

## Research Article

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### Author for correspondence:

\*Ángel Gálvez,

Email: [angel.galvez@uv.es](mailto:angel.galvez@uv.es)

# Artificial grass in parks as a potential new threat for urban bird communities

David Sánchez-Sotomayor<sup>1,2</sup> , Antonio Martín-Higuera<sup>1,2</sup> ,

José A Gil-Delgado<sup>1</sup> , Ángel Gálvez<sup>1\*</sup>  and Edgar Bernat-Ponce<sup>1</sup> 

<sup>1</sup>Cavanilles Institute of Biodiversity and Evolutionary Biology, University of Valencia, c/ Catedrático José Beltrán, 2, 46980, Paterna, Spain and <sup>2</sup>Bird and Mammal Section, Asociación Iberoza, Madrid, Spain

## Summary

Green areas are key habitats for urban avifauna. Urban parks stand out from other anthropic habitats especially in providing trophic resources for many bird species. Consequently, modifications of these green zones can imply major changes in urban biodiversity. Potential pernicious urban remodelling is taking place in parks of eastern Spain because natural grass is being replaced with artificial grass to save water and to avoid management. This study aimed to determine whether remodelled parks with artificial grass harbour lower avian diversity (alpha, beta and gamma diversity) than traditional parks with natural grass. We surveyed 21 parks with artificial grass and 24 parks with natural grass in 18 towns of the Valencia Region in autumn 2020. In each park, we carried out 5-minute and 25-m radius point counts for determining bird species and their abundance. The effects of park area and grass type on alpha diversity (species richness, Shannon diversity index, Pielou's Evenness and total abundance) were tested by means of GLMs. Differences in beta diversity and its components (nestedness and turnover) were also analyzed with the Bray-Curtis dissimilarity index. Gamma diversity was assessed by means of species accumulation curves. Finally, differences in community composition were tested by PERMANOVA and SIMPER tests. The parks with natural grass always harboured higher gamma diversity, species richness and abundance. Turnover was higher in parks with natural grass, whereas nestedness was higher in artificial grass parks. Differences in community composition were due mainly to abundance differences in common ground-feeding birds. We highlight that the trend of replacing natural by artificial grass in urban parks has harmful effects on urban bird communities and is a threat to bird conservation. Although artificial grass might save water, the effects on urban biodiversity should be carefully evaluated.

## Introduction

With increasing rural habitat conversion in urban areas, some parts of cities have developed and integrated green areas within their limits, among other human utilities (Ward *et al.* 2010, Swensen 2018). As a result of urbanization and expanding cities, these green areas became isolated patches in urban matrices, and progressively further away from natural and rural areas of city outskirts (Fernández-Juricic and Jokimäki 2001). Cities constantly change through refurbishing old buildings and developing new structures. As these traditional urban green areas are remodelled or even disappear, new park models emerge (Shaw *et al.* 2008, Bernat-Ponce *et al.* 2020).

Nowadays, urban green areas represent the main habitat for many bird species that are adapted to anthropization of the natural environment (Bernat-Ponce *et al.* 2018, Morelli *et al.* 2018). Urban parks provide numerous resources for birds, such as food supplies (partly formed by human food scraps), water, shelter, and nesting sites (Murgui and Hedblom 2017, Isaksson 2018). The grass and lawns in these habitats are important sources of food for many bird species (Savard *et al.* 2000). Furthermore, as small birds better tolerate human presence than their predators, especially aerial predators like raptors, urban areas can release them from some predation (Rebolo-Ifrán *et al.* 2017, but also see Negro *et al.* 2020). Nevertheless, these urban-adapted species face threats that have caused severe population declines in some species such as the House Sparrow *Passer domesticus* (Summers-Smith 2003, Mohring *et al.* 2021). Lack of insects and scarcity of seeds in urban parks (e.g. due to excessive mowing), several types of pollution, exposure to diseases or new predators, and invasive species, are some of the main causes suggested for bird population declines in urbanized areas (Beckerman *et al.* 2007, Schroeder *et al.* 2012, Weir 2015, Dadam *et al.* 2019). Several bird species can be affected by urbanization, as reflected in alterations to bird communities (Shultz *et al.* 2012, Banville *et al.* 2017). Besides, urban birds have been widely used as environmental bioindicators to understand how unintended urbanization consequences can affect the well-being of both citizens and wildlife, and to assess habitat quality and environmental change (Herrera-Dueñas *et al.* 2014, Pollack *et al.* 2017, Bernat-Ponce *et al.* 2021).

