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François Chiron, Romain Lorrillière, Carmen Bessa-Gomes, Piotr Tryjanowski, Joan Casanelles-Abella, et al.. How do urban green space designs shape avian communities? Testing the area–heterogeneity trade-off. Landscape and Urban Planning, 2024, 242, pp.104954. 10.1016/j.landurbplan.2023.104954 . hal-04297916

HAL Id: hal-04297916

<https://hal.science/hal-04297916>

Submitted on 21 Nov 2023

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Landscape and Urban Planning

How do urban green space designs shape avian communities? Testing the area-heterogeneity trade-off

--Manuscript Draft--

Manuscript Number:	LANDUP-D-23-00757R1
Article Type:	Research Paper
Section/Category:	Habitat, Wildlife and Biodiversity
Keywords:	urban ecosystems birds green space design specialisation
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Abstract:	<p>In cities, green areas are essential for biodiversity conservation, with land cover heterogeneity being a decisive factor. Yet, as heterogeneity increases for a given green area, the patch size of land covers automatically decreases, as the area available for individual species, especially habitat specialist species. This relationship, known as the area-heterogeneity trade-off, is expected to lead to a unimodal relationship between species richness and land cover heterogeneity, and contrasted effects between species according to their level of urban avoidance.</p> <p>We investigated the potential consequences of this trade-off on birds in green areas selected along an urban intensity gradient in six European cities. Using a European database on bird occurrences in nesting habitats, we defined a continuous gradient of urban avian avoidance.</p> <p>We confirmed the marked area-heterogeneity trade-off in urban green areas but found no effect of land cover heterogeneity on total avian richness at green area level. However, both land cover heterogeneity and patch size were positively associated with richness of urban avoider species, indicating that urban avoiders fared better in green areas with large and heterogeneous patches. Total richness was also higher in green areas surrounded by an urban matrix composed of a variety of land covers.</p> <p>To protect urban bird avoiders, which are most at risk in cities, green area managers</p>

	and urban planners should thus be aware that land cover heterogeneity is not a panacea if patch sizes are too small. To conserve avian richness, we stress the importance of maintaining large vegetated areas as well as heterogeneity in land covers within the urban matrix.
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Title

How do urban green space designs shape avian communities? Testing the area–heterogeneity trade-off

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1. Introduction

In cities, nature develops in all places but faces multiple pressures due to human activities that are more intense than in non-urban areas (Muratet & Chiron, 2021). Consequently, urbanisation has significant impacts on species distribution, ecological interactions and diversity of communities (Marzluff et al., 2008). At the city level, improving the suitability and permeability of urban habitats is essential in order to facilitate the movements of living organisms and maintain viable population sizes and complex ecological networks. In this regard, the meta-analysis of Beninde et al. (2015) stressed the importance of green space design strategies and identified land cover area and its heterogeneity as the main predictors of species richness of several taxa. For instance, species richness responded positively to area, with the strongest effects on birds followed by insects and plants. Land cover heterogeneity promotes species richness through resource use differences (i.e. niche partitioning) by increasing the opportunities for different living conditions and the coexistence of consumer species (Finke & Snyder, 2008). Habitat amount and heterogeneity (e.g. habitat diversity) have attracted much attention in ecology, as researchers seek to determine the ecological rules that shape populations and communities and objectively define the threshold or minimum viable area below which diversity decreases (Drinnan, 2005).

While the positive relationship between the heterogeneity or area of land cover and species richness is widely acknowledged, the effects of heterogeneity and area have rarely been tested together. However, these two factors are dependent. As heterogeneity increases, in the sense of diversity of different land-cover types, the total amount of a given land cover decreases within a landscape (Figs. 1a-b). In addition to the effect of compositional heterogeneity on the total amount of each land cover type, dividing a land cover patch into several units increases configurational heterogeneity and reduces the average patch size of land cover (Figs. 1c-d; Fahrig et al., 2011).

Ignoring these negative correlations, known as the “area–heterogeneity trade-off” (Allouche et al., 2012), may lead to mischaracterising the effects of land cover heterogeneity on species richness.

In this respect, Kadmon & Allouche (2012) suggested that environmental heterogeneity has a general unimodal rather than a positive effect on species richness. Using empirical data on birds, Allouche et al. (2012) validated this hypothesis and provided a comprehensive evaluation of the mechanisms underlying this pattern. For relatively low levels of heterogeneity, when area is not a limiting factor, species richness is expected to increase with heterogeneity due to species colonisation with new species adapted to the newly available niches (Figs. 1e-f). As heterogeneity continues to increase, richness is expected to decrease when the amount of effective area available for the individual species decreases, which possibly reduces population sizes and increases the likelihood of stochastic extinctions (Allouche et al., 2012). We expect that this unimodal relationship is particularly fitted to environments where the trade-off between habitat area (or mean patch area) and heterogeneity is accentuated by the limited space left for nature as in cities. In Europe, only 18% of urban areas are covered by vegetation, thus reducing the possibility for large habitat patches (Fuller & Gaston, 2009). Small elements like gardens are a major component of the green area proportion in many cities (Goddard et al., 2010). In addition to the small size of green spaces, stakeholders such as public managers and private landowners also promote a plurality of habitat conditions by dividing them into smaller lots covered by lawns, grasslands, dense or sparse tree cover and shrub layers in addition to aquatic habitats. The diversification of land cover types can thus split green spaces into very small fragments.

Despite the large diversity of green spaces in European cities, the purposes and practices behind the creation of land cover mosaics are common standards used by most city managers across the continent (Kohout & Kopp, 2020). These standards aim to provide people with cultural (e.g.,

aesthetics), social (e.g., recreational), economic and health benefits (Muratet & Chiron, 2021) while conserving urban biodiversity (Muratet et al., 2015). Managers of urban green spaces often assume that environmental heterogeneity increases species diversity (Aronson et al., 2017). It was therefore proposed that management actions should endeavour to increase the heterogeneity of habitat conditions to promote local species diversity (Kati et al., 2010). Based on this overview, decisions that promote greater heterogeneity in cities might thus lead to unintended results, especially if the overall size of the urban green spaces is small and species abundance is low.

The predicted effects of land cover heterogeneity on birds are unlikely to be uniform across species. As shown by Allouche et al. (2012), the position of the inflection point, i.e. the level of heterogeneity that maximizes richness in the unimodal relationship, depends on niche width of the species. Niche width is also termed “niche or ecological specialisation” (Devictor et al., 2010). If the patch area is not a limiting factor, land cover heterogeneity may increase richness of specialist species, and of species that specifically benefit from the juxtaposition of complementary habitats, such as woody structures for nesting or as shelter and open areas for foraging. On the contrary, generalists (i.e., species with broader niches) can persist in a wide variety of environmental conditions across space or time, are less sensitive to reduced habitat patch areas and less prone to local extinctions (Allouche et al., 2012). Richness of specialists are thus expected to increase with land cover heterogeneity (at constant patch size area), and with habitat patch size (at constant land cover heterogeneity) at a higher rate than generalists. In birds, habitat specialist species have lower dispersal abilities and cannot thrive in human-degraded ecosystems (Clavel et al., 2011). They are less abundant and decline at a greater rate than generalists along the urbanisation gradient (Guetté et al., 2017) such that avian specialisation is negatively associated with tolerance to urban stressors (e.g., pollution, fragmentation, low habitat quality) (Callaghan et al., 2020a). Testing the area–

heterogeneity trade-off in urban green spaces in terms of avian diversity and according to the degree of species urban avoidance is thus particularly relevant. To date, this has not been assessed in cities. Untangling the relative effects of area and heterogeneity on avian fauna is critical if we are to design urban green areas that can conserve the most urban-sensitive species.

In this study, we first investigated whether there is a trade-off between land cover heterogeneity and mean patch area in urban green spaces. We predicted a negative correlation between these two variables. We explored this relationship in six European cities from boreal to Mediterranean climates. Although landscape design, planting and green space management practices are often the same across cities, we expected some variations due to the geographical context and size of the cities in question (Alós Ortí et al., 2022). Second, we tested whether this trade-off shapes the avian communities and results in the expected unimodal relationship between avian species richness and land cover heterogeneity in urban green spaces. We predicted a higher bird richness at intermediate than at low and high heterogeneity levels. Finally, we evaluated the proportion of urban avoider species in cities, and if and how their traits explain the response of bird richness to mean patch area and land cover heterogeneity. We expected that urban avoider species (i.e., species that usually breed in natural or semi-natural habitats but not in human settlements or other artificial environments) would show a stronger positive relationship with mean patch area, and a stronger positive relationship with land cover heterogeneity than urban-adapted species, all other variables being equal.

2. Methods

2.1 Study areas

We sampled avian communities in six European cities: Almada and Lisbon (Portugal), Paris (France), Zurich (Switzerland), Poznan (Poland) and Tartu (Estonia). The selected cities represent a gradient of latitude and longitude to cover most of the climatic conditions in mainland Europe. To ensure the consistency of the spatial extent between cities, in some cases we used their full extent (e.g., Tartu) and in others only a section (e.g., Paris). In all cases, we verified that a density gradient of urban areas was represented by asking coordinators of the study for each city to select a spatial extent with low and high proportion of impervious areas (Pinho et al., 2021).

2.2 Sampling site selection

To avoid heterogeneity between national databases, we used a pan-European land cover classification based on the European Urban Atlas (2018), which provides maps for urban areas across Europe (i.e., minimum mapping unit = 0.25 ha, minimum mapping width = 10 m). For consistency in land use intensity and typology across Europe, we only used the “Urban Green Areas” category (hereafter abbreviated UGA, Code 14100 in the Mapping Guide v6.2 for the Urban Atlas) to select the sampled sites. UGAs are public green areas for predominantly recreational use such as gardens, parks or castle parks, or forests and other green areas extending from the surroundings into urban areas.

Within the study areas, we selected UGAs according to two independent gradients: 1) UGA size and 2) structural connectivity with green elements (i.e., UGAs and other areas considered as favourable land covers for birds such as forest and low urban density with <30% impervious surface). We calculated the degree of connectivity between green areas using the Proximity Index (PI) in Fragstats within a 5 km radius from every selected UGA. The PI is the area of all nearby

green elements (m²) within the selected radius weighted by edge to edge distance squared to a specific UGA (McGarigal et al., 2012).

All possible UGAs were classified in six size classes and six PI classes equally splitted along the two gradients and similarly across cities to allow comparisons, leading to 36 possible combinations. Within these combinations and keeping a minimum distance of 500 m between UGAs (except for two sites in Zurich, and two sites in Tartu separated by 260 m and 170 m respectively), UGAs were selected randomly. Due to UGAs limitations, we used between 15 (Almada) and 35 (Paris) combinations (total of 176 UGAs, see Table 1 for more details).

2.3 Description of land cover at UGA and landscape levels

We computed land cover composition and configuration metrics at the local (within UGAs) and landscape levels, which were potentially drivers of the avian community.

At the local level, we produced a land cover map of each UGA using the photo-interpretation of high-resolution aerial images (i.e., 0.5 m) from 2015 with ESRI ArcMap 10.4. The photo-interpretation was performed at a scale of 1:600, allowing us to distinguish between the main land cover types within green areas (Alós Ortí et al., 2022). We classified land cover into 10 categories known to influence bird diversity: allotments gardens (i.e. parcels containing cultivated areas with fruits or vegetables, as well as a shed for tools and shelter), artificial surfaces (i.e. houses, roads and playgrounds), bare land, coniferous trees, broadleaved trees, lawns and grasslands, lakes and ponds, riparian vegetation, rivers and streams, and shrubs (Bino et al., 2008). In each UGA, we mapped all patches of land cover. Aerial images did not allow us to precisely classify the extent of all land cover types due to the overlapping vertical layers. Therefore, our land cover maps

provided accurate information about tree canopy (i.e. the upper layer) and the other layers not covered by trees. Based on this mapping, we measured UGA total area and the heterogeneity of land cover categories using Shannon's diversity index (Fahrig et al., 2011). The Shannon index is the most applied metric to capture the variety and the richness of land use/land cover and its characteristics in a given extent or spatial unit (Zhang et al., 2023). For UGA land cover configuration, we calculated mean patch area and contagion index, which measures the degree of aggregation of the same land cover within each UGA (Riitters et al., 1996). This index is computed based on the frequencies by which different pairs of land covers occur as adjacent pixels on a map. To describe land cover composition at the landscape level, we used all land cover classes of the Urban Atlas and calculated the land cover heterogeneity using Shannon's diversity index in a 500 m buffer distance from the edge of the UGAs. This distance is a good compromise to detect the maximum landscape effect on diversity and composition of urban bird communities (Melles et al., 2003; Chang et al., 2017; Villaseñor et al., 2021), while minimizing overlap between buffers and ensuring independence in landscape analysis (pairwise distances among UGAs were larger than 750 m in 99% of cases, Zuckerberg et al., 2020). Because buffer area depends on the UGA size and its shape, we controlled for the effect of the buffer area around each UGA on landscape heterogeneity. We used the residuals of the relationship between the land cover heterogeneity and buffer area in the statistical analyses. For landscape connectivity, we used the PI as calculated above (i.e., within a 5 km radius from every selected UGA). Except for the PI, all local and landscape metrics were calculated in R using the "landscapemetrics" package (Hesselbarth et al., 2019).

2.4 Bird sampling

In 2019, we counted birds using one point count per UGA, located as close as possible to its centroid on a location where the main land cover types (trees, bush, grassland, water) are present in a 50 m radius. In this manner, we collected information on potential bird species living in different types of land covers.

Within each country, we hired an experienced birdwatcher to count birds within 4 hours from daybreak to standardise the time of sampling along latitudes. Each plot was sampled twice to detect both more residential and more migratory bird species during the breeding season. First counts started early April (for Almada and Lisbon) and second counts ended late May (for Tartu and Poznan) with at least 3 weeks between the two counts for each city. We adapted the sampling day to the local weather conditions to avoid rain, fog and wind. All observations were conducted on weekdays from Monday to Friday. UGAs surveyed on the same day had the order of sampling randomised for the second survey to ensure that the time of the day did not influence species detection.

