FOLIA OECOLOGICA – vol. 50, no. 2 (2023), doi: 10.2478/foecol-2023-0012

Biodiversity in cities: the impact of biodiversity data across spatial scales on diversity estimates

C.X. Garzon Lopez*, Gabija Savickytė

Knowledge Infrastructures, Campus Fryslân, University of Groningen, Wirdumerdijk 34, 8911 CE, Leeuwarden, Netherlands

Abstract

Garzon Lopez, C.X., Savickytė, G., 2023. Biodiversity in cities: the impact of biodiversity data across spatial scales on diversity estimates. *Folia Oecologica*, 50 (2): 134–146.

The assessment and monitoring of biodiversity in urban areas has been shown to have enormous potential to inform integrative urban planning in cities. In this context, digital biodiversity repositories such as the Global Biodiversity Information Facility (GBIF) has been promoted for its central role in gathering and harmonizing biodiversity data worldwide, thereby facilitating these assessments and monitoring efforts. While GBIF data has been investigated for its potential at a large scale and in natural ecosystems, the question remains as to what extent, and in which context, is GBIF data applicable to urban biodiversity assessment and monitoring? In this study, we assessed the spatial patterns of biodiversity, by exploring species richness patterns in relation to land use types for three taxonomic groups (birds, mammals and arthropods) in three cities in The Netherlands (Rotterdam, Amsterdam and Groningen) at multiple spatial scales. We found significant variation in the effect of land uses on the species richness patterns, in terms of taxonomic group, spatial configuration and land cover type, and across spatial scales. Our study demonstrates the potential of GBIF data while highlighting the importance of the careful selection of one or multiple spatial scales, especially in relation to the taxonomic group characteristics and ecology and the spatial configuration of the cities studied.

Keywords

arthropods, birds, GBIF, mammals, Netherlands, spatial scale, urban biodiversity

Introduction

Rapid urbanization is one of the most prominent development trends over the last centuries (Elmqvist et al., 2013). The trend continues today: in Europe (European Union states) alone, 74% of the population currently resides in urban areas (European Union, 2016). This causes enormous economic, societal, infrastructural, and environmental pressures from and on urban environments (Lucertini and Musco, 2020; Seto et al., 2014). Cities are not only one of the major contributors to climate change, but they will also be greatly affected by it (Kumar, 2021; Balaban, 2012). Examples such as the urban heat island effect (UHI) and urban air quality degradation are prominent and visible in almost every city around the globe (Kumar, 2021; Balaban, 2012). However, the issue of climate change in urban areas has revealed that cities nowadays are facing multi-dimensional problems which are all amplified due to climate change and biodiversity loss (Chakraborty et al., 2019; Checker, 2011; Watkins et al., 2016; Alizadeh et al., 2022; Sicard et al., 2020). Social and economic inequal infrastructures become more visible when we talk about accessibility to cleaner air, public services, transportation, and green areas within the cities (Comber et al., 2008; Watkins et al., 2016; Chakraborty et al., 2019). Moreover, city areas are expected to reach 1.7 million Km^2

*Corresponding author:
e-mail: c.x.garzon@gmail.com

by 2050 (Zhou et al., 2019). This urban sprawl interacts with the surrounding rural and agricultural areas and has repercussions for natural ecosystems and its biodiversity which in turn can affect the health and the performance of these areas (Czamanski, 2008; De Carvalho and Szlafsztein, 2019). Furthermore, urbanization and city development have a paramount effect on biodiversity richness, species composition and ecosystem functioning in urban areas (Beninde et al., 2015; Lososová et al., 2016; La Sorte et al., 2014). This happens through landscape change-induced habitat loss as well as fragmentation of ecological networks in cities. While some urbanization effects might be negative, studies show that urban areas can also provide habitat and foster biodiversity if managed well (Aronson et al., 2014; Fournier et al., 2020; Kowarik, 2011; Ives et al., 2015; Lepczyk et al., 2017).

As the global biodiversity crisis deepens (Knapp et al., 2021; Guerry et al., 2021), urban environments have become crucial habitat providers for various species of plants and fungi and act as refuge for different avian, arthropod and mammal species from surrounding intensely managed landscapes (e.g. agricultural lands) (Baldock et al., 2015; Aronson et al., 2014, Fournier et al., 2020; Kowarik, 2011; Ives et al., 2015; Lepczyk et al., 2017). The Green Infrastructure (henceforth GI) plays a big role in providing such habitats in urban areas (Ferenc et al., 2014; Filazzola et al., 2019). A study by Sweet and colleagues, using data from Global Biodiversity Information Facility (GBIF), concluded that selected German cities from all states foster a considerable percentage (76%) of local biodiversity (Sweet et al., 2022). However, nature can thrive and thus provide us with valuable ecosystem services if certain habitat conditions are met (Beaugeard et al., 2020; Angold et al., 2006; Rudd et al., 2002). A study by Beaugeard et al. found that local urban biodiversity richness is highly benefited from the presence of green areas, proximity to the edge of the urban centre, and the proximity to green corridors. The presence of resource-rich green areas in urban contexts provides species with food and breeding habitats. Moreover, habitat patches with their edges next to contrasting, in this case urban, abiotic environments, are exposed to air, noise, and light pollution which negatively impacts the species living in that area (Driscoll et al., 2013). Lastly, movement and dispersal availability depend on the proximity to a green corridor (Beaugeard et al., 2020; Driscoll et al., 2013; Rudd et al., 2002).

The presence of high-quality GI and other similar nature-based solutions in cities is being increasingly referred to as a multifaceted tool able to mitigate the most pressing urbanization issues in the context of climate change (Sturiale and Scuderi, 2019; Gómez-Villarino et al., 2020; Madureira and Andresen, 2013). This involves installing, implementing or adapting green infrastructure into the existing city structure. Parks, green roofs, street vegetation, and tiny forests are some of the many examples of GI in cities (Liquete et al., 2015). The European Commission defines GI as a part of wider ecosystem services, which bring benefits not only to the natural environment but also to the wider population by cleaning the air, climate regulation, pollination, nutrient cycling, etc. (European Commission, 2019; Lai et al., 2018). In the urban context, GI provides all-around benefits for the city environment, infrastructure, and resident communities (Sturiale and Scuderi, 2019; Gómez-Villarino et al., 2020; Madureira and Andresen, 2013). The existence of high-quality GI, such as city parks, provides services for climate crisis adaptation firstly, by storm water management, which mitigates the effects of floods (Madureira and Andresen, 2013), and secondly by cleaning cycling nutrients, cleaning the air, and providing a cooling effect which helps to combat urban challenges such as air pollution and urban heat stress (Zardo et al., 2017). Moreover, the GI provides recreational spaces which are crucial for maintaining the social and personal well-being and health of local communities (Astell-Burt and Feng, 2019; Taylor and Hochuli, 2014; Annerstedt van den Bosch et al., 2015; Hegetschweiler et al., 2017). Different types of GI and their spatial arrangements provide different sets of benefits and are therefore relevant for understanding how biodiversity composition and ecology interact in urban settings to inform effective biodiversity sensitive urban designs (Garrard et al., 2018), including Nature Based solutions (Ronchi and Salata, 2022) and wildlife-inclusive cities (Apfelbeck et al., 2020).

However, despite the increasingly growing popularity of the use of GI as a tool for climate adaptation and mitigation, research focus on GI as the habitat of a large number of species that not only use urban environments as temporal or permanent habitat, but also perform ecological functions key for the efficiency of GI as a mitigation and adaptation hub has been limited (LaPoint et al., 2015; Apfelbeck, 2020; Schwarz, 2017; Loreau, 2001). While there are a number of research and reports investigating the connectivity, fragmentation of habitats as well as and the existence and provision of green corridors for wildlife in other spatial contexts such as natural reserves and agricultural areas, there is a clear lack of research in an urban context (van der Grift, 2005; Ovaskainen, 2012; Grashof-Bokdam, 1997).

The assessment and monitoring of biodiversity in urban areas has been performed until now through dedicated on-site studies and a small number of studies using digital biodiversity repositories. The question remains as to what extent an efficient and effective monitoring scheme could be implemented, one that not only facilitates comparisons across time and space, but also serves as an early change detection tool that complement local studies. The freely available biodiversity data provided by the Global Biodiversity Information Facility (GBIF) has been promoted for its central role, gathering and harmonizing biodiversity data worldwide, thereby facilitating the assessment and monitoring of biodiversity in multiple ecosystems (Proenca et al., 2017). As such, GBIF includes data from research and monitoring efforts as well as from citizen science, which have an even higher potential for biodiversity monitoring in urban areas (Li et al., 2019). While GBIF data has been investigated for its potential at large scales (national, global) and in natural ecosystems (Wolf, 2022; Sweet et al., 2022), the question remains as to what extent, and in which context, is GBIF data applicable to urban biodiversity assessment and monitoring. This study aims to fill the gap in research exploring the potential of GBIF data to identify drivers of species richness in urban areas across multiple spatial scales. With that aim, we measured bird, mammal and arthropods species richness in three Dutch cities and estimated the effect of land cover types and distance from the center of the city at multiple spatial scales.

