILABLE IN CITIES ON THE BODY CONDITION, NUTRITIONAL STATUS AND OXIDATIVE STRESS OF AN URBAN BIOINDICATOR PASSERINE

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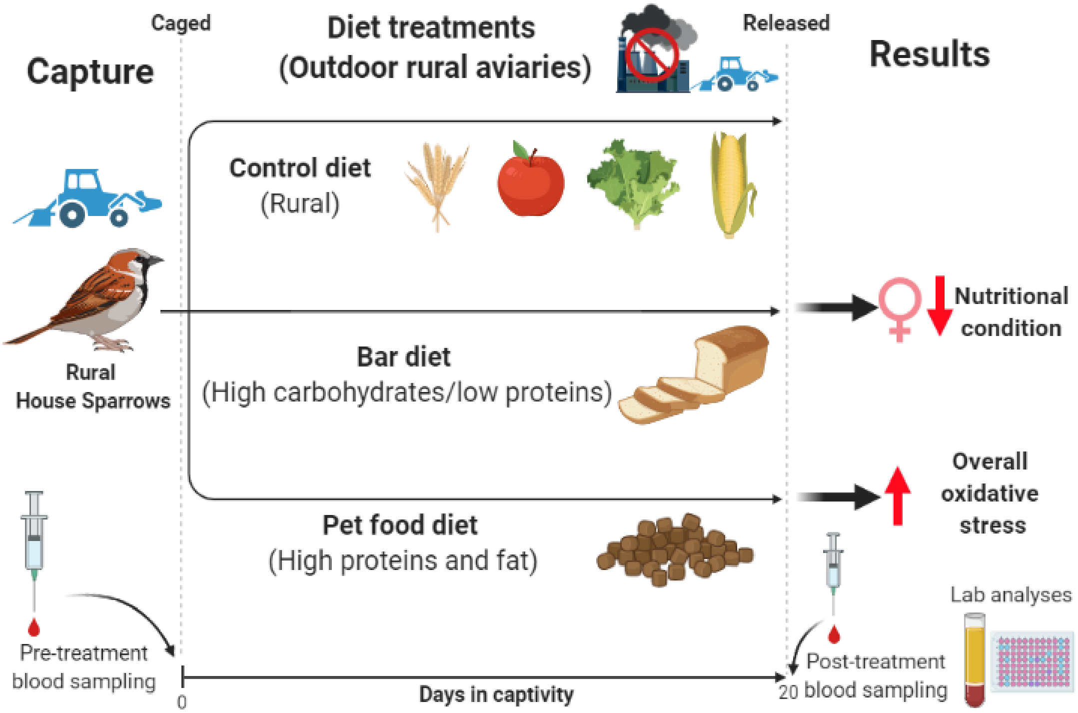
Abstract

Urban areas are constant predictable suppliers of anthropogenic processed food. The House Sparrow is an urban bioindicator species that has declined in its native area in the last few decades. Recent studies indicate the high oxidative stress level found in urban individuals, probably due to urban diet or pollutants, as a potential cause of decline. This study aimed to experimentally determine in captivity the effects on physical condition, plasma biochemical nutritional parameters and blood oxidative status of two abundant urban trophic resource types: snack-bar food leftovers and pet food. In a rural area in SE Spain, 95 House Sparrows were captured and kept in outdoor aviaries to exclude the potential effect of urban pollutants. Individuals were exposed for 20 days to one of three diet treatments: Control diet, similar to that available in rural areas (fruit, vegetables, grain mixture for poultry), Snack-bar diet (ultraprocessed snacks) or Cat food diet (dry pellets). Blood samples were collected before/after diet treatments. The relative change rates of 14 variables were obtained: physical condition (3); nutritional status (6); oxidant-antioxidant status (5). A PCA was run to find gradients of variables covariation. GLMMs were used to find the effect of diets on each selected PC. The snack-bar diet led to malnutrition, especially in females, and potential signs of anaemia (PC1). The Cat food diet (ultrarich in proteins) increased reactive oxygen species production, overall oxidative stress and albumin (PC2). The cat food diet led to higher uric acid production and haematocrit levels (PC6). Thus an unbalanced urban diet in an unpolluted environment alters body condition, nutritional physiology and oxidative stress in an urban bioindicator passerine. A synergic effect of low-quality diet and urban pollution can contribute to general House Sparrow decline in European cities.

Keywords

House Sparrow, pollution, physiology, reactive oxygen species, processed food, urban diet.

Graphical abstract



1. Introduction

Urbanisation involves fast landscape transformation (United Nations, 2018). This usually has negative effects on local biota by exposure to new stressors, such as new pollutants, food, disturbances, anthropogenic noise, predators, among others (Fernández-Juricic and Telleria, 2000; Woods et al. 2003; Francis and Barber, 2013; Herrera-Dueñas et al., 2017; Murgui and Hedblom, 2017; Cid et al., 2018, Townsend et al., 2019, Bernat-Ponce et al., 2021a). Consequently, cities are challenging environments for wildlife (Murgui and Hedblom, 2017; Isaksson, 2018), but some birds species (native and/or invaders), known as urban exploiters, have adapted to urban life and take advantage of cities (Eraud et al., 2007; Evans et al., 2011; Lepczyk and Warren, 2012; Oliveira et al., 2017). One of the main attractors of urban areas is constant human-induced food resources, provided directly by people as feeders, or indirectly by rubbish containers or bar terraces (Tortosa et al., 2002; Chace and Walsh, 2006; Reynolds et al., 2017; Bernat-Ponce et al. 2018, 2021b).

Cities are a constant supply of diverse human food. While rural birds feed mainly on crops, poultry food, wild seeds and invertebrates (Cramp and Perrins, 1994; Anderson, 2006), these trophic resources are less available in urban environments because green areas in cities are progressively disappearing (Peach et al., 2015; Bernat-Ponce et al., 2020). The most calorific and profitable resources in cities can be considered to be made up of the "junk food" that derives from scraps of human leftovers (Seress and Liker, 2015) found around rubbish containers or bar terraces

(Tortosa et al., 2002; Bokotey and Gorban, 2005; Bernat-Ponce et al., 2019, 2021b). Even though this kind of food may increase their overwinter survival and keep populations larger (Seress and Liker, 2015), the negative hidden effects on their body condition, behaviour, nutritional physiology, oxidative stress, breeding and general health can be significant and are still poorly known (Burt et al., 2021).

Urban food does not normally contain the required amounts of nutritional elements, such as amino acids and/or vitamins (antioxidants), for urban birds, rather excess fats, sugars, carbohydrates, or even proteins, which can alter some blood plasma biochemical parameters (Ishigame et al., 2006; Jones and Reynolds, 2008; Townsend et al., 2019). Alterations to uric acid, glucose, total protein and cholesterol levels in birds can lead to kidney disease, anorexia, starvation and heart disease, respectively (Macwhirter 1994; Rosskopf and Woerpel, 1996; Fudge, 2000; Harr, 2002). The nutritional characteristics of these anthropogenic scraps have switched at the same rhythm as human diet did, including food preservatives to adapt to a fast lifestyle (Simopoulos, 2002; Bonhommeau et al., 2013; Hartmann et al., 2013; Winson 2013). This industrially-processed food also has the potential to generate damaging substances and diseases for the humans and animals that feed on them, such as oxidative stress or cancer (Devaraj et al., 2008; Omoruyi and Pohjanvirta, 2014; Giraudeau et al., 2018).

A simple damage-centred biochemical definition of oxidative stress is the disturbance of the pro-oxidant–antioxidant balance in favour of the former, which causes potential damage (e.g. of lipids, proteins, DNA) by reactive oxygen species (ROS) (Sies, 1991; Costantini, 2019). Aerobic organisms have evolved several defence lines to avoid ROS toxicity, which vary from antioxidant enzymes to low-molecular-weight endogenous antioxidants (i.e. GSH) enhanced by dietary non-enzymatic antioxidants (i.e. carotenoids or vitamins) (Halliwell, 2007; Herrera-Dueñas et al., 2017). Birds obtain carotenoids in the wild from invertebrates, plants, seeds and/or fruit (Isaksson, 2009; Eeva et al., 2010). However, the availability of dietary antioxidants in a city can be low because green areas with these food resources are lacking. Living in a city, breathing polluted air and feeding on "junk food" (oxidant conditions) mean that unquenched free radicals may remain in the body and continue to cause oxidative reactions that, in turn, damage cellular components (Monaghan et al., 2009). If this oxidative situation continues over time, cells can stop working properly and no longer perform their functions, or apoptosis can even start that facilitates several diseases (Costantini and Verhulst, 2009; Isaksson, 2015).

The House Sparrow (*Passer domesticus*) is a clear example of an urban exploiter and an avian bioindicator (Anderson, 2006; Herrera-Dueñas et al. 2017). However, this species has sharply declined in European urban areas in the last few decades (De Laet and Summers-Smith, 2007; Shaw et al., 2008). Several hypotheses have been put forward, but the main reason for its decline in urban areas still seems unclear (Anderson, 2006). One of the most recent hypotheses is that pollution levels and shortage of antioxidants in available food in cities may unbalance House Sparrow's oxidative stress levels, which could affect their survival and breeding success (Herrera-Dueñas et al., 2017; Peach et al., 2018). The urban House Sparrow presents higher oxidative damage levels, greater antioxidant enzymes activity and a lower antioxidant capacity than those inhabiting rural areas (Herrera-Dueñas et al., 2017). Individuals living in highly urbanised areas present altered blood plasma biochemical parameters and a weaker immune system compared to their rural conspecifics (Pineda et al., 2013; Herrera-Dueñas et al., 2015). However, it is still poorly known if this oxidative stress situation is due mainly to city pollution or urban diet.

The aim of the present study was to determine in captivity the effects on physical condition, plasma biochemical nutritional parameters and blood oxidative status of two food types that House Sparrows find in abundance in urban ecosystems. The experiment was performed in a rural area with rural birds to eliminate the potential effect of pollutants and to, thus, isolate the effect of diet. Sparrows are frequently seen in cities around bars or restaurants and feed on leftover food (Haemig et al., 2015). Moreover, cat feeders are widespread in most cities, and are either particular citizens or come from the city's management of urban cat colonies (Rollinson et al., 2003; Galbraith et al. 2014; Gunther et al., 2016). Sparrows are regular visitors to cat feeders (pers. obs.) and feed on pet food as part of their urban diet (Bower, 1999; Lendvai and Chastel, 2008; Andrzejczak et al. 2021). These two urban food types are mostly opposite as far as their composition is concerned. However, birds in cities can eat both types. While a snack-bar restaurant diet is usually rich in refined carbohydrates and sugar, but poor in proteins, vitamins and aminoacids, cat food is rich in protein and usually supplemented with antioxidants, amino acids (i. e. taurine) and vitamins, with very different consequences for birds. We can expect differential effects of our experimental urban diets on individuals' physical condition and nutritional status (e.g. high-carbohydrate/low-protein diet that reduces body mass/high-protein diet and increases body mass; Peig and Green, 2009; Coogan et al. 2018). We can also expect each diet to alter different nutritional parameters (e.g. high-carbohydrate diet increases cholesterol, high-protein diet increases albumin; Herrera-Dueñas et al., 2014). If urban diets are the main driver or a codriver of the higher oxidative stress levels observed in urban individuals, we would find a higher oxidative stress status in them after diet treatments due to marked unbalances, excess or lack of basic nutritional elements. However, if air pollution is the main or the only potential driver, we would not expect to find any significant changes in blood oxidative biomarkers levels after running the two experimental urban diets compared to a control diet.

2. Material and methods

2.1 Study area, capture site and season of the year

We captured (mist-netting) House Sparrows in three close rural areas that represent the potential heterogeneity of habitats in the rural birds of the same study area. The procedure ensured a sufficient number of individuals for each experimental round. The three rural capture sites were located north of the Alicante province (Spain) (Figure 1): 1) Alcoy (38° 41' 46'' N; 0°

27’ 26’’ W); 2) Muro de Alcoy (38° 46’ 23’’ N; 0° 24’ 60’’ W); 3) La Sarga (38° 38’ 29’’ N; 0° 27’ 57’’ W). Birds were captured in the vicinity of henhouses. These three areas are characterised by good air quality (European Environment Agency, 2018; RVVCCA 2020) and availability of unprocessed food (crops and poultry/cattle food) (pers. obs.). Birds were captured during six sampling rounds between October 2018 and February 2019 (autumn/winter 2018-2019), spaced by at least 22 days.

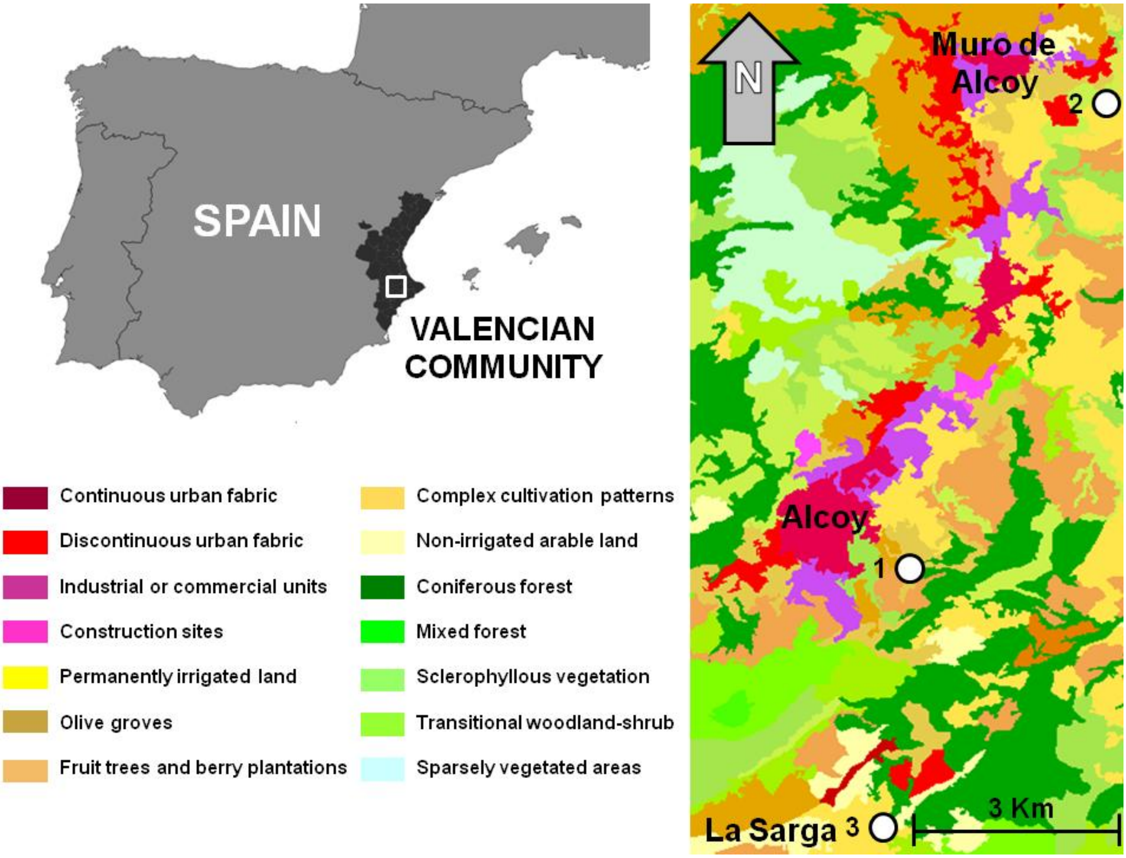


Figure 1. Location of the study area in the Valencian Community (SE Spain), the three capture sites and their surrounding habitats: site 1 in Alcoy; site 2 in Muro de Alcoy; site 3 in La Sarga. Land uses were collected from the Copernicus CLC 2018 (Copernicus Programme, 2020) and the general map was created with QGIS (2018).

2.2 Experimental design

The diet experiment was carried out in two symmetric outdoor aviaries (1.75 w X 2.5 h X 2.3 d = 10.06 m^2 each one), and was approved and supervised by the Generalitat Valenciana by keeping a similar space/individual ratio, as recommended by Innis et al. (1985) and Møller (1988). These aviaries were located in a private country house in La Sarga (Jijona, Spain) (38° 38’ 29’’ N; 0° 27’ 57’’ W), which is the aforementioned capture site 3. This small village is isolated from urban areas (the closest town is Alcoy, which is 5 km away) and free of the urban pollutants that derive from heavy traffic or close industries (European Environment Agency, 2018; RVVCCA, 2020). Each aviary was equipped with a rectangular platform (0.75 x 1.75 m) made of isolation materials and covered by tiles. They were located 1 m above the ground to be used as refuges and shelters. Aviaries also included tree branches, two feeders (one protected from rain), one drinker, one small bath, sand, nest boxes, as well as metal sheets on the roof to protect from direct sun and bad weather (heavy rain and strong winds).

We tested the effect of two experimental diets by simulating the food types frequently eaten by sparrows in cities. The Snack-bar diet (hereafter Bar) was composed of the processed food found in leftovers on snack-bar terraces (pers. obs.), which are usually poor in protein supplies, but high in carbohydrates and sugars: bread scraps, cheese doodles and croissants (Hacendado® Refs: 8 480000 832023; 8 480000 335739; 8 480000 820853) (Table 1). The second urban diet, dry cat food (hereafter the Cat food diet), was composed of crushed ox and chicken pellets

(Tandy® Ref: 0014125). The Cat food diet was selected because House Sparrows are regular visitors to the cat feeders at the University of Alicante (Spain) and several cities in the Alicante province (pers. obs.), and pet food forms part of their urban diet (Bower, 1999; Andrzejczak et al., 2021). This food represents an urban diet that is very rich in protein and fat, but is poorer in sugars and carbohydrates (Table 1). Our control diet simulated the food components available in the rural landscapes in our study area. This diet comprised a mixture of cereals for poultry provided by Cereales Pastor® (M. Gallinas, http://www.cerealespastor.com/mixturas/) as rural sparrows frequently feed around henhouses or stables. We added slices of tomato, lettuce and apple from the orchards located close to the outdoor aviaries. Invertebrates were not included in the control diet because House Sparrows do not feed on significant amounts of invertebrates in autumn-winter (Bernis, 1989).

Given the limited availability of the two outdoor aviaries, we experimented with only one of the urban diets (Bar or dry Cat food) and the Control diet (rural food) at the same time during all six rounds. Each experimental round in both aviaries lasted 20 complete days (Herrera-Dueñas, 2018). We studied urban diets during alternative rounds to maintain the climate similarities among them by alternating the pairs Bar/Control and Cat food/Control 3 times. We also alternated the aviary side used for the Control and experimental diets to ensure that every diet was tested on both sides. During each round, captured individuals were randomly assigned to either the urban diet treatment or the Control diet in the separate aviaries (total numbers of experimental individuals: Bar = 28 (14 males/14 females); Cat food = 21 (10 males/11 females); Control = 46 (32 males/14 females). The sex balance was maintained when captures allowed it by randomising the assignation of the individuals of each sex to each dietary treatment. Diet components were made available *ad libitum* in the correspondent aviary and were renewed every 2 days at the same time.

After captures, birds were transported in ringing cloth bags to a small house in the vicinity of the aviaries. There birds were ringed with official metal rings (SEO/BirdLife) and colour rings. They were weighed (±0.01 g) and their tarsi were measured using a calliper (±0.1 mm). Blood samples (~0.25 ml) were collected (~30 min after capture) by jugular venepuncture using 25G needles. The exact total extracted blood volume was annotated. Blood was collected in heparinised 0.5 ml Eppendorf tubes and kept at 2-8°C in a portable fridge before being transported to the laboratory (maximum 5 h). Eppendorf tubes were centrifuged for 10 min at 10,000 rpm to separate red blood cells (RBC) from plasma. The haematocrit % was determined (see the Haematocrit determination below), plasma was divided into several aliquots to avoid freeze-thaw cycles, and both plasma aliquots and RBC pellets were stored at -80°C until the analyses.

After the experimental treatment of each round, birds were extracted from aviaries and the same aforementioned proceeding was followed to measure, weigh and sample the post-treatment blood of the individuals captured from both aviaries. After processing, House Sparrows were set free in their original capture area.

2.3 Ethical procedures

The present study was carried out in accordance with all the applicable institutional, regional and national guidelines for the capturing, ringing, captivity, sample taking, caring and use of animals. Bird ringing was performed with the permission of the Centro de Migración de Aves of SEO/BirdLife (Spain) and the landowners of the capture areas. Individuals' blood sampling and capture were carried out with the supervision, formation and permission of the Centre de Recuperació de Fauna de la Santa Faç of the Direcció Territorial d'Alacant (Generalitat Valenciana - Conselleria d'Agricultura, Medi Ambient, Canvi Climàtic i Desenvolupament Rural) (Authorisation Reference Number: 7029/PM/vj).

2.4 Studied variables

Laboratory analyses were carried out over a 2-week period less than 6 months after the dietary experiments had ended. Biomolecule degradation was a minor issue because we froze samples at –80°C.

2.4.1 Body condition index

Each House Sparrow's body condition index before and after dietary treatments was calculated using the Scaled Mass Index (SMI) defined by Peig and Green (2009), and previously used by Herrera-Dueñas et al. (2017) and Beaugeard et al. (2019) for the studied species. This index adjusts the mass of all the individuals to that expected for a standard body size. The energy capital accumulated in the body as a result of feeding is a reliable indicator, which we assume is a measurement of an animal's health and quality (Peig and Green, 2009; Herrera-Dueñas, 2018):

$$SMI = M_i * \left(\frac{L_0}{L_i}\right)^{bSMA}$$

M_i and L_i are the body mass and tarsus length of individual i, respectively; L_0 is the arithmetic mean tarsus length value for the study population (L_0 = 18.84 mm, SE = 0.07, n=95); bSMA is the scaling exponent estimated by the standardised major axis (SMA) regression of lnM on lnL (bSMA = 1.57) of individuals, calculated with the "lmodel2" function of the "lmodel2" package (Legendre, 2018). The regression of lnM on lnL was performed with the body mass values prior to dietary treatments.