During each visit, the birdwatcher counted and identified all individuals heard or seen for 10 min. This duration is a minimum but represents a non-negligible investment of time that we considered as reasonable to detect a large proportion of bird species at sampling point and estimate bird responses to main local and landscape drivers (Voříšek, 2008). Birds were counted inside a 100 m radius area around the observer and within UGA's boundaries. Birds flying over the sampling point (transients) were recorded separately, as they neither arrived or left the sampled area during the point count nor clearly indicated their use of the area as a breeding territory. However, flying birds feeding or hunting in the air (i.e., swifts) were noted as observed within the 100 m radius. Species richness was the total number of species counted within a 100 m radius during the two visits for each UGA.

We used a constant-area sampling (i.e. inside a 100 m radius area) rather than an area-balanced approach with more sampling points in the largest UGAs for two reasons. First, we aimed at understanding influences of variables shaping the area - heterogeneity trade-off without the influence of sampling effort. Second, replicating count points in large green areas is cumbersome, demanding a lot of field work if we wanted to cover the full green area extent (e.g. one point count location for every 250 m by 250 m area). To avoid potential bias due to our sampling strategy, we verified that counting area (3.14 ha) covered the full extent of most UGAs (63% of them were smaller than this area) or was representative of the UGA land cover heterogeneity. For UGAs bigger than 3.14 ha, there was a good correlation in land cover heterogeneities between the UGA and the counting area levels, and a good correlation in mean patch areas between these two levels too (Pearson's correlation coefficient $\rho = 0.77$ and 0.97 respectively, $p < 0.01$). Bird sampling was thus conducted in an area representative of the whole UGAs studied.

As ambient noise during bird surveys could influence bird detection, we accounted for in statistical models (see below). During the first 4 minutes of counting birds, the birdwatcher collected noise data and measured the maximum sound volume in decibels (dB) in the environment for 1 minute in each of the four cardinal directions using a digital decibel metre (device Preciva®, measuring range: 30-130 dB; accuracy: ± 1.5 dB) set on a tripod at the height of 1 metre. We then calculated the mean "maximum sound volume" (dBmax) as we surveyed each point twice (temporal replicate). Any potential noise peak bias during the 1-minute sampling period would be smoothed by the second survey. Finally, we recorded temperature (in °C) and wind (Beaufort scale) to exclude potential observations collected during bad weather conditions.

2.5 Defining avian avoidance in urban areas

We built a continuous gradient of urban avian avoidance, which is consistent with the breeding ecology of species, and integrated a variety of habitat requirements for nesting (Keller et al., 2020). For that purpose, we used the life-history trait database of Storchová & Hořák (2018), which compiles information about bird occurrences in 15 nesting habitats: deciduous forest, coniferous forest, woodland, shrub, savanna, tundra, grassland, mountain meadows, reed, swamps, desert, freshwater, marine, rocks and human settlements. This information was derived from *The Birds of the Western Palearctic* handbook (Cramp, 2006) and covers most bird species breeding in the Western Palearctic. Habitat occurrences for four species (*Anser anser*, *Bubulcus ibis*, *Egretta garzetta*, *Nycticorax nycticorax*) are missing from Storchová & Hořák (2018). We removed them from the habitat preference analysis and regression models using this information. We then carried out principal component analysis (PCA function, R package FactoMineR; Husson et al., 2007) using information about bird occurrences in each nesting habitat. We evaluated the multivariate relationships between species coordinates on principal components and habitat variables to define the avian avoidance of urban areas (see Results).

Our approach differs from using expert knowledge to define species preferences to cities. Expert-based knowledge is valuable, but species ranking along a gradient of urban avoidance may vary between experts or countries, leading to oversimplified categories (“urban dweller” vs “urban avoider”). Even though Callaghan & colleagues (2020b) previously estimated urban avoidance for European birds, we could not use their database because 25 species included in our study were missing from it (21% of the total number found here).

2.6 Statistical analyses

2.6.1 Trade-offs between mean patch areas and land cover heterogeneity of UGAs

To evaluate the area-heterogeneity trade-offs in our study area, we fitted a linear mixed model with “UGA land cover heterogeneity” (Shannon’s index) and “UGA total area” (log-transformed) as predictors of “UGA mean patch area” (log-transformed) (R package lme4; Pinheiro et al., 2021). In this model, we used the whole dataset, i.e. all cities, and added “city” as a random factor to account for the nested design of the study (Zuur et al., 2009). To assess the robustness of these relationships at the city level, we separately repeated the analysis for each city using a linear model and the same predictors.

2.6.2 Local and landscape effects on bird richness

We built a generalised linear mixed model using a Poisson family (glmer, R package lme4; Pinheiro et al., 2021) with “UGA land cover heterogeneity” and its squared value as predictors of avian richness to test for non-linear effect of the area–heterogeneity trade-off. As fixed effects, we added other variables that might influence birds: “land cover heterogeneity of landscape” (Shannon’s diversity), “UGA total area” (log-transformed), “UGA mean patch area” (log-transformed) and “proximity index”. We included as a potentially confounding variable “maximum sound volume” (dBmax). We did not include the “UGA contagion index” from the list of predictors as it was significantly correlated with “UGA land cover heterogeneity” (Pearson’s correlation coefficient $\rho = -0.72$ with p-value < 0.01 ; Appendix H). We used the whole dataset and added “city” as a random factor. Additionally, we repeated this analysis for each city separately using generalised linear models (glm) with the same predictors.

2.6.3 Influence of urban avoidance on the response of species richness to local and landscape variables

Based on the PCA, we assigned each bird species to a coordinate on the second component (PC2) representing their level of urban avoidance (Appendix A). We then categorised species into three terciles corresponding to minimum, intermediate and high values of urban avoidance along PC2. To define terciles, we didn't equally split the urban avoidance gradient because species number would have been very unbalanced making statistical analysis problematic. Our splitting minimises species number differences among terciles which resulted in 37, 45 and 33 species for minimum, intermediate and high values of urban avoidance respectively.

For all UGAs, we calculated the avian richness of terciles as the number of species belonging to each urban avoidance tercile, and used it as the response variable in a glmer using a Poisson family. As fixed effects, we included "UGA land cover heterogeneity" and its squared value to test for its non-linear effect, "UGA total area" (log-transformed), "UGA mean patch area" (log-transformed), "urban avoidance tercile", "proximity index", "land cover heterogeneity of landscape" and "maximum sound volume". We incorporated the interactive effects between tercile group and "UGA land cover heterogeneity", and between tercile group and "mean patch area" into the model to assess the effect of urban avoidance on the response of species richness to UGA variables. We added "city" as a random factor to account for the nested design of the study. Finally, we computed a chi-squared (χ^2) test on the response variable (i.e., avian richness of terciles) to assess variability in the distribution of tercile avian richness between cities.

Prior to fitting the models described above, we standardised the explanatory variables by subtracting the mean and dividing by the standard deviation. We assessed statistical significance

using the ‘Anova’ function (type II, with χ^2 and Z statistics for the Poisson and Gaussian families, respectively) and performed Tukey HSD post-hoc tests (‘TukeyHSD’ function) to compare avian richness between cities and terciles.

To check for multicollinearity between explanatory variables, we computed variance inflation factors (VIF) using *check_collinearity* function (R package performance). To alleviate moderate level of multicollinearity ($VIF < 4$) between some explanatory variables (Zuur et al., 2009), we then applied a hierarchical partitioning procedure (R packages ‘hier.part’ for glm and ‘partR2’ for glmer) to obtain independent contribution of each explanatory variables based on semi-partial R^2 and quantified its uncertainty using parametric bootstrapping (Walsh & McNally, 2008; Stoffel et al., 2021). For all models, we statistically verified that the residuals met the assumptions of homoscedasticity and normality using DHARMA package (Hartig, 2021). For Poisson models, absence of residuals’ overdispersion was checked (‘testDispersion’ function). We also graphically confirmed the spatial independence of the residuals by computing a spline correlogram using the ‘spline.correlog’ function (package ncf; Bjørnstad, 2019). Graphs were generated using ‘ggplot2’ package in R (Wickham, 2019; version 3.3.3). All analyses were performed in R version 4.1.2 (R Core Team, 2021).

3. Results

In total, we recorded 119 nesting bird species (7523 individuals) in the 176 green areas studied across the six European cities (see the list of species in Appendix B). Mean bird richness per UGA was 14.7 species (between 4 and 28, Table 1).

3.1 Relationships between mean patch area, UGA land cover heterogeneity and avian richness

Results from our linear models (Appendix C) confirmed a significant area-heterogeneity trade-off in urban green areas, as mean patch area in UGAs was negatively associated with land cover heterogeneity (Appendix I). Additionally, mean patch area was positively associated with UGAs' total area, as expected. These relationships were consistent across cities.

Species richness was positively correlated with UGA total area and land cover heterogeneity at the landscape level (Figs. 2b-c). Contrary to our prediction and after controlling for effects of these two latter variables we found no effect of UGA land cover heterogeneity on avian species richness (neither linear nor unimodal) in a model combining all cities (Appendix D, Figs. 2a). At city level, we found a unimodal relationship between land cover heterogeneity and avian richness for Zurich only (Appendix E).

3.2 Continuum of urban avoidance trait in birds

Based on the trait database of Storchová & Hořák (2018), we found that the birds encountered in UGAs have nesting habitats encompassing all categories found in Europe from marine and coastal habitats to swamps and forests, with the exception of deserts. The most frequent habitats occupied by the observed bird species were deciduous forests (56 sp.), coniferous forests (41 sp.) and human settlements (35 sp.). The vast majority of species occupied different habitat types (2.6 on average), although 17 species (14%) were attributed to a single habitat. Among them, two species nest only on human settlements (*Streptopelia decaocto* and *Alopochen aegyptiaca*), making them strict urban dwellers (Fischer et al., 2015).

The PCA for the nesting habitat trait revealed three principal components, which explained 20.0%, 14.2% and 11.1% of the total variance, respectively (Figs. 3a-b, Appendix F). The first axis (PC1) separated forest and woodland species from marine and freshwater species (“forest-grassland and aquatic species” gradient). The second axis (PC2) separated species of human settlements and rocks from those living in reeds, swamps, shrubs and tundra, thus representing a gradient of “avian avoidance of urban areas”. Finally, the third axis (PC3) did not show any clear pattern; it will not be discussed further.

To assess the robustness of PC2 as an index of urban avoidance, we correlated it with a continuous measure of urban tolerance (i.e., “urban score”) calculated based on the eBird observations of bird occurrences along a gradient of urbanisation in Europe (Callaghan et al., 2020b). This score was calculated as the distributional response of species to VIIRS night-time lights, a proxy of human population density and urbanization. Although some species were lacking from the urban score (21% of our species list), the two indices were negatively correlated (Pearson’s correlation coefficient $\rho = -0.45$ with $p\text{-value} < 0.001$), thus validating the use of PC2 as an urban avoidance index. Our measure extended the calculation of urban avoidance to 25 other species in European cities.

3.3 Avoider species and their variation among cities

Based on PC2 coordinates (Fig. 3b), we grouped species into three terciles. Tercile 1 (PC2: -3.2 to -0.69, 37 species, low urban avoidance) consisted of urban dweller species such as *Columbia livia*, *Streptopelia decaocto* and *Apus apus* that nest on man-made structures, and urban utiliser species such as *Corvus corone* and *Columba palumbus* that are so-called generalist species. Tercile 2 (PC2: -0.69 to 0.11, 45 species, medium urban avoidance) grouped avoider species such as *Dryocopus*

martius and *Buteo buteo* that usually nest outside developed urban areas in forest patches but can self-sustain populations in large contiguous UGAs. This tercile also included a few urban utilisers like *Turdus merula* and *Anas platyrhynchos*. Tercile 3 (PC2: 0.11 to 5.65, 33 species, high avoidance) was composed of aquatic (e.g., *Remiz pendulinus*, *Cettia cetti*, *Acrocephalus schoenobaenus*) as well as shrub and grassland species (e.g., *Lululla arborea*, *Hippolais icterina*), which very rarely occur in urban areas. Although such species are generally absent from developed areas, they may persist in networks of natural areas embedded in an urban matrix (Fischer et al., 2015).

The number of species in the three terciles varied among cities ($\chi^2=39.18$, $df=10$, $p\text{-value}<0.001$). The proportion of species belonging to the high avoidance group (Tercile 3) was higher in Poznan and Tartu (28% of species) than in Zurich, Almada, Paris and Lisbon (between 10% and 18%, Fig. 4, Appendix G).

3.4 Influence of urban avoidance on the response of species richness to local and landscape variables

We found significantly contrasting responses of species richness to mean patch area of UGA depending on the level of urban avoidance, shifting from negative for species associated with human settlements (Tercile 1, low avoidance) to positive for urban avoider species (Terciles 2 and 3 for medium and high avoidance groups respectively, Fig. 5a, Table 2). We found similar relationships regarding the species richness response to land cover heterogeneity of UGA depending on the level of urban avoidance of birds. Relationships gradually shifted from negative to positive between species associated with human settlements (Tercile 1) and urban avoider species (Terciles 2 and 3) (Fig. 5b, Table 2). Variance contribution of UGA land cover

heterogeneity (including its interactive effect with urban avoidance level) to species richness variation ($R^2=0.031$, Table 2) was however lower than contribution of UGA mean patch area ($R^2=0.056$, Table 2).