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Currently, the development of new urban remodelling could have negative impacts on the wildlife that inhabits cities (Muller *et al.* 2010, Wright 2013). In some urban areas, natural vegetation is being replaced with synthetic alternatives, such as artificial grass, paving and rubber ground, to meet human requirements (Verbeeck *et al.* 2011, Bernat-Ponce *et al.* 2020). Moreover, artificial grass is located on a metal grid and concrete, which isolate it from the soil (Schmidt *et al.* 1990) and also increases exposure to plastic pollutants with potential harmful effects on wildlife health (Gall and Thompson 2015, Pochron *et al.* 2017). Many urban areas around the world, especially in arid and semi-arid climates, are adopting this urban planning measure to reduce the watering and management requirements of public green areas (New Yorkers for Parks 2006) by converting traditional parks into “domestic” versions. This replacement has pernicious effects on urban bird species, such as House Sparrow, due to lower availability of trophic resources including not only seeds and plant material but also invertebrates (Bernat-Ponce *et al.* 2020). Thus, we can expect major biological diversity loss in cities with this urban remodelling, which is measurable through birds as bioindicators.

In order to achieve appropriate biodiversity conservation, we need to correctly understand the different biological diversity components in urban areas. Taxonomic diversity does not only imply local (alpha) diversity, but also regional (gamma) diversity and the similarity/dissimilarity between local communities (beta diversity) (Whittaker 1972). The omission of any of these dimensions can lead to hasty conclusions. The direct or indirect effects of human activities on biological communities have been studied for decades, often focusing on alpha diversity and neglecting beta diversity (e.g. Perillo *et al.* 2017, Howes and Reynolds 2021). However, beta diversity is a key conservation planning element because it indicates whether diversity is concentrated in a few sites or spreads across many (Socolar *et al.* 2016, Cao *et al.* 2021).

Remodelling urban green areas with artificial grass to replace natural grass can drive reductions in alpha and gamma diversity through loss of resources for birds, especially seeds and invertebrates. We can expect this remodelling to give rise to beta diversity losses due to habitat homogenization. Additionally, finding a high nestedness component of beta diversity (the poorest site being a subset of the richest site; Baselga 2012) can indicate a highly human-disturbed habitat (Fernández-Juricic 2002). In any case, studying the change in species composition using nestedness and turnover (replacement of some species by others between sites; Baselga 2012) components of beta diversity in real systems is essential to understand the processes that shape species distributions and to establish appropriate management plans that enhance bird conservation in urban areas (Baselga 2012, González-Oreja *et al.* 2012b).

Even though different impacts of human activities on wildlife have been studied for years, the effects of recent urban remodelling activities are still poorly known. The consequences of replacing natural grass with artificial grass in parks have only been recently studied in population terms for House Sparrows (Bernat-Ponce *et al.* 2020). The present work is the first study to compare differences in biological diversity and community composition in the avifauna that inhabits remodelled and traditional urban parks. The present study aimed to explore if remodelled parks with artificial grass harbour lower avian diversity (alpha, beta and gamma) than traditional parks with natural grass. For this purpose, we hypothesize that: i) artificial grass parks will harbour less alpha and gamma diversity due to lower abundance and variety of food resources; ii) beta diversity will be lower in parks with artificial grass, due to lower

heterogeneity of food resources; iii) nestedness will be higher in parks with artificial grass, as a sign of habitat quality loss; iv) community composition will differ between both park types as an effect of urbanization on avifauna. The results of the present study will be key for bird conservation and the future urban planning for cities in many countries.