2.4.2 Haematocrit determination

In order to estimate the blood samples haematocrit before and after treatments, we followed a similar procedure to that in Owen (2011). After centrifuging the 0.5 ml Eppendorf tubes containing blood samples, the height (mm) of the total volume but also the maximum and minimum height (mm) of the RBC pellet were measured with a calliper (30.1 mm). To estimate the total volume of plasma and RBC, we calibrated a curve-relating volume (Ql) to height (mm), measured with the calliper in three empty and random 0.5 ml Eppendorf tubes. These three tubes were progressively filled with 25 Ql of water. The volume height in mm was measured each time with a calliper (30.1 mm) until the volume in the three tubes was 500 Ql (volume (Ql) vs. height (mm) correlation R^2 = 0.999).

This procedure allowed us to determine the volume (Ql) of plasma and RBC (using the mean height of the highest and lowest parts of the RBC pellet) contained in the Eppendorf tubes, and also the haematocrit % (Owen, 2011).

2.4.3 Nutritional status

We analysed the blood plasma aliquots obtained before and after treatments using the Spinreact© commercial kits adapted for small avian samples for the following parameters: total protein (Ref: 1001290), albumin (Ref: 1001023), cholesterol (Ref: 41020), triglycerides (Ref: 41032), uric acid (Ref: 41000), and glucose (Ref: 41010). Samples' absorbance was read with a Biochrom EZ Read 400 microplate reader at 540 nm for total proteins, at 630 nm for albumin, at 520 nm for uric acid, and at 505 nm for cholesterol, triglycerides and glucose. These nutritional parameters have been previously studied in urban and rural House Sparrow populations (Herrera-Dueñas et al., 2014). All the samples in these and the following analyses (below) were run in duplicate wells on a plate to determine intraplate consistency. Both readings were used to obtain a more accurate concentration value. For total protein, albumin, cholesterol, triglycerides, uric acid and glucose, the intraplate coefficient of variation (CV) was 2.46%, 4.65%, 5.01%, 4.44%, 5.01% and 3.29%, respectively (Hanneman et al., 2013).

2.4.5 Oxidant-antioxidant biomarkers

In order to evaluate House Sparrows' oxidative damage and both their non-enzymatic and enzymatic antioxidant status, we followed the established protocols for analysing blood redox status in the plasma and RBC of small birds (Costantini et al., 2012; Récapet et al., 2019). We measured one plasmatic marker of oxidative status and two markers of the antioxidant barrier, respectively: (1) reactive oxygen metabolites, mainly driven by organic hydroperoxides, acting as precursors of long-term oxidative damage with the d-ROMs Test (MC002 kit, Diacron International, Grosseto, Italy) (Costantini et al., 2012; Récapet et al., 2019); (2) non-enzymatic antioxidant capacity by the OXY-Adsorbent Test, which is the plasma antioxidant barrier (MC435, Diacron International, Grosseto, Italy) (Costantini et al., 2012); (3) concentration of the intracellular thiol groups (antioxidant barrier) in RBCs with the –SHp test (MC433, Diacron International) using 12.5 Ql of erythrocyte haemolysate (RBCs diluted 1:200 with distilled water) (Bernabucci et al. 2005; Costantini et al., 2012; Isaksson 2013). By the d-ROMs test and with OXY-Adsorbent test results, we calculated the oxidative status index (d-ROMs:OXY ratio), which indicates the pro- and antioxidants balance and is considered a good predictor of oxidative stress (hereafter OSI) (Costantini et al., 2006). The OXY and ROM assays captured several dietary macromolecules, such as OXY measures of dietary antioxidants (e.g. α-tocopherol) and ROM measures of peroxidised dietary amino acids and lipids (Isaksson 2013). Thus both assays are likely to be influenced by food quantity and/or quality and could, hence, influence circulating ROM and OXY levels (Costantini 2010; 2011; van de Crommenacker et al., 2011). Erythrocyte-SH was chosen because it is a good estimator of total intracellular glutathione, which is the most abundant and important intracellular antioxidant (van den Berg et al., 1992; Halliwell and Gutteridge, 2015). Absorbance was read with a Biochrom EZ Read 400 microplate reader at 505 nm for d-ROMs and OXY, and at 405 for SHp. For d-ROMs, OXY and –SHp, the intraplate coefficient of variation (CV) was 3.95%, 7.15% and 5.06%, respectively (Hanneman et al., 2013).

2.4.6 Total carotenoids

In order to evaluate the total carotenoids (dietary antioxidants) present in House Sparrow plasma before and after dietary treatments, plasma aliquots (20 µl) were diluted in 200 Ql of pure ethanol (1:10) in 1.5 ml centrifuge tubes. Tubes were vortexed for 30 seconds and centrifuged (4,000 rpm) in a refrigerated centrifuge MPW 65-R for 10 minutes at 4°C to precipitate flocculent protein (Romero-Haro and Alonso-Alvarez, 2014). Absorbance was determined in 50 µl of the supernatant at 446 nm in a Biochrom EZ Read 400 microplate reader. A standard curve of lutein was prepared by diluting 1 mg of lutein (Ref: 10010811 Cayman Chemical Company) in 10 ml of pure ethanol [100 Qg/ml] and performing 11 serial dilutions 1:2 (absorbance (446nm) vs. lutein concentration [µg/ml] correlation $R^2 = 0.999$). The standard curve of lutein was used to determine the total carotenoids concentration of the plasma samples. For total carotenoids, the intraplate coefficient of variation (CV) was 5.41% (Hanneman et al., 2013).

2.5 Statistical analyses

The 14 individually measured House Sparrow variables were divided into three groups: body mass, SMI and haematocrit % (physical condition);

albumin, total protein, uric acid, triglycerides, cholesterol and glucose (nutritional status); d-ROMs, OXY-Adsorbent, oxidative stress index, -SHp and carotenoids (oxidant-antioxidant status, hereafter OAS).

2.5.1 Principal component analysis (PCA) of the change rates of the measured variables

A principal components analysis (PCA) is an appropriate statistical method to identify the gradients defined by covariating variables by, in this case, simultaneously isolating the variation of interest across multiple nutritional, oxidative stress and body condition biomarkers (Hõrak and Cohen, 2010). Firstly, to analyse the effects of dietary treatments on the measured variables, we calculated the relative change rate between the start and end of treatments, defined as the log(final value/initial value) of each variable for every individual (Crawley, 2013). As the relative change rates in body mass and SMI were the same, the PCA only included the SMI. The variables change rates were standardised to avoid giving more weight to the most variable ones with the “scale” function. The PCA was performed with the “rda” function of the “vegan” package (Oksanen et al., 2020). To select a suitable number of PCs, we considered those whose cumulative proportion of explained variance reached 75%. We calculated the correlations of each studied variable with every PC to find the most significant elements for all the new defined variables (PCs). We took into account those variables with at least moderate to strong (>|0.5|) Pearson’s correlations in their PCs (function “cor” of the package “stats”; Ratner, 2009; Gust and D’journo, 2015; R Core Team, 2020).

In order to test if the gradients identified by PCA were affected by urban diets and other variables, PC scores were used as a response variable in the linear mixed models (LMMs) with Gaussian distribution to obtain the best model/s for each PC. The fixed factors included in the models were: diet treatment (Control (r = reference level of the factor); bar; Cat food), sex (female (r); male), capture site (Alcoy (r), Muro de Alcoy, Sarga), and the interaction between diet treatment and sex. The capture site was included in the models to control for potential differences due to individuals' origin (rural heterogeneity), even though it was not a variable of interest. The combination of sampling round (1-6) an aviary (left or right) was defined as a random factor (12 levels) to contemplate lack of independence of the individuals sharing the same aviary at the same time. The "glmmTMB" function of the "glmmTMB" package (Brooks et al., 2017) was used to fit models. The "Anova" function of the "car" package (Fox and Weisberg, 2019) was employed to check for the importance of the fixed effects. The "icc" function of the "performance" package (Lüdecke et al., 2020) was utilised to estimate the proportion of the variance in the change rates that was accounted for by the random effect (Conditional Intraclass Correlation Coefficient-cICC). All the combinations of variables (fixed terms and interaction) were checked to find the best or equally plausible models (ΔAICc < 2) with the lowest Second-Order Akaike Information Criterion (AICc). Only the models for which every variable was significant (or marginally significant) were selected. The AICc value was obtained with the "AICc" function of the "MuMIn" package (Bartoń, 2019). All the analyses were carried out in RStudio 4.0.0 (R Core Team, 2020).

The predictions of the best models that included diet were graphically represented with the “ggeffects” function for the influence of random effects (“ggeffects” package; Lüdecke, 2018).

2.5.2 Analysis of the relative change rates of the raw variables

The post-treatment and pretreatment raw values of the 14 measured variables were compared for each diet by parametric (for the normally distributed differences between pairs) or non-parametric (for the non-normally ones) paired samples tests. The functions “t.test” and “wilcox.test” from the “stats” package were used (R Core Team, 2020). When the variable sex or its interaction with treatment was significant in the LMMs, the post- and pre-treatment raw values were compared and disaggregated by sex.

In order to test if the relative change rates of these variables were affected by urban diets, relative rates were employed as a response variable in the LMMs with Gaussian distribution by following the previously described procedure to obtain the final models. Furthermore, the initial values of both the specific studied variable and body condition (initial SMI) of individuals were included as covariates in the LMMs, and both were standardised with the scale function to avoid fitting problems and to make the model interpretation easy. The initial SMI was not included in the set of LMMs for body mass.

Table 1. Nutritional composition of the three experimental dietary treatments (Control and two urban diets: Bar and Cat food) and their analytical components obtained and/or calculated with the information available on product labels, complemented with Perry et al. (2009) and Fatsecret (2019). An * indicates analytical components in dry weight.

Diet	**Composition**
Control	Broken corn (49%), wheat (27%), barley (10%), sorghum (10%), sunflower seeds (3%), calcium (1%), lysine (0.9%), methionine (<0.1%), sodium (0.07%). Tomato, lettuce and apple slices
Bar	Bread (50%), cheese doodles (25%), croissants (25%)
Cat food	Cereals, meat and animal subproducts (4% chicken, 4% ox), oils and fats, subproducts of plant origin, vegetables and mineral products. Vitamins, provitamins, preservatives, antioxidants, binders and colorants

Diet	**Analytical components (%/100g)**	
Control	*Poultry food*	Crude protein 13.8%, crude fibre <2%, crude oils and fats 3.4%, inorganic matter (ash) 6.8%, humidity 11.7%, calcium 1%, lysine 0.9%, methionine <0.1%, sodium 0.07%. Other components (mostly carbohydrates) aprox. 75% without humidity
	*Tomato**	Protein 17%, carbohydrates 75%, fats 9%, carotenoids (lutein) 32 *ug/100 g*
	*Lettuce**	Protein 30%, carbohydrates 62%, fats 8%, carotenoids (lutein) 171 *ug/100 g*
	*Apple**	Protein 2%, carbohydrates 95%, fats 3%, carotenoids (lutein) 15 *ug/100 g*
Bar	Bar mixture*	Protein 10%, carbohydrates 58.75%, fats 31.25%
Cat food	Ox pellets	Crude protein 30%, crude fibres 3.5%, crude oils and fats 10%, inorganic matter (ash) 10%, humidity 10%, other components 36.5%

3. Results

3.1 Gradients of covariation between the change rates of the physical, nutritional and oxidative stress variables

The first six PCs explained 76.6% of the total variance in the measured change rates of the studied variables (physical, nutritional, OAS) in House Sparrows. The correlations of the variable's change rates with each PC are found in Table 2. The variables that most correlated with PC1 were body condition (SMI), triglycerides, glucose, total proteins and haematocrit. Thus it can be interpreted as a nutritional condition gradient. All the relevant variables in this gradient correlated negatively with PC1 (higher PC1, worse nutritional condition). PC2 could be interpreted as an overall oxidative stress gradient because the most correlated variables were oxidative stress index, d-ROMs and albumin. These three PC2 variables correlated negatively with PC2 (the higher the PC2 score, the more marked the reduction in overall oxidative stress). Cholesterol and thiols (-SHp) were the variables that most correlated with PC3, which could be interpreted as circulating and intracellular antioxidants gradients. These two variables correlated negatively with PC3 (the higher PC3, the lower the levels of antioxidants). PC4 was related to the plasma antioxidant capacity gradient because the most correlated variables were OXY and albumin, both positively correlated with PC4 (the higher PC4, the higher plasma antioxidant capacity). PC5 correlated strongly and positively with carotenoids, but weakly and negatively with OXY. It represents a gradient of the carotenoids plasma level vs. the plasma antioxidant barrier (a higher

PC5 score means an increase in carotenoids and a decrease in OXY). PC6 represented a gradient of protein intake because it correlated positively with the uric acid and haematocrit levels (an increase in PC6, a rise in protein metabolism).

3.2 Effect of urban diets on the gradients of variables change

Urban diets had significant effects on four of the six gradients generated with the PCA (Table 3). The bar treatment decreased the nutritional condition (PC1) vs. the Control treatment (Table 3; Figure 2A). The physical condition of the males feeding on Cat food tended to increase, while the nutritional condition of the females feeding on the bar diet tended to diminish (Table 3; Figure 2A). The Cat food treatment increased overall oxidative stress (PC2) compared to the Control treatment (Table 3; Figure 2B). For the gradient of the carotenoids plasma level vs. the plasma antioxidant barrier (PC5), carotenoids considerably increased in the Control diet, while the plasma antioxidant barrier increased in the Cat food diet (Table 3; Figure 2C), especially in males. The Bar and Cat food dietary treatments tended to have an inverse effect on protein intake. Compared to the Control diet, the Cat food diet significantly increased the PC6 scores, especially in females, while the Bar diet tended to lower these scores values in males (Table 3; Figure 2D). The models' ICCc values always came very close to 0. Intracellular and circulating antioxidants (PC3) and plasma antioxidant capacity (PC4) were not affected by dietary treatments (Table 3). In general, the changes in the raw variables followed a similar pattern. They are found in the Appendix of the Supplementary Material.

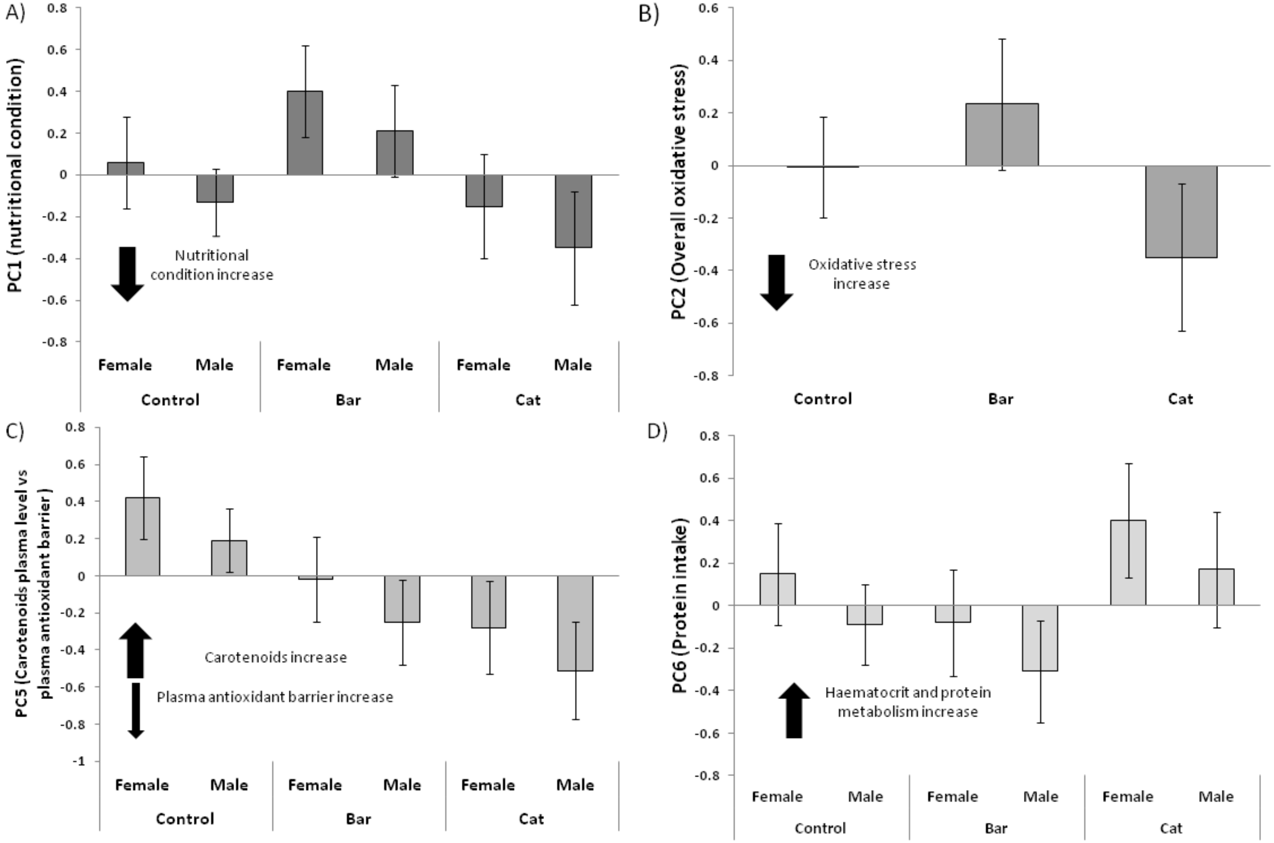
Table 2. Correlations of each studied variable with all six selected PCs (> 75% of the cumulative explained variance). The explained variance proportion of each PC is shown. The most relevant variables (r > 0.5) for each PC appear in bold

Variable	PC1 (21.34%)	PC2 (17.52%)	PC3 (11.65%)	PC4 (10.20%)	PC5 (8.31%)	PC6 (7.58%)
SMI	**-0.748**	-0.196	0.263	-0.220	0.033	-0.165
Haematocrit	**-0.595**	0.176	-0.159	-0.141	0.064	**0.511**
Albumin	-0.175	**-0.549**	-0.137	**0.516**	-0.126	-0.166
Uric acid	-0.321	-0.293	0.179	-0.196	-0.359	**0.611**
Total protein	**-0.666**	0.004	-0.386	0.373	-0.145	-0.188
Glucose	**-0.687**	0.066	0.219	-0.303	-0.129	-0.420
Triglycerides	**-0.691**	-0.313	0.311	-0.031	0.051	-0.029
Cholesterol	-0.373	-0.060	**-0.751**	0.239	-0.004	0.169
d-ROMs	0.240	**-0.899**	0.099	0.087	-0.060	0.021
OXY	0.029	0.202	0.378	**0.551**	**-0.507**	0.025
Oxidative stress index	0.210	**-0.928**	-0.078	-0.166	0.171	0.008
-SHp	0.028	-0.050	**-0.579**	-0.463	-0.200	-0.219
Carotenoids	-0.303	0.057	0.130	0.333	**0.748**	0.081

Table 3. Summary of the best GLMMs for each PC involving the physical, nutritional and oxidative stress variables of the rural House Sparrows submitted to experimental diets. The plausible models with all the variables included for being significant (or marginally significant) are ordered from the best to the worse (AICc) for each response variable. Only those with significant variables are shown. Estimated values are provided and significant p values are marked by (*) = p < 0.05; (**) = p < 0.01; (***) = p < 0.001. The marginally significant variables are marked by (·) = p < 0.1.

Response variable	AICc	ICCc	Interc.		Treatment				Sex		Capture site			
					Bar		Cat food		Male		Muro		Sarga	
PC1	156.8	0.002	-0.056		0.338	**	-0.215		-0.194	·	1.066	***	0.393	***
	157.6	0.011	-0.193	*	0.377	**	-0.161				1.023	***	0.403	**
PC2	173.6	0.034	-0.007		0.240		-0.343	*						
PC3	157.9	0.049	0.141	·							-1.019	***	-0.578	***
PC4	174.3	0.067	-0.103								0.194		0.573	**
PC5	162.8	0	0.420	***	-0.438	***	-0.704	***	-0.229	*				
	164.5	0	0.261	**	-0.394	**	-0.654	***						
PC6	175.9	0	0.149		-0.226		0.251	·	-0.234	·				
	176.1	0	0.085		-0.253	·	0.288	·	-0.225	·	0.084		0.344	*
	177.2	0	-0.071		-0.213		0.340	*			0.039		0.358	*
	177.3	0	-0.014		-0.180		0.303	*						

Figure 2. Graphs of the predictions (mean395%CI) of the best GLMM for each PC with diet treatment as a significant (or marginally significant) factor: A) PC1 (nutritional condition); B) PC2 (overall oxidative stress); C) PC5 (carotenoids plasma level vs. plasma antioxidant barrier); D) PC6 (protein intake).



4. Discussion

The experimental urban diets herein tested were able to alter the physical condition and plasma biochemical nutritional parameters of rural House Sparrows over a 20-day captivity period in a non-urban-polluted environment. Cat food had a significant effect on individuals' oxidative stress. With the PCA, we identified six gradients of change in the studied variables, which explained more than 75% variance, and allowed us to simultaneously isolate the variation of interest across the multiple nutritional, oxidative stress and body condition biomarkers of the study. However, only four gradients were significantly related to the experimental urban diets.

The main identified gradient was related to the general nutritional condition because it correlated with the change rates of body condition (SMI), triglycerides, glucose, total proteins, and haematocrit. The Bar diet reduced individuals' nutritional condition, especially for females, compared to the Control diet, while the body condition of the individuals that fed the Cat food diet did not change, or even tended to increase. Glucose and triglycerides are pre-fat storage nutritional elements obtained from diet. Low levels of these elements and/or plasma total proteins and haematocrit are associated with a poorer physical condition, low-fat reserves, anaemia, fasting, malnutrition and nutritional deficiencies of minerals (Leclercq, 1983; Wannamethee and Shaper, 1994; Acquarone et al., 2002; Sánchez-Guzmán et al., 2004; Fair et al., 2007; Pineda et al., 2013; Herrera-Dueñas et al., 2014). The low fat and protein contents in the Bar diet herein used could have prevented individuals' daily needs being met. Thus increased competition could explain the worse general nutritional condition of females because males were the more dominant House Sparrow sex (Liker and Barta, 2001).