4. Discussion

4.1 Area–heterogeneity trade-off and their effect on avian richness

Our results highlight that the mean patch area and land cover heterogeneity of UGAs are not independent factors: as heterogeneity increases, land cover patch size decreases. This pattern is the consequence of dividing UGA area into different land cover types (i.e., heterogeneity of composition) and the same land cover into small fragments (i.e., heterogeneity of configuration or fragmentation). This relationship was consistent across cities and confirms that despite the “manipulated nature” of UGAs, their design is governed by the same rules as in less human-dominated environments (Parris, 2018). The area–heterogeneity trade-offs thus apply to urban landscapes, which underscores the need to study patch area and land cover heterogeneity in the same framework of analysis if we want to disentangle their roles in avian diversity.

Contrary to our expectations, we found neither positive nor unimodal relationships between land cover heterogeneity and avian species richness within UGAs, as proposed by Allouche et al. (2012). Despite the different approaches used for measuring the heterogeneity of local conditions in UGAs (e.g., through land cover richness, vegetation structure or its complexity, presence of microhabitats), empirical studies generally observed a positive correlation between heterogeneity and species richness (Husté et al., 2006; Nielsen et al., 2014; González-Oreja et al., 2012). This relationship was illustrated for various taxonomic groups such as plants, birds, mammals and

reptiles (Cornelis & Hermy, 2004; Garden et al., 200 ; Matthies et al., 2017). In these studies, part of the species-heterogeneity relationships can be attributed to larger parks that tend to harbour a greater number of habitat units (e.g., lawn, forest, pond, etc.) and hence greater species richness than smaller ones (Cornelis & Hermy, 2004), but this relationship was also found in small parks (between 0.1 and 2 ha in Pachuca city, Mexico, Carbó-Ramírez & Zuria 2011). Controlling the effect of land cover heterogeneity by UGA area (total or mean patch size) in our study should thus not entirely explain the gap between our results and previous studies.

The species-heterogeneity relationship is an essential element of niche-based theories of species diversity and a key concept of the unimodal relationship presented by Allouche et al. (2012): environmental heterogeneity should provide suitable conditions for a larger number of species with different ecological requirements, thereby increasing the local species pool (Chase & Leibold, 2003). No relationship with heterogeneity may result from the reduced land cover diversity in the studied UGAs, which hampered our ability to detect changes in avian communities. This is unlikely because our sampling clearly evidenced variability in the land cover types (Table 1). The absence of a relationship might also arise from the relatively limited occurrence of key habitats such as aquatic environments. The occurrence of waterbodies such as lakes, ponds, river streams and riparian habitats has plummeted in urban ecosystems due to significant degradation from drainage and changes in the hydrological regimes of cities (Moggridge et al., 2014). In our sampling, these environments were underrepresented compared with terrestrial ones. However, 34% of sampled species (n=40) were associated with one or several aquatic habitats, thus indicating that these species were not absent from our sampling.

Finally, birds may only occupy a few patches among those accessible for nesting in UGAs, thus making the relationship between bird richness and land cover heterogeneity inaccurate at the UGA

level. For instance, ground nesting birds rarely reproduce in open urban habitats like lawns and grasslands because of multiple threats, which could affect the behaviour, reproduction and survival of chicks and adults (e.g., higher disturbance caused by pedestrians, domestic animals, or management actions like grass cutting, (Fernandez-Juricic & Jokimäki, 2001)). In urban woodlands, cleared understory vegetation (no shrub or herbaceous vegetation layers, and dead wood being removed) may increase disturbances and predation risks and limit food availability (Crocì et al., 2008).

Avian richness was positively influenced by the total area of UGAs. Larger UGAs increase the effective area available for individual species, thereby increasing population sizes and decreasing the likelihood of stochastic extinctions (Allouche et al., 2012). Our results confirmed the species-area relationship: a well-known pattern already observed in birds, mammals and plants within UGAs and in more natural ecosystems (Nielsen et al., 2014; Matthies et al., 2017). Interestingly, avian richness also increased with land cover heterogeneity in the neighbouring landscape. The land cover heterogeneity of the adjacent matrix is related to the diversity of terrestrial and aquatic natural habitats. Birds nesting in these habitats may spill over into UGAs and use them as foraging, nesting or refuge sites (Mayorga et al., 2020).

4.2 Urban avoider species richness is greater in large and diversified green patches of land covers

The share of urban avoider species (terciles 2 and 3) in terms of species richness was greater in urban green areas with large land cover patches than in small ones. Richness of urban avoiders can simply be explained by habitat preferences and distribution of selected habitats among small and large green patches.. Also, urban avoiders are less able to persist in small habitat patches (high edge-to-area ratio), because they prefer the interior of large patches, similar to many specialists of

forest and aquatic ecosystems (Fernandez-Juricic, 2001). Within-patch habitat quality is unlikely to be constant as UGA size and heterogeneity change. On the contrary, the number of urban adapted species (tercile 1, urban dwellers and utilisers) decreased with patch size within UGAs. Urban dwellers can persist in urban environments independently of the natural areas, while urban utilisers can exploit resources supplied by human activities (Fischer et al., 2015). The latter can be widely distributed and locally abundant in cities (Ducatez et al., 2018). Being mostly generalist, they can thrive in fragmented environments, because they are able to nest and feed in various environments.

Competitive interactions between urban adapted and avoider species may also explain these relationships. Urban adapted species may gradually disappear in large natural patches to avoid increasing competition with urban avoiders, which are specialist birds (Stade et al., 2021). This process could partly explain the segregation of habitat specialist and generalist birds between large and small patches. In small and fragmented patches, urban adapted birds can be widespread and behaviourally dominant species (Ducatez et al., 2018). They can also exclude rarer and more subordinate species such as urban avoiders through competitive interference (Martin & Bonier, 2018).

Although we did not find any effect of land cover heterogeneity of UGAs on total avian richness, we evidenced its positive effect on the richness of urban avoider species. Because these latter are often habitat specialists, land cover heterogeneity can provide the suitable conditions for more species with different ecological requirements, thereby increasing their local species pool (Chase & Leibold, 2003). Since land cover heterogeneity is negatively correlated with mean patch area of UGAs, effect of land cover heterogeneity on urban avoider species cannot be analysed apart from

the effect of mean patch size. We thus revealed a trade-off on species richness depending on species urban tolerance.

4.3 Relevance of green space design for bird conservation

We reported a relatively rich bird diversity (21% of European breeding bird species, Keller et al., 2020), thus making UGAs relevant areas for bird conservation. Most of these areas correspond to parks, renowned as urban biodiversity hotspots (Nielsen et al., 2014). It was therefore crucial to understand how designing UGAs for nurturing a diversity of bird species. Achieving this target is the result of a compromise between land cover surface area and its heterogeneity.

Although a recent analysis on the importance of natural areas recognised the value of small spaces for biodiversity conservation (Riva & Fahrig, 2022), we highlight the necessity to put effort into protecting large UGAs composed of big enough land cover patches to conserve bird richness. Compared to small patches, large areas provide better protection for urban avoiders, which are most at risk globally (Clavel et al., 2011). Even though small green areas such as woodlots and public or private gardens are widespread and sometimes the only natural areas in cities, large areas are scarce and regularly exposed to urban development pressure (Fuller & Gaston, 2009).

Land cover diversification of UGAs (e.g., mixing woodlots, grasslands, waterbodies, and so on.) may increase the potential number of ecological niches and hence species richness of local assemblages (Callaghan et al., 2019). Restoring or protecting different land cover types is achievable in large UGAs where space is not a limiting resource compared to smaller ones. In small UGAs however, land cover diversification should not be regarded as a panacea by urban planners (Kati et al., 2010) because it could lead to a drastic reduction in the average amount of a

given habitat and its effective area available for specialist birds or birds that exploit complementary habitats. This may unintentionally reduce the occurrence of the most endangered species in cities. For birds, a minimum area of 4.4 ha and of 53.3 ha were described to prevent the loss of urban adapters and of area-sensitive species like urban avoiders (Beninde et al., 2015). We thus recommend limiting land cover heterogeneity on small UGAs to maintain green patches as large as possible.

To optimize biodiversity at city level, activities of design and management should be coordinated all across a city. Actions should diversify ecological conditions across green areas and not just within. In small areas, maximizing the coverage of a specific habitat type but different across UGAs would maintain urban avoiders populations in local species assemblages and enhance beta-diversity. Actions should target ecological context from which UGAs are originate to preserve their uniqueness as city tend to homogenize local conditions (Groffman et al., 2014). Although this could work for biodiversity, designing UGAs dominated by a few land cover types in small UGAs could be negatively perceived by users if it does not take into consideration preference of people for some green area characteristics (e.g. presence of open areas, playgrounds, flowerbeds, waterbodies, tree cover, see Palliwoda and Priess, 2021 and Madureira et al., 2018). A key challenge for UGA design is thus combining human needs with ecological requirements for preserving and enhancing avian communities especially green area sensitive species.

Finally, our study pointed out the importance of maintaining a variety of land cover types in the urban matrix surrounding green areas, which appears to be as important as within UGA heterogeneity for conserving local bird assemblages. The ongoing urban densification process observed in some European cities (Li et al., 2022) can thus pose a threat by reducing land cover diversity (e.g. through the loss of private gardens or rare habitats) and richness of bird species in

UGAs. Conservation strategies should thus promote actions at the city level to conserve land cover heterogeneity and high quality habitats closed to public green areas.

5. Conclusions

We evidenced the marked land cover area-heterogeneity trade-off in urban green areas. We showed the predominant role of green area size on avian richness. Urban avoiders thrive in urban green areas with large and heterogeneous patches but are at risk in smaller ones unless they exhibit large habitat patches. We suggest the importance of coordinating design and management actions at the city levels across green areas to optimise bird conservation depending on UGA size and their heterogeneity level. We also highlight the need to confront ecological theories with empirical data across different bioclimatic zones to better understand urban ecosystems and advise stakeholders with robust and ecologically informed knowledge.

Our study opens the door to future research questions. We intentionally used a trait to quantify species preference to urban habitats and indirectly to the notion of habitat specialisation, which lies at the heart of niche theory. To go further, future research should investigate the species colonisation/extinction processes in land cover patches. For this purpose, long-term studies are required on persistence across small patches versus large ones (or across homogeneous versus heterogeneous). Also, the role of species interaction in the distribution of urban generalists and specialists (avoiders) along a urban gradient remains to be explored. Whether the increase in generalists is due to the relaxed interspecific competition from specialist species is unknown. Although our study assessed the trade-off in different biogeographical contexts, whether the area-heterogeneity relationship hold true in more specific environments (e.g. aquatic or terrestrial) has not yet been explored.

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Table 1: Summary table of the urban green areas (UGAs) and landscape areas in the six cities investigated in this study as well as the avian species richness expressed as mean, minimum and maximum values. Land cover heterogeneity of UGAs and landscapes corresponds to the Shannon diversity index of habitat types respectively within UGAs (i.e., ten land cover categories) and within a 500-metre buffer around the UGAs (i.e., land use categories of the Urban Atlas). The contagion index measures the degree of aggregation of the same land cover within each UGA. dBmax is the maximum number of decibels in the environment (dBmax) during bird counts in each UGA. The proximity index measures the degree of green area isolation and is the sum of the area of all nearby green spaces divided by the square of the distance to the focal green area. Climate data were collected for the variables of temperature (Mean temp.) and precipitation (Annual prec.) between 1991 and 2021 and are issued from the European Centre for Medium-Range Weather Forecasts (ECMWF, source www.ecmwf.int, access 23/06/2022). All coordinate data for the cities are taken from the OpenStreetMap project (source <https://www.openstreetmap.fr>, access 23/06/2022).

Table 2: Results of regression model analysing the influence of urban avoidance (tercile groups) on the response of species richness to local variables of UGAs and landscape (500 m buffer around each urban green area, UGA) across the six cities. Our model controlled for maximum noise (dBmax) and included 'city' as random effect to account for possible consistent differences among cities. (UGA land cover heterogeneity)² is the squared term of UGA land cover heterogeneity to test for a non-linear relationship. We reported the slope estimate, its standard error (Std. Error), degree of freedom (df), the p-value (model without the interaction effect), and semi-partial R² (part R²) values that provide an estimate of the variance explained by a fixed effect independent of other predictors. Total variance explained by the model was 0.692 (Marginal R² value). Significant variables are highlighted in bold in the table (*p<0.05, **p<0.01, ***p<0.001). Statistical tendencies (0.050<p<0.010) are followed by a full stop (.).