Materials and methods

Study sites

We selected Amsterdam, Rotterdam and Groningen as the three case study cities as they represented three different city profiles regarding size, population and economic flows in the Netherlands. In this study we used land-use and biodiversity observation data only from the urban core region in order to avoid the misrepresentation and data bias in the suburban and peri-urban areas.

Amsterdam

Amsterdam is one of the biggest and well-known cities of the Netherlands with a core urban area of 712.3 Km^2 and a population of 907,976 within the whole municipality. In 2011, the city council of Amsterdam published a structural vision "Amsterdam 2040: economically strong and sustainable". As a part of the environmental vision of the development plan, the "Ecological vision" report focusing on ecology, biodiversity and green connectivity was developed (Gemeente Amsterdam, 2012). The development plan underlines that it drifts from previous GI development strategies, as it also focuses on environmental sustainability for local flora and fauna, and not only as recreation spaces for Amsterdam inhabitants (Gemeente Amsterdam, 2012). This marks an important shift, as it requires additional attention towards establishing and strengthening the infrastructure required for fostering biodiversity in a functioning ecological network for species movement and dispersal.

Rotterdam

With a core urban area of 585.8 Km^2 and a population count of 651,157 (population density 2,963/Km^2 in the whole municipality), this city is known for one of the largest ports in Europe. In 2018, the municipality of Rotterdam released an environmental program as a part of the college targets 2018–2022 and since then more than 20 ha of greenery have been added to the public spaces such as streets and squares (Gemeente Rotterdam, 2022). While this initiative has been very successful, it was the only environmental goal in the college targets 2018–2022 (Gemeente Rotterdam, 2018; Gemeente Rotterdam, 2022).

Groningen

Groningen has a core urban area of 147.3 Km^2 and population of 233,218 (population density 1,246/Km^2). In 2018, the development plan of the Green plan Groningen (Groenplan Groningen in Dutch) development plan focuses on planning and developing physical environments of Groningen. The agenda covers multiple disciplines such as urban growth and housing, economic development, community health, sustainable energy transition, climate-proof, and livable Groningen (Gemeente Groningen and Strootman Landshapsarchitecten, 2020). The latter includes implementing greening initiatives such as extending the tree cover (planting 1,000 trees a year) and implementing 30,000 m^2 of new green areas by creating and repurposing currently unused (gray infrastructure) space (Gemeente Groningen and Strootman Landshapsarchitecten, 2020). Moreover, in collaboration with residents, municipality launched a "Groningen climate-proof" initiative by encouraging (and subsidizing) citizens and private and business entities to invest in climate adaptation – green roofs, green facades, planting and adoption of trees and tiny forests in private and public areas, preserving rainwater and public and private urban garden initiatives (Gemeente Groningen, 2018). Furthermore, the municipality aims to not only plant more plants in general, but also more diverse plant species in order to increase the biodiversity and resilience of GI in the city (Gemeente Groningen and Strootman Landshapsarchitecten, 2020). With the help of these initiatives, the municipality strives to strengthen the ecological network in Groningen which would provide direct benefits for flora, fauna, local residents as well as climate adaptation.

Methods

In order to estimate biodiversity levels in the chosen cities, we used publicly available land cover data from the Copernicus Urban Atlas 2018 (European Union, Copernicus Land Monitoring Service 2018) from each of the cities. This data set includes several land use classes that were reclassified, using the QGIS software (QGIS Development Team, 2020), into 4 different land use classes for each city. This was categorized in line with the European Commission's established GI definitions. Thus, the classification was done as follows (Fig. 1):

- Class 1 – gray built up areas which include categories of the continuous urban fabric of varying density, including sports and leisure areas,
- Class 2 – green areas which included green urban areas, forests, herbaceous vegetation associations, and wetlands,
- Class 3 – agriculture areas included arable land, permanent crops, pastures with complex, and mixed cultivation patterns, and orchards.

Then the land use classes were rasterized and imported into GRASS GIS (GRASS Development Team, 2021) to calculate distance from the core of the cities to its periphery and perform a GI connectivity and patch metrics analysis.

For the multiscale analysis of biodiversity, polygon (hexagon) grids of 3 sizes – 0.2, 0.5 and 1 Km^2 per hexagon were created in order to estimate the spatial distribution of green areas in relation to species occurrences

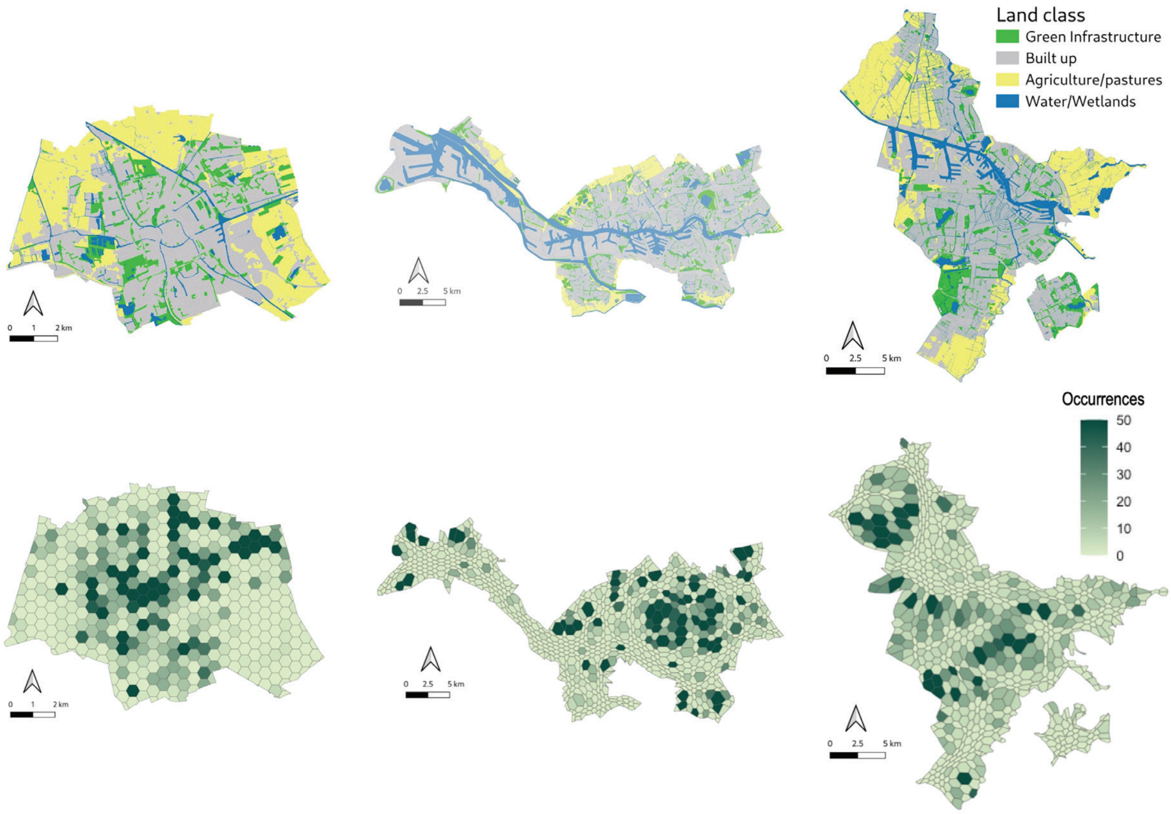

Fig. 1. Maps, from left to right, of Groningen, Rotterdam and Amsterdam with the resulting land classification (above) and a cartogram (0.5 Km^2 grid cell size) with the GBIF occurrences used in the analyses, from 2017 to 2021, for the three taxonomic groups studied (Aves, Arthropoda and Mammalia). The colour bar corresponds to the number of occurrences per grid cell, and the shape deformation corresponds to differences in sapling effort, accounted for as the number of years sampled. In this case, smaller cells indicate undersampling while bigger cells indicate oversampling.

and richness (Fig. 1). The raster and vector layers were cropped to city boundaries to contain only the urban core of the selected cities in order to focus on the urban green infrastructure connectivity while reducing the effect of rural areas surrounding the city.