## Methods

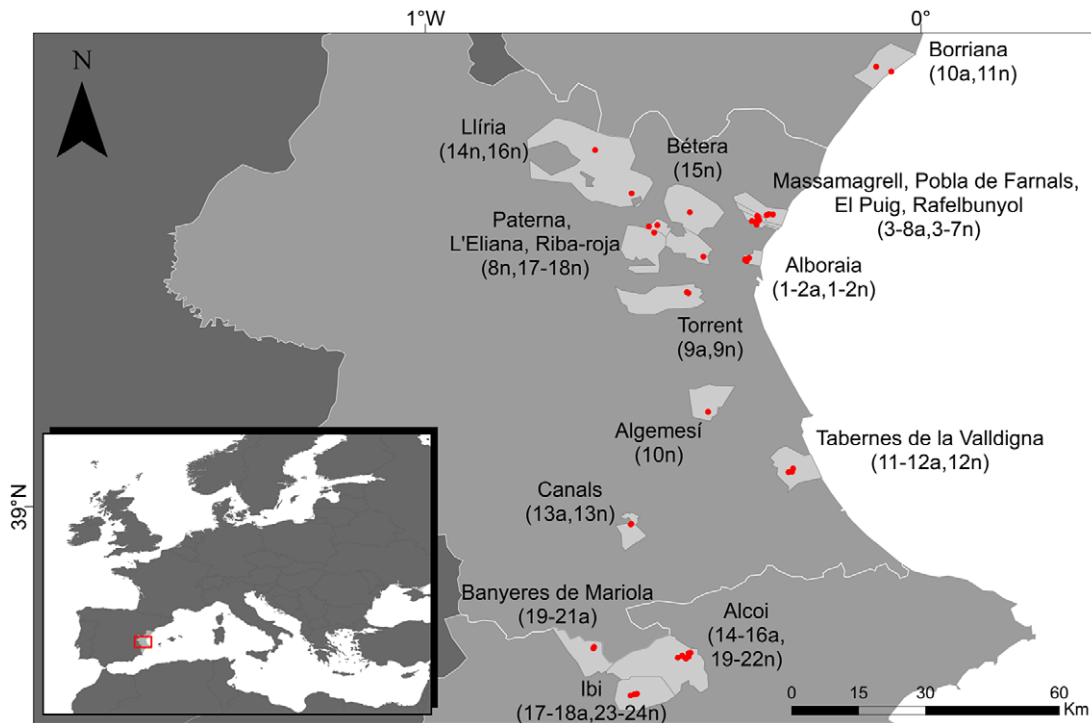
### Study area

We selected 45 urban parks (24 with natural grass, 21 with artificial grass) from 18 towns across eastern Spain (Valencia Region) with a maximum distance between them of 146 km (Figure 1). We highlight that the artificial grass trend in the urban parks in this region started in the past decade, and the available number of parks with this grass type is still low, but quickly increasing. These localities are characterized by scarce annual rainfall ( $456 \pm 4.12$  mm) and warm annual average temperatures ( $16.5 \pm 1.78^\circ\text{C}$ ), which are distinctive of the Mediterranean climate (Fick and Hijmans 2017).

### Data collection

Each park was sampled once by a point-count in autumn 2020 (Gibbons and Gregory 2006). We surveyed bird communities between 08h00 and 10h00 as human activity at these times is low and bird activity is high. Point-counts lasted 5 minutes and comprised a 25-m radius (Gibbons and Gregory 2006). By sight and hearing, we noted the presence and abundance of bird species effectively using each park (e.g. birds flying were not considered; de la Hera 2019). In the Valencia region, parks are a continuous supply of plant and invertebrate food resources throughout the year due to a design that includes a mixture of native and non-native plant species. In contrast, trophic resources in the urban matrix are expected to be lower, as in other regions (Gilbert 1989, Murgui 2007a, Zhou and Chu 2012). For this reason, the search for food sources is an important basis of interactions between birds and urban parks, even during non-breeding seasons such as autumn and winter (Bernat-Ponce *et al.* 2018, de la Hera 2019, Bermúdez-Cavero *et al.* 2021). We did not carry out sampling during the breeding season, but the relative diversity in urban parks in autumn, with presence of wintering species, strongly correlates with the diversity in spring (de la Hera 2019). As most parks (26 of 45), especially those with artificial grass (20 of 21), cover less than 1 ha, we carried out one point-count per park to establish similar sampling conditions independently of the park area. We included peripheral zones for the parks smaller than the sampling area (0.2 ha) by enlarging the surface area considered for the census. Sampling was not performed on rainy or windy days to avoid differences due to weather conditions.