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City	Number of UGAs	Urban green area mean (min;max)				Landscape (500 m) mean (min;max)		City climate		City coordinates		Species richness mean (min;max)		
		Area (ha)	Mean patch area (ha)	Land cover heterogeneity	Number of habitat types	Contagion index	dBmax (decibels)	Land cover heterogeneity	Proximity index	Mean temp. (°C)	Annual prec. (mm)		Longitude (deg.)	Latitude (deg.)
Almada	15	6.7 (0.4;43.2)	0.1 (0.02;0.36)	1.2 (0.4;1.8)	5.3 (2;10)	59.8 (48.5;68.7)	64.9 (60.1;71.1)	1.6 (1.2;2.0)	3.1 (0.2;10.0)	16.5	591	38°40'00"N	9°0'00"W	14.5 (9;22)
Lisbon	31	5.3 (0.4;31.4)	0.2 (0.03;0.70)	1.1 (0.1;1.7)	5.0 (2;8)	61.1 (49.8;88.3)	66.9 (61.6;81.9)	1.7 (1.1;2.0)	2.8 (0.4;9.1)	16.7	591	38°43'N	9°08'W	14.6 (4;22)
Paris	35	3.2 (0.2;18.5)	0.1 (0.02;0.42)	0.9 (0.4;1.7)	4.6 (2;7)	63.4 (43.4;83.7)	67.9 (52.9;119.0)	1.4 (0.7;2.0)	3.4 (0.5;9.5)	11.7	720	48°51'24"N	2°21'07"E	12.6 (6;19)
Poznan	32	7.8 (0.4;108.7)	0.2 (0.03;2.41)	0.9 (0.3;1.3)	4.3 (2;8)	63.0 (44.9;88.4)	68.9 (59.7;72.8)	1.6 (0.8;2.1)	3.8 (0.7;9.4)	9.7	649	52°24'00"N	16°55'00"E	18.3 (10;28)
Tartu	32	5.3 (0.3;32.0)	0.4 (0.03;3.48)	0.8 (0.0;1.4)	4.6 (1;7)	68.7 (50.1;100)	69.9 (49.5;64.2)	1.7 (1.2;2.0)	2.9 (0.6;6.4)	6.4	727	58°22'44"N	26°43'12"E	14.6 (8;23)
Zurich	31	3.8 (0.2;28.2)	0.1 (0.02;0.14)	1.1 (0.2;1.4)	4.8 (3;7)	60.5 (48.9;86.0)	70.9 (54.4;70.2)	1.8 (1.5;2.3)	4.1 (1.5;10.2)	9.7	1460	47°22'40"N	8°32'28"E	13.8 (6;18)

Table 2: Results of regression model analysing the influence of urban avoidance (tercile groups) on the response of species richness to local variables of UGAs and landscape (500 m buffer around each urban green area, UGA) across the six cities. Our model controlled for maximum noise (dBmax) and included ‘city’ as random effect to account for possible consistent differences among cities. (UGA land cover heterogeneity)² is the squared term of UGA land cover heterogeneity to test for a non-linear relationship. We reported the slope estimate, its standard error (Std. Error), degree of freedom (df), the p-value (model without the interaction effect), and semi-partial R² (part R²) values that provide an estimate of the variance explained by a fixed effect independent of other predictors. Total variance explained by the model was 0.692 (Marginal R² value). Significant variables are highlighted in bold in the table (*p<0.05, **p<0.01, ***p<0.001). Statistical tendencies (0.050<p<0.010) are followed by a full stop (.).

Predictor of species richness	Estimate	Std. Error	Chisq-value	df	p-value	part R ²
(Intercept)	1.980	0.123	40.240	1	<0.001***	
Mean patch area (log)	0.064	0.045	2.070	1	0.150	<0.001
UGA total area (log)	0.103	0.036	8.385	1	0.004**	0.004
UGA land cover heterogeneity	0.021	0.031	0.043	1	0.510	<0.001
(UGA land cover heterogeneity) ²	-0.012	0.015	0.732	1	0.392	<0.001
Proximity index	0.003	0.021	0.018	1	0.893	<0.001
Landscape land cover heterogeneity	0.056	0.024	5.519	1	0.019*	0.005
dBmax	-0.050	0.028	3.346	1	0.067.	0.005
Urban avoidance			527.052	2	<0.001***	<0.001
Tercile 1	1.980	0.123				
Tercile 2	-0.338	0.042				
Tercile 3	-1.424	0.062				
UGA land cover heterogeneity: Urban avoidance			6.390	2	0.041*	0.031
UGA land cover heterogeneity (tercile 1)	-0.020	0.040				
UGA land cover heterogeneity (tercile 2)	0.060	0.046				

	UGA land cover heterogeneity (tercile 3)	0.166	0.067				
Mean patch area (log): Urban avoidance				19.925	2	<0.001***	0.056
	Mean patch area (log) (tercile 1)	-0.045	0.049				
	Mean patch area (log) (tercile 2)	0.182	0.049				
	Mean patch area (log) (tercile 3)	0.242	0.070				

Figure 1: Area–heterogeneity trade-offs in urban ecosystems.

When the urban green area (UGA) is limited (urban parks are perfect examples of such non-extensible zones), the area–heterogeneity trade-off (a) describes the decrease in the area of various land covers expected with an increase in land cover heterogeneity (i.e., compositional heterogeneity) (Allouche et al., 2012). For example, starting from a green space area ‘A’ with two equally distributed land covers (black and dark grey squares; $A/2$ each), increasing land cover heterogeneity to four land cover types decreases the area of each one ($A/4$ in this example). (b) In our sample of 176 UGAs from six European cities (see their distribution on the inserted map; Lisbon and Almada overlap at this scale), we can empirically confirm the expected relationship displayed in (a). Note that the y-axis in (b) is the area of land covers of a given UGA standardised by its total area, which accounts for variability in the sampled UGAs (i.e., the relationship in (a) is expected at a constant area). (c) Theoretically, while compositional heterogeneity is independent from configurational heterogeneity [6], both co-occur in European UGAs. Thus, the reduction in patch area (i.e., measure of configurational heterogeneity) for all the types of land cover present here occurs when land cover heterogeneity (i.e., compositional heterogeneity) increases. (d) This is verified by the relationship displayed in (c) in the studied urban green areas. Note that the y-axis in (d) is the mean patch size of a given UGA divided by its total area to account for variability in our UGAs (i.e., the relationship in (c) is expected at constant area). (e) Land cover heterogeneity allows for the colonisation of new species adapted to the newly available niches. Concurrently, the decreasing area of each land cover type (see (a-b)) can push species to local extinction, and species requiring the largest effective area of a given land cover are at the highest risk (Allouche et al., 2012 ; Ben-Hur & Kadmon, 2020). (f) The area–heterogeneity trade-off thus leads to the hypothesis that species richness has a unimodal response to land cover heterogeneity: when reaching the level of heterogeneity at which extinction and colonisation are equal, further land cover diversification decreases species richness. The exact types of relationships depicted here (a, c, d, e) can differ depending on the studied ecosystem and species (Ben-Hur & Kadmon, 2020), as illustrated by the differences between panels (a) vs (b) and (c) vs (d). While the area–heterogeneity trade-off and its implications are still under debate (Carnicer et al., 2013), here we investigate whether the area–heterogeneity trade-off holds true in UGAs, leading to a unimodal response of species richness in avian communities to land cover heterogeneity.

Figure 2: Effects of urban green area (UGA) and landscape (500 m buffer around each UGA) variables on species richness of avian communities for all cities (a). We computed the estimates and confidence interval (CI) based on the regression models presented in Appendix D. (UGA land cover heterogeneity)² is the squared term of UGA land cover heterogeneity. dBmax is the maximum sound volume during bird counting. We provide estimated mean relationships between species richness and total area of UGAs (b ; $P=0.004$) and landscape land cover heterogeneity of UGAs (c ; $P=0.020$). We used different dot shapes for each city: Pa, Ta, Lx, Po, Al and Zu stand for the cities of Paris, Tartu, Lisbon, Poznan, Almada and Zurich, respectively.

Figure 3: Principal component analysis with (a) variables and (b) individuals (i.e., species coded as the first three letters of the genus and species; see Appendix A for a summary of the species names) with regard to nesting bird habitat preferences. This analysis positioned the species along a gradient of urban avoidance on PC2 from low avoidance (low value) to high avoidance (high value). The blue colour gradient indicates the contribution of variables to PC1 and PC2 (a), and the different colours in (b) indicates the species according to terciles 1 (low urban avoidance; green), 2 (medium urban avoidance; orange) and 3 (high urban avoidance; purple).

Figure 4: Avian species richness for the different levels of avian urban avoidance (terciles) in the six cities. For each city, we pooled the bird species belonging to terciles 1 (low avoidance; green), 2 (medium avoidance; orange) and 3 (high avoidance; purple). We ordered the cities according to the total number of species from the least to the most. We provided the proportion of species per tercile in white.

Figure 5: Responses of species richness to the mean patch area (a) and land cover heterogeneity of UGAs (b) according to the level of avian urban avoidance: low urban avoidance (green, tercile 1), medium avoidance (orange, tercile 2) and purple (high avoidance, tercile 3). Grey shaded bands indicate are 95% confidence intervals estimated from the generalized linear mixed model (Table 2).

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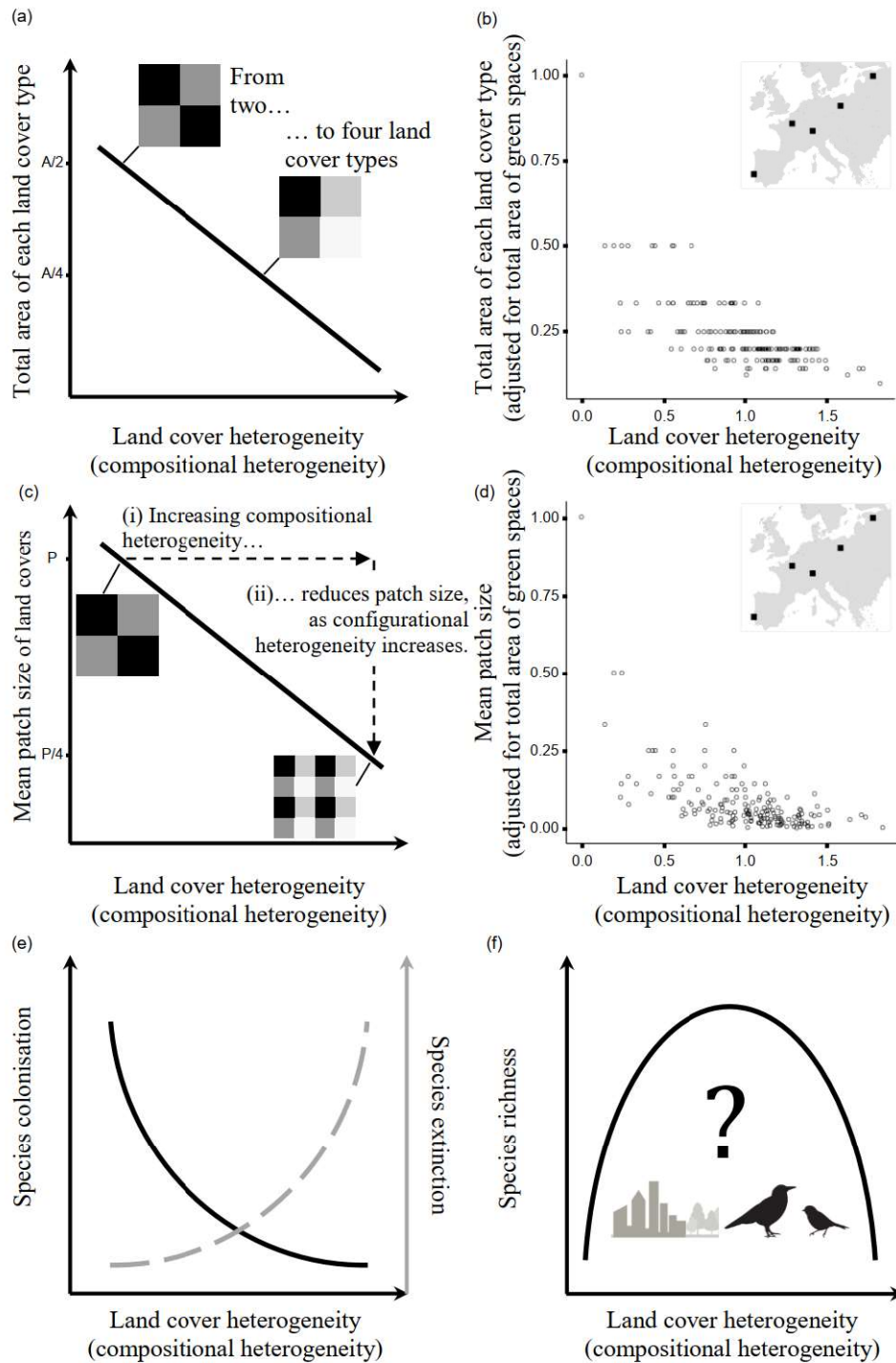
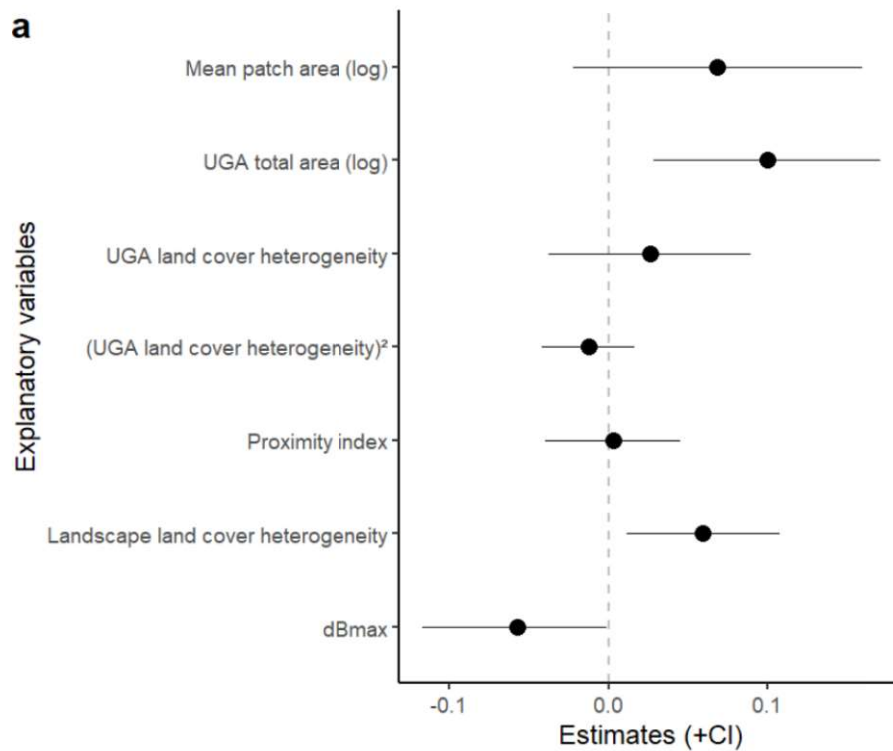