Thereafter, publicly available biodiversity data from the Global Biodiversity Information Facility website was selected and downloaded (GBIF.org, GBIF.orgA, GBIF.orgB, 2022). For the purpose of this study, and to reduce sampling bias (Hughes et al., 2020), we group occurrence data from species in three large groups: arthropods, birds, and mammals. We do not explore plant biodiversity as it is also affected by urban planning (e.g. municipality-driven selection of species in GI), and Reptilia or Amphibia, as their distribution is largely affected by water bodies and our focus was on the impact of land uses in urban areas and distance to core. The downloaded data covered species occurrence records from 2017 to 2021 from Amsterdam, Groningen and Rotterdam. The data was explored to remove duplicates and synonyms; and reduce spatial uncertainty by limiting the observations to those that occur with uncertainty lower than 500 meters. Using the three grid sizes, we calculated species occurrences and species richness per group. To further account for the sampling bias, we reduced spatial clustering (Beck et al., 2014); the grid cells with no observations were excluded and the maximum richness was randomly set to 50 species. Additionally, for each grid, we calculated metrics such as distance to the core of the cities and number of pixels per land class.

The statistical analyses were performed using a model averaging approach at each spatial scale. We fitted a global weighted logistic regression model (i.e., Class 1 + Class 2 + Class 3 + distance to core) taking each of the taxonomic groups as the response variable (i.e., mammals, birds and arthropods) and calculated the Akaike weight (AIC*wv*) across all fitted models for each variable *v* (Burnham and Anderson, 2002). An AIC*wv* value can be interpreted as a normalized relative likelihood representing the fit of the model, facilitating model averaging. Thus, to estimate relative importance of each explanatory variable, we summed the AIC*wv* across all models in which the variable occurred. Higher AIC*wv* values indicate a higher importance of that variable relative to the other variables. Such model averaging allows assessments to be based on multiple models, as well as to mitigate the bias in parameter estimation that may occur selecting a single best model (Burnham and Anderson, 2002). The set of models with a delta AIC value higher than four was selected for each analysis to approximate the true model (Burnham and Anderson, 2002). All the analyses were performed in R software (R Core Team, 2020) using packages MuMIn

(Bartoň, 2020), arm (Gelman and Su, 2020), and spdep (Bivand et al., 2013) for model averaging; and ggplot2 (Wickham, 2016) for the figures.

Results

The focal cities selected had less than 10% of its area devoted to Green Infrastructure (Amsterdam = 4.1%, Rotterdam = 6.9% and Groningen = 7.2%). Groningen differed from Amsterdam and Rotterdam in its higher percentage of land cover with pastures/agriculture (Groningen = 29%, Amsterdam = 8% and Rotterdam = 14%). In terms of GI patch metrics the cities have similar mean GI patch sizes, between 0.9 (Groningen and Rotterdam) and 0.11 Km^2 (Amsterdam). Groningen has the smallest number of GI patches (902) which might be related to its smaller area; yet Rotterdam (4,150) has 1,000 more patches than Amsterdam (2,939) despite its smaller size. The shape of the patches is another relevant metric, as areas with less edge have more core area and are less affected by surrounding stressors like noise, air and light pollution. The shape index metric provides a measurement of the relation between total area and edge of the patches, where 1 would be a 1:1 ratio area: edge and higher values reflect a larger edge to area ratio. All cities have a large extent of GI edg

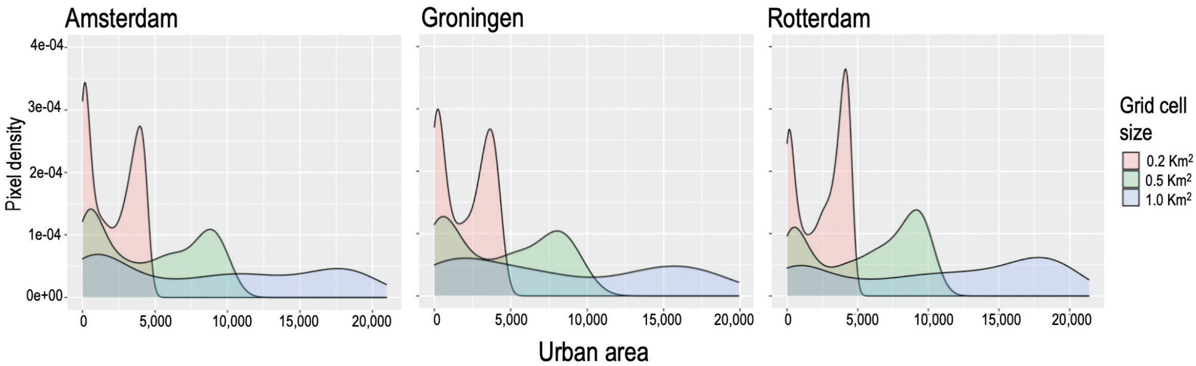

Fig. 2. Histogram of the distribution of urban class pixels across grid cell sizes.

Table 1. GBIF occurrences included in this study. Institutions refer to the main organizations/institutions providing the data and percentage citizen science (%) refers to the percentage of total occurrences that were gathered using citizen science projects. Natural History Museum Rotterdam (NMW), Finnish Biodiversity information facility (LAJI), Centre de Recerca Ecologica i Aplicacions Forestals (CREAF).

City	Group	Species	Institutions	Citizen science (%)	Total occurrences
Groningen	Mammalia	25	iNaturalist, naturgucker, NMR, Unie Van Waterschappen, Regelink	2.74	1.676
	Arthropoda	505	iNaturalist, naturgucker, Mosquito Alert	100.00	1.971
	Aves	193	eBird, Movebank, naturgucker, iNaturalist, Regelink, xeno-canto, LAJI	99.63	5.415
Amsterdam	Mammalia	21	iNaturalist, naturgucker, NMR Unie Van Waterschappen, Regelink	1.43	2.579
	Arthropoda	615	NMR, iNaturalist, naturgucker	87.82	2.110
	Aves	108	iNaturalist, naturgucker, NMR, Unie Van Waterschappen, Regelink, Naturalis, LAJI, Movebank	8.30	1.393
Rotterdam	Mammalia		iNaturalist, NMR, Walvisstrandingen	49.02	51
	Arthropoda	107	CREAF, iNaturalist, Mosquito Alert	73.13	536
	Aves	213	iNaturalist, NMR, LAJI, Naturalis, xeno-canto, eBird	98.32	11.568

es in relation to the total area of the GI, but Amsterdam and Rotterdam have values above 30 in shape index, while Groningen has a slightly lower value (shape index = 20). The assessment of spatial heterogeneity in all cities was, as expected, higher with small grid cell size, however heterogeneity was almost completely lost at 1 Km^2 for all the cities (Fig. 2).

A total of 27,299 species occurrences were included in the estimates of biodiversity (Amsterdam = 6,082, Rotterdam = 12,155 and Groningen = 9,062 occurrences, Table 1). In terms of taxonomic groups, the birds accounted for more than 50% of the total occurrences and comprised the majority of observations for all the cities surveyed (67.3% birds, 15.8% arthropods, 16.9% mammals). Spatial distribution of occurrences is clustered for all cities and at all scales. Using the grid cells approach, we found grids of 0.2 Km^2 with more than 500 occurrences but a large number of grid cells having no more than 20 observations and at least 50% of the cells with no occurrences in all cities and for all taxonomic groups, except in the case of arthropods in Rotterdam (30% cells with no occurrences) and birds in