We measured 10 environmental variables of the surveyed parks within the point-count area, in addition to total park area and the grass type. Distance from the point count to the outskirts of the town (m) and park area (ha) were measured using Google Earth Pro 7.3.2.5776. We considered six variables related to park vegetation, including arboreal species richness and abundance (count data), presence of shrubs lower than 1 m height, presence of shrubs higher than 1 m, presence of exotic vegetation species and presence of vegetation with fruit (e.g. dates, olives, acorns). We also considered other environmental variables: presence of supplementary anthropogenic sources of food (e.g. bar terraces) and presence of children's play areas. Finally, we also recorded the presence of



**Figure 1.** Map of the study area with the sampled towns and park locations in the Valencia Region. Names of places include the list of parks with artificial grass (1a-21a) and natural grass (1n-24n).

sources of water as a nominal variable at four levels: no water, fountains, ponds, both ponds and fountains (see Table S1 in the online supplementary material for a summary of environmental characterization and park areas by type of grass).

Replacing natural grass with artificial grass might have differential effects depending on the species' trophic ecology. Besides considering the whole bird community in every park, we also divided species into two complementary subsets according to their main feeding habits. The first subset, 'ground species', included species that primarily fed on the ground, like House Sparrow, Blackbird *Turdus merula* and Eurasian Collared-Dove *Streptopelia decaocto*. The second subgroup ('other species') consisted of species that fed sporadically on the ground, including aquatic birds like Mallard *Anas platyrhynchos*. Some birds like Great Tit *Parus major* were included in both categories because they can feed on both the ground and in bushes/trees in a similar proportion (Lack 1971; Table S2).

#### Taxonomic diversity measurement

We measured gamma diversity as the species accumulation (Tuomisto 2010), showing species accumulation curves with increasing efforts (in terms of the number of surveyed parks and sampled area) in both park types using the "specpool" function of the "vegan" package (Oksanen *et al.* 2019). Different alpha diversity measures were calculated for each park using "vegan" R package: species richness (hereafter S) by the "specnumber" function), the Shannon diversity index (hereafter H) by the "diversity" function), Pielou's evenness (hereafter J) by  $H/\log(S)$ ; Magurran 2004) and total bird abundance (A). Finally, we evaluated the beta diversity of bird communities that inhabited the parks with both grass types. For this purpose, we partitioned beta diversity into turnover

(species substitution between sites), and nestedness (different number of species between sites) components, as well as total beta diversity (sum of turnover and nestedness components), comparing pairs of parks of the same grass type, using the "beta.pair.abund" function with the Bray-Curtis dissimilarity index in the "betapart" package (Baselga *et al.* 2021).

#### Statistical analysis

Firstly, to discard misleading results due to spatial autocorrelations of parks with the same grass type, we employed Moran's I index (the "moran.test" function in "spdep" R package; Bivand and Wong 2018). To check for any environmental differences between point-count areas of parks with natural or artificial grass, we ran a PERMANOVA test (function "adonis2" in "vegan" package) with the aforementioned environmental variables using grass type as group variable, but excluding the variable Park Area, which was used in further analyses. We obtained the gamma diversity of the complete set of urban parks in the study area and separately for each grass type.

In order to determine the effect of the grass type in alpha diversity measures, we performed Generalized Linear Mixed Models (the "glmmTMB" function in the "glmmTMB" R package; Brooks *et al.* 2017). To control for species/area effects (Chamberlain *et al.* 2007), we included the park area and its interaction with grass type as covariables. We included town as a random variable to control for variation among localities (e.g. park management). Interaction was removed when not significant. To check for significant differences in turnover, nestedness and the total beta diversity between the two park types (independent factor), we used Generalized Linear Models (the "glm" function in the "stats" package; R Core Team 2020). All these analyses were carried out for all

the species altogether, but also for both ground species and other species groups.

Differences in species composition between the parks with natural and artificial grass were evaluated by a PERMANOVA test (Anderson *et al.* 2017). We used a SIMPER test to quantify the contribution of each species on the differences in species composition between types of parks, measured as a percentage of the total dissimilarity between both groups of parks (Clarke 1993; “simper” function in the “vegan” package). All the analyses were performed in R 4.0.2 (R Core Team 2020).