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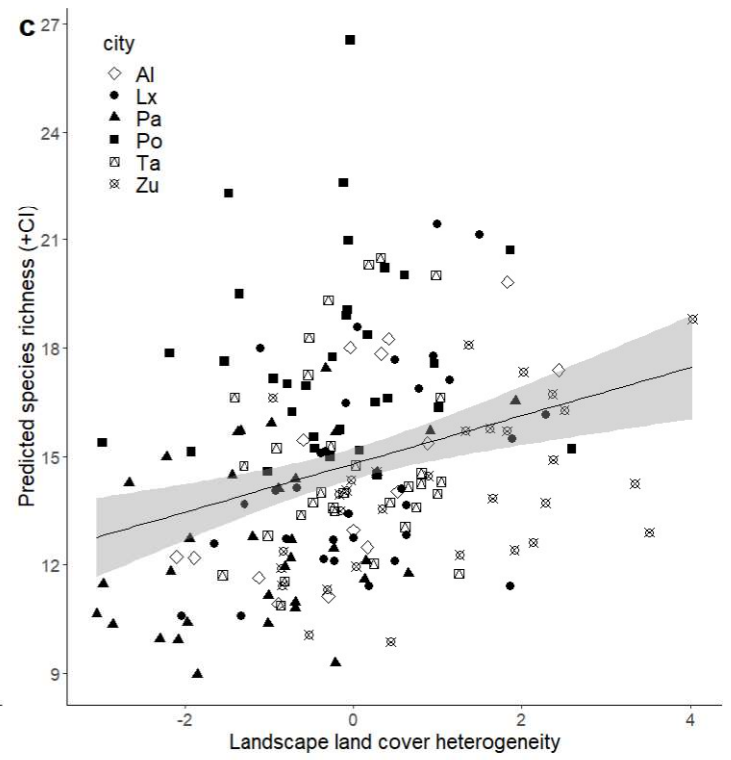
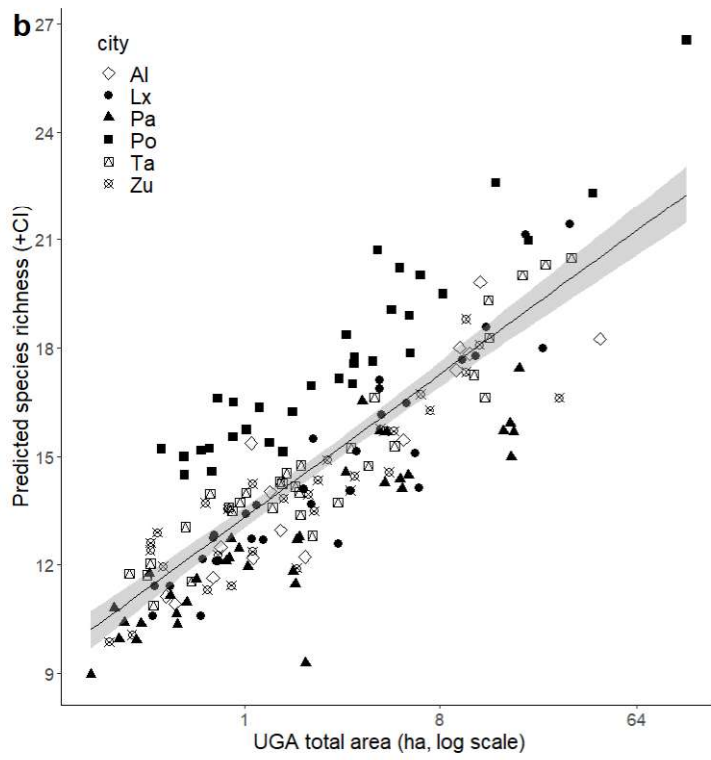


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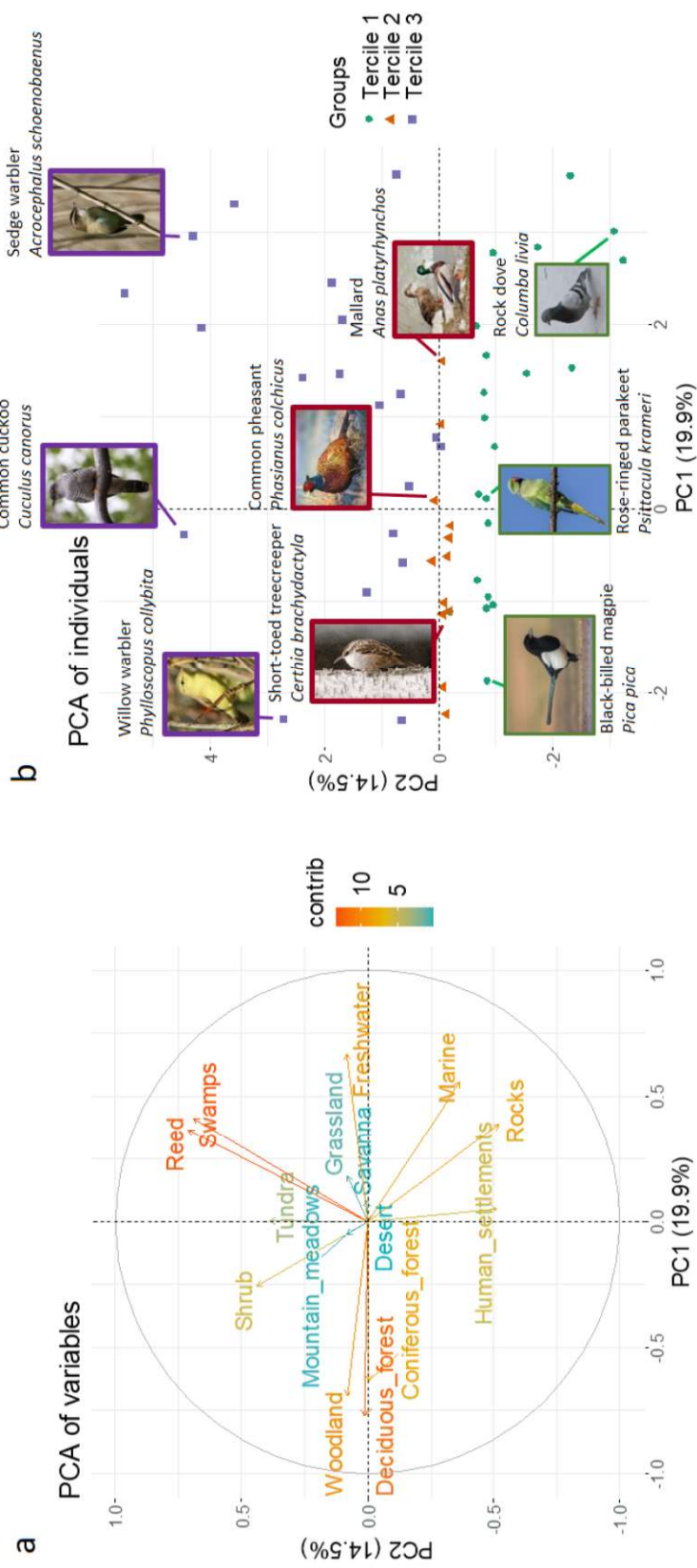


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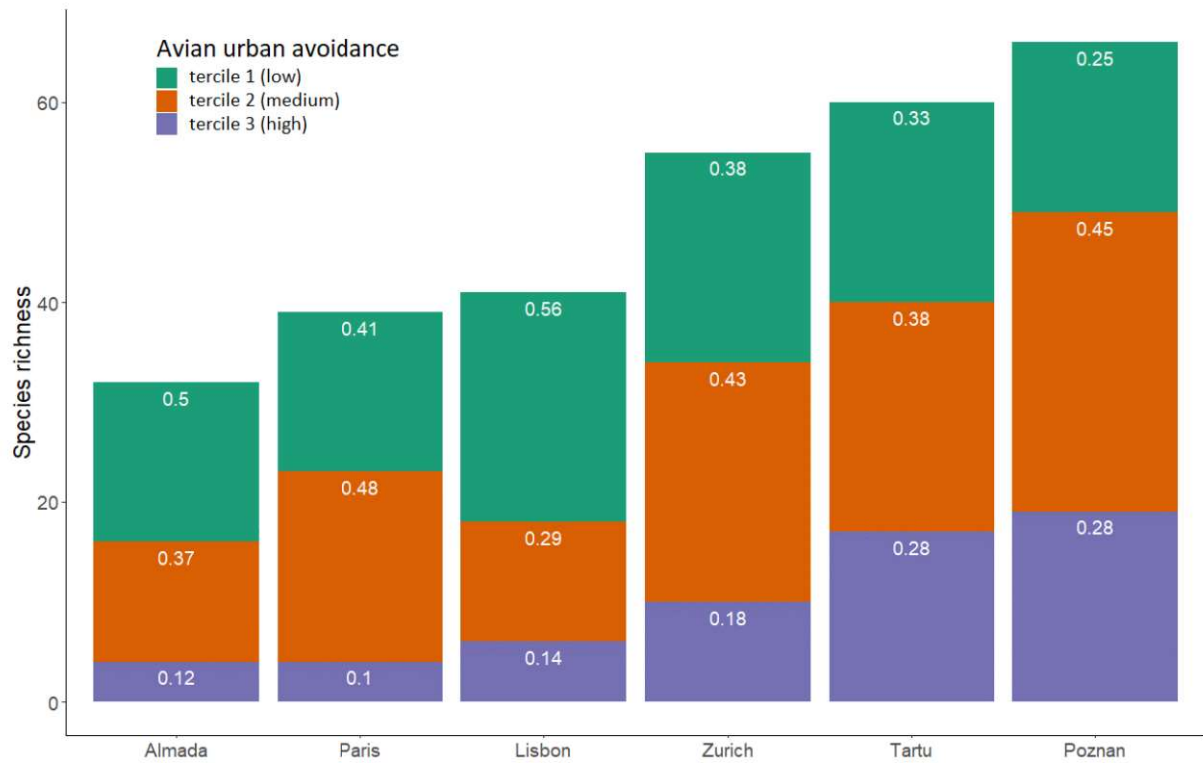
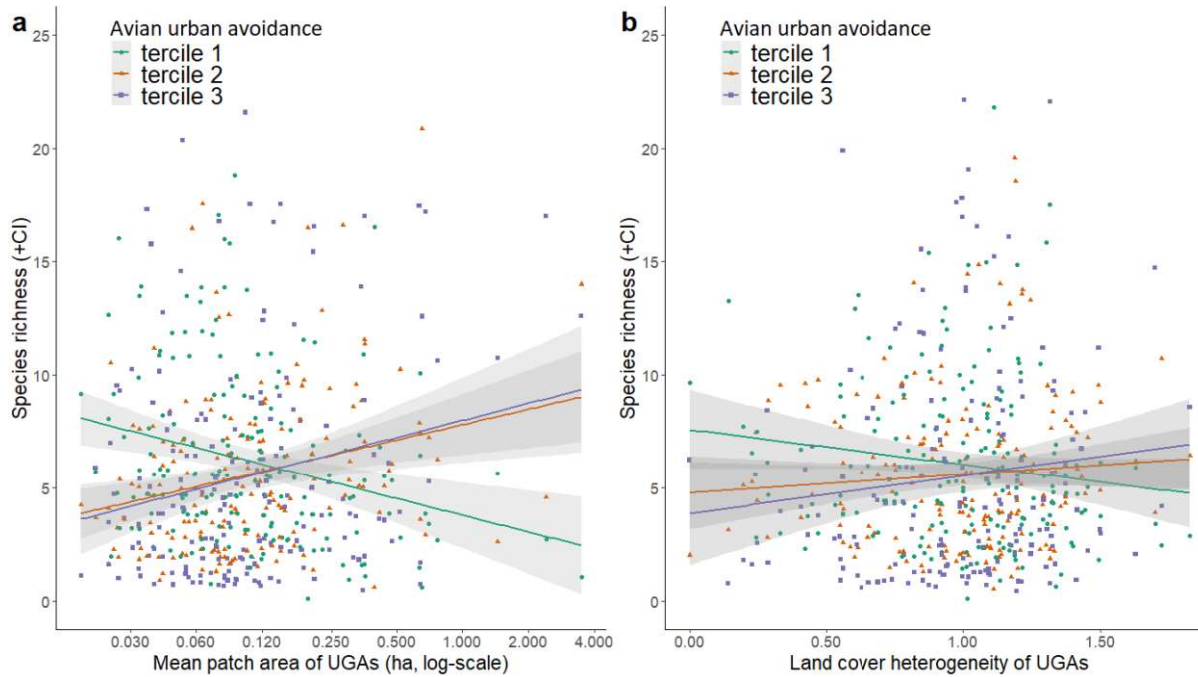


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Appendix A: Correlation coefficients between bird nesting habitats and the first three principal components (PCs).

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Appendix C: Results of the regression models between mean patch area, land cover heterogeneity and area of UGAs.

Appendix D: Results of the regression model analysing the effects of landscape (500 m buffer around each urban green area, UGA) and UGA variables on species richness of avian communities using dataset from all cities.

Appendix E: Results of full regression models analysing the effects of landscape (500 m buffer around each urban green area, UGA) and UGA variables on species richness of avian communities in the six studied cities.

Appendix F: Correlation coefficients between bird nesting habitats and the first three principal components (PCs).

Appendix G: Tukey HSD (Honestly Significant Difference) post-hoc tests.

Appendix H: Correlogram plot of the predictors for all cities pooled together.

Appendix I: Relationship between mean patch area and land cover heterogeneity (Shannon's index of land cover types) of the studied urban green areas.

Appendix A: Correlation coefficients between bird nesting habitats and the first three principal components (PCs). Data on bird occurrences in 15 nesting habitats in Europe were compiled by Storchová & Hořák (2018). Occurrence data derive from *The Birds of the Western Palearctic* handbook (Cramp, 2006). p-value: ns: non-significant, *p<0.05, **p<0.01, ***p<0.001.