The second gradient for the explained variance was related to the variation in the overall oxidative stress, as evidenced by its correlation with OSI, d-ROMs and albumin. The Bar diet had no significant effect on this gradient, but Cat food increased oxidative stress in both sexes vs. the Control diet. This effect could result from excessive protein and amino acids intake, which could be linked with increased albumin production, overeating and ROS generation (d-ROMs) (Walker and Johnston, 1971; Samoylenko et al., 2013), which are markers of protein perioxidation (Costantini, 2010; Costantini, 2011; Récapet et al., 2019). An increase in d-ROMs is also linked with higher overall oxidative status if the plasma antioxidant barrier (OXY) remains stable or if it does not increase to fight the pro-oxidant situation (OSI = d-ROMs/OXY ratio). Furthermore, albumin is the commonest indicator of nutrition status (Sekine et al., 2013), but is also an extracellular antioxidant agent (–SH supplier) because this protein has ligand-binding capacities that confer it antioxidant properties (e.g. binding Cu(II) and Fe(II) ions to avoid deleterious hydroxyl radical formation). It also works as a scavenger (quencher/shock absorber) that absorbs ROS and prevents them from attacking other essential proteins (Stinefelt et al., 2005; Roche et al., 2008) by acting as the first line of defence against ROS (Candiano et al., 2009). So an increase in albumin could derive from high protein intake, and could act as a defence reaction against high ROS production, or both at the same time. However, it would seem that ROS production was not compensated by the free radicals scavenging/absorber capacity of the large amount of albumin produced with this diet.

The third and fourth gradients were similar for the explained variance and both also correlated with the oxidative stress-related variables. However, they were not affected by the experimental urban diets. PC3 was related to change in intracellular (–SHp) and circulating (cholesterol) antioxidants, as shown by a positive association with both variables. Cholesterol could act as a circulating antioxidant because hydrogen peroxide is needed to produce it (Schroepfer, 2000; Casagrande et al., 2014,) and it also seems key for GSH (gluthatione/-SHp) synthesis and productivity in mammals (Chevallier et al., 2020). A high protein intake with low carbohydrates would be linked with increased cholesterol and intracellular cysteine levels (e.g. RBC), which are associated with high antioxidant GSH levels (gluthatione/-SHp) (Deneke, 2001). No effect of the Cat food diet appeared. This finding suggests that this experimental diet contained enough carbohydrates. PC4 was related to plasma antioxidant capacity because it correlated with OXY (plasma antioxidant barrier) and albumin (endogenous antioxidant protein with scavenger and absorber functions). As OXY also measures “scavengers” antioxidants, and the “shock absorbers” that include proteins (e.g. albumin), the covariation herein found could be expected. These gradients suggest that the experimental urban diets, including that ultrarich in protein, did not reduce either endogenous antioxidants production or their plasma antioxidant capacity (i.e. circulating dietary antioxidants).

The fifth gradient was related mainly to the change in carotenoids in plasma and secondarily to the plasma antioxidant barrier gradient (OXY). The individuals fed the Control diet showed increased carotenoids, while those that ate the other diets, especially Cat food, tended to present an increased antioxidant barrier (OXY), particularly males. As diet can explain variation in plasma carotenoids (Cohen et al. 2009), the effect of the Control diet suggests that it was richer in carotenoids than wild rural birds' usual diet, likely because wild House Sparrows gained less access to, or preferred, vegetables than what we assumed when we designed our Control diet. We expected to find a positive covariation between these variables because carotenoids are assumed to be dietary antioxidants in birds and the plasma antioxidant barrier (OXY) includes dietary antioxidants. However, we should contemplate the weak correlation (close to 0.5) of OXY with PC5 and be cautious when interpreting this PC. This fifth gradient showed a potential negative association between OXY and carotenoids, which suggests that these pigments might not act as significant dietary antioxidants in birds as other studies have found (Costantini et al., 2006, 2007; Costantini and Møller, 2009). Results are often conflicting when different dietary antioxidants are simultaneously considered, which implies compensation mechanisms between different antioxidant defence types (Cohen et al., 2009), which could be the case in this study. Hence further studies should be conducted to disentangle the significance of these pigments and their relation to OXY in wild birds' oxidative stress.

Finally, the sixth gradient correlated with rising haematocrit and uric acid rates. The Cat food and Bar diets had opposite effects on this gradient because the former was associated with positive values and the latter with negative values in PC6. The differential protein intake and nutrient availability in the Control, Bar and Cat diets could explain the different effects they had on this gradient (Jenni-Eiermann and Jenni 1998; Sánchez-Guzmán et al., 2004). Both haematocrit and uric acid indicated nutritional status and low levels would correlate with a poor physical condition (Plaza and Lambertucci 2018). However, the increment in uric acid could also indicate metabolic disorders, potential kidney damage and gout as a result of high (> 30%) protein intake (Chandra et al., 1985; Gartrell et al., 2003), which might also be related to high levels in haematocrit (Mayer and Donnelly 2013). For this reason, the apparently good health condition observed in the Cat food diet individuals with high haematocrit could mask actual problems deriving from increased uric acid, which could threat individuals' future well-being (Plaza and Lambertucci 2018).

Even though the Bar diet tended to reduce one of the most important avian blood antioxidants, such as uric acid (PC6), oxidative stress levels (PC2) did not increase in this diet as expected. Antioxidant levels rose in the Cat food diet (e.g. OXY, albumin, uric acid), as did oxidative stress, which suggests several complex independent biochemical ways for fighting pro-oxidant environments. However, a strong effect of high protein intake (e.g. leading to an increase in ROS, albumin, uric acid and haematocrit) could confound if there was a true antioxidant response or excessive protein intake had a side effect (Cohen et al., 2007; Costantini, 2011). Gradients PC3 and PC4, mainly related to dietary antioxidant capacity, were not affected by the high-protein Cat food diet. This implies that the response to an oxidant environment could be a more plausible explanation to interpret our results. In the dual-faceted scenario of antioxidant defence vs. oxidative damage (Beaulieu and Costantini, 2014), the Cat food diet represents a situation in which a significant increase in oxidative stress (PC2) is not compensated by a rise in the antioxidant capacity-related variables (PC3 and PC4). Furthermore, the apparent increase in the plasma antioxidant barrier (PC5) in the Cat food diet should be carefully interpreted because it should be related more to a non-increase in carotenoids than to a real OXY increase. Although some individual circulating antioxidants increased (e.g. uric acid (PC6; Appendix Table1 and Figure 2A), albumin (PC2), cholesterol (Appendix Figure 2E) and oxidative damage (PC2, d-ROMs and OSI) seemed to weigh more heavily in the balance (Herrera-Dueñas et al. 2017), which highlights the potential effect of urban diet on birds' oxidative stress.

Despite the present study obtaining most of the expected results, we acknowledge some limitations that should be taken into account to evaluate potential consequences for wild urban populations (Calisi and Bentley 2009). Captivity *per se* might induce stress, which can alter physiological and oxidative stress responses to urban diets compared to wild birds (Dickens and Bentley, 2014). The wide variety and heterogeneity of anthropogenic food in cities, especially for the Bar diet (e.g. bread, snacks, ham, chicken cheese, chips, etc.), could not be fully represented in the diets included in this study (MacGregor-Fors et al., 2020). The Bar diet is only available on bar/restaurant terraces. These urban elements offer anthropogenic food resources with different geographical, seasonal and daily patterns in distinct countries. Frosty days in the Mediterranean Region are scarce and bar terraces are set up all year round (Bernat-Ponce et al., 2018; 2021b). They might be restricted to warmer months in northern countries, which might seasonally limit Bar diet availability (Haemig et al., 2015). Even though food resources can abound more in warmer months when terraces are open, terrace activity might also determine food availability peaks (e.g., breakfast, lunch). Furthermore, Bar food composition may vary with most consumed meals in different countries, and even their presence might be restricted to certain urban sectors in cities (e.g. city centres, business streets, public transport stations).

Our results show that, in captivity, a strict nutritionally-poor Bar diet can reduce House Sparrows' physical condition, especially females, which might derive from the males dominating the experiment. However, this can also occur in the wild if food is scarce and females have to compete with males for limited resources. This hypothetical scenario could even imply lower survival, a weaker immune system and a worse body condition for urban female House Sparrows, even in unpolluted habitats (Cornet et al., 2014). It could result in male-biased sex ratios in small or declining populations, which could have significant pernicious effects on this declining species' breeding performance, long-term stability and demographical parameters (Shaw et al., 2008; Morrison et al., 2016).

A pet food diet in urban areas is expected to be similar (rich in protein and fats) across countries to that herein employed. However, city individuals are likely to combine such food with other natural or anthropogenic elements (e.g. snack-bar food, seeds, insects) (Andrzejczak et al., 2021). A pet food diet might be found mainly in neighbourhoods with individual houses and backyards, probably far from the main areas with bar terraces. This suggests that the distribution of both experimental diets in a city is likely to be irregular, which could have differential effects on House Sparrows between countries, cities, or even between populations, in the same city (Andrzejczak et al. 2021).

We found that a strict cat food diet in captivity tends to improve body condition, but increases oxidative stress, even with a greater antioxidant barrier. In the present study, more oxidative damage (d-ROMs > 30%; Appendix Figure 3A) found in the House Sparrows fed Cat food was above the difference in a similar oxidative damage marker (TBARS;, a marker of lipid peroxidation; Halliwell and Gutteridge, 2015; Costantini, 2016) between rural and urban wild individuals (TBARS >6.5-15%) reported by Herrera-Dueñas et al. (2014; 2017). Therefore, our results suggest that Cat food in an unpolluted environment is enough to generate even bigger differences in oxidative stress biomarkers than those found in the wild between rural and urban individuals. However, the negative effects of a pet food diet may be compensated in wild populations because this urban food may be available when other food resources (e.g. insects) are scarce, and also when large high amounts of protein are required (Andrzejczak et al., 2021). So the demographic consequences of these diets in the wild are not clear and should be evaluated by conducting further research. The potential effects of urban diets on breeding parameters in captivity in unpolluted environments might be one of the next steps to be evaluated, and also if the body condition and oxidative stress parameters alter in rural birds under healthy food conditions, but with the presence of some urban pollutants (e.g. NO_X).

5. Conclusions

Our results experimentally show that the urban diet type eaten by House Sparrows affects their general nutritional condition and oxidative stress in a rural outdoor environment free of urban pollutants. Unbalanced urban diets (nutritionally poor) can worsen physical condition, by lowering the general body condition and haematocrit, and alter the nutritional parameters linked with malnutrition (low levels of triglycerides, uric acid, total proteins, glucose), especially in females. A high-protein unbalanced urban diet increased both ROS levels (d-ROMs) and the plasma antioxidant barrier (OXY, albumin, uric acid), but not enough to avoid oxidative stress. This increase in uric acid could result in health risks (e.g. gout, kidney disease). Interestingly, our results show that the general condition and the oxidative stress of the rural individuals reared from eggs to adults in a wild "healthy" rural environment can be profoundly altered if urban food constraints come into force and are isolated from pollution. However, we do not know the long-term demographic consequences (e.g. survival, breeding) that these diets can have on wild populations. More research is needed to explore the effects of long-term exposure to urban diets and to complete birds' oxidative stress puzzle in cities (urban diet/pollution). A synergic effect of urban low-quality diet and pollution might boost general House Sparrow decline in European cities, and the results presented herein are a promising start to disentangle the importance of the urban diet role on the decline of this urban bioindicator.

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Data statement

Data available on request from the authors.

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Supplementary material

APPENDIX

1. Raw variables relative change rates after dietary treatments

Urban diets had significant effects on five of the 14 sampled variables when their pre-post treatment relative change rates were compared to those of the control diet (Appendix Table 1). Capture site was significant for several variables, where the Muro and Sarga change rates took the same sign and usually differed significantly from the Alcoy ones. The initial value of each studied variable was important for their change rate, with the lowest and highest values tending to homogenise to the mean value after dietary treatments. However, only one interaction, treatment with sex in triglycerides, was significant (Appendix Table 1). The results of the GLMMs for physical condition, nutritional status and oxidant-antioxidant status, but the raw values of the mean pre-post dietary treatments of the studied variables and their pre-post paired-comparisons, can be found in the Appendix in Table 1 and Figures 1-3.

1.1 Physical condition

The plausible GLMMs showed that body mass diminished after dietary treatments in captivity (Appendix Table 1). The Bar diet tended to reduce body mass ($p = 0.059$) compared to the Control diet, while the individuals that ate the Cat food diet obtained similar values to the control ones. In general, males significantly lost less body mass or their values remained compared to females for all three diets. Body condition, calculated as SMI, tended to decrease with captivity similarly to body mass ($p = 0.082$; Appendix Table 1; Appendix Figure 1). The SMIs of males did not significantly reduce in comparison to females for any experimental diet. Haematocrit significantly reduced with the Bar diet (Appendix Table 1; Appendix Figure 1).

1.2 Nutritional status

The total protein, glucose and cholesterol variables were not affected by urban diets compared to the Control one. However, these three variables responded differently to captivity. The total protein relative change rate was not affected by the experiment, while glucose and cholesterol levels significantly rose with captivity.

The albumin values were affected by dietary treatments, and significantly lowered with the Bar diet and marginally, but significantly, increased with the Cat food diet vs. the Control diet. Uric acid lowered in the Control diet and the Bar diet tended to further lower these levels (marginally significant). However, the the pre-treatment levels remained with the Cat food diet compared to the control diet birds, which significantly lost uric acid, and the Bar diet individuals tended to lose even more than the control birds (p = 0.095; Appendix Figure 2). Triglycerides were also affected by treatment, which significantly decreased in the females on the Bar diet vs. males. The triglycerides values of the males on the Bar diet increased (Bar treatment x sex male interaction; coefficient = 0.175; p = 0.032; Appendix Table 1 and Figure 1).

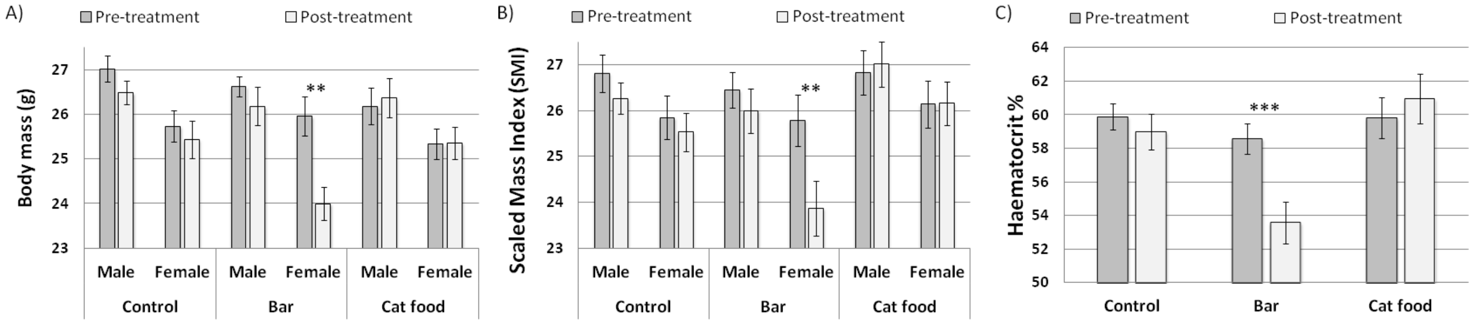
1.3 Oxidant-Antioxidant status

Only one of the five variables was affected by dietary treatments. Intracellular thiols (-SHp) were reduced by captivity, while the OXY values (plasma barrier to hypochlorite-induced oxidation) had increased after the three dietary treatments. Total carotenoids increased in the rural diet group (control), but remained stable after the two urban dietary treatments.

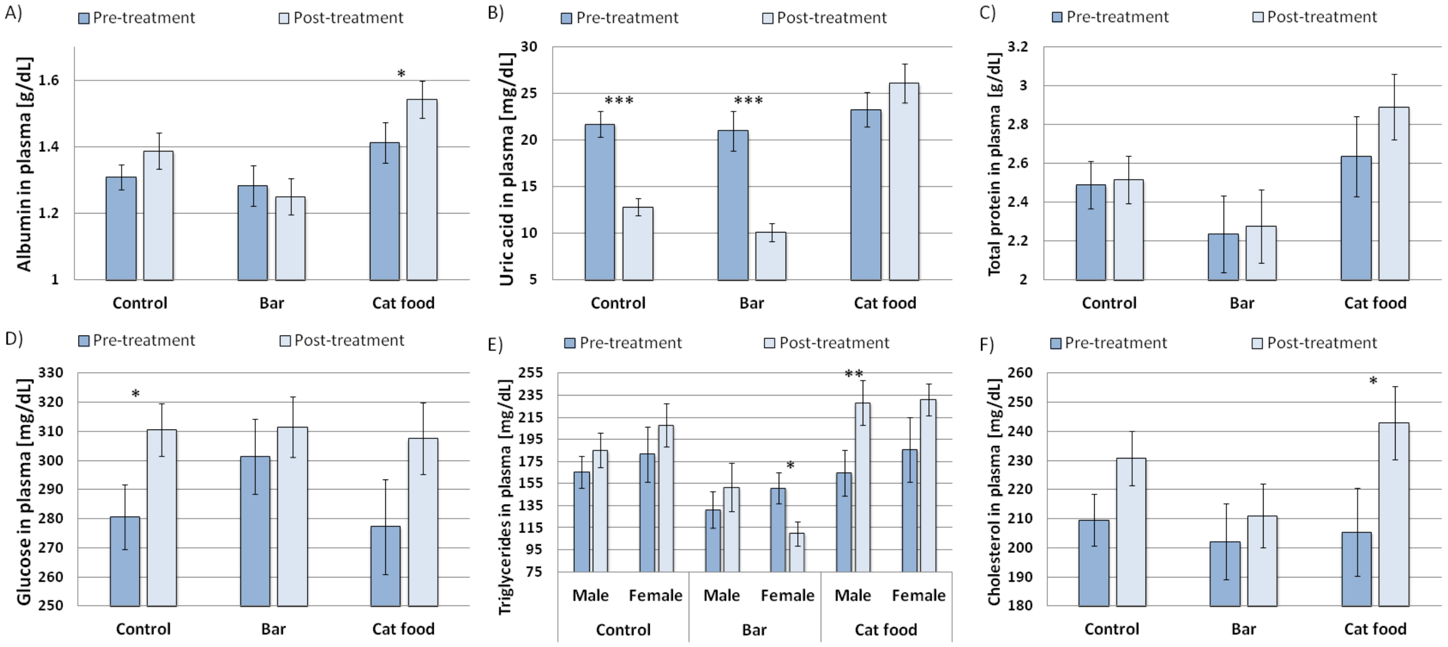
Appendix Table 1. Results of the best GLMMs for the change rates of each studied variable in House Sparrows to analyse the effects of dietary treatments. Plausible models with all the included significant (or marginally significant) variables are ordered from the best to the worse (AICc). Only those with significant variables are shown. Estimates values are provided and significant p values are marked by (*) = $p < 0.05$; (**) = $p < 0.01$; (***) = $p < 0.001$. The marginally significant variables are marked by (·) = $p < 0.1$. SMI was not included in the body mass and SMI GLMMs. The treatment with sex interaction was significant only in the triglycerides model, and the interaction estimates and p-value are shown in the text. Those variables that were significant in the ANOVA analysis but not in the GLMM conditional model are indicated in italics, The treatment Bar and sex male interaction (GLMM) in the triglycerides variable was significant ($p = 0.032$), but the global interaction in the ANOVA analysis was not ($p = 0.09$).

Group	Response variable	AICc	ICCc	Interc.	Treatment Bar	Treatment Cat food	Sex Male	SMI	Init. value	Capture site Muro	Capture site Sarga
Physical condition	**Body mass**	-454.9	0.157	-0.012 *					-0.008 **	-0.019	-0.028 **
		-453.5	0.050	-0.010 ·	-0.012 ·	0.005	0.014 **		-0.008 **	-0.027 *	-0.024 **
	Scaled Mass Index	-454.2	0.195	-0.010 ·					-0.007 **	-0.020	-0.034 **
		-452.5	0.078	-0.008	*-0.012*	*0.008*	0.012 **		-0.007 **	-0.028 *	-0.029 ***
	Haematocrit (%)	-300.9	0.076	0.008	-0.040 **	0.004			-0.030 ***	-0.063 **	-0.059 **
		-299.8	0.194	-0.001					-0.031 ***	-0.055 *	-0.068 **
Nutrirional status	**Albumin**	-256.7	0.100	-0.008	-0.046 *	0.047 ·			-0.064 ***	0.082 *	0.148 ***
	Uric acid	-17.2	0.004	-0.238 ***	-0.051 ·	0.316 ***			-0.176 ***		
	Total protein	-107.4	0.084	0.018					-0.076 ***		
	Glucose	-204.3	0.077	0.038 **					-0.102 ***		
	Triglycerides	-52.2	0.021	0.119 ***	-0.165 ***	0.079			-0.098 ***	-0.307 ***	-0.155 **
	(Interaction)	-50.6	0	0.128 **	-0.248 ***	0.060	*-0.020*		-0.096 ***	-0.294 ***	-0.148 **
	Cholesterol	-119.0	0.058	0.053 **					-0.114 ***		
Oxidative stress status	**d-ROMs Test**	9.2	0.150	-0.017					-0.242 ***	0.156	0.273 *
		10.2	0.212	0.057					-0.237 ***		
	OXY-Adsorbent Test	-110.2	0	0.058 ***					-0.111 ***	-0.126 *	-0.074 *
	Oxidative stress index	37.9	0.100	-0.074					-0.243 ***	0.268 ·	0.304 **
	SHp Test	-327.7	0.062	-0.023 ***					-0.025 ***		
	Total carotenoids	275.7	0	1.166 ***	-1.106 ***	-1.104 ***			-0.537 ***	-1.280 **	-0.113
		277.3	0	1.077 ***	-1.071 ***	-1.131 ***		-0.207 *	-0.580 ***		

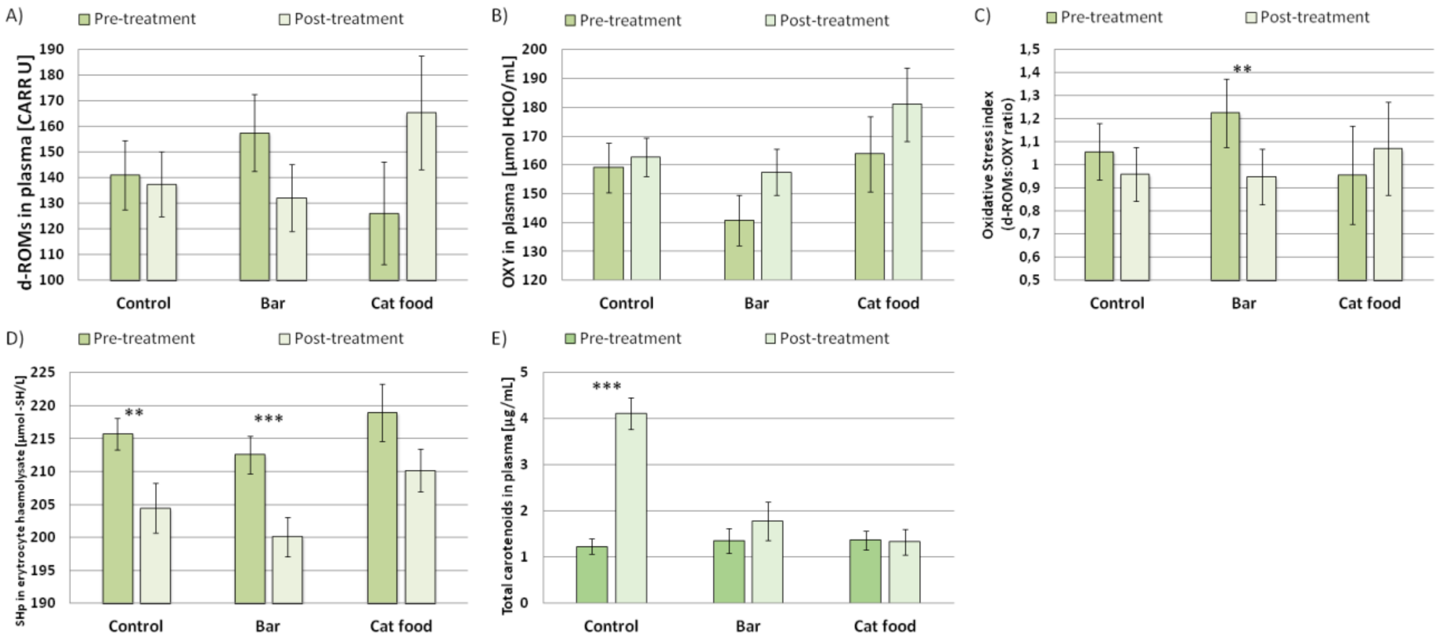
Appendix Figure 1. The pre- and post-treatment mean values (3SE) after the three dietary treatments (Control, Bar, Ccat food) of the studied physical condition variables in House Sparrows: A) Body mass (g); B) SMI; C) haematocrit (%). The paired samples p values are marked by (*) = p < 0.05; (**) = p < 0.01; (***) = p < 0.001.