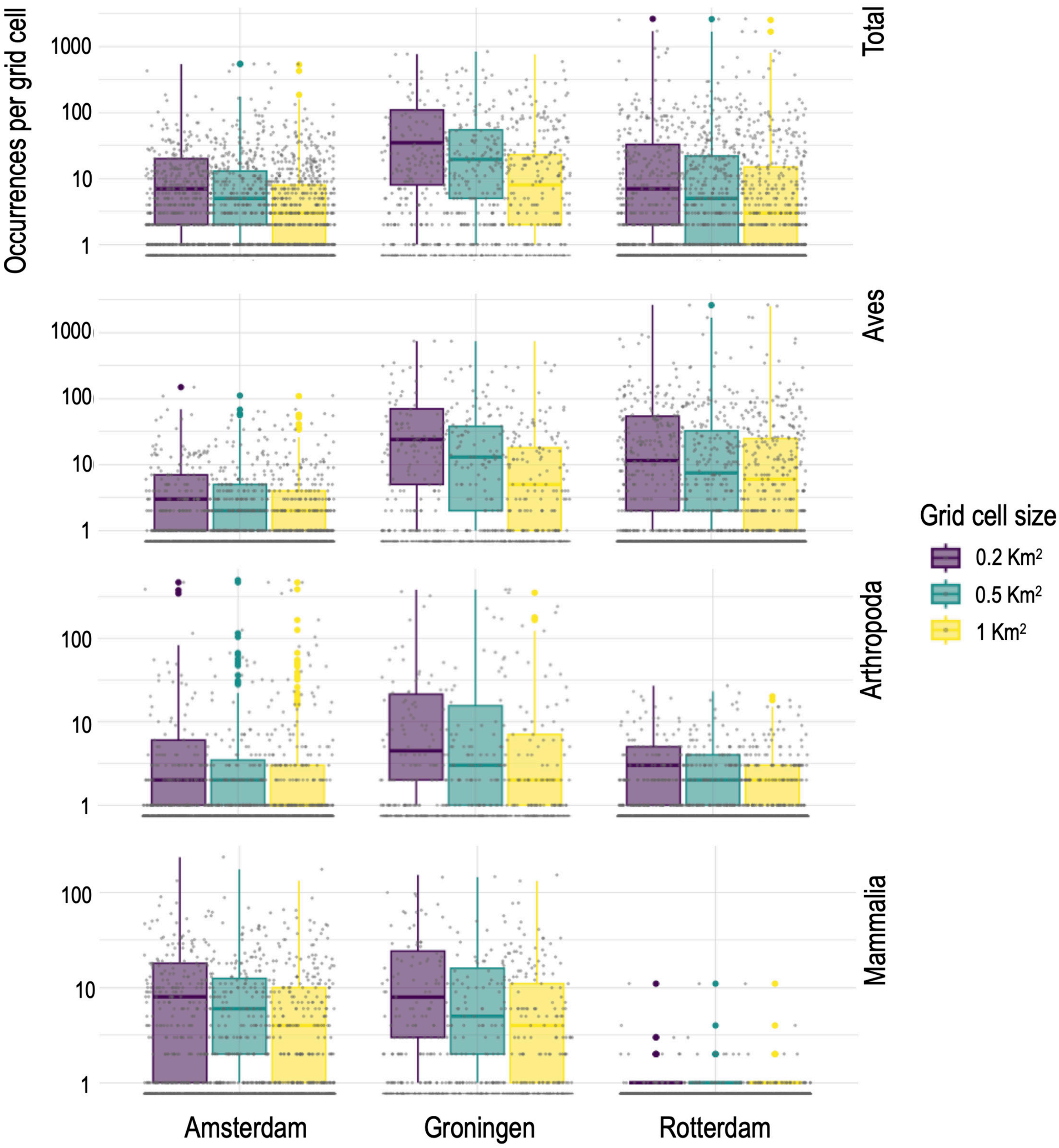

Fig. 3. Distribution of occurrences per grid cell size and across cities.

Amsterdam (38% cells with no occurrences) (Fig. 3).

Determinants of species richness across scales

Species richness changed across scales for all taxonomic groups and cities. In the case of Groningen, models explaining bird species richness, specifically, that include green cover were more significant explaining species richness at all scales. Mammal species richness was positively affected by urban cover at all scales (Fig. 4), while agriculture/pasture was only significantly negative at medium and large scales (0.5 and 1.0 Km^2). In the case of arthropods, agriculture/pasture cover and distance to the core negatively affected species richness at all scales, while urban cover negatively affected species richness at medium scales (0.5 Km^2).

In the case of Amsterdam, models including all variables best explained species richness in all the groups, however, the direction and significance of the effect varied with scale. Arthropod species richness was negatively correlated with all the variables but the significance of the effect varied with scale, especially in the case of distance to

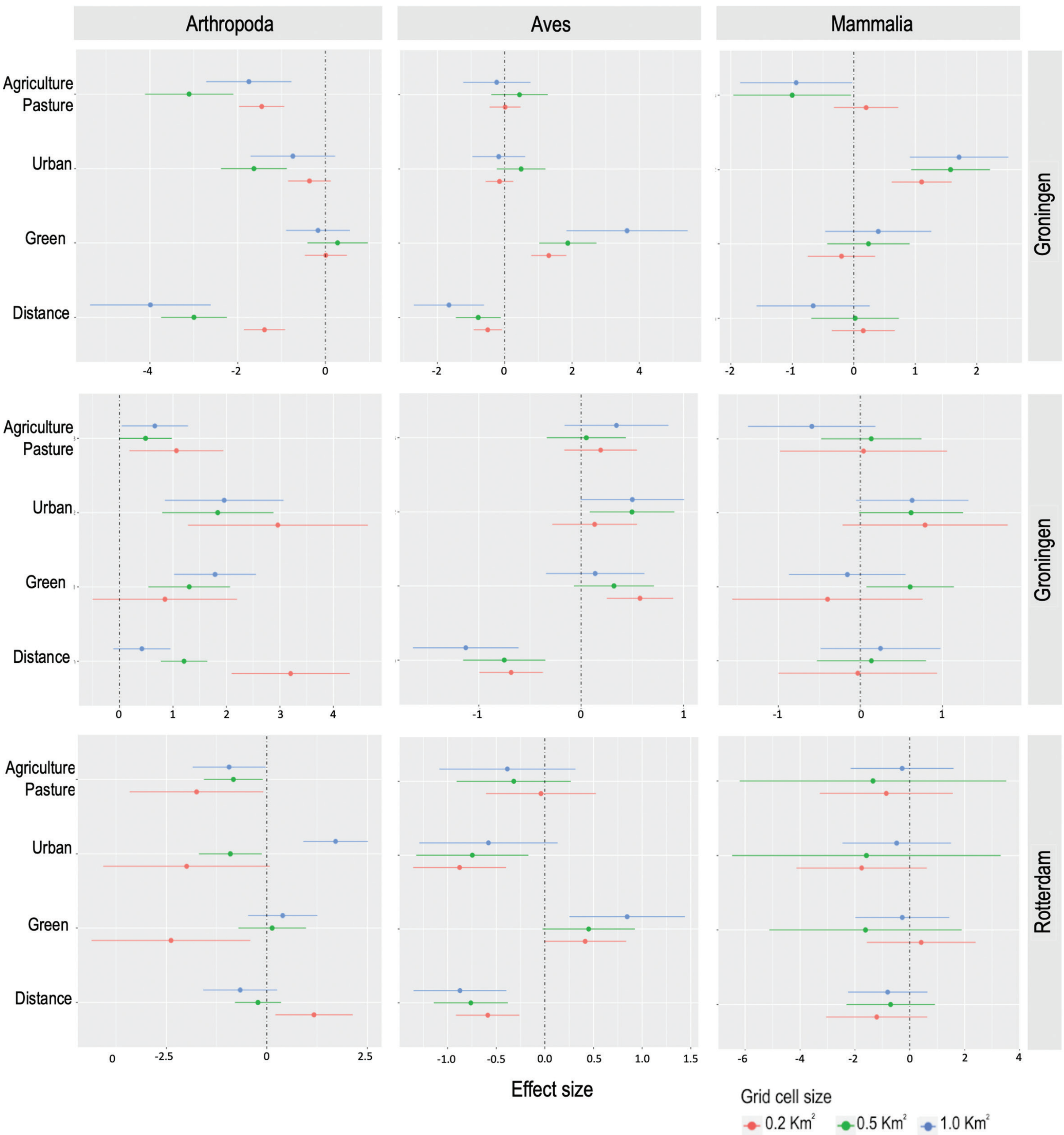

Fig. 4. Model-averaged effect sizes of the type of cover (urban, green and agriculture/pasture) and distance to the core of the city) at three spatial scales modeled against arthropod, bird (Aves) and mammal species richness. The coefficients correspond to the relative variable importance (i.e., w + v), and the bars correspond to the 95% confidence intervals. The model means are represented by the dashed vertical line.

the core. Bird species richness in Amsterdam was best explained at small scale, despite larger within variable variation in significance, with all variables having a positive effect except for distance to the core that was significantly negatively correlated. Species richness in mammals was not significantly correlated to the variables explored (Fig. 4).