Habitats / Principal Components (% of variance)	PC 1 (20.1%)	PC 2 (14.2%)	PC 3 (11.1%)
Grassland	0.22*	ns	0.67***
Shrub	-0.23**	0.42***	0.29**
Rocks	0.33***	-0.52***	ns
Reeds	0.35***	0.70***	ns
Swamps	0.42***	0.66***	ns
Marine	0.50***	-0.34***	-0.30**
Coniferous forest	-0.65***	ns	-0.34***
Freshwater	0.67***	ns	-0.35***
Woodland	-0.70***	ns	0.29**
Deciduous forest	-0.79***	ns	-0.24**
Human settlements	ns	-0.50***	0.22*
Mountain meadows	ns	ns	0.31***
Savanna	ns	ns	0.60***
Tundra	ns	0.40***	ns

Appendix B: List of the 119 bird species recorded in six European cities (within a total of 176 urban green areas) along with their urban avoidance value (PC2, this study), tercile group (1: low urban avoidance, 2: medium urban avoidance, 3: high urban avoidance), species occurrence (% of sites occupied) and number of individuals counted per city. NA: information not available.

Species (latin name)	Species code	Urban avoidance index (PC2)	Tercile group	Occurrence (%)	Number of individuals counted					
					Almada	Lisbon	Paris	Poznan	Tartu	Zurich
<i>Acanthis cannabina</i>	acacan	0.15	3	6.7	0	0	0	18	0	0
<i>Accipiter nisus</i>	accnis	0.09	2	1	0	0	0	1	0	0
<i>Acrocephalus arundinaceus</i>	acraru	3.64	3	1.5	0	0	0	3	0	0
<i>Acridotheres cristatellus</i>	acrcri	-0.25	2	1.5	5	0	0	0	0	0
<i>Acrocephalus dumetorum</i>	acrdom	0.73	3	1.5	0	0	0	0	2	0
<i>Acrocephalus palustris</i>	acrpal	1.64	3	2.1	0	0	0	3	1	0
<i>Acrocephalus schoenobaenus</i>	acrsch	4.36	3	2.1	0	0	0	1	2	0
<i>Acrocephalus scirpaceus</i>	acrsci	2.01	3	3.1	0	0	1	4	0	2
<i>Aegithalos caudatus</i>	aegcau	0.73	3	5.1	0	4	3	1	0	7
<i>Alopochen aegyptiaca</i>	aloaeg	-0.78	1	2.1	0	6	0	0	0	0
<i>Anas platyrhynchos</i>	anapla	0.01	2	16.4	0	15	11	26	12	15
<i>Anas strepera</i>	anastr	-0.04	2	1	0	0	0	2	0	0
<i>Anser anser</i>	ansans	NA	NA	1	0	12	0	0	0	0
<i>Apus apus</i>	apuapu	-2.27	1	36.9	14	15	57	83	15	53
<i>Apus melba</i>	apumel	-2.27	1	2.1	0	0	0	0	0	11
<i>Apus pallidus</i>	apupal	-3.15	1	8.2	15	110	0	0	0	0
<i>Ardea cinerea</i>	ardcin	0.01	2	2.1	0	0	1	0	0	2
<i>Aythya fuligula</i>	aytful	0.01	2	1	0	0	0	1	0	0
<i>Branta canadensis</i>	bracan	-0.78	1	1.5	0	0	2	1	0	0
<i>Bubulcus ibis</i>	bubibi	NA	NA	1.5	0	5	0	0	0	0
<i>Buteo buteo</i>	butbut	0.09	2	3.1	2	2	0	0	0	1
<i>Carduelis carduelis</i>	carcar	-0.78	1	39.5	28	46	4	18	23	40
<i>Carduelis chloris</i>	carchl	0.03	2	52.3	42	45	8	0	35	14
<i>Carduelis spinus</i>	carspi	-0.02	2	3.6	0	0	0	0	0	1
<i>Cecropis daurica</i>	cecdau	-1.48	1	1	3	0	0	0	0	0
<i>Certhia brachydactyla</i>	cerbra	0.09	2	36.4	10	29	27	14	0	17
<i>Certhia familiaris</i>	cerfam	-0.02	2	2.6	0	0	0	0	4	0
<i>Cettia cetti</i>	cetcet	4.15	3	1	1	0	0	0	0	0
<i>Chroicocephalus ridibundus</i>	chrrid	0.76	3	3.1	0	13	0	0	0	46

<i>Cisticola juncidis</i>	cisjun	1.07	3	3.1	3	3	0	0	0	0
<i>Coccothraustes</i>	coccoc	-0.09	2	13.8	0	0	0	7	29	1
<i>coccothraustes</i>										
<i>Columba livia</i>	colliv	-3.20	1	53.8	274	435	117	106	29	28
<i>Columba oenas</i>	coloen	0.09	2	2.1	0	0	5	0	0	0
<i>Columba palumbus</i>	colpal	-0.70	1	72.3	11	50	147	101	30	67
<i>Corvus corone</i>	corcor	-0.88	1	66.7	21	7	55	35	80	76
<i>Corvus frugilegus</i>	corfru	-0.70	1	6.7	0	0	0	18	19	0
<i>Corvus monedula</i>	cormon	-0.88	1	21	0	0	3	55	36	4
<i>Cuculus canorus</i>	cuccan	4.39	3	2.6	0	0	0	5	0	0
<i>Cyanistes caeruleus</i>	cyacae	-0.78	1	68.2	15	45	93	0	36	61
<i>Delichon urbicum</i>	delurb	-2.27	1	13.3	28	21	0	18	0	1
<i>Dendrocopos major</i>	denmaj	-0.80	1	13.3	1	3	4	11	6	5
<i>Dendrocopos medius</i>	denmed	0.01	2	4.6	0	0	0	1	7	0
<i>Dendrocopos minor</i>	denmin	0.01	2	1.5	0	0	0	2	0	0
<i>Dryocopus martius</i>	drymar	-0.02	2	1	0	0	0	2	0	0
<i>Egretta garzetta</i>	egrgar	NA	NA	1	0	1	0	0	0	0
<i>Emberiza calandra</i>	cmbcal	-0.25	2	1	0	1	0	0	0	0
<i>Emberiza citrinella</i>	embcit	0.58	3	1	0	0	0	1	0	0
<i>Emberiza schoeniclus</i>	embsch	5.65	3	1	0	0	0	1	0	0
<i>Erithacus rubecula</i>	erirub	0.81	3	43.6	9	29	42	3	20	23
<i>Estrilda astrild</i>	estast	1.80	3	1	0	2	0	0	0	0
<i>Falco tinnunculus</i>	faltin	-1.35	1	6.2	3	4	0	4	0	2
<i>Ficedula albicollis</i>	ficalb	-0.09	2	1	0	0	0	0	1	0
<i>Ficedula hypoleuca</i>	fichyp	0.09	2	9.7	0	0	0	4	16	0
<i>Ficedula parva</i>	ficpar	-0.02	2	1	0	0	0	0	1	0
<i>Fringilla coelebs</i>	fricoe	-0.80	1	59	0	6	38	47	66	89
<i>Fulica atra</i>	fulatr	0.01	2	2.1	0	0	2	2	0	7
<i>Gallinula chloropus</i>	galchl	0.11	3	3.1	0	0	8	0	0	0
<i>Garrulus glandarius</i>	gargla	-0.02	2	17.9	7	20	10	6	1	5
<i>Hippolais icterina</i>	hipict	0.73	3	9.2	0	0	0	5	11	1
<i>Hirundo rustica</i>	hirrus	-1.03	1	14.4	31	28	0	0	1	2
<i>Jynx torquilla</i>	jyntor	0.01	2	3.1	0	0	0	0	5	0
<i>Larus argentatus</i>	lararg	-3.15	1	9.2	0	0	2	0	25	0
<i>Larus canus</i>	larcana	-0.87	1	2.6	0	0	0	0	5	0
<i>Larus fuscus</i>	larfus	-1.66	1	2.1	1	4	0	0	0	0
<i>Larus michahellis</i>	larmic	-2.16	1	7.2	6	12	0	0	0	1
<i>Locustella naevia</i>	locnae	2.11	3	1	0	0	0	0	0	1
<i>Lophophanes cristatus</i>	lopcri	-0.02	2	3.6	6	0	0	0	0	1
<i>Lullula arborea</i>	lularb	0.58	3	1	0	0	0	1	0	0
<i>Luscinia luscinia</i>	luslus	0.63	3	4.1	0	0	0	2	6	0

<i>Luscinia megarhynchos</i>	lusmeg	0.63	3	5.6	0	0	0	12	0	0
<i>Mergus merganser</i>	mermer	0.11	3	1	0	0	0	0	0	3
<i>Milvus migrans</i>	milmig	0.99	3	2.1	0	0	0	0	0	3
<i>Milvus milvus</i>	milmil	-0.36	2	2.1	0	0	0	0	0	4
<i>Motacilla alba</i>	motalb	-0.83	1	26.7	7	29	0	0	24	10
<i>Motacilla cinerea</i>	motcin	0.73	3	1	0	0	0	0	0	1
<i>Muscicapa striata</i>	musstr	-0.70	1	5.1	0	0	0	0	4	7
<i>Nycticorax nycticorax</i>	nycnyc	NA	NA	1	0	1	0	0	0	0
<i>Oriolus oriolus</i>	oriori	0.01	2	3.6	0	0	0	5	1	0
<i>Parus major</i>	parmaj	-0.70	1	78.5	7	25	103	54	50	83
<i>Passer domesticus</i>	pasdom	-0.16	2	57.9	104	181	84	97	30	58
<i>Passer montanus</i>	pasmon	0.73	3	21	0	0	0	44	59	1
<i>Periparus ater</i>	perate	-0.12	2	11.8	6	18	0	0	0	0
<i>Phalacrocorax carbo</i>	phacar	-0.87	1	2.6	0	0	1	0	0	3
<i>Phasianus colchicus</i>	phacol	-0.14	2	1.5	0	0	0	2	0	0
<i>Phoenicurus ochruros</i>	phooch	-2.27	1	22.6	7	16	3	4	0	31
<i>Phoenicurus phoenicurus</i>	phopho	0.01	2	12.3	0	0	0	9	18	1
<i>Phylloscopus collybita</i>	phycol	-0.02	2	26.2	0	0	15	10	16	33
<i>Phylloscopus sibilatrix</i>	physib	-0.02	2	4.1	0	0	0	0	9	0
<i>Phylloscopus trochilus</i>	phytro	3.03	3	14.9	0	0	0	9	8	18
<i>Picus canus</i>	piccan	0.01	2	1	0	0	0	1	0	0
<i>Pica pica</i>	picpic	-0.70	1	34.4	0	0	42	52	4	16
<i>Picus viridis</i>	picvir	-0.02	2	8.2	0	0	7	7	0	2
<i>Podiceps cristatus</i>	podcri	0.01	2	1	0	0	0	0	0	1
<i>Podiceps ruficollis</i>	podruf	0.01	2	1	0	0	0	2	0	0
<i>Poecile palustris</i>	poepal	-0.09	2	2.1	0	0	2	2	0	1
<i>Prunella modularis</i>	prumod	0.81	3	13.3	0	0	27	0	2	0
<i>Psittacula krameri</i>	psikra	-1.20	1	21.5	0	58	53	0	0	0
<i>Pyrrhula pyrrhula</i>	pyrpyr	-2.01	1	3.6	0	0	0	0	7	0
<i>Recurvirostra avosetta</i>	recavo	-0.87	1	1	0	10	0	0	0	0
<i>Regulus ignicapilla</i>	regign	0.09	2	10.3	1	6	4	1	0	10
<i>Regulus regulus</i>	regreg	-0.12	2	5.1	0	0	7	0	2	1
<i>Remiz pendulinus</i>	rempen	3.64	3	1	0	0	0	0	1	0
<i>Serinus serinus</i>	serser	-0.80	1	39.5	50	115	0	22	1	10
<i>Sitta europaea</i>	siteur	-0.02	2	21.5	0	0	10	10	13	17
<i>Sterna hirundo</i>	stehir	-0.87	1	2.6	0	0	0	0	4	0
<i>Streptopelia decaocto</i>	strdec	-0.98	1	37.4	62	39	2	38	0	12
<i>Sturnus unicolor</i>	stuuni	-0.93	1	22.1	46	128	0	0	0	0
<i>Sturnus vulgaris</i>	stuvul	-0.83	1	49.7	0	0	53	102	43	39
<i>Sylvia atricapilla</i>	sylatr	-0.02	2	58.5	13	36	40	25	7	94
<i>Sylvia borin</i>	sylbor	0.73	3	4.6	0	0	0	4	4	0

<i>Sylvia communis</i>	sylcom	0.63	3	3.1	0	0	0	2	3	0
<i>Sylvia curruca</i>	sylcur	0.63	3	15.9	0	0	0	22	10	0
<i>Sylvia melanocephala</i>	sylmel	0.52	3	13.8	18	18	0	0	0	0
<i>Thectocercus acuticaudatus</i>	theacu	-1.20	1	2.1	0	6	0	0	0	0
<i>Troglodytes troglodytes</i>	trotro	0.09	2	28.7	15	33	44	2	1	3
<i>Turdus iliacus</i>	turili	0.81	3	2.6	0	0	0	0	4	0
<i>Turdus merula</i>	turner	0.03	2	84.1	84	229	73	62	28	95
<i>Turdus philomelos</i>	turphi	0.03	2	16.4	0	0	6	22	7	6
<i>Turdus pilaris</i>	turpil	0.09	2	29.2	0	0	0	87	77	0

Appendix C: Results of the regression models between mean patch area, land cover heterogeneity and area of UGAs. We reported the slope estimate, standard error (Std. Error) and degree of freedom (df) for each explanatory variable. Significant variables are highlighted in bold in the table (*p<0.05, **p<0.01, ***p<0.001).