Appendix Figure 2. The pre- and post-treatment mean values (3SE) after the three dietary treatments (Control, Bar, Cat food) of the studied nutritional variables in House Sparrows: A) albumin [g/dL]); B) uric acid [mg/dL]); C) total protein [g/dL]; D) glucose [mg/dL]; E) triglycerides [mg/dL]); F) cholesterol [mg/dL]). The paired samples p values are marked by (*) = $p < 0.05$; (**) = $p < 0.01$; (***) = $p < 0.001$.



Appendix Figure 3. The pre- and post-treatment mean values (3SE) of the studied House Sparrows' oxidant-antioxidant biomarkers: A) d-ROMs [CARR U]); B) OXY [Qmol HCLO/mL]); C) oxidative Stress index [d-ROMs:OXY ratio]); D) -SHp [Qmol -SPh/L]); E) total Carotenoids [Qg/mL]. The paired samples p values are marked by (*) = p < 0.05; (**) = p < 0.01; (***) = p < 0.001.



Recreational noise pollution of traditional festivals reduces the juvenile productivity of an avian urban bioindicator

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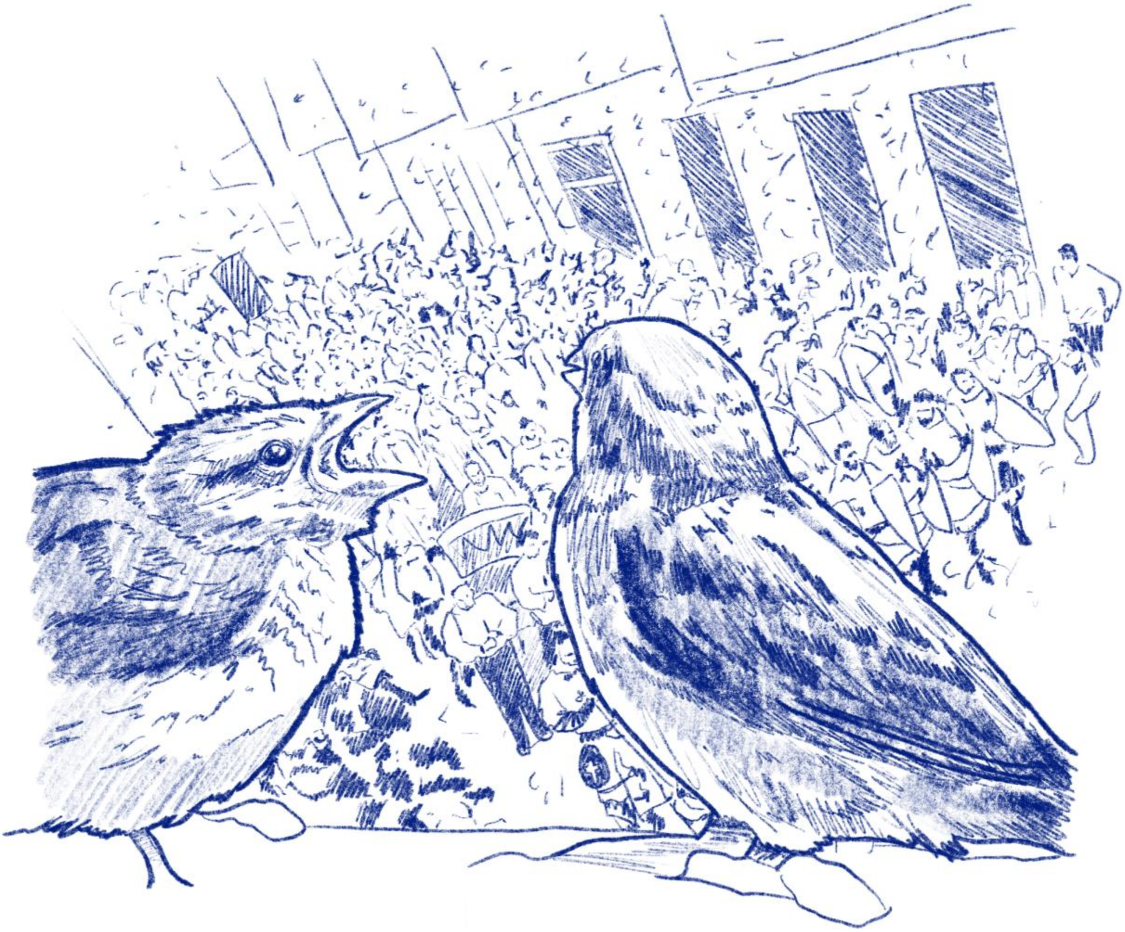

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Chapter 7

RECREATIONAL NOISE POLLUTION OF TRADITIONAL FESTIVALS REDUCES THE JUVENILE PRODUCTIVITY OF AN AVIAN URBAN BIOINDICATOR

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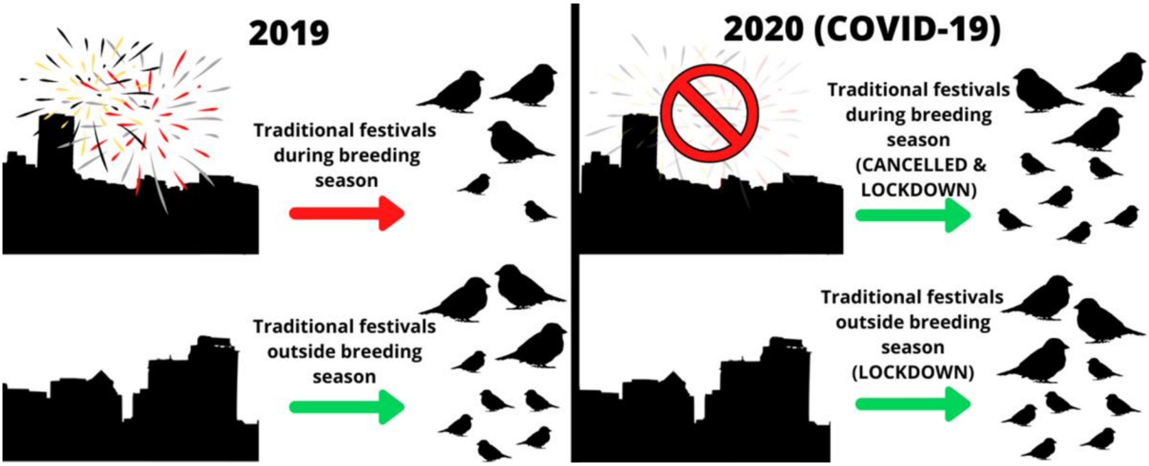
Abstract

Noise is a pollutant of emergent concern for ecologists and conservation biologists. Recreational noise pollution, especially unpredictable and intermittent sounds, and its effects on wildlife and biodiversity have been poorly studied. Researchers have paid very little attention to the effect of noisy traditional festivals (fireworks and powder-guns). This study aimed to explore the effect of these recreational activities on the juvenile productivity of an urban avian bioindicator: the house sparrow. We studied five pairs of localities in the Valencia Region (E Spain) with noisy traditional festivals. Each pair was composed of one locality with festivals during the breeding season and the closest similar locality, but with festivals outside the reproductive period (controls). Both locality types were sampled twice each spring (May–June of 2019 and 2020). Sampling dates were selected as 15 and 30 days after noisy festivals ended, while the control localities were sampled 1 day after the census of their correspondent town pair with noisy breeding season festivals. The ratio of the juveniles/adults detected during surveys in the influence area of festivals (100-m buffer around the parades route) was used as a house sparrow breeding success proxy. Data were analysed using GLMM: year (2019/2020), festivals season (breeding/non-breeding), survey (15/30 days), and their interactions were included as fixed factors. Pair of localities and locality nested within the pair were random factors. In 2019, juvenile productivity was lower in the towns with noisy traditional festivals during the breeding season than in the control towns. The 2020 festivals were cancelled due to COVID-19. In spring 2020, house sparrow juvenile productivity was the same in both town groups. Lockdown did not increase this species' juvenile productivity in the control localities in 2020 *versus* 2019.

Keywords

COVID-19, Fireworks, House sparrow, Recreational noise, Pollution

Graphical abstract



Introduction

Anthropogenic noise is an almost omnipresent environmental stressor in an urbanising world. Environmental noise pollution is a form of air pollution (Goines and Hagler, 2007) and is a threat for not only human health and well-being, but also for wildlife (Meillère et al., 2015; Merrall and Evans, 2020). It has recently become an important factor for ecologists and conservation biologists to study (Bowles, 1995; Gill and Sutherland, 2000). Increasing population and urbanisation are often linked with higher levels of different noises, usually related to human transport, especially road traffic (Barber et al., 2010) and industry (Habib et al., 2007; Slabbekoorn, 2013; Sordello et al., 2020). These sources of human-made sounds are the most widely studied ones, and this worldwide pollutant and its negative effects on wildlife and biodiversity are clearer every year (Meillère et al., 2015; Davies et al., 2017; Sordello et al., 2020). However, certain sources of noise and their implications for the conservation of species and wildlife are poorly studied and are still not completely understood. For example, very few studies have dealt with recreational noise sources and breeding performance (Sordello et al., 2020)

There are two main types of anthropogenic noise: chronic/frequent (usually predictable), such as road traffic or airports, and temporal/intermittent (usually unpredictable), such as fireworks or alarms (Barber et al., 2010; Shamoun-Baranes et al., 2011; Sordello et al., 2020). Both kinds of noise can have different impacts on wildlife because they can affect habitat use, communication, biology/physiology, reproduction, behaviour, or even the whole ecosystem (Francis and Barber, 2013; Sordello et al., 2020). These effects can differ depending on the main noise features, such as intensity, frequency, or even time of day (Francis and Barber, 2013).

Most research into anthropogenic noise effects on wildlife have focused on the consequences of chronic noise on avian communication (Halfwerk et al., 2011a; Leonard and Horn, 2012; Brumm and Zollinger, 2013; Gil and Brumm, 2014). Even though noise pollution reduces growth and body condition in several bird species (Injaian et al., 2018; Phillips et al., 2018), behavioural changes are the commonest effects of this pollutant on animals. Among them, Francis and Barber (2013) highlighted four categories: changes in temporal patterns (sleep, routines, activity) (Kight and Swaddle, 2011); modification of spatial distributions or even movements (Francis et al., 2011a); decreased foraging or provisioning efficiency (Leonard and Horn, 2012) linked with increased vigilance and anti-predator behaviour (Quinn et al., 2006; Merral and Evans, 2020), and changes in mate attraction and/or territorial defence (Halfwerk et al., 2011a). These behavioural changes could have major effects on species' breeding success, but noise impacts on wildlife reproduction are not well-documented (Sordello et al., 2020).

The reduction of parental care due to noise pollution, linked with decreased nest attendance (Schroeder et al., 2012; Meillère et al., 2015) and consequent poor incubation conditions (alterations in developmental temperatures), has been widely documented (Nord and Nilsson, 2011;

DuRant et al., 2013). These anthropogenic noise-derived situations can result in marked alterations to early (pre-hatching) and late life stages (post-hatching) (Ardia et al., 2010), such as mismatching embryonic energy use (Booth, 1987; Eiby and Booth, 2008; DuRant et al., 2011), reduced immunity (Ardia et al., 2010; DuRant et al., 2012a) and/or mismatching metabolic and growth rates (Booth, 1987; Olson et al., 2006; DuRant et al., 2012b; Injaian et al., 2018). Chronic urban noise effects on birds are related to smaller clutches (Halfwerk et al., 2011b), higher embryonic mortality rates (Potvin and MacDougall-Shackleton, 2015); shorter telomeres in nestlings and fledglings (Dorado-Correa et al., 2018), fewer fledglings (Schroeder et al., 2012) and a general reduction in their biological fitness (Kight and Swaddle, 2011; Meillère et al., 2015).

Although urban individuals can adapt their behaviour to chronic noisy environments (Slabbekoorn and Peet, 2003; Mockford and Marshall, 2009), behavioural adaptation to intermittent and unpredictable loud noises, such as alarms or pyrotechnics that are often used to scare wildlife (Mott and Boyd, 1995; Berge et al., 2007), might be more difficult and could have strong negative effects on these populations.

In lots of towns and cities in the Mediterranean area of Spain, and other countries like Italy, France or Croatia, traditional local festivals include parades of Moors and Christians. They represent the battles of past ages between both armies in these localities, but they are also usually linked with religious aspects. These parades have been exported to other countries, especially to Latin America (Catalá-Pérez, 2012).

Eastern Spain is the Mediterranean area with more localities (185 all year round) that perform these specific representations (Catalá-Pérez, 2012). Worldwide however, but especially in this region, other traditional festivals with similar recreational fireworks take place, such as the "mascletà" of "Fallas" in Valencia and "Hogueras" in Alicante. Extreme air pollution, due to increased levels of PM10, PM2.5, and other pollutants, is associated with massive firework celebrations and is recognized as an important health and an environmental issue (Moreno et al., 2007; Andradottir and Thorsteinsson, 2019). Several studies have documented the increase of PM2.5 and PM10 due to firework festivals. However, this increase lasts for just several hours and its concentration drops to control sites levels in less than 24 hours (Seidel and Birnbaumb, 2015; Lai and Brimblecombe, 2017; Hoyos et al., 2020). Also, some studies describe the effect of particulate matter on birds' health (e.g. Gorriz et al., 1994; Steyn and Maina, 2015) but individuals were exposed to high levels of these pollutants for months.

In the Valencia Region, Moors and Christians festivals are described by Catalá-Pérez (2012) as a "waste of gunpowder" where more than 5,000 kg are spent. It is used in fireworks, blunderbusses and harquebus (powder-guns) to simulate historic battles during parades that move along historic town centres (Supplementary Material Fig. 1). Besides, the powder-guns of parades can produce sonic booms of between 120-150 dB (Lloret, 2017; Maeso et al., 2018). These sonic booms are defined as impulse or impact noises because they are very brief sounds or short bursts with a sound pressure rise of at least 40 dB in 0.5 seconds, or even faster, that may occur singly or as a series of events (Tandon, 2003).

The effects of recreational noise on historic buildings, urban heritage and pet welfare have been studied (Bolster, 2012; Lloret, 2017). In addition, Shamoun-Baranes et al. (2011) recorded immediate behavioral responses of wild birds to high festival noise levels from fireworks. However, to our knowledge, no studies have been carried out to evaluate the effects of these recreational, acute, temporal and intermittent boom sounds on urban birds breeding success. Furthermore, we consider that these sounds can be especially negative in spring, the breeding season of most urban birds.

This research aimed to study the potential pernicious effect of powder-gun sonic booms and fireworks on the breeding success of an avian urban bioindicator that nests in human constructions, the house sparrow (*Passer domesticus*). We hypothesise that this noise pollution would reduce house sparrow juvenile productivity (juveniles/adults ratio) in those localities with traditional festivals in the middle of the breeding season (April-May) and we would, therefore, find a lower juveniles/adults ratio compared to the control localities (with the same kind of festivals, but outside the breeding season). Furthermore, we explore if there exists a consistent increase of particulate matter pollution derived from fireworks in a sample of localities in the Valencian region to assess the potential effect of these air pollutants on juvenile productivity. Due to the global COVID-19 pandemic, these festivals were cancelled in 2020, which created the opportunity to check the effect of suppressing recreational noise in the same towns. We predicted that if noisy festivals during the breeding season were the main cause of reduced breeding performance, the juveniles/adults ratio would be the same in both groups of towns after the prohibition in 2020. In addition, we checked if this 2020 lockdown had a positive effect on house sparrow juvenile productivity in those localities that hold no festivals in spring.

Material and methods

Study area

The present study was carried out in ten south-eastern Spanish localities distributed in the provinces of Alicante (9) and Valencia (1) (Fig. 1). Every locality runs important events with powder-guns and fireworks (recreational impulse-type noise) as part of their traditional Moors and Christians festivals. These festival parades walk around the town's most centric streets. These localities were distributed into five pairs of similar localities, which were selected according to the following criteria: i) five localities were selected where parades are held in the middle of the house sparrow breeding season (April-May) (Bernis, 1989; Anderson, 2006) (noisy-breeding localities); ii) for each noisy-breeding locality, we selected one locality located at less than 20 km (range 3-18 km), whose festivals are held outside the breeding season (non-noisy breeding localities). These localities were established as the control areas (Table 1). Localities within a pair were located in the same bioclimatic stage, so no differences in breeding phenology were expected between them (Costa, 1982; Rivas-

Martínez and Armaiz, 1984). These 10 localities were visited twice between May and June in both 2019 and 2020.

Survey design and bird sampling

The Moors and Christians parade routes in each locality (noisy and control) were obtained thanks to the public information of the Festivals Associations. Their direct areas of influence (calculated as a buffer area of 100 m around and along routes; Supplementary Material Fig. 2) were established in QGIS (2018). We considered that a 100-metre buffer area was enough to detect the effects of parades on the species.

The house sparrow breeding season in eastern Spain starts in March and ends by August (Murgui, 2016). The incubation period of this species ranges between 13 and 19 days (Pardo, 1982; Veiga, 1990; Moreno-Rueda and Soler, 2002), while nestlings stay in nests for almost 15 days (Pardo, 1982). Fledglings are fed by parents for 10–14 days after fledging, and entirely rely on parents for food for the first 7 days after leaving the nest (Weaver, 1942; Summers-Smith, 1963). They become more independent of adults between 7–14 days after fledging and move in flocks (Summers-Smith, 1963). Therefore, festivals with recreational noise can affect juvenile productivity in four breeding stages: adults pairing, nest construction, incubation, nestlings and fledglings.

Each noisy locality was surveyed twice along their parade's areas of influence each year. The first survey was carried out 15 days after the festivals ended (survey date: 10 May - 6 June). The second survey was performed 30 days after festivals finished (15 days after the first survey; 25

May - 19 June). This survey proceeding allowed us to take into account the species' breeding phenology, its reproductive peak of activity and the potential effects of festival noise pollution on different breeding stages, especially incubation and nestlings. The peak of the clutches number in this species occurs usually 1 month after the start of the laying period, late March in the proximity of our study area (Gil-Delgado et al., 1979; Murgui, 2016), and overlaps in most cases with spring festival dates (late in April to mid-May). During this period, most breeding pairs would be involved in incubation while a few would be feeding small nestlings. The localities with festivals outside the breeding season (control localities) were sampled the day after surveying the noisy locality of its pair, also along their parade's area of influence. Therefore, the control localities were also sampled twice yearly, with a difference of 15 days between surveys, approximately 15 (11 May - 7 June) and 30 days (26 May - 20 June) after the noisy festivals of its pair locality.

Surveys started 15 minutes after the official sunrise time and always finished before 10:30h on dry, still (wind speed < 10km/h) mornings as house sparrows are more active during that period (Anderson, 2006). During each sampling session, the noisy-influenced area was exhaustively surveyed by the same observer, who walked and searched contacts with house sparrows. Sex (in adults), bird age (adult/juvenile) and location were recorded in the sampling area. Differences between males/females and juveniles/females can be found in Summers-Smith (1963), Cramp and Perrins (1994), Peach et al. (2013), and Jenni and Winkler (2020). Juveniles/females are increasingly difficult to separate later in summer as juveniles mature (Peach et al., 2013). However, our study was carried out in spring (May-June), which minimised this issue.

In order to correctly identify individuals, especially juveniles/females, we used binoculars and a reflex camera with a telephoto lens that allowed the photo-identifying of doubtful bird age through a series of pictures. To avoid age misclassification, the identification of juvenile/females was not based on an only-trait criterion.

The surveys in the noisy-breeding and non-noisy breeding (control) localities were carried out in 2019 and 2020. Due to the global COVID-19 pandemic and lockdown restrictions, the Moors and Christians festivals were cancelled in all the towns in 2020. These restrictions created the opportunity to check the effect of suppressing recreational noise in the same towns (noisy breeding localities), but also the lockdown effect on house sparrow breeding productivity in the control localities.