For Rotterdam, models including all variables, best explained bird and arthropods species richness but the effect significance changed with scale. Arthropods species richness significantly declined with agriculture/pasture cover at the small scale (0.2 Km^2), while at large scale (1.0 Km^2), models including urban cover were positively related with species richness. Models including green cover (positive effect) and distance to the core (negative effect) best explained bird species richness at all scales, while urban cover was only negatively related at small and medium scales. In the case of arthropod species richness, similar to the case of Amsterdam and Groningen, variation across scales was higher, with distance to the core having a significant positive effect and, agriculture/pasture and urban cover had a negative effect at the smaller scale (Fig. 4).

Discussion

As urban expansion continues (Chen et al., 2020), biodiversity monitoring and management becomes of paramount importance to tackle ecosystem services and well-being of communities living in cities around the world (Aronson et al., 2017). This study focuses on the applicability of GBIF data at multiple scales to assess species richness patterns in relation to land cover spatial heterogeneity. Our study demonstrates the importance of spatial scale, in terms of resolution, when assessing the effect of land cover type on species richness and the relevance of GBIF data coverage. We found that, despite the clustered distribution of species occurrences in the three cities studied (approx. 50% of the cells had no occurrences), species richness in the three taxonomic groups investigated (birds, arthropods and mammals) is significantly affected by land cover type and distance to the core of the city, but the direction and strength of the effect varies across spatial scales. Specifically, we found that the spatial arrangement of land cover types, especially GI cover, and the taxonomic group studied, together with the spatial scale, have a significant effect on species richness.

Biodiversity observations in cities

Biodiversity data quality, quantity and coverage are key to reduce biases and uncertainty in assessments of change in ecosystem integrity across the world (Jansen et al., 2022; Rocchini and Garzon-Lopez, 2017). In this context, The Netherlands is one of the countries with the largest number of biodiversity observations in Europe (Wetzel et al., 2018), and as such, it has great potential to provide robust assessment. This is even more expected from cities, where, given its accessibility, more biodiversity data, compared to remote areas, is collected (Barbosa et al., 2013). We found that, while data quantity is higher, this does not improve data quality and area coverage for the three cities studied. In Amsterdam, Rotterdam and Groningen species observations were highly clustered, resulting in more than 50% of the urban areas with zero biodiversity observations. In accordance with other studies, the bigger cities (Amsterdam and Rotterdam) had significantly more data than the smaller city (Groningen) confirming the biases in data collection efforts in smaller cities (Kendal et al., 2020).

From data to assessments

Biodiversity data is often used to identify drivers of species distribution via species distribution models (SDMs) or to assess the rate of change in relation to anthropogenic disturbances. The outcomes of those analyses are conditioned by the characteristics of the biodiversity data used (Hughes et al., 2021). Such biases in biodiversity data due to uneven sampling intensity have often been tackled by resampling at larger scales (Dyderski et al., 2018). We examined whether and to what extent this approach was possible in cities. We found that decreasing spatial resolution resulted in a significant reduction of heterogeneity in the distribution of urban cover, a key determinant of species distribution in cities, to an extent that spatial heterogeneity was already lost at the 1 Km^2 grid size in our study.

Species distribute in relation to how they use resources and are used as a resource; this is determined by their size and patterns of aggregation, as well as their mobility (Condit, 2000; Braaker et al., 2014; Beaugeard et al., 2021). Consequently, the scale at which the effect of city heterogeneity can be assessed varies from one species to another. In this study, we focused on the exploration of the potential of GBIF data to assess the extent at which environmental variables in urban areas drive the spatial arrangement of species richness, an indicator of biodiversity, at multiple spatial scales. We found that in the three cities studied, the strength and direction of the variables driving species richness also depend on the taxonomic group and the spatial scale explored. The variation in the determinants of species richness per taxonomic group was so relevant, that in some cases it shifted from significantly positive to negative within the same taxonomic group. Previous research have explored the effect of land cover types for biodiversity in urban areas using varying grid sizes from 0.1 to 50 Km finding contrasting results on the direction of the effect. Studies performed at 1 Km^2 – a scale explored in our study – found mammal (Gallo et al., 2017; Hursh et al., 2023) and arthropod richness (Fenoglio et al., 2020) declined with urban cover, in line with our findings in the case of arthropods and contrary to our findings in the case of mammals for two of our cities (Groningen and Amsterdam), and species richness increased with the size of green areas (Cooper et al., 2021), in line with our findings for just one of cities explored (Rotterdam). Finally, studies performed at higher resolution (0.01 Km^2) found a general trend in increase of species richness with increasing distance to the core of the city (Sweet et al., 2022; Aznarez et al., 2022). The variation in the selection of scales and the outcomes on the patterns of biodiversity further demonstrate the importance of exploring the drivers of biodiversity at multiple scales, and strengthening of digital biodiversity repositories to allow for research at smaller scales and the exploration of the effect of spatial scales at the species level. Previous studies have highlighted the importance of the type and characteristics of green spaces as an important factor for biodiversity (Das et al., 2023; Grade et al., 2022; Xie et al., 2023; Lepczyk et al., 2017) which might explain the differences in the direction of the effect of urban and green areas across cities.

Conclusions

The increase of green infrastructure in cities across Europe (Xu et al., 2022), has moved urban areas closer to the goal of creating biodiversity inclusive cities as expected from the Sustainable Development Goals (SDG) (Opoku, 2019). In this study, we have shown that the autoecology of organisms surveyed, even at the large grouping categories (i.e., mammals, birds and arthropods) (Concepción et al., 2015), and the spatial scale(s) of the assessments are critical to identify the effect of urban and green infrastructure in cities. As well as the importance of the GI to biodiversity. In this context, developing integrative biodiversity assessments that accurately inform the design and management of biodiversity-inclusive cities requires the recognition of the ecology of species as an important factor driving ecological dynamics and responses to stress at multiple scales (Garzon-Lopez et al., 2015). We propose a multiscale approach that includes: i) the tracking of heterogeneity, ii) biodiversity data coverage, and iii) and the ecology of the species, all of them across spatial scales.

Future research should further evaluate the potential of GBIF data to explore the role of these environmental variables, as well as other variables relevant in urban areas, such as distance to water bodies and gray infrastructure types, shaping biodiversity patterns at multiple scales and accounting for the ecology at the species of functional levels.

References

Alizadeh, M.R., Abatzoglou, J.T., Adamowski, J.F., Prestemon, J.P., Chittoori, B., Akbari Asanjan, A., Sadegh, M., 2022. Increasing heat stress inequality in a warming climate. *Earth's Future*, 10 (2): e2021EF002488. https://doi.org/10.1029/2021ef002488

Angold, P.G., Sadler, J.P., Hill, M.O., Pullin, A., Rushton, S., Austin, K., Small, E., Wood, B., Wadsworth, R., Sanderson, R., Thompson, K., 2006. Biodiversity in urban habitat patches. *Science of the Total Environment*, 360 (1– 3): 196–204. https://doi.org/10.1016/j.scitotenv.2005.08.035

Annerstedt van den Bosch, M., Mudu, P., Uscila, V., Barrdahl, M., Kulinkina, A., Staatsen, B., Swart, W., Kruize, H., Zurlyte, I., Egorov, A.I., 2015. Development of an urban green space indicator and the public health rationale. *Scandinavian Journal of Public Health*, 44 (2): 159–167. https://doi.org/10.1177/1403494815615444

Apfelbeck, B., Snep, R., Hauck, T.E., Ferguson, J., Holy, H., Jakoby, M., MacIvor, J., Schär, L., Taylor, M., Weisser, W.W., 2020. Designing wildlife-inclusive cities that support human-animal co-existence. *Landscape and Urban Planning*, 200: 103817. https://doi.org/10.1016/j.landurbplan.2020.103817

Aronson, M.F., La Sorte, F.A., Nilon, C.H., Katti, M., Goddard, M.A., Lepczyk, C.A., Warren, P.S., Williams, N.S., Cilliers, S., Clarkson, B., Dobbs, C., Dolan, R., Hedblom, M., Klotz, S., Kooijmans, J.L., Kühn, I., Macgregor-Fors, I., McDonnell, M., Mörtberg, U., Pysek, P., Siebert, S., Sushinsky, J., Werner, P., Winter, M., 2014. A global analysis of the impacts of urbanization on bird and plant diversity reveals key anthropogenic drivers. *Proceedings of the Royal Society B. Biological Sciences*, 281 (1780): 20133330. https://doi.org/10.1098/rspb.2013.3330

Aronson, M., Lepczyk, C.A., Evans, K.L., Goddard, M.A., Lerman, S.B., MacIvor, J.S., Nilon, C.H., Vargo, T., 2017. Biodiversity in the city: key chal lenges for urban green space management. *Frontiers in Ecology and the Environment*, 15 (4): 189–96. https://doi.org/10.1002/fee.1480.