Level ^a	Mean patch area predictor (log)	Estimate	Std. error	F-value	df	p-value
All cities (n=176, 0.666)	UGA land cover heterogeneity	-0.498	0.037	183.160	1	<0.001***
	Area of UGAs (log)	0.666	0.035	356.920	1	<0.001***
Almada (n=15, 0.740)	UGA land cover heterogeneity	-0.612	0.123	24.769	1	<0.001***
	Area of UGAs (log)	0.719	0.129	31.012	1	<0.001***
Lisbon (n=31, 0.736)	UGA land cover heterogeneity	-0.300	0.083	13.008	1	<0.001***
	Area of UGAs (log)	0.817	0.089	84.749	1	<0.001***
Paris (n=35, 0.781)	UGA land cover heterogeneity	-0.427	0.063	46.398	1	<0.001***
	Area of UGAs (log)	0.487	0.052	88.890	1	<0.001***
Poznan (n=32, 0.792)	UGA land cover heterogeneity	-0.617	0.089	47.890	1	<0.001***
	Area of UGAs (log)	0.578	0.075	59.415	1	<0.001***
Tartu (n=32, 0.903)	UGA land cover heterogeneity	-0.739	0.065	130.760	1	<0.001***
	Area of UGAs (log)	0.884	0.066	178.470	1	<0.001***
Zurich (n=31, 0.363)	UGA land cover heterogeneity	-0.277	0.092	9.010	1	<0.01**
	Area of UGAs (log)	0.383	0.090	17.917	1	<0.001***

^aNumber of UGAs and adjusted R² value

Appendix D: Results of the regression model analysing the effects of landscape (500 m buffer around each urban green area, UGA) and UGA variables on species richness of avian communities using dataset from all cities (176 UGAs with complete land cover information). Relationships were controlled for the maximum sound volume level (dBmax) and included ‘city’ as a random effect to account for possible consistent differences among cities (generalised linear mixed model). “(UGA land cover heterogeneity)²” is the squared term of UGA land cover heterogeneity to test for a non-linear relationship. We reported the slope estimate, standard error (Std. Error) and degree of freedom (df) for each explanatory variable. We added semi-partial R² (part R²) values that provide an estimate of the variance explained by a fixed effect independent of other predictors. Significant variables are highlighted in bold in the table (*p<0.05, **p<0.01, ***p<0.001). Marginally significant statistical tendencies (0.050<p<0.010) are followed by a full stop (.). Non-significant two-way interactions or quadratic terms (i.e. UGA land cover heterogeneity²) were removed from the models in order to better estimate single effects of predictors.

Level ^a	Predictor of species richness	Estimate	Std. Error	Chisq-value	df	p-value	part R ²
All cities (n=176, 0.297)	dBmax	-0.047	0.028	2.944	1	0.086.	0.025
	UGA total area (log)	0.103	0.036	8.289	1	0.004**	0.018
	Mean patch area (log)	0.068	0.045	2.347	1	0.126	0.008
	Proximity index	0.005	0.021	0.049	1	0.825	0.000
	Landscape land cover heterogeneity	0.055	0.024	5.400	1	0.020*	0.031
	UGA land cover heterogeneity	0.024	0.032	0.562	1	0.454	0.002
	(UGA land cover heterogeneity) ²	-0.014	0.015	0.853	1	0.356	0.004

^aNumber of UGAs and marginal R² value

Appendix E: Results of full regression models analysing the effects of landscape (500 m buffer around each urban green area, UGA) and UGA variables on species richness of avian communities in the six studied cities. Relationships were controlled for the maximum noise level (dBmax) (generalised linear model). Almada was omitted because of the small sample size. (UGA land cover heterogeneity)² is the squared term of UGA land cover heterogeneity to test for a non-linear relationship. We reported the slope estimate, standard error (Std. Error) and degree of freedom (df) for each explanatory variable. Significant variables are highlighted in bold in the table (*p<0.05, **p<0.01, ***p<0.001). Marginally significant statistical tendencies (0.050<p<0.010) are followed by a full stop (.). We also reported semi-partial R² (part R²) values that provide an estimate of the variance explained by a fixed effect independent of other predictors.

Level ^a	Predictor of species richness	Estimate	Std. Error	Chisq-value	df	p-value	part R ²
Lisbon (n=31, 0.756)	dBmax	-0.250	0.103	6.217	1	0.013*	0.027
	UGA total area (log)	0.333	0.115	8.171	1	0.004**	0.041
	Mean patch area (log)	-0.193	0.123	2.440	1	0.118	0.015
	Proximity index	0.029	0.047	0.370	1	0.543	0.001
	Landscape land cover heterogeneity	-0.011	0.063	0.033	1	0.857	0.001
	UGA land cover heterogeneity	-0.329	0.225	2.142	1	0.143	0.006
	(UGA land cover heterogeneity) ²	0.020	0.035	0.330	1	0.565	0.000
Paris (n=35, 0.677)	dBmax	-0.025	0.038	0.440	1	0.507	0.003
	UGA total area (log)	0.159	0.091	3.070	1	0.080.	0.034
	Mean patch area (log)	0.031	0.164	0.037	1	0.848	0.014
	Proximity index	0.064	0.061	1.067	1	0.302	0.001

	Landscape land cover heterogeneity	0.057	0.061	0.851	1	0.356	0.010
	UGA land cover heterogeneity	0.167	0.283	0.350	1	0.554	0.005
	(UGA land cover heterogeneity) ²	0.021	0.053	0.156	1	0.693	0.001
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Poznan (n=32, 0.624)	dBmax	-0.015	0.132	0.013	1	0.909	0.000
	UGA total area (log)	0.053	0.075	0.496	1	0.481	0.030
	Mean patch area (log)	0.210	0.107	3.842	1	0.050*	0.030
	Proximity index	-0.020	0.052	0.149	1	0.699	0.001
	Landscape land cover heterogeneity	0.090	0.071	1.640	1	0.200	0.004
	UGA land cover heterogeneity	0.432	0.308	1.952	1	0.162	0.006
	(UGA land cover heterogeneity) ²	-0.010	0.055	0.032	1	0.858	0.002
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Tartu (n=32, 0.536)	dBmax	-0.087	0.089	0.952	1	0.329	0.009
	UGA total area (log)	0.151	0.139	1.178	1	0.278	0.015
	Mean patch area (log)	-0.043	0.149	0.083	1	0.773	0.010
	Proximity index	-0.060	0.077	0.614	1	0.434	0.004

	Landscape land cover heterogeneity	0.106	0.092	1.340	1	0.247	0.004
	UGA land cover heterogeneity	-0.356	0.392	0.822	1	0.365	0.002
	(UGA land cover heterogeneity) ²	-0.039	0.050	0.617	1	0.432	0.002
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Zurich (n=31, 0.692)	dBmax	-0.027	0.093	0.083	1	0.774	0.001
	UGA total area (log)	0.086	0.084	1.060	1	0.303	0.02
	Mean patch area (log)	0.056	0.139	0.165	1	0.684	0.006
	Proximity index	0.047	0.056	0.692	1	0.406	0.004
	Landscape land cover heterogeneity	0.049	0.049	0.975	1	0.323	0.010
	UGA land cover heterogeneity	0.006	0.272	0.001	1	0.982	0.005
	(UGA land cover heterogeneity) ²	-0.119	0.057	4.555	1	0.033*	0.022
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^aNumber of UGAs and marginal R² value

Appendix F: Correlation coefficients between bird nesting habitats and the first three principal components (PCs). Data on bird occurrences in 15 nesting habitats in Europe were compiled by Storchová and Hořák (2018). Occurrence data derive from *The Birds of the Western Palearctic* handbook (Cramp, 2006). p-value: ns: non-significant, *p<0.05, **p<0.01, ***p<0.001.

Habitats / Principal Components (% of variance)	PC 1 (20.1%)	PC 2 (14.2%)	PC 3 (11.1%)
Grassland	0.22*	ns	0.67***
Shrub	-0.23**	0.42***	0.29**
Rocks	0.33***	-0.52***	ns
Reeds	0.35***	0.70***	ns
Swamps	0.42***	0.66***	ns
Marine	0.50***	-0.34***	-0.30**
Coniferous forest	-0.65***	ns	-0.34***
Freshwater	0.67***	ns	-0.35***
Woodland	-0.70***	ns	0.29**
Deciduous forest	-0.79***	ns	-0.24**
Human settlements	ns	-0.50***	0.22*
Mountain meadows	ns	ns	0.31***
Savanna	ns	ns	0.60***
Tundra	ns	0.40***	ns

Appendix G: Tukey HSD (Honestly Significant Difference) post-hoc tests comparing the avian richness between tercile groups (1 or low urban avoidance, 2 or medium urban avoidance and 3 or high urban avoidance) and cities. We ranked the results according to the p-value.

Comparison (tercile group:city)	Difference	Lower	Upper	p-value adjusted
1:Lisbon-3:Almada	6.731	4.482	8.981	0.000
1:Paris-3:Almada	5.781	3.574	7.988	0.000
1:Paris-3:Lisbon	5.921	4.157	7.685	0.000
1:Poznan-2:Lisbon	3.395	1.593	5.197	0.000
1:Poznan-2:Paris	4.579	2.829	6.328	0.000
1:Poznan-3:Almada	7.417	5.179	9.655	0.000
1:Poznan-3:Lisbon	7.556	5.754	9.359	0.000
1:Poznan-3:Paris	7.436	5.687	9.185	0.000
1:Tartu-3:Almada	5.409	3.159	7.658	0.000
1:Tartu-3:Lisbon	5.548	3.732	7.365	0.000
1:Tartu-3:Paris	5.428	3.664	7.191	0.000
1:Tartu-3:Poznan	3.804	2.002	5.607	0.000
1:Zurich-3:Almada	5.989	3.740	8.239	0.000
1:Zurich-3:Lisbon	6.129	4.313	7.946	0.000
1:Zurich-3:Paris	6.008	4.244	7.772	0.000
1:Zurich-3:Poznan	4.385	2.583	6.187	0.000

1:Zurich-3:Tartu	4.710	2.893	6.526	0.000
2:Paris-1:Lisbon	-3.893	-5.657	-2.129	0.000
2:Poznan-3:Almada	5.354	3.116	7.592	0.000
2:Poznan-3:Lisbon	5.494	3.692	7.296	0.000
2:Poznan-3:Paris	5.373	3.624	7.122	0.000
2:Tartu-3:Almada	4.312	2.062	6.561	0.000
2:Tartu-3:Lisbon	4.452	2.635	6.268	0.000
2:Tartu-3:Paris	4.331	2.567	6.095	0.000
2:Zurich-1:Poznan	-3.524	-5.326	-1.722	0.000
2:Zurich-3:Lisbon	4.032	2.216	5.849	0.000
2:Zurich-3:Paris	3.912	2.148	5.675	0.000
3:Almada-1:Almada	-6.133	-8.745	-3.522	0.000
3:Lisbon-1:Almada	-6.273	-8.522	-4.024	0.000
3:Lisbon-1:Lisbon	-6.871	-8.687	-5.054	0.000
3:Lisbon-2:Almada	-4.473	-6.722	-2.224	0.000
3:Lisbon-2:Lisbon	-4.161	-5.978	-2.345	0.000
3:Paris-1:Almada	-6.152	-8.359	-3.945	0.000
3:Paris-1:Lisbon	-6.750	-8.514	-4.986	0.000

3:Paris-1:Paris	-5.800	-7.510	-4.090	0.000
3:Paris-2:Almada	-4.352	-6.559	-2.145	0.000
3:Paris-2:Lisbon	-4.041	-5.804	-2.277	0.000
3:Poznan-1:Almada	-4.529	-6.767	-2.291	0.000
3:Poznan-1:Lisbon	-5.127	-6.929	-3.325	0.000
3:Poznan-1:Paris	-4.177	-5.926	-2.428	0.000
3:Poznan-1:Poznan	-5.813	-7.600	-4.025	0.000
3:Poznan-2:Poznan	-3.750	-5.538	-1.962	0.000
3:Tartu-1:Almada	-4.854	-7.103	-2.604	0.000
3:Tartu-1:Lisbon	-5.452	-7.268	-3.635	0.000
3:Tartu-1:Paris	-4.501	-6.265	-2.738	0.000
3:Tartu-1:Poznan	-6.137	-7.939	-4.335	0.000
3:Tartu-1:Tartu	-4.129	-5.946	-2.313	0.000
3:Tartu-2:Poznan	-4.075	-5.877	-2.272	0.000
3:Zurich-1:Almada	-6.144	-8.393	-3.895	0.000
3:Zurich-1:Lisbon	-6.742	-8.558	-4.925	0.000
3:Zurich-1:Paris	-5.792	-7.556	-4.028	0.000
3:Zurich-1:Poznan	-7.427	-9.230	-5.625	0.000

3:Zurich-1:Tartu	-5.419	-7.236	-3.603	0.000
3:Zurich-1:Zurich	-6.000	-7.817	-4.183	0.000
3:Zurich-2:Almada	-4.344	-6.593	-2.095	0.000
3:Zurich-2:Lisbon	-4.032	-5.849	-2.216	0.000
3:Zurich-2:Poznan	-5.365	-7.167	-3.563	0.000
3:Zurich-2:Tartu	-4.323	-6.139	-2.506	0.000
3:Zurich-2:Zurich	-3.903	-5.720	-2.087	0.000
1:Zurich-2:Paris	3.151	1.387	4.915	0.000
2:Lisbon-3:Almada	4.022	1.772	6.271	0.000
2:Tartu-1:Poznan	-3.105	-4.907	-1.303	0.000
2:Zurich-3:Almada	3.892	1.643	6.142	0.000
2:Paris-1:Paris	-2.943	-4.652	-1.233	0.000
2:Paris-3:Lisbon	2.978	1.214	4.742	0.000
3:Paris-2:Paris	-2.857	-4.567	-1.148	0.000
3:Tartu-2:Tartu	-3.032	-4.849	-1.216	0.000
3:Almada-2:Almada	-4.333	-6.945	-1.722	0.000
3:Zurich-2:Paris	-2.849	-4.613	-1.085	0.000
2:Zurich-1:Lisbon	-2.839	-4.655	-1.022	0.000