## Results

We did not find any significant spatial autocorrelation on the distribution of parks with grass types (Moran I Statistic =  $-0.128$ ;  $p = 0.866$ ). The environmental features of point-count areas of both park types with natural and artificial grass were not significantly different from one another according to the PERMANOVA test ( $F = 0.694$ ,  $df = 44$ ,  $p = 0.518$ ).

Bird species accumulation increased with effort in both park types, but it was always higher in parks with natural grass than in those with artificial grass (Figure 2). In the 45 surveyed parks, we obtained a gamma diversity of 31 bird species (Tables 1 and 2). In with artificial grass, gamma diversity was 18 species, 35.7% less than in parks with natural grass where species accumulation was 28. Neither the park area nor the park area with grass type interaction was significant in most of the alpha diversity measurements, except for abundance, in the three species groups in our GLMMs (Table 1; Tables S3-S5 for complete results). Only species richness and abundance of the other species group were significantly positively affected by the park area or its interaction with grass type had a significant positive effect on species richness and abundance of the other species group, and almost a marginally significant positive effect on all species richness ( $P = 0.055$ , Table S3). In contrast, the natural grass showed a positive effect on species richness (S) and abundance (A) for the three bird groups: all, ground, and other species. Nevertheless, the Shannon Diversity Index (H) was only positively affected by the natural grass when analyzing all birds together or the ‘ground’ group. We found no significant differences in Pielou’s evenness (J) for any of the three bird groups according to grass type.

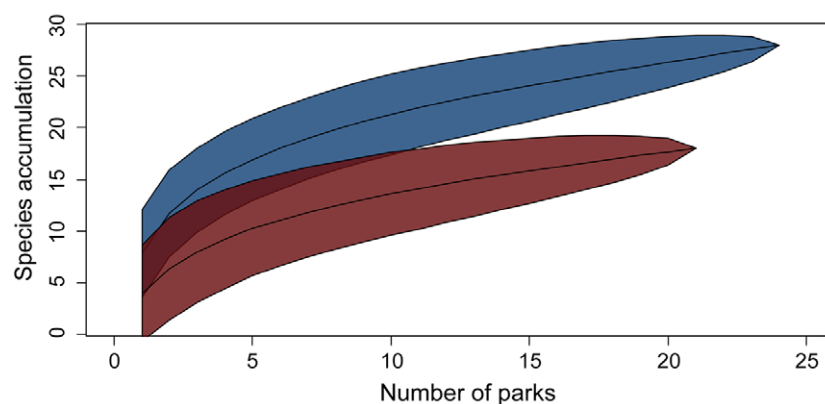
Turnover was the main component of beta diversity (measured as dissimilarity), always higher than nestedness, in every analysis

**Table 1.** Summary of the GLMMs carried out to study the effect of grass type and park area (controlled by town locality as random effect) on four alpha diversity parameters of the three bird groups in 45 parks in the Valencian Region: Species richness (S), Shannon-Diversity Index (H), Pielou’s Evenness (J) and Abundances (A). \* represents  $P$  value  $< 0.05$ . See Tables S3-S5 for complete model outputs.

Diversity parameter	Bird groups	GLMM parameter			
		Intercept	Park Area	Grass (Natural)	Park Area: Grass (Natural)
S	All	1.335*	0.104	0.559*	–
	Ground	1.262*	0.101	0.481*	–
	Others	–0.369	0.195*	0.953*	–
H	All	1.157*	0.096	0.397*	–
	Ground	1.154*	0.061	0.327*	–
	Others	0.716*	0.093	–0.037	–
J	All	1.468*	0.032	0.122	–
	Ground	1.384*	0.051	0.212	–
	Others	1.802	–0.063	0.058	–
A	All	2.709*	–0.237	0.336*	0.339*
	Ground	2.615*	–0.273	0.272*	0.408*
	Others	0.509*	0.377*	0.982*	–

(Table 2). Results of GLM showed no differences in total beta diversity in any bird group according to the type of grass. However, turnover was significantly higher in parks with natural than artificial grass, in the ‘all bird’ group and ‘ground’ group. In contrast, these two groups showed significantly higher nestedness in parks with artificial grass, compared to parks with natural grass (Table 3; Table S6 for complete results).