Astell-Burt, T., Feng, X., 2019. Urban green space, tree canopy, and prevention of heart disease, hypertension, and diabetes: a longitudinal study. *The Lancet Planetary Health*, 3, Suppl. 1: S16. https://doi.org/10.1016/s2542-5196(19)30159-7

Aznarez, C., Svenning, J.-C., Taveira, G., Baró, F., Pascual., U., 2022. Wildness and habitat quality drive spatial patterns of urban biodiversity. *Landscape and Urban Planning*, 228: 104570. https://doi.org/10.1016/j.landurbplan.2022.104570.

Balaban, O., 2012. Climate change and cities: a review on the impacts and policy responses. *Metu Journal of the Faculty of Architecture*, 29 (1): 21–44. https://doi.org/10.4305/metu.jfa.2012.1.2

Baldock, K.C.R., Goddard, M.A., Hicks, D.M., Kunin, W.E., Mitschunas, N., Osgathorpe, L.M., Potts, S.G., Robertson, K.M., Scott, A.V., Stone, G.N., Vaughan, I.P., Memmott, J., 2015. Where is the UK's pollinator biodiversity? The importance of urban areas for flower-visiting insects. *Proceedings of the Royal Society B. Biological Sciences*, 282 (1803): 20142849. http://doi.org/10.1098/rspb.2014.2849

Barbosa, A.M., Pautasso, M., Figueiredo. D., 2013. Species–people correlations and the need to account for survey effort in biodiversity analyses. *Diversity and Distributions*, 19 (9): 1188–97. https://doi.org/10.1111/ddi.12106.

Bartoň, K., 2020. *MuMIn: Multi-Model Inference*. [cit. 2023-02-14]. https://CRAN.R-project.org/package=MuMIn

Beaugeard, E., Brischoux, F., Angelier, F., 2020. Green infrastructures and ecological corridors shape avian biodiversity in a small French city. *Urban Ecosystems*, 24 (3): 549– 560. https://doi.org/10.1007/s11252-020-01062-7

Beaugeard, E., Brischoux, F., Angelier, F., 2021. Green infrastructures and ecological corridors shape avian biodiversity in a small French city. *Urban Ecosystems*, 24 (3): 549–60. https://doi.org/10.1007/s11252-020-01062-7.

Beck, J., Böller, M., Erhardt, A., Schwanghart, W., 2014. Spatial bias in the GBIF database and its effect on modeling species' geographic distributions. *Ecological Informatics*, 19: 10–15. https://doi.org/10.1016/j.ecoinf.2013.11.002.

Beninde, J., Veith, M., Hochkirch, A., 2015. Biodiversity in cities needs space: a meta-analysis of factors determining intra-urban biodiversity variation. *Ecology Letters*, 18 (6): 581–592. https://doi.org/10.1111/ele.12427

Bivand, R. S., Pebesma, E., Gomez-Rubio, V., 2013. *Applied spatial data analysis with R.* 2nd ed. New York: Springer. [cit. 2023-02-11]. http://www.asdar-book.org/

Braaker, S., Ghazoul, J., Obrist, M.K., Moretti, M., 2014. Habitat connectivity shapes urban arthropod communities: the key role of green roofs. *Ecology*, 95 (4): 1010–1021. https://doi.org/10.1890/13-0705.1

Burnham, K.P., Anderson, D.R., 2002. *Model selection and multimodel inference: a practical information-theoretic approach*. 2nd ed. New York: Springer. 488 p.

Chakraborty, T., Hsu, A., Manya, D., Sheriff, G., 2019. Disproportionately higher exposure to urban heat in lower-income neighborhoods: a multi-city perspective. *Environmental Research Letters*, 14 (10): 105003. https://doi.org/10.1088/1748- 9326/ab3b99

Checker, M., 2011. Wiped out by the "Greenwave": environmental gentrification and the paradoxical politics of urban sustainability. *City & Society*, 23 (2): 210–229.https://doi.org/10.1111/j.1548-744x.2011.01063.x

Chen, G., Li, X., Liu, X., Chen, Y., Liang, X., Leng, J., Xu, X., Liao, W., Qiu, Y., Wu, Q., Huang, K., 2020. Global projections of future urban land expansion under shared socioeconomic pathways. *Nature Communications*, 11 (1): 537. https://doi.org/10.1038/s41467-020-14386-x

Comber, A., Brunsdon, C., Green, E., 2008. Using a GIS-based network analysis to determine urban greenspace accessibility for different ethnic and religious groups. *Landscape and Urban Planning*, 86 (1): 103–114. https://doi.org/10.1016/j.landurbplan.2008.01.002

Concepción, E.D., Moretti, M., Altermatt, F., Nobis, M.P., Obrist, M.K, 2015. Impacts of urbanisation on biodiversity: the role of species mobility, degree of specialisation and spatial scale. *Oikos*, 124 (12): 1571–82. https://doi.org/10.1111/oik.02166.

Condit, R., 2000. Spatial patterns in the distribution of tropical tree species. *Science*, 288 (5470): 1414–1418. https://doi.org/10.1126/science.288.5470.1414.

Cooper, D.S., Wood, E.M., Katz, N.D., Superfisky, K., Osborn, F., Novoselov, A., Tarczynski, J., Bacasen, L.K., 2021. Large cities fall behind in "Neighborhood Biodiversity". *Frontiers in Conservation Science,* 2: 734931. https://www.frontiersin.org/articles/10.3389/fcosc.2021.734931

Czamanski, D., 2008. Urban sprawl and ecosystems — can nature survive? *International Review of Environmental and Resource Economics*, 2 (4): 321–366. https://doi.org/10.1561/101.00000019

Das, M., Das, A., Pereira, P., 2023. Developing an integrated urban ecological efficiency framework for spatial ecological planning: a case on a tropical mega metropolitan area of the global south. *Geoscience Frontiers*, 14 (1): 101489. https://doi.org/10.1016/j.gsf.2022.101489

De Carvalho, R.M., Szlafsztein, C.F., 2019. Urban vegetation loss and ecosystem services: the influence on climate regulation and noise and air pollution. *Environmental Pollution*, 245: 844–852. https://doi.org/10.1016/j.envpol.2018.10.114

Dyderski, M.K., Paź, S., Frelich, L.E., Jagodziński, A.M., 2018. How much does climate change threaten European forest tree species distributions? *Global Change Biology*, 24 (3): 1150–63. https://doi.org/10.1111/gcb.13925.

Driscoll, D.A., Banks, S.C., Barton, P.S., Lindenmayer, D.B., Smith, A.L., 2013. Conceptual domain of the matrix in fragmented landscapes. *Trends in Ecology and Evolution*, 28 (10): 605–613. https://doi.org/10.1016/j.tree.2013.06.010

Elmqvist, T., Redman, C.L., Barthel, S., Costanza, R., 2013. History of urbanization and the missing ecology. In *Urbanization, biodiversity and ecosystem services: challenges and opportunities*. Dordrecht: Springer, p. 13–30. http://dx.doi.org/10.1007/978-94-007-7088-1_2

European Union, Copernicus Land Monitoring Service 2018, European Environment Agency (EEA). *Urban Atlas* 2018. [cit. 2023-02-02]. https://land.copernicus.eu/local/urban-atlas/urban-atlas-2018

European Union, 2016. *Urban Europe: statistics on cities, towns and suburbs: 2016 edition*. Eurostat Statistical Books. Luxembourg: Publications Office. [cit. 2023-02-27]. https://op.europa.eu/publication/doi/10.2785/91120

European Commission, 2019. *Communication from the Commission to the European Parliament, the European Council, the Council, the European Economic and Social Committee and the Committee of the Regions*. 640. [cit. 2023-02-08]. https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:52019DC0640&from=EN

Fenoglio, M.S., Rossetti, M.R., Videla, M., 2020. Negative effects of urbanization on terrestrial arthropod communities: a meta-analysis. *Global Ecology and Biogeography*, 29 (8): 1412–29. https://doi.org/10.1111/geb.13107

Ferenc, M., Sedláček, O., Fuchs, R., Dinetti, M., Fraissinet, M., Storch, D., 2014. Are cities different? Patterns of species richness and beta diversity of urban bird communities and regional species assemblages in Europe. *Global Ecology and Biogeography*, 23 (4): 479–489.