Assessment of air pollution derived from fireworks festivals

The Valencian Government has a web of vigilance and control of environmental pollution with 74 monitoring stations in 42 localities all over the region (RVVCCA, 2021). However, only one out of the ten study localities with Moors and Christians parades had a monitoring station (Alcoy). In order to take into account the potential effect of pollutants derived from festivals fireworks we selected 15 monitoring stations of 15 localities (Supplementary Table 1) that had important festivals in 2019, with similar fireworks displays during them (Moors and Christians, Fallas, Magdalena, Hogueras, Mascletà), and obtained daily measurements of

PM10 (Qg/m^3) in them. Only 7 of these localities had PM2.5 records, thus we extracted also daily PM2.5 values of these localities to evaluate in each of them the correlation between PM10 and PM2.5 values, a relationship that has been already found in other studies (Lai and Brimblecombe, 2017; Munir et al., 2017). To study if fireworks festivals increased significantly air pollution levels we calculated mean daily PM10 values during one week before the start of the festivals and the mean daily values of PM10 during the festivals (mean duration of festivals: 4.33 days; range = 3-9 days). We calculated the relative growth rate of mean daily PM10 values due to fireworks festivals, measured as log(festivals mean/week before festivals mean) (Crawley, 2013), which can be found in the Supplementary Material (Supplementary Table 1).

Statistical analysis

We calculated the ratio juveniles/adults of each studied locality during both visits and in each year as a house sparrow breeding productivity proxy (Du Feu and McMeeking, 1991; Cave et al., 2009; Morrissette et al., 2010; Peach et al., 2013). As recently fledged birds of the same familiar flock cannot be considered independent data, and are easily distinguishable from independent juveniles, all the fledglings in the same family flock were considered to be only one first-year bird contact to calculate the juveniles/adults ratio. The dataset of the present study can be found in the Supplementary Material (Supplementary Table 2).

We used a General Linear Mixed Model (GLMM) to evaluate the effects of recreational festivals noise (powder-guns/fireworks) on house sparrow juvenile productivity. The dependent variable was the juveniles/adults ratio. The model included the following fixed factors: year of study (2019 or 2020), festivals season (breeding or outside breeding) and survey (first or second: 15 or 30 days after the spring festivals, respectively). The triple interaction and double interactions between the three fixed factors were also included. Pair identity and locality, the latter nested within the former, were included as random factors in the model. We employed the function “glmmTMB” of the R package “glmmTMB” (Brooks et al., 2017) to fit the GLMM with Gaussian distribution. We calculated the conditional intraclass correlation (ICCc) to estimate the proportion of the variance in the ratio that was accounted for by random effects with function “icc” of the “sjstats” package (Lüdecke, 2020).

To evaluate the effects of recreational festivals on particulate matter pollution we fitted a General Linear Model (GLM) with Gaussian distribution and the relative growth rate of PM10 as dependent variable using the selected localities in Supplementary Table 1. We assessed the effect of the type of festival (Moors and Christians or others), by testing if including this variable as factor with two levels, improved the model reducing the AICc (“AICc” function of the “MuMIn” package (Barton, 2020)). The effect of festivals on PM10 change was assessed through the significance of the intercept. Finally, to check if between towns variability in PM10 pollution derived from fireworks increased variability in juvenile productivity in towns with festivals in spring, we compared coefficients of variation of juveniles/adults ratio between both types of towns in both 2019 surveys. We used the “asymptotic_test” function of the “cvequality” package (Marwick and Krishnamoorthy, 2019). The analyses were carried out in R 4.0.0 (R Core Team, 2020).

Fig. 1 Location of the 10 studied localities (five pairs) selected to study the effects of recreational powder-guns/fireworks noise on house sparrow juvenile productivity in the Valencia Region (SE Spain). Pairs of localities are highlighted in the same colour and symbol.

Table 1 The geographical, demographical and climate characteristics of the selected towns to study the effects of recreational powder-guns/fireworks noise on house sparrow juvenile productivity in the Valencia Region (SE Spain). This table was produced by the authors using the data provided by the Festivals Associations, AVAMET (2020), and the Spanish National Statistics Institute (Instituto Nacional de Estadística, 2020).

Pair	Breeding category	Locality	Coordinates	Festivals	Sampled area (ha)	Number of Inhabitants (2019)	Temperature (°C) (2019)			Rainfall mm (2019)	Altitude m.a.s.l.
							Min.	Mean	Max.		
1	Noisy	Banyeres de Mariola	38°42’57”N 0°39’26”W	22-25 Apr.	26.00	7 068	-4.5	14.7	37.4	418.8	816
	Non-noisy	Bocairent	38°45’57”N 0°36’46”W	1-6 Feb.	24.54	4 195	-5.7	14.8	38.3	899.0	680
2	Noisy	Onil	38°37’46”N 0°40’26”W	22 Apr.-1 May	29.14	7 507	-5.3	-	37.6	462.8*	697
	Non-noisy	Castalla	38°35’48”N 0°40’15”W	1-4 Sept.	36.38	10 124	-	-	-	426.0	675
3	Noisy	Muro de Alcoy	38°46’47”N 0°26’10”W	10-13 May	29.21	9 324	-5.9	16.3	40.2	350.4	410
	Non-noisy	Cocentaina	38°44’42”N 0°26’26”W	9-12 Aug.	28.21	11 511	-3.5	16.3	40.4	518.4	430
4	Noisy	Petrer	38°29’00”N 0°46’00”W	14 May[1]	28.67	34 276	-2.4	16.6	39.5	369.8	462
	Non-noisy	Villena	38°38’06”N 0°51’57”W	4-9 Sept.	51.15	33 964	-3.9	16.1	39.7	313.3	505
5	Noisy	Alcoy	38°41’54”N 0°28’25”W	4-6 May[1]	43.60	58 994	-2.6	16.3	39.0	460.2	562
	Non-noisy	Ibi	38°37’38”N 0°34’31”W	13-15 Sept.	24.71	23 489	-1.6	16.0	36.5	444.4	755

[1] Exact dates vary slightly every year

* Data from 2018

\- Data unavailable

Results

GLMM showed that the interaction between year and season of festivals was significant (Table 2; Fig. 2a). In 2019, the juveniles/adults ratio was significantly higher in the localities with festivals outside the breeding season than in those with festivals during the breeding season (Table 2). However, the ratios in both studied locality types (noisy and non-noisy) were similar in 2020 when festivals were cancelled (Fig. 2b; Table 2). The effect of the time of surveys was significant. In the second survey, 30 days after festivals, significantly more juveniles (higher juveniles/adults ratio) were found compared to the first one (15 days after festivals) (Fig. 2; Table 2). No interaction between survey time and any other variable was significant, which indicates that the survey time effect was similar in both years and for both towns with or without festivals during the breeding season. The ICCc of the nested random effects was 0.406, which suggests that 41% of variance in the juveniles/adults ratio could be attributed to particular differences between pairs and localities. Pearson correlations between PM2.5 and PM10 daily values were significant ($p < 0.001$) in all localities and higher than 0.9 in four of them, thus PM10 levels were a good proxy of PM2.5 values. The best model included only the intercept (0.003 ± 0.023 SE), which was nor significant ($p = 0.888$), showing that fireworks during festivals were not consistently associated to change in PM10 pollution. The type of festival did not have an effect on PM10 change during festivals ($p = 0.172$, $\Delta AICc = 1.143$). Between towns variability of juvenile productivity of house sparrows was similar in both groups of towns in both surveys (15 days Test statistic = 0.022, p = 0.882; 30 days Test statistic = 0.445, p = 0.505).

Table 2 The GLMM model fitted to evaluate the effect of powder-guns/fireworks of recreational festivals on the juvenile productivity of the house sparrow in the Valencia Region (SE Spain). Significant *p* values are marked by (*) = p < 0.05; (**) = p < 0.01; (***) = p < 0.001.

GLMM terms	**Estimate**	**SE**	***p***
Intercept	0.073	0.018	<0.001***
Fixed factors			
YEAR 2019	-0.051	0.012	<0.001***
FESTIVALS SEASON OUTSIDE	-0.012	0.014	0.395
VISIT 30 DAYS	0.054	0.012	<0.001***
Interaction of fixed factors			
YEAR 2019 * FESTIVALS SEASON OUTSIDE	0.046	0.017	0.008**
YEAR 2019 * VISIT 30 DAYS	-0.015	0.017	0.374
FESTIVALS SEASON OUTSIDE * VISIT 30 DAYS	0.014	0.017	0.424
YEAR 2019 * FESTIVALS SEASON OUTSIDE * VISIT 30 DAYS	0.010	0.025	0.672
Random factor (1 | PAIR/LOCALITY)	**Variance**	**Std. Dev.**	
LOCALITY:PAIR	0.00014	0.01165	
PAIR	0.00108	0.03267	
Residual	0.00037	0.01933	

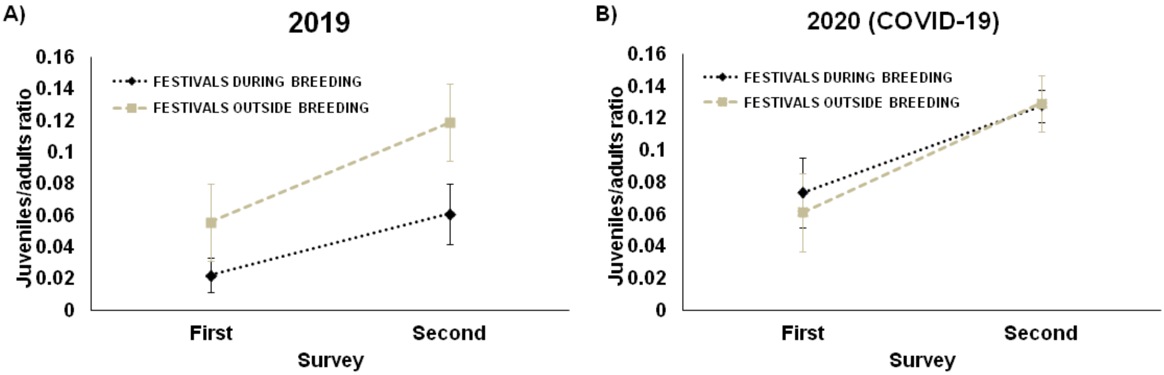


Fig. 2 The juveniles/adults ratio (±SE) in the studied localities. Surveys were conducted 15 (first survey) and 30 (second survey) days after festivals in the noisy-breeding localities and 1 day after of its noisy pair locality in the non-noisy breeding localities. Diamonds (festivals during breeding) denote the localities with noisy-breeding, while squares (festivals outside breeding) depict the control localities with the same festivals, but outside the breeding season. Note that in 2020, recreational noisy-festivals were only carried out in Bocairent and they were outside the breeding season (early February).

Discussion

Our results showed that the recreational noise pollution generated by the impulse-type noise of traditional festivals in spring is associated to reduced juvenile productivity, measured with the juveniles/adults ratio, of an urban avian bioindicator. In 2019, we found significant differences in the ratios in both sets of localities. The localities that held powder-guns/fireworks festivals during the breeding season reached significantly lower ratios in both surveys compared to the localities with the same festivals, but held outside the breeding season. Air pollution, measured through daily PM10, did not consistently increase during fireworks festivals in 2019 in a sample of Valencian localities. In 2020, when traditional festivals were cancelled due to COVID-19 restrictions, the juveniles/adults ratio of localities that held festivals in spring the previous year were similar to the control localities in both surveys. This 2020 result supports the pernicious effect of the studied recreational noise pollution on the species' juvenile productivity, and hence on its breeding success.

Peaks of air pollution due to fireworks last for just some hours (Seidel and Birnbaumb, 2015; Hoyos et al., 2020), while more prolonged exposures to air pollution are needed to produce pernicious effects on birds' health (Gorriz et al., 1994; Steyn and Maina, 2015). Our PM10 pollution analysis in Valencian localities with fireworks showed that levels of this pollutant during festivals are very variable between towns. In half of the analysed localities PM10 levels are even lower during festivals than the previous week. Part of this variability may be associated to meteorological conditions, mainly winds that may disperse air contaminants quickly (Fortelli et al., 2016; Hoyos et al., 2020). In addition, festivals entail a reduction of some industrial activities, which close for holidays, and traffic, since some streets are closed and part of the population spends these days out of town. This reduction of polluting activities might compensate or even overcompensate the pollution produced by fireworks through the "holiday effect" (Tan et al., 2009; Tan et al., 2013) and explain another part of the observed variability of PM10. If air pollution was determinant for the reduction in juvenile productivity, a larger variability of this variable could be expected in localities with festivals in spring, but in fact, variability of juvenile productivity was the same as in towns with festivals out of the breeding season. Taking all these factors into account, the effect of particulate matter pollution produced by spring fireworks on the reduction of house sparrow juvenile productivity should be weak and, probably, undetectable at our study scale. Thus, the observed reduction in productivity is better explained by the effect of impulse-type noise pollution. However, detailed physiological studies carried out with adults, fledglings, or nestlings before and after festivals could help to understand the potential effects of short-term air pollutants, such as PM2.5, derived from fireworks (Li et al., 2021).

More juveniles per adult were found in the second survey than in the first in both sets of localities and years. This result was expected given the species' phenology in the study area as the peak of clutches occurs around mid-April (Gil-Delgado et al., 1979; Anderson, 2006; Murgui, 2016). Consequently, the number of contacts with juveniles was expected to increase between mid-May and June, when the second survey was carried out (Peach et al., 2013). Some early-breeding pairs were expected to lay a second/third clutch, which contributed to the late appearance of juveniles (Gil-Delgado et al., 1979). However in 2019, the juveniles/adults ratio in the second survey of noisy localities (30 days after festivals) did not reach the value obtained for the control localities, which shows that recreational noise had a strong effect on populations that was detected even 1 month after festivals had ended.

This diminished breeding productivity can be derived from pernicious effects of this source of environmental pollution in four stages: 1) pairing of adults and nest construction; 2) egg laying, incubation and hatching; 3) nestlings; 4) fledglings. When considering the species' phenology in the study area and the festival dates (late-April to mid-May), most of the pairs that bred during festivals would be incubating eggs and a minority would be feeding small nestlings (Gil-Delgado et al., 1979; Alonso, 1984; Veiga, 1990; Murgui et al., 2016). We can expect nests with small nestlings at that time to produce the juveniles counted in the first survey, while the fledglings produced by nests containing eggs during festivals would be counted in the second survey. Therefore, if festival noise affected only nests in the incubation stage, we would expect the juvenile/adult ratio to lower only in the second survey. However, if noise affected only nests in the nestling stage, we would expect a decrease to take place only in the first survey. We found a similar drop in the juvenile/adult ratio for both surveys, which supports that this impulse-type noise pollution affects both nests with eggs or nestlings.

Several mechanisms have been proposed to explain how impulse-type noise could affect eggs. Sonic booms similar to fireworks/powder-guns were cited as causing eggs cracking (Austin et al., 1970) but this mechanism is considered unlikely by other studies (Cogger and Zegarra, 1980; Bowles et al., 1991; Ting and Garrelick, 2002). Parental behavioural changes in the incubation stage can be a more plausible cause of declining breeding productivity (Bowles, 1995). Thus recreational noise could have led to abandonment of nests, smaller clutches, less egg parental care or frequently interrupted incubation (Anderson, 2006; Halfwerk et al., 2011b; Schroeder et al., 2012; Meillère et al., 2015), which could worsen incubation conditions, alter developmental temperatures (e.g. eggs cooling), increase embryonic mortality, and reduce innate immunity and body condition of future nestlings (Ardia et al., 2010; Nord and Nilsson, 2011; DuRant et al., 2013; Potvin and MacDougall-Shackleton, 2015). A recent study linked anthropogenic noise with hatch failure and reduced fitness of several bird species across a whole continent (Senzaki et al., 2020). Owing to the high intensity and frequency of impulse noise during the festivals in our study area, all these effects could have synergistically acted. Nonetheless, we cannot determine the relative importance of each one.

Nestling survival may be negatively affected by noise-induced behavioural changes in adults or nestlings themselves. Adults might be frightened away to quieter areas or spend more time being alert while foraging. Hence they would need to invest more time and energy to provide the same amount of food for nestlings, which would weaken their provision efficiency and increase the young starvation risk (Leonard and Horn, 2012; Schroeder et al., 2012). Recreational noise could also reduce the fitness of the affected populations by acoustically masking parent-offspring communication (Schroeder et al., 2012). Furthermore, noise, like any other disturbance, could also affect nestlings' behaviour by prematurely advancing fledgling, increasing accidental nest-falling, or even involving the probability of being pushed out of nests by another chick (Anderson, 2006).

Recent fledglings are the last stage that could also be affected by noise because they still depend on parents. However due to species' phenology and festival dates, no fledglings were expected to be present in these localities during spring festivals. In other localities that organise festivals late in spring or early in summer, noisy disturbances could lead dependent fledglings to lose parental care, and thus it could increase their exposure to predation or starvation (Francis and Barber, 2013). They could also experience decreased immune response (Kight and Swaddle, 2011) and be forced to use unfavourable habitats (Blickley et al., 2012; Sordello et al., 2020).

Pernicious effects of traditional festival noise on breeding productivity are also plausible on other urban birds that nest in buildings, such as barn swallows (*Hirundo rustica*) and house martins (*Delichon urbicum*), or in open nests of parks and gardens, such as serins (*Serinus serinus*), blackbirds (*Turdus merula*) and greenfinches (*Chloris chloris*), but more research is needed (Kight et al., 2012; Patón et al., 2012).

Our study was also useful for studying an environmental-jolt (Meyer, 1982), the effect of the lockdown imposed by the Spanish government between March and June 2020, on the breeding productivity of an urban avian exploiter. We expected better breeding productivity due to limited road traffic, fewer pedestrians, better parent–offspring communication, increased communication distance, and decreased pollution (Schroeder et al., 2012; Derryberry et al., 2020; Rupani et al., 2020; Shilling and Waetgen, 2020). On the other hand, lockdown could limit their breeding productivity due to the reduced food availability linked to human activities (Anderson, 2006; Bernat-Ponce et al., 2018). However, the control localities, without spring festivals, showed very similar juveniles/adults ratios in 2019 without lockdown and in 2020 with lockdown. Hence we found no evidence that

COVID-19 had an effect on normal house sparrow breeding success. Similarly, Gordo et al. (2021) reported that the abundance of urban birds in Catalonia (NE Spain) did not increase during lockdown compared to previous years.

The final effects of COVID-19 crisis on birds and wildlife in general will likely depend on the duration of the special measures that all countries have adopted to limit the mobility of people and numerous economic activities. On the short-term, behavioral differences were expected and several studies have found better detectability, improved communication, migration changes and uncommon behaviours (e.g. rural species roaming in the city centres) during lockdown (Bar, 2020; Derryberry et al., 2020; Gordo et al., 2021). Changes in bird abundance might take longer to occur, and Gordo et al. (2021) did not found such an effect in Catalonia (NE Spain) and, in line with that study, we did not found any effect of lockdown on house sparrows average juvenile productivity. However, if the restrictions associated to COVID-19 pandemic are prolonged in time, their effects on urban bird abundance could be stronger and easier to detect. These changes in abundance might be both, negative due to the decrease of anthropogenic food (e.g. decrease of feral pigeons; Soh et al., 2021), or positive due to the reduction of human disturbances, sensory pollutants, such as artificial light (Bustamante-Calabria et al., 2021) and noise as well as other pollutants (e.g. Bar, 2020). It is likely that the sign or intensity of this change be distinct for different groups of species. We hypothesize that prolonged COVID-19 restrictions effects would be more pernicious for avian urban exploiters, such as house sparrows, which rely more on anthropogenic food sources than for avian urban adapters, which mostly feed on natural resources even though take advantage of urban habitats, such as blackbirds (Blair, 1996; Croci et al., 2008; Soh et al., 2021). On the other hand, both groups of species would be benefited by the reduction in noise and chemical pollutants. Therefore, future studies on the effects of COVID-19 restrictions on birds should address these issues and develop ways to disentangle the relative role of these factors for different species.

To our knowledge, this is the first time that the juvenile productivity of a bird species is studied in the field in relation to impulse noise pollution from recreational activities with fireworks and powder-guns. Fireworks have been used worldwide in traditional celebrations for centuries and presently they are still very important in recreational and cultural activities (Werrett, 2010). Even though they are object of admiration and leisure for humans, the effects of their environmental pollution on wildlife are still poorly known (Shamoun-Baranes et al., 2011; Andradottir and Thorsteinsson, 2019). Although present research is necessarily conditioned by the specific characteristics of the festivals and towns studied, the negative effects of the recreational activities with extreme noise pollution on urban wildlife herein described can also be plausible in similar recreational festivals worldwide. Therefore, more research is needed to address the demographic effects of recreational fireworks on wildlife in other regions and along the year.

Traditional festivals are a cultural heritage that cannot be eliminated merely for their negative impact on birds and/or other urban wildlife, but their impact should be evaluated and understood. Nowadays, urban birds are negatively affected by changes in the structure of cities and resources availability (Shaw et al., 2008; Isaksson, 2018; Bernat-Ponce et al., 2020). Thus the effects of such traditional festivals would be added to these other negative effects. They may be mitigated by improving conditions for birds in cities, especially by developing green urban policies and reducing other sources of environmental pollution, such as urban light, a sensory pollutant that increases the clutch failure in house sparrows (Senzaki et al., 2020). To address these issues, social and natural scientists must collaborate with politicians to focus on this environmental problem through interdisciplinarity, organizational theory, and team science (Palmer et al., 2016).

Our study extends scarce current knowledge about the effects of recreational, unpredictable and acute noise pollution on wildlife. However, more complex and profounder studies should be carried out to disentangle which specific suggested mechanisms might lie behind the observed declining breeding productivity. Studies on the long-term effect of urban noise on birds have been focused on chronic noise, which creates a new selection pressure on birds that use acoustic signals to achieve reproductive success. This seems to create two groups of species: one that can adapt their signals to the competing chronic noise singing at a higher pitch, and another that cannot (Slabbekoorn and Peet, 2003; Mockford and Marshall, 2009; Francis et al., 2011b). But, due to the unpredictability and short duration (few days) of fireworks or powder-guns recreational sounds, the development of adaptative behaviours (e.g. changing vocal frequency, nocturnal singing, habituation, predation risk compensation) to these sounds seems more unlikely than to chronic urban noise or loud but predictable sounds (Mott and Boyd, 1995; Quinn et al., 2006; Fuller et al., 2007; Gross et al., 2010; Francis et al., 2011b). Therefore, the mitigation of the effects of this recreational impulse-type noise should be better based in the development and implementation of policies relative to find suitable places to launch fireworks according to season or other practical aspects. These new urban policies should be developed based on further studies, ideally carried out with the collaboration of the firework industry, which should be concerned with this environmental problem.