We found significant differences in the species composition of the whole bird community between park types in the PERMANOVA tests for the ‘all species’ group ( $F = 3.439$ ,  $df = 44$ ,  $P = 0.001$ ). According to the SIMPER tests, only five species (House Sparrow, Eurasian Collared-Dove, Common Chiffchaff *Phylloscopus collybita*, Common pigeon *Columba livia*, Eurasian Blackbird) were responsible for more than 50% of variation in



**Figure 2.** Bird species accumulation ( $\pm$  CI) in the urban parks with artificial (red;  $n = 21$ ) and natural (blue  $n = 24$ ) grass in eastern Spain during the wintering season.

**Table 2.** Summary of the studied diversity parameters according to grass type in the surveyed parks and the three bird groups in 45 parks in the Valencian Region. Mean ± SE of alpha ( $\alpha$ ; species richness S, Shannon-Diversity Index H, Pielou's Evenness J and Abundances A), beta ( $\beta$ ; total, turnover and nestedness) and gamma (species accumulation) diversities.

Grass type	Diversity parameter	Bird groups		
		All	Ground	Others
<b>Both types (n = 45)</b>				
	$\alpha_S$	6.02 ± 0.44	5.38 ± 0.41	1.67 ± 0.21
	$\alpha_H$	1.36 ± 0.09	1.26 ± 0.09	0.42 ± 0.07
	$\alpha_J$	0.83 ± 0.02	0.81 ± 0.03	0.58 ± 0.07
	$\alpha_A$	21.55 ± 1.96	19.71 ± 1.90	5.73 ± 1.00
	$\beta_{total}$	0.68 ± 0.01	0.66 ± 0.01	0.81 ± 0.01
	$\beta_{turnover}$	0.50 ± 0.01	0.42 ± 0.01	0.68 ± 0.02
	$\beta_{nestedness}$	0.18 ± 0.01	0.22 ± 0.01	0.12 ± 0.01
	$\gamma$	31	22	15
<b>Natural grass (n = 24)</b>				
	$\alpha_S$	7.79 ± 0.45	6.79 ± 0.46	2.46 ± 0.26
	$\alpha_H$	1.69 ± 0.08	1.56 ± 0.07	0.69 ± 0.09
	$\alpha_J$	0.84 ± 2.35	0.84 ± 0.02	0.81 ± 0.05
	$\alpha_A$	29.08 ± 2.35	25.92 ± 2.53	9.04 ± 1.49
	$\beta_{total}$	0.66 ± 0.01	0.64 ± 0.01	0.81 ± 0.01
	$\beta_{turnover}$	0.53 ± 0.01	0.47 ± 0.01	0.68 ± 0.02
	$\beta_{nestedness}$	0.12 ± 0.01	0.16 ± 0.01	0.13 ± 0.01
	$\gamma$	28	22	14
<b>Artificial grass (n = 21)</b>				
	$\alpha_S$	4.00 ± 0.52	3.76 ± 0.53	0.76 ± 0.21
	$\alpha_H$	0.98 ± 0.14	0.91 ± 0.15	0.12 ± 0.07
	$\alpha_J$	0.82 ± 0.04	0.76 ± 0.59	0.19 ± 0.11
	$\alpha_A$	12.95 ± 1.97	12.62 ± 1.96	1.95 ± 0.72
	$\beta_{total}$	0.71 ± 0.01	0.68 ± 0.01	0.82 ± 0.04
	$\beta_{turnover}$	0.45 ± 0.02	0.37 ± 0.02	0.72 ± 0.06
	$\beta_{nestedness}$	0.26 ± 0.02	0.31 ± 0.02	0.10 ± 0.03
	$\gamma$	18	16	7

**Table 3.** Summary of the GLMs carried out to study the effect of grass type on beta diversity of the three bird groups in 45 parks in the Valencian Region: Total beta diversity ( $\beta_{total}$ ) and turnover ( $\beta_{turnover}$ ) and nestedness ( $\beta_{nestedness}$ ) components of beta diversity. \* represents p value < 0.05. See Table S6 for complete model outputs.