3:Tartu-2:Lisbon	-2.742	-4.558	-0.925	0.000
2:Tartu-3:Poznan	2.708	0.905	4.510	0.000
2:Paris-1:Almada	-3.295	-5.502	-1.088	0.000
2:Lisbon-1:Lisbon	-2.710	-4.526	-0.893	0.000
1:Tartu-2:Paris	2.571	0.807	4.334	0.000
2:Poznan-2:Paris	2.516	0.767	4.265	0.000
2:Zurich-3:Tartu	2.613	0.796	4.429	0.000
1:Poznan-2:Almada	3.083	0.845	5.321	0.000
3:Tartu-2:Almada	-3.054	-5.303	-0.804	0.000
3:Poznan-2:Lisbon	-2.417	-4.220	-0.615	0.000
2:Tartu-1:Lisbon	-2.419	-4.236	-0.603	0.001
2:Paris-3:Almada	2.838	0.631	5.045	0.001
2:Zurich-3:Poznan	2.288	0.486	4.091	0.001
3:Poznan-2:Almada	-2.729	-4.967	-0.491	0.003
2:Zurich-1:Zurich	-2.097	-3.913	-0.280	0.007
2:Poznan-1:Poznan	-2.063	-3.850	-0.275	0.008
1:Tartu-1:Poznan	-2.008	-3.810	-0.206	0.013
1:Zurich-2:Lisbon	1.968	0.151	3.784	0.019

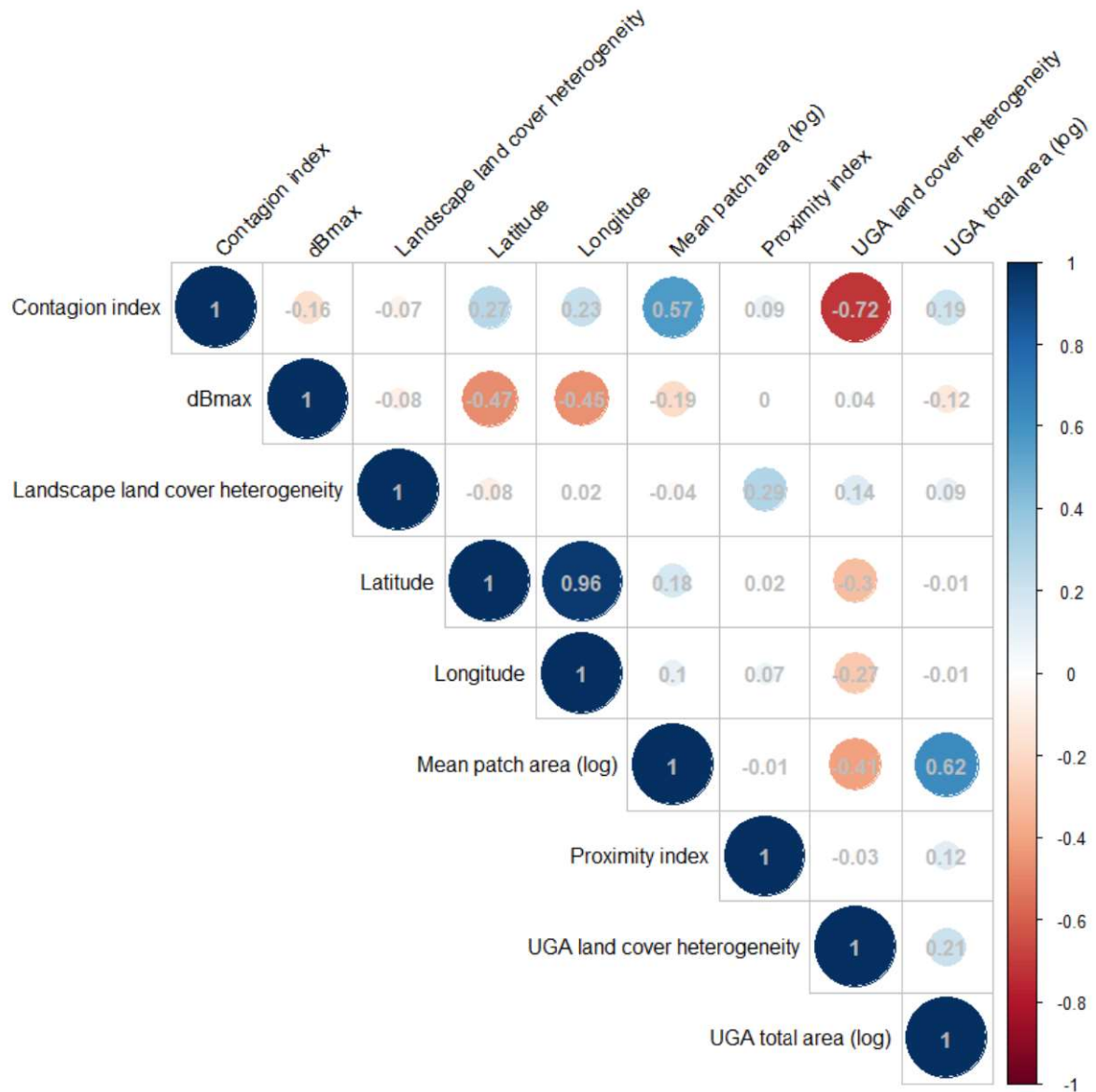
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1:Zurich-2:Tartu	1.677	-0.139	3.494	0.111
3:Zurich-3:Poznan	-1.615	-3.417	0.187	0.143
3:Tartu-2:Paris	-1.559	-3.322	0.205	0.160
2:Tartu-2:Paris	1.474	-0.290	3.238	0.240
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2:Tartu-1:Paris	-1.469	-3.233	0.295	0.245
2:Zurich-2:Poznan	-1.462	-3.264	0.341	0.289
2:Tartu-1:Almada	-1.822	-4.071	0.428	0.291
1:Zurich-1:Poznan	-1.427	-3.230	0.375	0.331
3:Tartu-3:Lisbon	1.419	-0.397	3.236	0.355

2:Poznan-1:Lisbon	-1.377	-3.179	0.425	0.398
1:Tartu-2:Lisbon	1.387	-0.429	3.204	0.399
2:Poznan-2:Lisbon	1.333	-0.470	3.135	0.461
3:Tartu-3:Paris	1.299	-0.465	3.062	0.469
1:Zurich-2:Almada	1.656	-0.593	3.905	0.469
1:Tartu-1:Lisbon	-1.323	-3.139	0.494	0.491
3:Poznan-3:Almada	1.604	-0.634	3.842	0.521
3:Zurich-3:Tartu	-1.290	-3.107	0.526	0.538
3:Poznan-2:Paris	-1.234	-2.983	0.515	0.551
2:Almada-1:Almada	-1.800	-4.411	0.811	0.595
2:Paris-2:Almada	-1.495	-3.702	0.712	0.627
2:Paris-2:Lisbon	-1.183	-2.947	0.580	0.644
1:Paris-2:Almada	1.448	-0.759	3.655	0.683
2:Tartu-1:Tartu	-1.097	-2.913	0.720	0.805
2:Zurich-2:Paris	1.054	-0.709	2.818	0.817
2:Tartu-2:Poznan	-1.042	-2.845	0.760	0.854
1:Poznan-1:Almada	1.283	-0.955	3.521	0.863
3:Tartu-3:Almada	1.280	-0.970	3.529	0.871

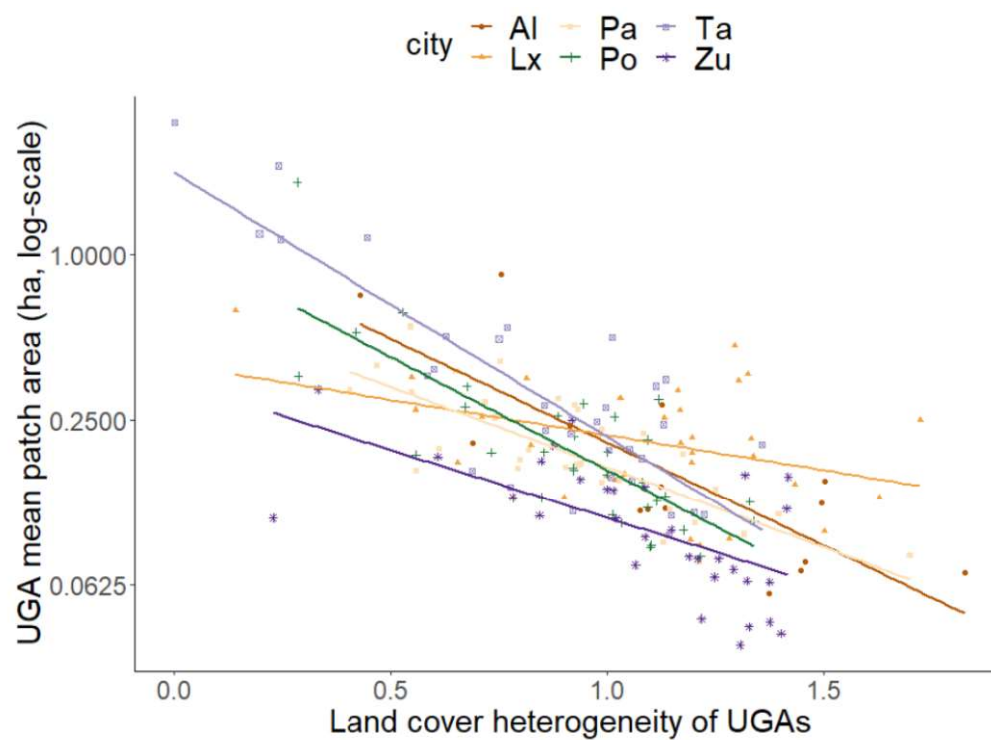
1:Paris-1:Lisbon	-0.950	-2.714	0.814	0.915
1:Tartu-2:Almada	1.075	-1.174	3.325	0.970
2:Poznan-2:Almada	1.021	-1.217	3.259	0.981
1:Zurich-1:Lisbon	-0.742	-2.558	1.075	0.994
1:Poznan-1:Lisbon	0.685	-1.117	2.488	0.997
1:Zurich-2:Poznan	0.635	-1.167	2.437	0.999
2:Poznan-1:Almada	-0.779	-3.017	1.459	0.999
1:Tartu-1:Almada	-0.725	-2.974	1.525	1.000
1:Zurich-1:Tartu	0.581	-1.236	2.397	1.000
1:Lisbon-1:Almada	0.598	-1.651	2.847	1.000
2:Poznan-1:Paris	-0.427	-2.176	1.322	1.000
2:Zurich-2:Tartu	-0.419	-2.236	1.397	1.000
1:Tartu-1:Paris	-0.372	-2.136	1.391	1.000
2:Zurich-2:Almada	-0.441	-2.690	1.808	1.000
3:Tartu-3:Poznan	-0.325	-2.127	1.478	1.000
1:Paris-1:Almada	-0.352	-2.559	1.855	1.000
1:Tartu-2:Poznan	0.054	-1.748	1.857	1.000
1:Zurich-1:Almada	-0.144	-2.393	2.105	1.000

1:Zurich-1:Paris	0.208	-1.556	1.972	1.000
2:Lisbon-2:Almada	-0.312	-2.561	1.938	1.000
2:Tartu-2:Almada	-0.022	-2.271	2.228	1.000
2:Tartu-2:Lisbon	0.290	-1.526	2.107	1.000
2:Zurich-2:Lisbon	-0.129	-1.946	1.687	1.000
3:Lisbon-3:Almada	-0.140	-2.389	2.110	1.000
3:Paris-3:Almada	-0.019	-2.226	2.188	1.000
3:Paris-3:Lisbon	0.121	-1.643	1.885	1.000
3:Zurich-3:Almada	-0.011	-2.260	2.239	1.000
3:Zurich-3:Lisbon	0.129	-1.687	1.946	1.000
3:Zurich-3:Paris	0.008	-1.756	1.772	1.000

Appendix H: Correlogram plot of the predictors for all cities pooled together. The colour and intensity of the circles are proportional to the correlation coefficients in grey. Except for latitude, only the contagion index, which measures the degree of aggregation of the same land cover within each UGA, was above the high correlation threshold ($\rho > 0.7$ with p-value < 0.01) (Zuur et al., 2009). We thus removed this latter variable as it was highly correlated with UGA land cover heterogeneity.



Appendix I: Relationship between mean patch area and land cover heterogeneity (Shannon's index of land cover types) of the studied urban green areas (n=176 UGAs). The different slopes represent the relationship for each city. Al, Pa, Ta, Lx, Po and Zu stand for the cities of Almada, Paris, Tartu, Lisbon, Poznan and Zurich, respectively. Model results are presented in Table S3.



Data accessibility

Data are available on request

Conflict of interest declaration

The authors declare that they have no known competing financial interests or personal relationships that have or could be perceived to have influenced the work reported in this article.

Acknowledgements

We thank the municipalities of Greater Paris, Lisbon, Almada, Poznan, Tartu and Zurich for their participation in the BioVEINS project. We are grateful for the help provided by the birdwatchers in each city. We thank Victoria Grace for editing the English and improving quality of the manuscript.



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Funding

This work was supported by the ERA-Net BiodivERsA project “BioVEINS: Connectivity of green and blue infrastructures: living veins for biodiverse and healthy cities” (H2020 BiodivERsA32015104). J.C.A. was supported by the Swiss National Science Foundation (project 31BD30_172467), A.V.M. by the University of Antwerp, P.T. by Polish funding (NCN/2016/22/Z/NZ8/00004), M.A.O. by the European Social Fund’s Dora Plus Programme and B.M. by the Research Foundation – Flanders (FWO) (1S84819N). The funding sources had no involvement in the study design or execution.