Conclusions

Noisy spring festivals reduced the juvenile productivity of an avian urban bioindicator, the house sparrow. In 2019, juvenile productivity was lower in the towns with noisy traditional festivals during the breeding season than in the control towns with festivals out of the breeding season. Festivals noise seems to negatively affect both nests with eggs or nestlings. Juvenile productivity in towns with spring festivals rose to equal the productivity of control towns in 2020, when festivals were cancelled everywhere due to COVID-19. However, lockdown did not increase juvenile productivity of this species in the control localities in 2020, thus COVID-19 restrictions did not improved average house sparrow breeding success. Traditional festivals are a cultural heritage that cannot be eliminated but their effects on urban wildlife should be mitigated with urban green measures and the reduction of other pollutants.

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Supplementary Material

Fig. 1. Alardo's day in Alcoy (SE Spain) in spring while firing powder-guns during traditional festivities.

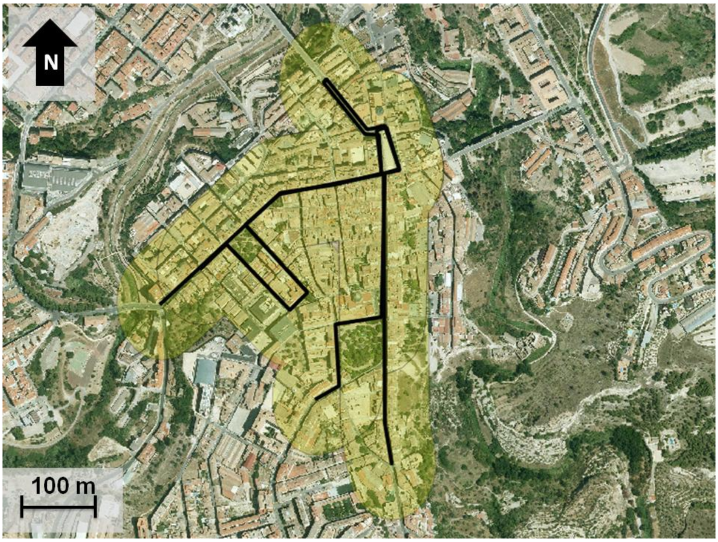

Fig. 2. Example of the noisy-influence area (shaded in yellow) of the Moors and Christians festivals in a studied locality (Alcoy). The black line denotes the festivals route.

Table 1. Relative growth rate of PM10 due to fireworks festivals, measured as log(festivals mean/week before festivals mean) in 15 localities of the Valencia Region. M&C is referred to Moors and Christians festivals while Others include Fallas, Magdalena, Mascletà and Hogueras.

Locality	Festival type	Mean of daily PM10 (μg/m^3) values during pre-festivals week	Mean of daily PM10 (μg/m^3) values during festivals	Relative growth rate
Alicante	Others	21.57	23.20	0.032
Albalat del Tarongers	Others	8.86	7.67	-0.063
Alcoy	M&C	13.43	14.00	0.018
Alzira	Others	18.57	17.25	-0.032
Burjassot	Others	31.00	33.00	0.027
Burriana	Others	6.71	8.25	0.089
Castellón	Others	27.29	30.56	0.049
Caudete de las Fuentes	M&C	9.00	6.20	-0.162
Elda	M&C	12.29	18.20	0.171
Elche	M&C	31.86	31.20	-0.009
Gandía	Others	19.25	22.20	0.062
Paterna	Others	19.40	19.50	0.002
Quart de Poblet	M&C	19.14	15.33	-0.096
Sagunto	M&C	16.00	12.00	-0.125
Valencia	Others	29.14	35.50	0.086

Table 2. Raw counts in 2019 and 2020 breeding seasons of individual males, females, adults, and juveniles of House Sparrows and number of contacts with juveniles (fledglings in the same family flock were considered to be only one contact) in each studied locality and survey. Breeding category: noisy includes localities with festivals during the breeding season; non-noisy are localities with festivals out of the breeding season). The ratio Juveniles/Adults ratio (see Material and Methods) was the dependent variable to study the effect of recreational noise on the juveniles.

Pair	Locality	Breeding category	Survey	Males		Females		Adults		Juveniles		Juveniles contacts		Juv/Ad ratio	
				2019	2020	2019	2020	2019	2020	2019	2020	2019	2020	2019	2020
1	Banyeres de Mariola	Noisy	1	30	40	19	16	49	56	1	0	1	0	0.0204	0
			2	36	42	17	21	53	63	3	11	3	8	0.0566	0.1270
1	Bocairent	Non-noisy	1	38	42	13	17	51	59	1	0	1	0	0.0196	0
			2	41	36	12	13	53	49	9	7	5	5	0.0943	0.1020
2	Onil	Noisy	1	97	95	33	23	130	118	0	9	0	6	0	0.0509
			2	120	81	38	34	158	115	5	23	4	13	0.0253	0.1130
2	Castalla	Non-noisy	1	100	112	38	33	138	145	0	9	0	5	0	0.0345
			2	96	90	38	36	134	126	16	18	10	13	0.0746	0.1032
3	Muro de Alcoy	Noisy	1	67	49	26	22	93	71	6	12	6	8	0.0645	0.1127
			2	54	48	16	17	70	65	12	13	9	10	0.1286	0.1539
3	Cocentaina	Non-noisy	1	51	45	14	14	65	59	14	15	6	7	0.0923	0.1187
			2	47	37	14	23	61	60	21	19	11	11	0.1803	0.1833
4	Petrer	Noisy	1	52	40	22	18	74	68	1	9	1	8	0.0135	0.1177
			2	50	46	21	16	71	62	10	17	5	9	0.0704	0.1452
4	Villena	Non-noisy	1	86	105	28	21	114	126	21	21	15	15	0.1316	0.1191
			2	79	92	25	35	104	127	28	36	18	20	0.1731	0.1575
5	Alcoy	Noisy	1	58	54	25	17	83	71	1	8	1	6	0.0120	0.0845
			2	63	60	23	12	86	72	2	14	2	7	0.0233	0.0972
5	Ibi	Non-noisy	1	45	45	13	16	58	61	2	2	2	2	0.0345	0.0327
			2	46	57	11	14	57	71	6	9	4	7	0.0702	0.0986

Chapter 8

General discussion

The results of the present thesis show that Valencian Community towns have become challenging environments for the urban House Sparrow. Even though this species was and is still widely distributed in this region (Gil-Delgado & Mora 2015, Bernat-Ponce 2021), Valencian urban areas have changed in the last decades through new urbanization trends that alter sparrows habitat with pernicious consequences for their populations (Vera & Escandell 2016). The first part of the thesis, consisting of Chapters 2, 3, 4, and 5, explored the main habitat of the species (Bernat-Ponce et al. *in press*), those local factors that determine its prevalence, and the urban structure changes that are altering the species habitat in the Valencian localities (Bernat-Ponce et al. 2018, 2020, 2021a). Furthermore, the second part of the thesis, consisting of Chapters 6 and 7, explored two environmental elements, anthropogenic processed food and noise pollution (Chapter 6, Bernat-Ponce et al. 2021b/Chapter 7), which could also contribute to the decline of the species in this Eastern region of the Iberian Peninsula (Vera & Escandell 2016). The results of this thesis mainly encompass the "lack of food" hypothesis to explain the decline of the species (Chapter 1, Summers-Smith 2003, MacLeod et al. 2006, Shaw et al. 2008), but their interrelation will be widely discussed onwards.

First, Bernat-Ponce et al. (*in press*/Chapter 2) and Bernat-Ponce et al. (2018/Chapter 3), deal with the habitat features of the main ecosystem where House Sparrows thrive, the urban areas. In Chapter 2, Bernat-Ponce et al. (*in press*) reviewed those features of the contemporary cities that have affected the urban avifauna from prehistory to the present. First human permanent settlements were linked to the development of agriculture 10,000 years ago in different areas of the planet (Harari 2014). During this period the incipient pre-urban areas started the present link between birds and cities (Negro et al. 2020). Since then, cities have attracted birds for centuries (Bernat-Ponce et al. *in press*/Chapter 2). Furthermore, modelling the historical urbanization process of bird species in the early urban areas has resulted very simplistic (Negro et al. 2020): the current paradigm is that hole nesting species became urban as soon as they found cavities in buildings and there was food for them in the city itself or the surroundings (Summers-Smith 1988).

The House Sparrow, as an urban exploiter, thrives with urbanization (Croci et al. 2008, Bernat-Ponce et al. *in press*/Chapter 2). Furthermore, it became one of the best species to describe the urbanization process of birds as it is adapted and pre-adjusted to exploit anthropic ecosystems while is practically absent in other environments (Summers-Smith 1988, Negro et al. 2020). However, for several reasons, cities are becoming less attractive for the birds during the last decades (Chapter 1, Bernat-Ponce et al. *in press*/Chapter 2). Current urban changes are so intense and quick that some urban avian species are not able to adapt and even several ones are declining, such as the House Sparrow (Chapter 1, De Laet & Summers-Smith 2007). The current paradigm of the urbanization process of the species (cavities in buildings and food availability in the city) has been altered during the last decades. So much so that it is almost extinct in

London, Glasgow, and Ghent (Summers-Smith 2003) or it is sharply declining in Paris and Valencia (Murgui & Macias 2010, Mohring et al. 2021). However, as stated in Chapter 1, there is not a clear answer yet to explain this generalized decline of populations in urban areas of the Western Palearctic while different factors might be occurring in varied geographical areas or cities typologies.

Chapter 3 (Bernat-Ponce et al. 2018) explored those main urban factors that enhance or limit the abundance of the focal species in the study area where populations are declining (Vera & Escandell 2016). The thesis required this regional knowledge as the urban structure between Spain and other European countries are different and their urban changes might disparately affect the species. Furthermore, this information would set the first steps to further development of fine-scale research of these relevant urban elements regarding House Sparrows abundance. As expected by previous studies, House Sparrows were more abundant in urban parks than in other elements of the urban matrix as these public green areas are essential foraging habitats for the species throughout the year (Murgui 2009, Murgui & Macias 2010). However, private gardens in Great Britain were the main feeding habitats for House Sparrows (Chamberlain et al. 2007, Shaw et al. 2011). For that reason, sparrows in the study area of the thesis would be strongly dependent on urban parks while changes in their management might have pernicious effects on local populations that could contribute to the decline of the species. Therefore, fine-scale research was needed to understand the potential role of potential new green areas management trends on the species.

In line with Chapter 3 results (Bernat-Ponce et al. 2018), Chapter 4 focused on exploring the new management and planning of green urban areas carried out during the thesis in some Valencian towns (Bernat-Ponce et al. 2020). Several urban parks of the study area were remodelled replacing natural or semi-natural substrates (grass or soil) with impervious surfaces (artificial grass, concrete, or rubber) aiming to save water and reduce management costs. These remodelations may make green urban habitats less suitable for urban biodiversity due to drastic reductions in the availability of natural food resources (Verbeeck et al. 2011, Warhurst et al. 2014, Bernat-Ponce et al. 2020/Chapter 4). The fine-scale research carried out in the Chapter 4 of the thesis showed that in short term (less than 5 years), abundance indexes of House Sparrows at 10 remodelled parks were reduced by 60% while in 22 non-remodelled parks these indexes only decreased by 15% (Bernat-Ponce et al. 2020/Chapter 4). This study describes for the first time the significant negative effects of artificial park remodelling on an urban bioindicator bird species, probably due to drastic reductions of natural trophic resources. The results of this chapter (Bernat-Ponce et al. 2020) are in close accordance with one of the main suggested reasons that explain the global decline of the House Sparrow in urban areas, the lack of food hypothesis (Robinson et al. 2005, Vincent 2005, Anderson 2006, MacLeod et al. 2006, Shaw et al. 2008). Besides, it sheds more light on an additional unreported problem for the species, green urban areas remodelling (Figure 1).

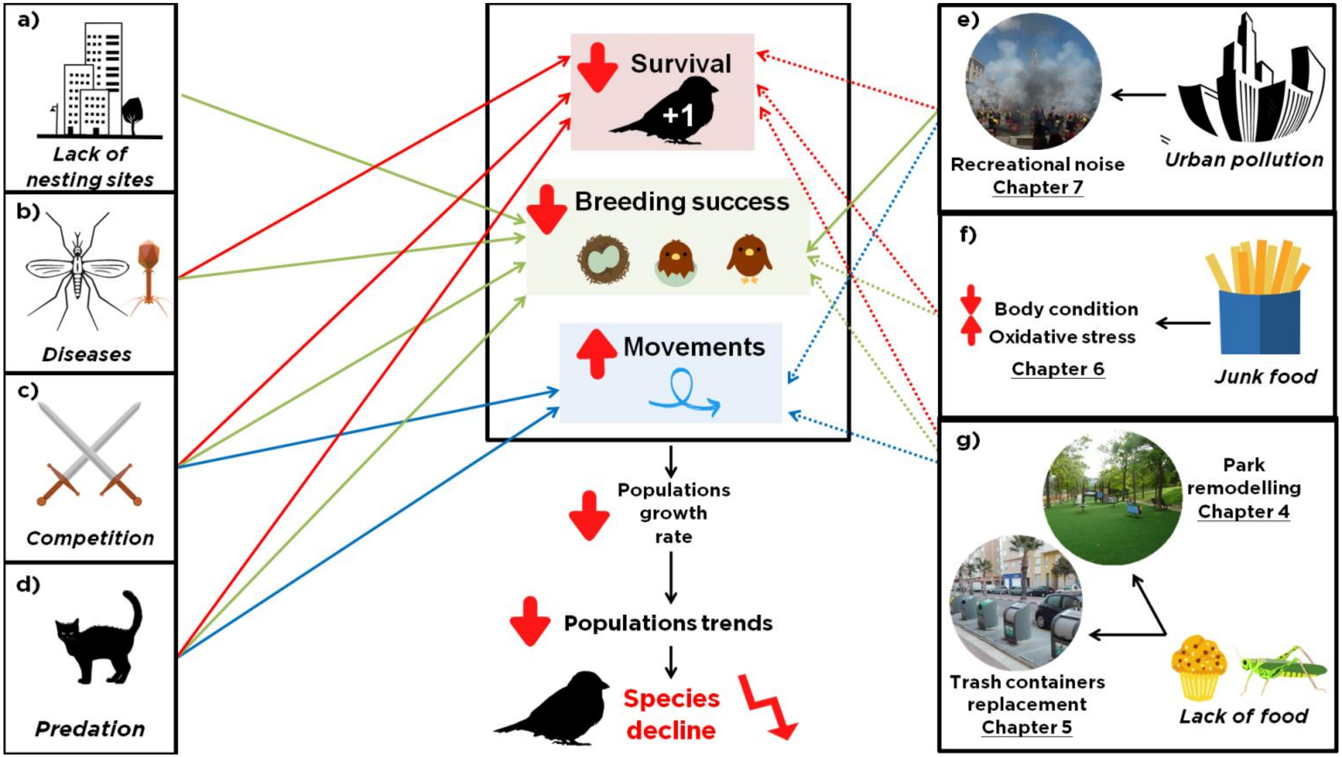

Figure 1. Summary of the main suggested hypotheses and some of their potential effects that would explain the decline of the House Sparrow (Chapter 1). The hypotheses of the right side (e, f, g) have been completed with Chapter 4, 5, 6, and 7 findings. Their potential (dashed lines) and proved (solid lines) effects on the species (survival, breeding success, and movements) are shown.

Chapter 3 (Bernat-Ponce et al. 2018) also found that House Sparrows in the Valencian Region were more abundant around surface trash containers, especially in winter when natural food is scarcer. These elements of the urban matrix offer a predictable and continuous supply of anthropogenic food resources around them (Bokotey & Gorban 2005). However, the increasing European trend of replacing surface trash containers with underground containers (Nilsson 2011, ISWA 2013) could diminish trophic resources availability which could be reflected in reduced abundance indexes of sparrows. Chapter 5 of the thesis (Bernat-Ponce et al. 2021a), focused on a second fine-scale study about the change in urban waste management policies, showed that the replacement would be pernicious for urban populations of House Sparrows (Figure 1). Underground trash containers presented lower abundances of the species when compared with the traditional surface trash containers, probably due to a lack of food in their surroundings (Bernat-Ponce et al. 2021a).

The regional (Bernat-Ponce et al. 2018/Chapter 3) and the two fine-scale studies (Bernat-Ponce et al. 2020, 2021a/Chapters 4 and 5) of the thesis show that urban structure changes and new urban planning measures are contributing to the lack of food resources for the species in cities (Chapter 1, Figure 1). The results of these three chapters link these trophic resources availability reduction in urban areas with plummeting recorded House Sparrows abundance indexes but with unexplored effects on already declining sparrow's trends (Figure 1, Vera & Escandell 2016).

The second part of the thesis focused on two environmental factors, anthropogenic processed food (Chapter 6) and noise pollution (Chapter 7), that might have pernicious effects on local populations of the species. House Sparrows are strongly dependent of anthropogenic food resources (Anderson 2006, Herrera-Dueñas et al. 2015) and this thesis highlights two scarcely studied urban elements that are key suppliers of low-quality processed food: trash containers and bar terraces (Bernat-Ponce et al. 2018, 2021a/Chapters 3 and 5). Furthermore, cities are polluted areas (Herrera-Dueñas et al. 2017, Bernat-Ponce et al. *in press*) and the Chapter 3 (Bernat-Ponce et al. 2018) described that House Sparrows were less abundant in urban areas with heavy traffic and pollution (Peach et al. 2018). Furthermore, urban House Sparrows present higher levels of oxidative stress when compared to rural conspecifics (Herrera-Dueñas et al. 2017). But no clues are available about if this oxidative stress might be mainly due to urban pollution from traffic (Bernat-Ponce et al. 2018/Chapter 3, Peach et al. 2018) or to low-quality urban diet found in bar terraces or trash containers (Bernat-Ponce et al. 2018, 2021a/Chapters 3 and 5). Chapter 6 of the thesis showed that the general condition and the oxidative stress of the rural individuals reared from eggs to adults in a wild "healthy" rural environment can be profoundly altered if urban food constraints come into force and are isolated from pollution. A synergic effect of urban low-quality diet and pollution might be boosting the general decline of House Sparrows in European cities. However, the integration of the Chapter 6 results with the findings of the Chapters 3 and

5 of this thesis are a promising start to disentangle the importance of the urban diet role on the generalized decline of this urban bioindicator.

The second environmental factor analysed in the thesis (Chapter 7) was urban noise. In European cities, the anthropogenic noise (e.g. traffic) is one of the major urban pollutants (Bernat-Ponce et al. *in press*/Chapter 2) and its effects on birds have been widely studied, especially regarding avian communication (Halfwerk et al. 2011, Leonard & Horn 2012, Brumm & Zollinger 2013, Gil & Brumm 2014). Therefore, and given that may be expected that urban House Sparrows are, in some way, adapted to chronic urban noise (Angelier et al. 2016), Chapter 7 focused on the effects of intermittent and unpredictable noise on the breeding performance of the species. In the Valencian Community, there are plenty of traditional noisy festivals characterized by intense fireworks and powder guns activities, such as the Moors and Christians parade (Catalá-Pérez 2012, Lloret 2017, Maeso et al. 2018), whose effects on wildlife are unknown. Recreational noise from these traditional festivals was able to reduce the juvenile productivity of the studied species, negatively affecting both nests with eggs or nestlings, which were reflected in lower juveniles/adults ratios in localities with festivals in spring (Bernat-Ponce et al. 2021b/Chapter 7). Potential explanations to the findings have been widely described in Bernat-Ponce et al. (2021b/Chapter 7). Even though lockdown changes could have been positive for the species due to the reduction of human disturbances (Bar 2020, Bustamante-Calabria et al. 2021), it was not reflected in the study (Bernat-Ponce et al. 2021b/Chapter 7). This lockdown scenario also meant a reduction of anthropogenic food availability on the streets due to a drastic reduction in outdoor human activities (Soh et al. 2021). Thus, for an urban exploiter species highly-dependent on these resources (e.g. bar terraces) in the Valencia Region (Bernat-Ponce et al. 2018, 2021a/Chapters 3 and 5) this might have compensated the positive effects of pollutants reduction. It is important to highlight that the proven negative effects of recreational noise could even be added to those derived from urban changes and new planning measures, adding an extra problem to the already existing ones with unknown effects on the species trend (Figure 1).

The thesis findings present a series of limitations. Most findings were related to local abundance indexes (Chapters 3, 5, and 7) and not to population trends (only Chapter 4, short-term local populations' trends) which could limit the extrapolations to their real effects on the species decline (Figure 1). Also, the studied urban changes might not be occurring in every locality of the Valencian Community, Spain, or other countries, limiting their applicative utility. Furthermore, individual urban changes studied in the thesis, such as park remodelling or replacement of surface trash containers, might not be independent of other urban changes that have not been studied in this work, such as new buildings trends. Several combinations of urban changes might affect differently to the abundance indexes and the species trend. However, the thesis individual findings were sufficiently clear to expect similar outcomes in other locations if these studied urban changes would occur. It is also important to highlight that the wide variety and heterogeneity of anthropogenic food in cities, especially for bar diet, could not be fully represented in the diets used in

Chapter 6 (MacGregor-Fors et al. 2020, Andrzejczak et al. 2021). Furthermore, anthropogenic diets may present different compositions, or even varied geographical, seasonal, and daily patterns in different countries. Finally, regarding the last chapter of the thesis, it is remarkable that the House Sparrow is also declining in localities without noisy festivals during spring (Vera & Escandell 2016). Besides, these recreational festivals have been celebrated for more than 100 years in some localities when sparrow populations were not declining. For that reason, the recreational noise can be another stick in the wheel but not a key factor for its global decline (Figure 1).