Diversity parameter	GLM parameter	Bird groups		
		All	Ground	Others
$\beta_{total}$	Intercept	0.890*	0.877*	1.515*
	Grass (Natural)	-0.216	-0.297	-0.064
$\beta_{turnover}$	Intercept	-0.211	-0.536*	0.945*
	Grass (Natural)	0.371*	0.434*	-0.191
$\beta_{nestedness}$	Intercept	-1.037*	-0.871*	-2.199*
	Grass (Natural)	-0.930*	-0.792*	0.301

species composition between parks with both grass types due to the mean abundance differences.

**Discussion**

The environmental features of sampled point-counts of both park types (artificial vs. natural grass) were homogenous, except for their grass type. This was probably because urban parks in the Valencia Region are similar in design terms irrespective of their size (Murgui 2007b, 2010). However, the new trend of replacing natural grass with artificial grass, such as concrete, plastic grass or rubber, makes a huge difference in the current design of urban green areas (Verbeeck *et al.* 2011, Bernat-Ponce *et al.* 2020). The bird gamma diversity in our 45 studied parks was higher than, or at least similar to, those found in equivalent studies conducted in other parts of the world (e.g. Malaysia, northern Spain, Mexico, India), and reached



31 observed species (Carbó-Ramírez and Zuria 2011, Kale *et al.* 2018, de la Hera 2019). However, gamma diversity was higher in the parks with natural grass (28 species) than in those with artificial grass (18 species). Nevertheless, we acknowledge that gamma diversity can be affected by the different park area of traditional and artificial parks (Nielsen *et al.* 2014). Indeed, our alpha diversity results suggest an effect of park size on species richness, at least for the other birds group. Additional studies could shed light on this issue.

Bird species richness was higher in the three bird groups in parks with natural than in those with artificial grass. Shannon diversity was also higher in parks with natural grass (only non-significant in the group of 'other' birds), which is a consequence of not only the aforementioned higher species richness but also of similar Pielou's evenness between both park types. This study found no significant relation between park area and these two alpha diversity measures, even though several studies link species richness with the area of the studied parks (Fernández-Juricic and Jokimäki 2001, Chamberlain *et al.* 2007). We consider that employing fixed radius point-counts for sampling (Gibbons and Gregory 2006, Buckland *et al.* 2015), rather than complete counts with walking routes (Murgui 2010), could explain the general absence of park size effects on bird alpha diversity (except for the abundance component). Thus, we cannot rule out that larger parks, which may act as islands in an urban matrix, can harbour higher species richness and abundance. In fact, our results revealed that increasing effort (in terms of the number of surveyed parks and, consequently, an increasing surveyed area) implied greater species accumulation (Arrhenius 1921, Fernández-Juricic and Jokimäki 2001, Chamberlain *et al.* 2007). Moreover, we found higher bird abundances in point-counts located in larger parks with natural grass, even though the grass type also had a statistically significant effect on bird abundance. These patterns were found in the three studied bird groups: i) all birds together; ii) ground foragers; iii) other species.

Loss of species in insular patches tends to be balanced out by increased abundance of the remaining species, known as the compensatory density effect (MacArthur *et al.* 1972, McGrady-Steed and Morin 2000). When taking parks as patches, we did not find this compensatory density effect in the parks with artificial grass and reduced species richness. The abundance of each remaining species did not generally increase with reduced species richness in artificial grass parks, and the abundance in these remodelled parks also decreased. Replacing natural grass with artificial grass might reduce the carrying capacity of these urban green areas by cutting back available resources, especially trophic ones. Natural grass and weeds provide a wide range of resources for wild birds (especially granivorous and insectivorous species), such as seeds, insects, edaphic invertebrates, and water, as well as grass itself (Newton 1972, Iglesias *et al.* 1993).