Filazzola, A., Shrestha, N., MacIvor, J.S., 2019. The contribution of constructed green infrastructure to urban biodiversity: a synthesis and meta-analysis. *Journal of Applied Ecology*, 56 (9): 2131–2143. https://doi.org/10.1111/1365-2664.13475

Fournier, B., Frey, D., Moretti, M., 2020. The origin of urban communities: from the regional species pool to community assemblages in city. *Journal of Biogeography*, 47 (3): 615–629. https://doi.org/10.1111/jbi.13772

Gallo, T., Fidino, F., Lehrer, E.W, Magle, S.B., 2017. Mammal diversity and metacommunity dynamics in urban green spaces: implications for urban wildlife conservation. *Ecological Applications*, 27 (8): 2330–41. https://doi.org/10.1002/eap.1611

Garrard, G.E., Williams, N.S.G., Mata, L., Thomas,J., Bekessy, S.A., 2018. Biodiversity sensitive urban design. *Conservation Letters*, 11 (2): e12411. https://doi.org/10.1111/conl.12411.

Garzon-Lopez, C.X., Jansen, P.A., Bohlman, S.A., Ordonez, A., Olff, H., 2014. Effects of sampling scale on patterns of habitat association in tropical trees. *Journal of Vegetation Science*, 25 (2): 349–362. https://doi.org/10.1111/jvs.12090.

GBIF.org (26 July 2022) GBIF Occurrence Download https://doi.org/10.15468/dl.8xmjx9

GBIF.orgA (26 July 2022) GBIF Occurrence Download https://doi.org/10.15468/dl.k9gn29

GBIF.orgB (20 July 2022) GBIF Occurrence Download https://doi.org/10.15468/dl.s6uz7x

Gelman, A., Su, Y.-S. 2020. *arm: data analysis using regression and multilevel/hierarchical models*. [cit. 2023-02-02]. https://CRAN.R-project.org/package=arm

Gemeente Amsterdam, 2012. *Ecologische visie, ecologie, biodiversiteit en groene verbindingen in Amsterdam* [Ecological vision, ecology, biodiversity and green space connectivity in Amsterdam]. [cit. 2023-02-15]. https://www.publicspaceinfo.nl/media/uploads/files/AMSTERDAM_2012_0014.pdf

Gemeente Groningen, Strootman Landshapsarchitecten, 2020. *Groenplan Groningen. Gemeente Groningen. 2018. Dit kun je doen - Groningen Klimaatbestendig. Groningen Klimaatbestendig* [This is what you can do – Climate proof Groningen]. [cit. 2023-02-15]. https://groningenklimaatbestendig.nl/dit-kun-je-doen/

Gemeente Rotterdam, 2018. *Collegetargets 2018–2022. Gemeente Rotterdam*. [cit. 2023-02-15]. https://www.rotterdam.nl/bestuur-organisatie/collegetargets-2018-2022/

Gemeente Rotterdam, 2022. *De finish Rotterdam gaat voor groen 2018–2022* [The finish of Rotterdam goes green]. [cit. 2023-02-15]. https://www.kanbouwen.nl/wp-content/uploads/2022/04/Gemeente-Rotterdam-Rotterdam-gaat-voor-groen-De-Finish.pdf

Gómez-Villarino, M.T., Gómez Villarino, M., Ruiz-Garcia, L., 2020. Implementation of urban green infrastructures in peri-urban areas: a case study of climate change mitigation in Madrid. *Agronomy*, 11 (1): 31. https://doi.org/10.3390/agronomy11010031

Grade, A.M., Warren, P.S., Lerman, S.B., 2022. Managing yards for mammals: mammal species richness peaks in the suburbs. *Landscape and Urban Planning*, 220 (April): 104337. https://doi.org/10.1016/j.landurbplan.2021.104337

Grashof-Bokdam, C., 1997. Forest species in an agricultural landscape in the Netherlands: effects of habitat fragmentation. *Journal of Vegetation Science*, 8 (1): 21–28. https://doi.org/10.2307/3237238

GRASS Development Team, 2021. *Geographic Resources Analysis Support System (GRASS) Software, Version 8.0*. Electronic document. Open Source Geospatial Foundation. [cit. 2023-02-28]. http://grass.osgeo.org

Guerry, A.D., Smith, J.R., Lonsdorf, E., Daily, G.C., Wang, X., Chun, Y., 2021. *Urban nature and biodiversity for cities*. Policy Briefing. Global Platform for Sustainable Cities, World Bank. Washington, DC.

Hursh, S. H., Bauder, J.M., Fidino, M., Drake, D., 2023. An urban cast of characters: landscape use and cover influencing mammal occupancy in an American midwestern city. *Landscape and Urban Planning*, 229: 104582. https://doi.org/10.1016/j.landurbplan.2022.104582

Hegetschweiler, K.T., de Vries, S., Arnberger, A., Bell, S., Brennan, M., Siter, N., Olafsson, A.S., Voigt, A., Hunziker, M., 2017. Linking demand and supply factors in identifying cultural ecosystem services of urban green infrastructures: a review of European studies. *Urban Forestry & Urban Greening*, 21: 48–59. https://doi.org/10.1016/j.ufug.2016.11.002

Hughes, A., Orr, M., Ma, K., Costello, M., Waller, J., Provoost, P., Zhu, C., Qiao, H., 2021. Sampling biases shape our view of the natural world. *Ecography*, 44: 1259–1269. https://doi.org/10.1111/ecog.05926.

Ives, C.D., Lentini, P.E., Threlfall, C.G., Ikin, K., Shanahan, D.F., Garrard, G.E., Bekessy, S.A., Fuller, R.A., Mumaw, L., Rayner, L., Rowe, R., Valentine, L.E., Kendal, D., 2015. Cities are hotspots for threatened species. *Global Ecology and Biogeography*, 25 (1): 117–126. https://doi.org/10.1111/geb.12404

Jansen, J., Woolley, S.N.C., Dunstan, P.K., Foster, S.D., Hill, N.A, Haward, M., Johnson, C.R., 2022. Stop ignoring map uncertainty in biodiversity science and conservation policy. *Nature Ecology & Evolution*, 6 (7): 828–29. https://doi.org/10.1038/s41559-022-01778-z

Kendal, D., Egerer, M., Byrne, J.A., Jones, P.J., Marsh, P., Threlfall, C.G., Allegretto, G., Kaplan, H., Nguyen, H.K., Pearson, S., Wright, A., 2020. City-size bias in knowledge on the effects of urban nature on people and biodiversity. *Environmental Research Letters*, 15 (12): 124035.

Knapp, S., Aronson, M.F., Carpenter, E., Herrera-Montes, A., Jung, K., Kotze, D.J., La Sorte, F.A., Lepczyk, C.A., MacGregor-Fors, I., MacIvor, J.S., Moretti, M., 2021. A research agenda for urban biodiversity in the global extinction crisis. *Bioscience*, 71 (3): 268–279.

Kowarik, I., 2011. Novel urban ecosystems, biodiversity, and conservation. *Environmental Pollution*, 159 (8–9): 1974–1983. https://doi.org/10.1016/j.envpol.2011.02.022

Kumar, P., 2021. Climate change and cities: challenges ahead. *Frontiers in Sustainable Cities*, 3. https://doi.org/10.3389/frsc.2021.645613

La Sorte, F.A., Tingley, M.W., Hurlbert, A.H., 2014. The role of urban and agricultural areas during avian migration: an assessment of within-year temporal turnover. *Global Ecology and Biogeography*, 23 (11): 1225–1234. https://doi.org/10.1111/geb.12199

Lai, S., Leone, F., Zoppi, C., 2018. Implementing green infrastructures beyond protected areas. *Sustainability*, 10