To further develop and understand the potential effects of the thesis findings on the decline of the species, new complete studies and projects should be carried out. First, to understand the real effects of urban parks remodelling and surface trash containers replacement on the species mid-term trend; survival rate and breeding success should be compared with control populations with traditional management of parks and urban waste systems. This would allow knowing if these urban structure changes, besides affecting local abundance indexes, are able to alter population trends and boost the decline in affected localities. However, this research would need to take into account other urban changes that might mask the real effects of parks remodelling and trash containers replacements. For example, changes in nesting sites availability in the localities during the study period should be controlled to obtain reliable results in these suggested studies. Furthermore, some localities might present several urban changes at the same time and it should also be taken into account. But it is important to highlight that the thesis findings can also be extrapolated to develop further studies in other bird species. For example, the reduction of trophic resources due to park remodelling found in Chapter 4 (Bernat-Ponce et al. 2020) might be affecting other bird species, such as insectivorous ones, and more detailed research would be required in the future. Furthermore, to disentangle the role of urban diet on the decline of the House Sparrow more captivity studies should be carried out. Some research suggestions would include additional and more varied diets but also offering healthy food in polluted environments and experiments in lab conditions. Finally, for a better understanding of the effects of intermittent and loud recreational noises on the juvenile productivity of the species, more research is needed. For example, they should be focused on figuring out the specific behavioural (e.g. parental neglecting, premature fledgling, parent-offspring communication) and physiological causes that explain the juvenile productivity reduction (e.g. eggs cooling, increased embryonic mortality, reduced innate immunity) through fieldwork and lab research (Halfwerk et al. 2011, Leonard & Horn 2012, Brumm & Zollinger 2013, Gil & Brumm 2014).

Despite the aforementioned limitations of the study, this thesis sheds light on some novelty factors and unreported urban changes that alter the urban habitat of a declining bioindicator species. This thesis develops more deeply the hypothesis of the lack of food and links it with reduced abundance indexes of the species (Figure 1). These urban structure changes can be lowering their survival, breeding success, or even increasing their routine movements, which implies assuming higher risks

(Figure 1). Furthermore, the additional factors explored in the thesis, such as recreational noise and urban diet, but also those reported by other authors (Chapter 1), such as urban pollution, predation, competition, diseases, or lack of nesting places, can also play a role and all together derive in reduced population growth rates (Figure 1). This situation could be reflected in the negative regional trends reported by Vera and Escandell (2016) (Figure 1). However, these trends were not addressed in this work. What seems clear is that the decline of the House Sparrow in the Valencian Community is not probably attributable to one and only factor but to a synergy of them that, to a greater or lesser extent, significantly affect the local abundances, populations, and finally their trends (Figure 1, Anderson 2006). However, it should not be overlooked that what might be a decline key factor in a specific region might not weigh the same in other regions of Spain or the Western Palearctic.

Applications of the thesis findings for urban management friendly with the House Sparrow

Even though the thesis was focused on a specific study region, its findings will be useful to set up some urban management measures for the conservation of the House Sparrow throughout almost all over urban areas, especially in the Mediterranean region. However, it is important to highlight that the applicability of these measures may be limited due to economic, political, or urban limitations. Nevertheless, these management policies and measures should induce urban planners to design new models of cities that are respectful with our feathered neighbors and urban biodiversity in general, which undoubtedly will also benefit citizens' health. The following section includes a list of general and specific urban planning measures derived from the thesis findings that are considered essential for the House Sparrow conservation.

General measures for the management of urban areas:

-Given the importance of urban green areas for feeding and sheltering the species, localities should prioritize maintaining and preserving these pre-existing traditional green areas (natural food suppliers) in the cities.

-New green urban policies and laws should be developed to ensure the availability of natural food resources and shelter for the House Sparrow. These measures are essential as this thesis shows a strong link of the species by anthropogenic unhealthy food resources proceeding from trash containers and bar terraces, with poorly understood effects on their physiology and populations trends.

-Localities should create an interconnected urban green web in the city matrix to ensure efficient connectivity between parks to allow them urban birds to disperse, feed, shelter, and move when needed.

-Given the reduced abundance indexes of the species in the most crowded streets (e.g high buildings, traffic), localities should reduce road capacity for traffic and/or pedestrianize city centres to control air pollution and limit disturbances.

-Establish mandatory compensatory measures, such as increasing green areas or providing bird feeders by the municipalities, for already carried out pernicious urban plans that reduce food availability for birds.

-Urban public plans, remodelations, and urban structure changes, should be first assessed by environmental studies to limit and/or counter their pernicious effects on urban birds.

Specific measures for the management of urban green areas:

-Reduce the use of artificial substrates in green areas.

-Increasing the size, number, connectivity, and heterogeneity of green urban areas to work as green step-stones.

-Delimiting specific areas of urban parks for dogs to reduce disturbances for birds.

-Let native ruderal plants colonize parks and allow wild-growing of grass/weeds with patchy mowing to ensure green areas heterogeneity.

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Main conclusions of the thesis

Chapter 2. Effects of the features of contemporary occidental cities on urban avifauna

1. In the last decades, urbanization has grown exponentially on the planet and it will be one of the main threats to bird conservation.

2. Current occidental cities attract birds mainly due to the presence of abundant and predictable trophic resources, reduction of predators' diversity, or the existence of nesting places, among other factors.

3. The model of a contemporary city may become unattractive to birds and indeed some urban species are declining in the present due to: loss of green areas, pollution, reduction of natural trophic resources, new predators/competitors, and new building trends.

Chapter 3. Factors affecting the abundance of House Sparrows *Passer domesticus* in urban areas of southeast of Spain

4. The main causes of urban House Sparrow decline in southeast Spain may be related to urban land use and infrastructure changes.

5. House Sparrows were most abundant in parks and urban areas where rubbish bins were available as their foraging sites. The presence of rubbish bins, park area, and proximity to other parks favoured their abundance in these urban green areas.

6. The development of high-rise buildings, high human population density, and road traffic negatively impacted House Sparrow abundance.

Chapter 4. Replacement of semi-natural cover with artificial substrates in urban parks causes a decline of house sparrows *Passer domesticus* in Mediterranean towns

7. Currently urbanisation processes include urban park remodelling, which involves the replacement of semi-natural substrates (natural grass, bare soil) with artificial and impervious ones (pavement, concrete, areas for dogs, artificial grass).

8. House Sparrow populations in 22 non-remodelled parks declined by about 15% between 2015 and 2018, while the populations in the 10 remodelled parks declined by more than 60% during the same period.

9. Park remodelling might be linked to a reduction in both habitat suitability and availability of trophic resources, which could harm urban House Sparrows populations and possibly other species as well.

Chapter 5. Effect of replacing surface with underground rubbish containers on urban House Sparrows *Passer domesticus*

10. The link of the House Sparrow with anthropic human food is still important in modern urbanised areas and is patent in surface trash containers.

11. House Sparrows were more abundant close to surface than to underground rubbish containers. The presence of other urban features (bar terraces, private gardens, mature trees) interacting with the rubbish containers also influenced the abundance of House Sparrows.

12. The replacement of above-ground rubbish containers with underground ones may deprive House Sparrows' resources, which could lead to the decline of this species, especially in urban areas with little green cover.

Chapter 6. Experimental effect of anthropogenic food resources available in cities on the body condition, nutritional status and oxidative stress of an urban bioindicator passerine

13. The urban diet type eaten by House Sparrows affects their general nutritional condition and oxidative stress in a rural outdoor environment free of urban pollutants

14. Unbalanced urban diets (nutritionally poor) can worsen the physical condition, by lowering the general body condition and haematocrit, and alter the nutritional parameters linked with malnutrition (low levels of triglycerides, uric acid, total proteins, glucose), especially in females. A high-protein unbalanced urban diet increased both ROS levels (d-ROMs) and the plasma antioxidant barrier (OXY, albumin, uric acid), but not enough to avoid oxidative stress.

15. A synergic effect of urban low-quality diet and pollution might boost general House Sparrow decline in European cities, and the results

presented herein are a promising start to disentangle the importance of the urban diet role on the decline of this urban bioindicator.

Chapter 7. Recreational noise pollution of traditional festivals reduces the juvenile productivity of an avian urban bioindicator

16. Noisy spring festivals reduced the juvenile productivity of an avian urban bioindicator, the house sparrow.

17. In 2019, juvenile productivity (juveniles/adults ratio) was lower in the towns with noisy traditional festivals during the breeding season than in the control towns with festivals out of the breeding season. Festivals noise seems to negatively affect both nests with eggs or nestlings.

18. Juvenile productivity in towns with spring festivals rose to equal the productivity of control towns in 2020, when festivals were cancelled everywhere due to COVID-19. However, lockdown did not increase juvenile productivity of this species in the control localities in 2020, thus COVID-19 restrictions did not improve average House Sparrow breeding success.

Resumen global ampliado en castellano

1. Introducción

1.1. El gorrión común, nuestro vecino emplumado

El gorrión común *Passer domesticus* (Linnaeus, 1758) es un ave marrón de pequeño tamaño y una de las especies más reconocibles en todo el mundo. Además, ha estado ligada a los asentamientos humanos desde la Edad de Bronce (Ericson et al. 1997). Originaria del Medio Este, el desarrollo de la agricultura hace 10.000 años, desencadenó su expansión por comensalismo a la mayoría de de Eurasia y norte de África (Anderson 2006).

Hoy en día, el gorrión común presenta una amplia distribución por casi en todo el mundo, siempre estrechamente vinculado a la presencia humana, tanto en áreas rurales como urbanas (Summers-Smith 1988, Cramp y Perrins 1994, BirdLife International 2021). Este amplio rango de distribución es una consecuencia de procesos antropogénicos, como la liberación intencionda de individuos en áreas no nativas (Summers-Smith 1988, Cramp y Perrins 1994). El gorrión común ha sido considerado tradicionalmente como un modelo de especie explotadora urbana, favorecida por la urbanización y experta en obtener beneficios de estos cambios, y es sitios antropizados donde alcanza sus densidades más altas (Blair 1996, McKinney 2002, Anderson 2006, Kark et al. 2007, Seress y Liker 2015). Pero, a pesar de que el gorrión común está ampliamente distribuido y es abundante en todo el mundo, sus poblaciones han experimentado fuertes disminuciones durante el último siglo y en las primeras décadas del actual (BirdLife International 2021).

1.1. El declive del gorrión común

El declive del gorrión común se detectó por primera vez en Gran Bretaña en la década de 1920, particularmente en áreas urbanizadas. Este hecho coincidió con la sustitución del caballo como fuerza motriz por el uso del motor de combustión interna (Summers-Smith 2003). Durante las últimas décadas del siglo XX y principios del XXI, el gorrión común ha experimentado fuertes declives en áreas rurales y urbanas en todo su área nativa, el Paleártico Occidental (Gil-Delgado et al. 2002, Summers-Smith 2003, Chamberlain et al. 2005, Anderson 2006, Shaw et al. 2008, Dadam et al. 2019, Mohring et al. 2021). Desde 1980 hasta 2017, la especie disminuyó un 63% en toda Europa (EBCC 2019). Aunque en España el descenso de la especie no es tan marcado en comparación con otros países, se encuentran un 10% menos de gorriones en 2019 que en 1998 (SEO/BirdLife 2020). Sin embargo, en el este del país, concretamente en la Comunitat Valenciana, las poblaciones han disminuido más de un 20% entre 1998 y 2015 (Vera y Escandell 2016). Por estos motivos, esta región es una de las mejores zonas para estudiar el declive del gorrión común en España.

A pesar de que el declive del gorrión común se ha producido tanto en áreas urbanas como en zonas rurales, fue en estas últimas donde se detectó por primera vez (Summers-Smith 2000, Crick et al. 2002, Summers-Smith 2003, Robinson et al. 2005, De Laet y Summers-Smith 2007). Los cambios en las prácticas agrícolas y su intensificación parecen estar detrás del declive global detectado en áreas rurales y agrícolas (Gil-Delgado et al. 2002, Hole et al. 2002, Anderson 2006). Este conjunto de cambios ha reducido la disponibilidad de recursos tróficos, tanto de semillas, que sustentan a las aves durante todo el año, como de invertebrados, que son requeridos para alimentar a los pollos y juveniles en primavera (Summers-Smith 1988, Anderson 2006).

Sin embargo, no existe un consenso claro sobre una sola causa o grupo de causas que puedan explicar las disminuciones reportadas en áreas urbanas. Estos declives han sido de tal magnitud que incluso se ha llegado a la práctica extinción del gorrión común en algunos centros de ciudades europeas importantes (Summers-Smith 2003, Anderson 2006). Para explicar estas fuertes tendencias negativas se han expuesto varias hipótesis como: aumento de la depredación (Woods et al. 2003, Chamberlain et al. 2009, Bell et al. 2010), aumento de la competencia

(Skórka et al. 2016), pérdida de sitios de nidificación (Wotton et al. 2002, Moudrá et al. 2018), la reducción de áreas verdes y falta de alimento (McKinney 2002, Vincent 2005, Shaw et al. 2008, Peach et al. 2015), enfermedades (Anderson 2006, Dadam et al. 2019), dieta urbana de baja calidad (Herrera-Dueñas et al. 2015), contaminación urbana (Herrera-Dueñas et al. 2017, Peach et al. 2018) y nuevos estresores como la luz, el ruido y la radiación (Balmori y Hallberg 2007, Schroeder et al. 2012, Meillère et al. 2015).

Cada década da lugar a nuevas contribuciones para aclarar por qué los gorriones han disminuido tan drásticamente en el Paleártico Occidental, su área nativa. Sin embargo, parece improbable que una de las causas anteriormente mencionadas, por sí misma, pueda ser la única responsable de que las poblaciones caigan en picado de esa forma. Hoy en día, la hipótesis más aceptada es que el declive global de la especie podría deberse a una combinación de factores, algunos de ellos con más relevancia que otros (según la zoan geográfica), que pueden actuar sinérgicamente en contra de la conservación de este explotador urbano.

1.3 El gorrión común como bioindicador de salud urbana

El gorrión común es un ave extremadamente sedentaria que presenta un área de campeo urbana reducida y con muy baja plasticidad (Vangestel et al. 2010). Las especies de aves silvestres son sensibles a los contaminantes y los cambios rápidos en sus hábitats, lo que las convierte en componentes estructurales importantes del ecosistema (Swaileh y Sansur 2006, Kekkonen et al. 2011, Berglund et al. 2014). Por lo tanto, el estado físico del gorrión común, la condición nutricional, la abundancia, los parámetros reproductores y sus movimientos son un buen reflejo de la calidad del hábitat donde vive y lo convierte en un bioindicador útil para determinar la salud de las zonas urbanas.

1.4 Justificación y objetivos de la tesis

El principal objetivo de esta tesis es arrojar más luz sobre los factores potenciales que explican que una de las aves urbanas más vinculadas al ser humano, el gorrión común, esté disminuyendo drásticamente en las zonas urbanas de la Comunidad Valenciana (este de España). La presente tesis también pretende sugerir medidas paisajísticas y urbanísticas útiles para la conservación de la especie en zonas urbanas del este de España. No obstante, también se pretende que el conocimiento generado sea aplicable a otras partes de España y Europa. Este proyecto de investigación cubre el estudio de la especie desde una perspectiva ecológica integradora que incluye la selección de hábitat, fisiología nutricional, estrés oxidativo, ecología de la conservación, éxito reproductivo y dinámica de poblaciones.

Esta tesis se distribuye en 8 capítulos. El Capítulo 1 es una visión general introductoria a la biología de la especie y a su situación actual en Europa, España y la Comunidad Valenciana, el área geográfica donde se desarrolló la tesis doctoral. En la introducción se explora en detalle el declive de la especie, especialmente el referente de las áreas urbanas, y sus principales causas potenciales encontradas en la literatura. El Capítulo 2 también es una sección introductoria centrada en describir los efectos de las características de las ciudades contemporáneas en la avifauna urbana desde una visión más holística y general. El capítulo 3 tuvo como objetivo detectar cuáles son los factores urbanos determinantes de la abundancia del gorrión común en la Comunidad Valenciana. En el Capítulo 4 se exploraron los efectos de la reurbanización de los parques urbanos sobre el gorrión común. En el Capítulo 5, se investigó sobre los efectos potenciales de reemplazar los contenedores de basura de la superficie por otros subterráneos, situación que podría reducir los recursos alimenticios antropogénicos y bajar los índices de abundancia de la especie. En el Capítulo 6 se llevó a cabo un experimento en cautividad para estudiar si el mayor estrés oxidativo que se encuentra en los individuos urbanos podría deberse a la dieta urbana (alimentos procesados) o a los contaminantes de la ciudad. El Capítulo 7 se centró en la exploración del efecto potencial del ruido recreativo urbano (festividades tradicionales con fuegos artificiales) en la reproducción de aves urbanas utilizando el gorrión común como modelo. El Capítulo 8 recoge y discute la información de toda la tesis y también incluye medidas de ordenación y gestión urbanística para frenar el declive de esta especie bioindicadora en la Comunidad Valenciana. Asimismo, también incluye las conclusiones principales de cada uno de los capítulos que forman parte de la tesis.

2. Capítulos de la tesis doctoral

En esta sección se presenta un resumen de cada uno de los artículos/capítulos incluidos en la tesis doctoral.

2.1 CAPÍTULO 2. Efectos de las características de las ciudades occidentales contemporáneas sobre la avifauna urbana

Desde los primeros asentamientos humanos permanentes del Holoceno, hace 10 000 años, hasta las ciudades contemporáneas, las aves han cohabitado con los seres humanos. En las últimas décadas la urbanización ha crecido exponencialmente en el planeta y, en 2030, más del 60% de la población mundial vivirá en zonas urbanas. En función de su tolerancia a la urbanización las aves se clasifican en tres categorías: evitadoras, adaptadoras o explotadoras urbanas. Las ciudades occidentales contemporáneas atraen a las aves por la presencia de recursos tróficos abundantes y predecibles, la reducción en la diversidad de depredadores o la provisión de estructuras donde ubicar los nidos, entre otras. Sin embargo, la urbanización es uno de los mayores problemas actuales para la biodiversidad y el modelo de ciudad contemporánea puede dejar de ser atractivo para las aves e incluso causar el declive a ciertas especies ligadas a medios urbanos. Algunas razones que explican este proceso son: la gestión urbana y la pérdida de zonas verdes, la contaminación, la comida antropogénica y las nuevas tendencias arquitectónicas. Un cambio en el modelo de ciudad contemporánea que proteja la biodiversidad, aunque es un reto difícil, es posible siguiendo ejemplos como el de la infraestructura verde y sostenibilidad ambiental de la ciudad de Vitoria-Gasteiz (España).

2.2 CAPÍTULO 3. Factores que influyen en la abundancia de gorriones comunes *Passer domesticus* en zonas urbanas del sureste de España

El objetivo de este artículo fue explorar los factores bióticos y abióticos que inciden en la abundancia de gorriones comunes urbanos en la Comunidad Valenciana. Además, se quiso inferir en las causas que podrían explicar el declive urbano y proponer medidas potenciales que lo detengan e incrementen las poblaciones de la especie. La abundancia de gorriones comunes urbanos se analizó en relación con 17 variables del paisaje mediante la realización de 181 puntos de escucha (50 parques urbanos/38 escuelas/93 calles) por temporada en seis localidades y durante cinco temporadas (1 época de reproducción y 4 no reproducción) en el sureste de España. Los parques urbanos también fueron descritos por 21 variables adicionales. Se utilizaron Análisis de Partición Jerárquica (HP) para identificar los factores urbanos favorecidos o evitados por la especie en cada temporada. Los gorriones comunes eran más abundantes en parques y otras áreas urbanas donde los contenedores de basura estaban disponibles como sitios de alimentación. Los contenedores de basura, el área y la proximidad a otros parques incrementaron su abundancia en los parques. Además, los gorriones eran menos abundantes en calles concurridas, áreas de tráfico denso y zonas de edificios de gran altura. Los gorriones comunes están presentes en gran abundancia donde se alimentan, pero escasean en áreas contaminadas, concurridas y densamente urbanizadas de la ciudad. Los cambios en la estructura de estos hábitats urbanos específicos pueden ser la base de las disminuciones documentadas en la Comunidad Valenciana y España. Se necesita una planificación y gestión urbana verde para garantizar la existencia de áreas verdes heterogéneas con suficiente cobertura de vegetación que proporcionen una alta disponibilidad de alimentos naturales frente a los restos de alimentos antropogénicos. El mantenimiento de edificios que ofrezcan huecos para ubicar los nidos y una menor contaminación del aire podría mejorar las ciudades de manera adecuada para sustentar a las poblaciones de gorrión común.

2.3 CAPÍTULO 4. La sustitución de la cubierta seminatural por sustratos artificiales en los parques urbanos provoca una disminución de los gorriones comunes *Passer domesticus* en ciudades mediterráneas

Muchos pueblos y ciudades europeas han sufrido procesos de reurbanización en las últimas décadas. Sin embargo, sus efectos sobre la biodiversidad urbana han sido poco estudiados. Actualmente los procesos de urbanización incluyen la remodelación de parques urbanos, que implica la sustitución de sustratos seminaturales (césped natural, suelo desnudo) por otros artificiales (pavimento, hormigón, áreas para perros, césped artificial). Nuestro objetivo fue explorar, por primera vez, los efectos de la remodelación de parques sobre las tendencias y las abundancias de un explotador urbano en declive, el gorrión común. Se obtuvo un índice de abundancia de gorriones comunes mediante puntos de escucha en 32 parques urbanos de cuatro municipios de la Comunidad Valenciana (España) en cuatro veranos (2015-2018). De los parques estudiados, 10 fueron remodelados durante el período de estudio. Antes de la remodelación, las tendencias y la abundancia de gorriones comunes en ambos grupos de parques eran similares y estables en promedio, lo que sugiere que no existía un sesgo entre ambos grupos de parques. Sin embargo, la abundancia de gorrión común se redujo significativamente en los parques afectados por obras de remodelación mientras que en los no remodelados se mantuvo estable. La remodelación de parques urbanos podría estar relacionada con una reducción tanto en la idoneidad del hábitat como en la disponibilidad de recursos tróficos, lo que podría dañar las poblaciones urbanas de gorriones comunes y posiblemente también otras especies. Por lo tanto, se necesitan urgentemente nuevas políticas de gestión de parques y medidas de planificación urbana para preservar a los gorriones comunes urbanos.