We unexpectedly found similar total beta diversity in the artificial grass parks and the green areas with natural grass. Beta diversity is not as related to gamma diversity as alpha diversity is (Socolar *et al.* 2016), so one must not interpret similar conservation values derived from similar beta diversities. When we analyzed the partition of this beta diversity, we found turnover to be its main component in both park types. Nevertheless, we observed opposite patterns in the proportions of turnover and nestedness components according to grass type. While the bird communities that inhabited the parks with natural grass displayed a significant higher proportion of turnover than in artificial grass parks, nestedness was significantly higher in the parks with artificial grass than in natural grass parks, which could imply more disturbed

patches (Fernández-Juricic 2002). However, species and abundance replacements play a key role in bird communities of both urban park types. This species substitution is more marked in natural grass parks, which could be due to a greater dissimilarity in rare species between parks. Parks with higher heterogeneity of resources (e.g. food, water) and conditions (e.g. humidity) due to extra supplies provided by natural grass can gather more heterogeneous bird communities (Mexia *et al.* 2018, Souza *et al.* 2019). In addition, abundance of common species is extremely variable, which also contributes to the turnover component (e.g. parks dominated by House Sparrow, Eurasian Collared Dove or Monk Parakeet *Myiopsitta monachus*). In contrast, the nestedness component is significantly higher in parks with artificial grass thanks to both bird species and abundance losses in this grass type. These losses can be explained by lower resource availability and heterogeneity in artificial grass, which lead to lower carrying capacity (González-Oreja *et al.* 2012a, Souza *et al.* 2019). In other words, the higher turnover component in natural grass parks indicates that these green areas harbour more occasional species. Contrarily, the higher nestedness component in the artificial grass parks points to a diversity loss, where poorest parks (in terms of species richness and abundances) are subsets of the richest ones (Socolar *et al.* 2016, Cao *et al.* 2021). In addition, the effect of park type on beta diversity components is also significant for the group of ground birds. Hence, the effects of changing natural to artificial grass would be more detrimental for species feeding on the ground, as they consume plants, seeds, or arthropods, which are resources that are replaced by plastic elements (Newton 1972, Weir 2015, Bernat-Ponce *et al.* 2020).

Significant differences between bird communities of both park types were mainly a consequence of common bird species' abundance loss. The bird species which contributed more to the differences between park types were those that fed directly on grass as an important trophic resource, such as House Sparrow, Eurasian Collared-Dove and Domestic Pigeon. Our results agree with other studies, which have found that urban parks with natural grass are key for bird presence and abundance (Fernández-Juricic *et al.* 2001, Morelli *et al.* 2018). Nevertheless, these differences in species composition are caused not only by birds that feed directly on ground vegetation, but also by insectivorous birds. Species like Eurasian Blackbird and Common Chiffchaff are indirectly affected by loss of the invertebrate communities associated with natural grass and soil because their main food resources are arthropods, earthworms, and snails (Cramp 1992, Iglesias *et al.* 1993, Cramp and Perrins 1994), especially during the breeding season (Pagani-Núñez *et al.* 2011). Even for exclusively granivorous species like Carduelines, pernicious effects could be extremely relevant because their trophic resources are not available on artificial grass (Newton 1972, Gil-Delgado *et al.* 2009).

Our results highlight that the current trend of replacing natural grass with artificial grass in urban parks in several European countries, especially Mediterranean ones, can pose a threat to the conservation of urban avifauna and bird communities. Thus, those areas where natural grass is replaced with concrete may present similar bird declines and community alteration (Verbeek *et al.* 2011, Bernat-Ponce *et al.* 2020). We consider that this grass replacement will spread to more countries and regions due to global change consequences (e.g. global warming, drought) to save water. Thus we believe that urban planning should take into account the consequences of this landscape modification because it harms urban biological communities of which birds are suitable bioindicators. We suggest prioritizing the traditional design and

management when remodelling urban green areas (Bernat-Ponce *et al.* 2018, Mexia *et al.* 2018). In arid and semi-arid regions, where water can be a limiting resource, traditional park designs with natural grass can be replaced with native trees, bushes and grass species that are adapted to scarce water conditions. This would benefit biodiversity (including invertebrates), compensate the water savings gained from artificial grass and reduce the consequent heat island effect (Domene *et al.* 2005, Yaghoobian *et al.* 2010). In addition, natural green urban areas also provide services for humans by improving our physical and mental health, and increasing sustainability (Phillips 1993, Alcock *et al.* 2014).

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