(10): 3544. https://doi.org/10.3390/su10103544
LaPoint, S., Balkenhol, N., Hale, J., Sadler, J., Ree, R., 2015. Ecological connectivity research in urban areas. *Functional Ecology*, 29 (7): 868–878. https://doi.org/10.1111/1365-2435.12489
Lepczyk, C.A., Aronson, M.F.J., Evans, K.L., Goddard, M.A., Lerman, S.B., MacIvor. J.S., 2017. Biodiversity in the city: fundamental questions for understanding the ecology of urban green spaces for biodiversity conservation. *BioScience*, 67 (9): 799–807. https://doi.org/10.1093/biosci/bix079
Li, E., Parker, S.S, Pauly, G.B., Randal, J.M., Brown, B.V., Cohen, B.S., 2019. An urban biodiversity assessment framework that combines an urban habitat classification scheme and citizen science data. *Frontiers in Ecology and Evolution*, 7. https://doi.org/10.3389/fevo.2019.00277.
Liquete, C., Kleeschulte, S., Dige, G., Maes, J., Grizzetti, B., Olah, B., Zulian, G., 2015. Mapping green infrastructure based on ecosystem services and ecological networks: a Pan-European case study. *Environmental Science & Policy*, 54: 268–280. https://doi.org/10.1016/j.envsci.2015.07.009
Loreau, M., Naeem, S., Inchausti, P., Bengtsson, J., Grime, J.P., Hector, A., Hooper, D.U., Huston, M.A., Raffaelli, D., Schmid, B., Tilman, D., 2001. Biodiversity and ecosystem functioning: current knowledge and future challenges. *Science,* 294 (5543): 804–808.
Lososová, Z., Čeplová, N., Chytrý, M., Tichý, L., Danihelka, J., Fajmon, K., Láníková, D., Preislerová, Z., Řehořek, V., 2016. Is phylogenetic diversity a good proxy for functional diversity of plant communities? A case study from urban habitats. *Journal of Vegetation Science*, 27 (5): 1036–1046. https://doi.org/10.1111/jvs.12414
Lucertini, G., Musco, F., 2020. Circular urban metabolism framework. *One Earth*, 2 (2): 138–142. https://doi.org/10.1016/j.oneear.2020.02.004
Madureira, H., Andresen, T., 2013. Planning for multifunctional urban green infrastructures: promises and challenges. *Urban Design International*, 19 (1): 38–49. https://doi.org/10.1057/udi.2013.11
Opoku, A., 2019. Biodiversity and the built environment: implications for the Sustainable Development Goals (SDGs). *Resources, Conservation and Recycling*, 141 (February): 1–7. https://doi.org/10.1016/j.resconrec.2018.10.011
Ovaskainen, O., 2012. How to develop the nature conservation strategies for the Netherlands? *De Levende Natuur*, 114 (2): 59–62.
Proka, A., Hisschemöller, M., Loorbach, D., 2018. Transition without conflict? Renewable energy initiatives in the Dutch energy transition. *Sustainability*, 10 (6): 1721. https://doi.org/10.3390/su10061721
QGIS Development Team, 2020. *QGIS Geographic Information System*. Open Source Geospatial Foundation. [cit. 2023-02-24]. http://qgis.org
R Core Team, 2020. *R: a language and environment for statistical computing*. R Foundation for Statistical Computing, Vienna, Austria. [cit. 2023-02-06]. https://www.R-project.org/
Rocchini, D., Garzon-Lopez, C.X., 2017. Cartograms tool to represent spatial uncertainty in species distribution. *Research Ideas and Outcomes*, 3: e12029
Ronchi, S., Salata, S., 2022. Insights for the enhancement of urban biodiversity using nature-based solutions: the role of urban spaces in green infrastructures design. In Mahmoud, I.H., Morello, E., Lemes de Oliveira, F., Geneletti, D. *Nature-based solutions for sustainable urban planning: greening cities, shaping cities*. Contemporary Urban Design Thinking. Cham: Springer International Publishing, p. 47–68. https://doi.org/10.1007/978-3-030-89525-9_3
Rudd, H., Vala, J., Schaefer, V., 2002. Importance of backyard habitat in a comprehensive biodiversity conservation strategy: a connectivity analysis of urban green spaces. *Restoration Ecology*, 10 (2): 368–375. https://doi.org/10.1046/j.1526- 100x.2002.02041.x
Schwarz, N., Moretti, M., Bugalho, M.N., Davies, Z.G., Haase, D., Hack, J., Hof, A., Melero, Y., Pett, T.J., Knapp, S., 2017. Understanding biodiversity-ecosystem service relationships in urban areas: a comprehensive literature review. *Ecosystem Services*, 27: 161–171. https://doi.org/10.1016/j.ecoser.2017.08.014
Seto, K.C., Dhakal, S., Bigio, A., Blanco, H., Delgado, G.C., Dewar, D., Huang, L., Inaba, A., Kansal, A., Lwasa, S., McMahon, J.E., Müller, D.B., Murakami, J., Nagendra, H., Ramaswami, A., 2014. Human settlements, infrastructure and spatial planning. In Edenhofer, O., Pichs-Madruga, R., Sokona, Y., Farahani, E., Kadner, S., Seyboth, K., Adler, A., Baum, I., Brunner, S., Eickemeier, P., Kriemann, B., Savolainen, J., Schlömer, S., von Stechow, C., Zwickel, T., Minx, J. C. (eds). *Climate change 2014: mitigation of climate change: Contribution of Working Group III to the Fifth assessment report of the Intergovernmental Panel on Climate Change*. Cambridge, United Kingdom and New York, NY, USA: Cambridge University Press, p. 923–1000. https://doi.org/10.1017/CBO9781107415416.018
Sicard, P., Agathokleous, E., Marco, A.D., Paoletti, E., Calatayud, V., 2021. Urban population exposure to air pollution in Europe over the last decades. *Environmental Sciences Europe*, 33: 28. https://doi.org/10.1186/s12302-020-00450-2
Sturiale, L., Scuderi, A., 2019. The role of green infrastructures in urban planning for climate change adaptation. *Climate*, 7 (10): 119. https://doi.org/10.3390/cli7100119
Sultana, M., Storch, I., 2021. Suitability of open digital species records for assessing biodiversity patterns in cities: a case study using avian records. *Journal of Urban Ecology*, 7 (1): juab014. https://doi.org/10.1093/jue/juab014
Sweet, F.S., Apfelbeck, B., Hanusch, M., Garland Monteagudo, C., Weisser, W.W., 2022. Data from public and governmental databases show that a large proportion of the regional animal species pool occurs in cities in Germany. *Journal of Urban Ecology*, 8 (1): juac002. https://doi.org/10.1093/jue/juac002
Taylor, L., Hochuli, D.F., 2014. Creating better cities: how biodiversity and ecosystem functioning enhance urban residents' wellbeing. *Urban Ecosystems*, 18 (3): 747–762.

https://doi.org/10.1007/s11252-014-0427-3

van der Grift, E.A., 2005. Defragmentation in the Netherlands: a success story? *GAIA – Ecological Perspectives for Science and Society*, 14 (2): 144–147. https://doi.org/10.14512/gaia.14.2.16

Watkins, S.L., Mincey, S.K., Vogt, J., Sweeney, S.P., 2016. Is planting equitable? An examination of the spatial distribution of nonprofit urban tree-planting programs by canopy cover, income, race, and ethnicity. *Environment and Behavior*, 49 (4): 452–482. https://doi.org/10.1177/0013916516636423

Wetzel, F.T., Bingham, H.C., Groom, Q., Haase, P., Kõljalg, U., Kuhlmann, M., Martin, C.S., Penev, L., Robertson, L., Saarenmaa, H., Schmeller, D.S., Stoll, S., Tonkin, J.D., Häuser, C.L., 2018. Unlocking biodiversity data: prioritization and filling the gaps in biodiversity observation data in Europe. *Biological Conservation*, 221 (May): 78–85. https://doi.org/10.1016/j.biocon.2017.12.024

Wickham, H., 2016. *ggplot2: elegant graphics for data analysis*. Use R! Springer-Verlag New York.

Wolf, J.M., Jeschke, J.M., Voigt, C.C., Itescu, Y., 2022. Urban affinity and its associated traits: a global analysis of bats. *Global Change Biology*, 28: 5667–5682. https://doi.org/10.1111/gcb.16320

Xie, Ch., Wang, J., Haase, D., Wellmann, T., Lausch, D., 2023. Measuring spatio-temporal heterogeneity and interior characteristics of green spaces in urban neighborhoods: a new approach using gray level co-occurrence matrix. *Science of the Total Environment*, 855: 158608. https://doi.org/10.1016/j.scitotenv.2022.158608.

Xu, F., Yan, J., Heremans, S., Somers, B., 2022. Pan-European urban green space dynamics: a view from space between 1990 and 2015. *Landscape and Urban Planning*, 226 (October): 104477. https://doi.org/10.1016/j.landurbplan.2022.104477

Zardo, L., Geneletti, D., Pérez-Soba, M., Van Eupen, M., 2017. Estimating the cooling capacity of green infrastructures to support urban planning. *Ecosystem Services*, 26: 225–235. https://doi.org/10.1016/j.ecoser.2017.06.016

Zhou, Y., Varquez, A. C. G., Kanda, M., 2019. High-resolution global urban growth projection based on multiple applications of the SLEUTH urban growth model. *Scientific Data*, 6 (1): article number 34. https://doi.org/10.1038/s41597-019-0048-z

Received March 6, 2023
Accepted May 31, 2023