2.4 CAPÍTULO 5. Efecto de sustituir los contenedores de basura de superficie por contenedores de basura subterráneos en el gorrión común *Passer domesticus* urbano

Los procesos de urbanización están aumentando en todo el mundo a un ritmo sorprendente que afecta a la vida silvestre de muchas maneras: cambia la estructura del hábitat, reduce los recursos y modifica la distribución, composición y abundancia de la biota local. En diferentes países, las técnicas de recolección de residuos urbanos están evolucionando y los contenedores de basura de superficie (receptáculos para almacenar temporalmente desechos domésticos antropogénicos del vecindario ubicados sobre el suelo en las calles) están siendo reemplazados por contenedores subterráneos (cajas metálicas con tolvas de acero que desembocan en grandes contenedores subterráneos) para mejorar las medidas de saneamiento, evitar malos olores y dificultar que los animales esparzan los desechos. Nuestro objetivo fue detectar si los gorriones comunes eran más abundantes cerca de los contenedores de basura de la superficie que cerca de los subterráneos. Registramos un índice de abundancia de gorriones comunes durante dos visitas en el invierno 2018-2019 mediante puntos de escucha ubicados en grupos de ambos tipos de contenedores (80 grupos de subterráneos y 85 de superficie) en ocho localidades del este de España. Modelamos el índice de abundancia según el tipo de contenedor de basura, y otras 14 variables ambientales en cuatro escalas: contenedor, edificios más cercanos, características urbanas cercanas y localidad en general, usando GLMMs. Los gorriones comunes eran más abundantes cerca de los contenedores de superficie que de los contenedores de basura subterráneos, lo que puede estar relacionado con una mayor disponibilidad de restos de alimentos. La presencia de otros elementos urbanos (terrazas de bares, jardines privados, árboles maduros) interactuando con los contenedores de basura también influyeron en la abundancia de gorriones comunes. La sustitución de los contenedores de basura de superficie con los subterráneos puede privar a los gorriones comunes de recursos tróficos, lo que podría conducir a la disminución de esta especie, especialmente en áreas urbanas con poca cubierta verde.

2.4 CAPÍTULO 6. Efecto experimental de los recursos alimentarios antropogénicos disponibles en las ciudades sobre la condición corporal, el estado nutricional y el estrés oxidativo de un paseriforme bioindicador urbano

Las áreas urbanas son proveedores constantes y predecibles de alimentos antropogénicos procesados. El gorrión común es una especie de bioindicador urbano que ha disminuido en su área nativa durante las últimas décadas. Estudios recientes apuntan al alto nivel de estrés oxidativo de los individuos urbanos, probablemente debido a la dieta urbana o a los contaminantes, como una posible causa de declive. Este estudio tuvo como objetivo determinar experimentalmente en cautiverio los efectos sobre la condición física, los parámetros nutricionales bioquímicos del plasma y el estado oxidativo de los gorriones alimentados con dos tipos de recursos tróficos urbanos abundantes: restos de comida de cafetería y pienso para mascotas. En una zona rural del sureste de España, se capturaron 95 gorriones comunes y se mantuvieron en aviarios al aire libre para excluir el efecto potencial de los contaminantes urbanos. Los individuos estuvieron expuestos durante 20 días a uno de tres tratamientos alimentarios: dieta control, similar a la disponible en áreas rurales (frutas, verduras, pienso de aves de corral), dieta de cafetería (comida ultraprocesada) o dieta de pienso seco para gatos. Se recolectaron muestras de sangre antes y después de los tratamientos alimentarios. Se obtuvieron las tasas de cambio relativo de 14 variables: condición física (3); estado nutricional (6); estado oxidante-antioxidante (5). Se utilizó un Análisis de Componentes Principales (PCA) para encontrar gradientes de covariación de variables. Se utilizaron GLMMs para conocer el efecto de las dietas en cada PC seleccionado. La dieta de cafetería provocó desnutrición, especialmente en las hembras, y posibles signos de anemia (PC1). La dieta de pienso de gatos (ultra rica en proteínas) aumentó la producción de especies reactivas de oxígeno, el estrés oxidativo general y la albúmina (PC2). Además, condujo a una mayor producción de ácido úrico y niveles de hematocrito (PC6). Así, una dieta urbana desequilibrada en un ambiente no contaminado altera la condición corporal, la fisiología nutricional y el estrés oxidativo en un paseriforme bioindicador urbano. Un efecto sinérgico de una dieta de baja calidad y la contaminación urbana puede contribuir al declive general del gorrión común en las ciudades europeas.

2.5 CAPÍTULO 7. La contaminación acústica recreativa procedente de las fiestas tradicionales reduce la productividad juvenil de un bioindicador aviar urbano

El ruido es un contaminante que cada vez despierta una mayor preocupación entre los ecólogos y biólogos de la conservación. La contaminación acústica recreativa, especialmente los sonidos impredecibles e intermitentes, y sus efectos sobre la biodiversidad han sido poco estudiados. Generalmente, los investigadores han prestado muy poca atención al efecto de las fiestas populares tradicionales con pirotecnia sobre la fauna salvaje. Este estudio tuvo como objetivo explorar el efecto de estas actividades recreativas sobre la productividad juvenil de un bioindicador aviar urbano: el gorrión común. Estudiamos cinco pares de localidades de la Comunidad Valenciana con fiestas populares ruidosas (Moros y Cristianos). Cada pareja estaba compuesta por una localidad con fiestas durante la época de cría y la localidad similar más cercana, pero con las fiestas fuera del período reproductivo (controles). Ambos tipos de localidad se muestrearon dos veces cada primavera (mayo-junio de 2019 y 2020). Las fechas de muestreo se seleccionaron 15 y 30 días después de que terminaran las fiestas populares. Las localidades control se muestrearon un día después del censo de su localidad par correspondiente con fiestas ruidosas en la temporada de reproducción. La proporción de juveniles/adultos detectados durante los censos en el área de influencia de las fiestas (100 metros alrededor de la ruta de los desfiles) se utilizó como un proxy del éxito reproductor del gorrión común. Los datos se analizaron utilizando un GLMM. Como factores fijos se incluyeron: año (2019/2020), temporada de celebración de las fiestas (durante la reproducción/fuera de la reproducción), muestreo (15/30 días después de las fiestas) y sus interacciones. La localidad anidada dentro del par de localidades fue definida como factor aleatorio. En 2019, la productividad de juveniles fue menor en las localidades con fiestas tradicionales ruidosas durante la temporada de cría que en las localidades control. Las festividades de 2020 fueron canceladas debido a la COVID-19. En la primavera de 2020, la productividad juvenil del gorrión común fue la misma en ambos grupos de localidades. El confinamiento domiciliario debido a la COVID-19 no aumentó la productividad de juveniles de esta especie en las localidades control en 2020 versus 2019.

3. Discusión general

Los resultados de la presente tesis muestran que los pueblos de la Comunidad Valenciana se han convertido en entornos desafiantes para el gorrión común urbano. A pesar de que esta especie estaba y sigue estando ampliamente distribuida en esta región (Gil-Delgado y Mora 2015, Bernat-Ponce 2021), las áreas urbanas valencianas han cambiado en las últimas décadas a través de nuevas tendencias urbanísticas que alteran el hábitat de los gorriones con consecuencias perniciosas para sus poblaciones. La primera parte de la tesis, que consta de los Capítulos 2, 3, 4 y 5, exploró el hábitat principal de la especie (Bernat-Ponce et al. e*n prensa*), aquellos factores locales que determinan su prevalencia y los cambios de estructura urbana que están alterando el hábitat de las especies en las localidades valencianas (Bernat-Ponce et al. 2018, 2020, 2021a). Además, la segunda parte de la tesis, que consta de los Capítulos 6 y 7, exploró dos elementos ambientales, alimentos procesados antropogénicos y contaminación acústica (Capítulo 6, Bernat-Ponce et al. 2021b/Capítulo 7), que también podrían contribuir a la disminución de la especie en esta región oriental de la Península Ibérica. Los resultados de esta tesis abarcan principalmente la hipótesis de la "falta de alimento" para explicar el declive de la especie (Capítulo 1, Summers-Smith 2003, MacLeod et al. 2006, Shaw et al. 2008), pero su interrelación será ampliamente discutida más adelante.

Primero, Bernat-Ponce et al. (*en prensa*/Capítulo 2) y Bernat-Ponce et al. (2018/Capítulo 3), tratan las características del ecosistema principal donde prosperan los gorriones comunes, las áreas urbanas. En el Capítulo 2, Bernat-Ponce et al. (*en prensa*) se revisaron aquellos rasgos de las ciudades contemporáneas que han afectado a la avifauna urbana desde la prehistoria hasta la actualidad. Los primeros asentamientos humanos permanentes estuvieron vinculados al desarrollo de la agricultura hace 10.000 años en diferentes áreas del planeta (Harari 2014). Durante este período, las incipientes áreas preurbanas iniciaron el vínculo actual entre aves y ciudades (Negro et al. 2020). Desde entonces, las ciudades han atraído aves durante siglos (Bernat-Ponce et al. *en prensa*/Capítulo 2). Sin embargo, por varias razones, las ciudades se están volviendo menos atractivas para las aves durante las últimas décadas (Capítulo 1, Bernat-Ponce et al. *en prensa*/Capítulo 2). Los cambios urbanos actuales son tan intensos y rápidos que algunas especies de aves urbanas no pueden adaptarse e incluso varias están disminuyendo, como el gorrión común (Capítulo 1, De Laet y Summers-Smith 2007). Todavía no hay una respuesta clara para explicar esta disminución generalizada de la especie en las áreas urbanas del Paleártico Occidental, mientras que diferentes factores pueden estar ocurriendo en diversas áreas geográficas o tipologías de ciudades.

El Capítulo 3 (Bernat-Ponce et al. 2018) exploró los principales factores urbanos que favorecen o limitan la abundancia de la especie focal en el área de estudio donde las poblaciones están disminuyendo (Vera y Escandell 2016). Como se esperaba por estudios anteriores, los gorriones comunes eran más abundantes en los parques urbanos que en otros elementos de la matriz urbana, ya que estas áreas verdes públicas son hábitats de alimentación esenciales para la especie durante todo el año (Murgui 2009, Murgui y Macias 2010). Por esa razón, los gorriones en el área de estudio de la tesis dependerían fuertemente de los parques urbanos y los cambios en su gestión podrían tener efectos perniciosos en las poblaciones locales que podrían contribuir al declive de la especie.

En línea con los resultados del Capítulo 3 (Bernat-Ponce et al. 2018), el Capítulo 4 se centró en explorar esa nueva gestión y planificación de áreas verdes urbanas llevadas a cabo durante la tesis en algunas localidades valencianas (Bernat-Ponce et al. 2020). Varios parques urbanos del área de estudio fueron remodelados reduciendo drásticamente la disponibilidad de recursos alimenticios naturales (Verbeeck et al. 2011, Warhurst et al. 2014, Bernat-Ponce et al. 2020/Capítulo 4). Los índices de abundancia de gorriones comunes en 10 parques remodelados se redujeron en un 60% mientras que en 22 parques no remodelados estos índices solo disminuyó en un 15% (Bernat-Ponce et al. 2020/Capítulo 4). Este estudio describe por primera vez los importantes efectos negativos de la remodelación de parques urbanos con sustratos artificiales (cemento, césped artificial). Los resultados de este capítulo (Bernat-Ponce et al. 2020) concuerdan estrechamente con una de las principales razones sugeridas, y descritas en el Capítulo 1, para explicar el declive global del gorrión común en áreas urbanas, la hipótesis de la falta de comida y la pérdida de zonas verdes (Robinson et al. 2005, Vincent 2005, Anderson 2006, MacLeod et al. 2006, Shaw et al. 2008).

En el Capítulo 3 (Bernat-Ponce et al. 2018) también se detectó que los gorriones comunes en la Comunidad Valenciana eran más abundantes en presencia de contenedores de basura de superficie, especialmente en invierno cuando la comida natural es más escasa. Sin embargo, la creciente tendencia europea de reemplazar los contenedores de basura de superficie por contenedores subterráneos (Nilsson 2011, ISWA 2013) podría disminuir la disponibilidad de recursos tróficos, y esto podría reflejarse en reducidos índices de abundancia de gorriones. El Capítulo 5 de la tesis (Bernat-Ponce et al. 2021a), centrado en un segundo estudio a escala fina sobre el cambio en las políticas de gestión de residuos urbanos, mostró que los contenedores de basura subterráneos presentaron menor abundancia de la especie en comparación con los contenedores de basura de superficie tradicionales, por lo que esta sustitución sería perniciosa para las poblaciones urbanas de gorrión común.

Los estudios regionales (Bernat-Ponce et al. 2018/Capítulo 3) y los dos estudios a escala fina (Bernat-Ponce et al. 2020, 2021a/Capítulos 4 y 5) de la tesis vinculan esta reducción de disponibilidad de recursos tróficos en áreas urbanas (cambios en la estructura urbana) con una disminución de los índices de abundancia de gorriones comunes registrados. No obstante, con efectos inexplorados sobre las documentadas tendencias poblaciones negativas de la especie (Vera y Escandell 2016).

La segunda parte de la tesis se centró en explorar dos factores ambientales, los alimentos procesados antropogénicos (Capítulo 6) y la contaminación acústica (Capítulo 7), que podrían tener efectos perniciosos sobre las poblaciones locales de la especie. Los gorriones comunes dependen en gran medida de los recursos alimentarios antropogénicos (Anderson 2006, Herrera-Dueñas et al. 2015) y presentan niveles más altos de estrés oxidativo en comparación con sus conspecíficos rurales (Herrera-Dueñas et al. 2017). Pero no se sabe si esta situación es debida a la contaminación urbana del tráfico (Bernat-Ponce et al. 2018/Capítulo 3, Peach et al. 2018) o a una dieta urbana de baja calidad como la que se encuentra en las terrazas de los bares o en la basura de los contenedores (Bernat-Ponce et al. 2018, 2021a/Capítulos 3 y 5). El Capítulo 6 de la tesis mostró que el estado general y el estrés oxidativo de los individuos rurales nacidos en un ambiente rural salvaje y saludable pueden alterarse profundamente si las restricciones alimentarias urbanas entran en vigor y se aíslan de la contaminación. Un efecto sinérgico de la dieta urbana de baja calidad y la contaminación podría estar impulsando el declive general de los gorriones comunes en las ciudades europeas. Sin embargo, la integración de los resultados del Capítulo 6 con los hallazgos de los Capítulos 3 y 5 de esta tesis es un comienzo prometedor para conocer el papel de la dieta urbana en el declive generalizado de este bioindicador urbano.

El segundo factor ambiental analizado en la tesis fue el ruido urbano (Capítulo 7), centrado en los efectos del ruido intermitente e impredecible sobre el éxito reproductivo de la especie. En la Comunidad Valenciana abundan las fiestas tradicionales y ruidosas caracterizadas por una intensa actividad de fuegos artificiales y pólvora, como los desfiles de Moros y Cristianos (Catalá-Pérez 2012, Lloret 2017, Maeso et al. 2018), cuyos efectos sobre la fauna se desconocen. El ruido recreativo de estos festivales tradicionales logró reducir la productividad juvenil de la especie estudiada, afectando negativamente tanto a los nidos con huevos como a los pollos (Bernat-Ponce et al. 2021b/Capítulo 7). Es importante resaltar que los efectos negativos comprobados del ruido recreativo podrían sumarse a otros derivados de cambios urbanísticos, como los descritos en los Capítulos 4 y 5 de la tesis, agregando un problema adicional para la especie.

Los hallazgos de la tesis presentan una serie de limitaciones, ya que la mayoría de resultados se relacionaron con índices de abundancia local (Capítulos 3, 5 y 7) y no con tendencias poblacionales (solo el Capítulo 4, tendencias de poblaciones locales a corto plazo en parques) que podrían limitar su extrapolación a los efectos reales sobre la disminución de la especie. Además, los cambios urbanos estudiados no están ocurriendo en todas las localidades de la Comunidad Valenciana, España u otros países, limitando su utilidad aplicativa o incluso pueden no ser independientes de otros cambios urbanos no estudiados en esta tesis. Sin embargo, los hallazgos individuales de la tesis son lo suficientemente claros como para esperar resultados similares en otros lugares si se dieran estos cambios urbanos estudiados. No obstante, para comprender mejor los efectos potenciales de los hallazgos de la tesis sobre el declive de la especie, se deberían realizar nuevos estudios y proyectos completos.

A pesar de las limitaciones del estudio anteriormente mencionadas, esta tesis arroja luz sobre algunos factores novedosos y cambios urbanos no reportados previamente que alteran el hábitat urbano de una especie bioindicadora en declive. Esta tesis desarrolla más profundamente la hipótesis de la falta de alimento sobre el declive del gorrión común y la vincula con índices de abundancia de la especie reducidos. Además, factores adicionales como el ruido recreativo, la dieta urbana, la contaminación urbana o los edificios altos también pueden estar desempeñando su papel en las tendencias regionales negativas reportadas por Vera y Escandell (2016), aunque no han sido estudiadas en esta tesis. Lo que parece claro es que el declive del gorrión común en la Comunidad Valenciana probablemente no sea atribuible a un único factor, sino a una sinergia de ellos que, en mayor o menor medida, afectan de forma significativa las abundancias locales, las poblaciones y finalmente sus tendencias (Anderson 2006).

4. Aplicaciones de los hallazgos de la tesis para una gestión urbana amigable con el gorrión común

Aunque la tesis se centró en una región de estudio específica, sus hallazgos son útiles para establecer algunas medidas generales de gestión de las áreas urbanas respetuosas con el gorrión común: preservar las áreas verdes tradicionales preexistentes en las ciudades, establecer nuevas leyes/ordenanzas verdes, potenciar una red urbana verde en la matriz de la ciudad, reducir el tráfico rodado y/o peatonalizar los centros de las ciudades y establecer medidas compensatorias obligatorias para los cambios urbanos que afecten a la biodiversidad. Además, todos los cambios en la estructura urbana deberían ser precedidos de estudios ambientales. Asimismo, la gestión de las zonas verdes urbanas debería abogar por reducir el uso de sustratos impermeables como el césped artificial, incrementar el tamaño, número, conectividad y heterogeneidad de las mismas con el objetivo de que puedan actuar como “piedras de paso” pero también como hábitats de calidad dentro de la matriz de asfalto de la ciudad.

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6. Conclusiones generales de la tesis doctoral

Capítulo 2

1. En las últimas décadas, la urbanización ha crecido exponencialmente en el planeta y será una de las principales amenazas para la conservación de las aves.

2. Las ciudades occidentales actuales atraen a las aves principalmente por la presencia de recursos tróficos abundantes y predecibles, la reducción de la diversidad de depredadores o la existencia de lugares de nidificación, entre otros factores.

3. El modelo de ciudad contemporánea puede volverse poco atractivo para las aves y, de hecho, algunas especies urbanas están disminuyendo en el presente debido a: pérdida de áreas verdes, contaminación, reducción de recursos tróficos naturales, nuevos depredadores/competidores y nuevas tendencias de construcción.

Capítulo 3

4. Las principales causas de la disminución del gorrión común urbano en el sureste de España pueden estar relacionadas con cambios en el uso del suelo y en la infraestructura urbana.

5. Los gorriones comunes eran más abundantes en parques y áreas urbanas donde los contenedores de basura estaban disponibles como sitios de alimentación. La presencia de contenedores, el área del parque y la proximidad a otros parques favorecieron su abundancia en estas zonas verdes urbanas.

6. El desarrollo de edificios de gran altura, la alta densidad de población humana y el tráfico rodado afectaron negativamente a la abundancia de gorriones comunes.

Capítulo 4

7. Actualmente los procesos de urbanización incluyen la remodelación de parques urbanos, que implica la sustitución de sustratos seminaturales (césped natural, suelo desnudo) por artificiales e impermeables (pavimento, hormigón, áreas para perros, césped artificial).

8. Las poblaciones de gorriones comunes en 22 parques no remodelados disminuyeron en aproximadamente un 15% entre 2015 y 2018, mientras que las poblaciones en los 10 parques remodelados disminuyeron en más del 60% durante el mismo período.

9. La remodelación del parque podría estar relacionada con una reducción tanto de la idoneidad del hábitat como de la disponibilidad de recursos tróficos, lo que podría dañar las poblaciones urbanas de gorriones comunes y posiblemente también otras especies.

Capítulo 5

10. El vínculo del gorrión común con los restos de comida antrópica sigue siendo importante en las zonas urbanizadas modernas y es patente en los contenedores de basura de superficie.

11. Los gorriones comunes eran más abundantes en los contenedores de basura de superficie que en los contenedores de basura subterráneos. La presencia de otras características urbanas (terrazas de bares, jardines privados, árboles maduros) interactuando con los contenedores de basura también influyó en la abundancia de gorriones comunes.

12. La nueva tendencia de sustitución de los contenedores de basura de superficie por otros subterráneos puede privar a los gorriones comunes de recursos tróficos, lo que podría conducir al declive de esta especie, especialmente en áreas urbanas con poca cubierta verde.

Capítulo 6

13. El tipo de dieta urbana que consumen los gorriones comunes afecta a su estado nutricional general y a su estrés oxidativo en un entorno rural libre de contaminación urbana.

14. Las dietas urbanas desequilibradas (nutricionalmente pobres) pueden empeorar la condición física, reducir el hematocrito y alterar los parámetros nutricionales relacionados con la desnutrición (niveles bajos de triglicéridos, ácido úrico, proteínas totales, glucosa), especialmente en las hembras. Una dieta urbana desequilibrada con alto contenido de proteínas aumentó tanto los niveles de ROS (d-ROMs) como la barrera antioxidante plasmática (OXY, albúmina, ácido úrico), pero no lo suficiente para evitar el estrés oxidativo.

15. Un efecto sinérgico de la dieta urbana de baja calidad y la contaminación podría impulsar el declive general del gorrión común en las ciudades europeas. Los resultados presentados en la tesis son un comienzo prometedor para determinar la importancia de la dieta urbana en el declive de este bioindicador urbano.

Capítulo 7

16. Las fiestas ruidosas en primavera redujeron la productividad juvenil de un bioindicador urbano aviar, el gorrión común.

17. En 2019, la productividad de juveniles (relación juveniles/adultos) fue menor en las localidades con fiestas tradicionales ruidosas durante la temporada de cría que en las localidades de control con fiestas fuera de la época de cría. El ruido de las fiestas parece afectar negativamente tanto a nidos con huevos como con pollos.

18. La productividad juvenil en las localidades con fiestas en primavera aumentó hasta igualar la productividad de las ciudades de control en 2020, cuando los festivales se cancelaron en debido a la COVID-19. Sin embargo, el confinamiento domiciliario no aumentó la productividad

juvenil de esta especie en las localidades de control en 2020, por lo que las restricciones de por la COVID-19 no mejoraron el éxito reproductor normal del gorrión común.