

# Biodiversity in cities. The effect of spatial scale and biodiversity data in heterogeneous landscapes.

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## Research Article

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# Abstract

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 online digital biodiversity databases. 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. 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. While GBIF data has been investigated for its potential at large scales (national, global) 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 have shown that the type of organisms surveyed -as well as its ecology and the spatial scale(s) of the assessments are critical to identify the effect of urban and green infrastructure in cities.

## 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 & 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 unequal 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<sup>2</sup> 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 & Szlafsztein, 2019). Furthermore, urbanization and city development has 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. And 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; CBD, 2012; Lepczyk et al., 2017).

As the global biodiversity crisis deepens (Knapp et al., 2021; Guerry et al., 2019), 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; CBD, 2012; Lepczyk et al., 2017). GI plays a big role in providing such habitats in urban areas (Ferenc et al., 2014; Filazzola, Shrestha, MacIvor, 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 green infrastructures (henceforth 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 & Scuderi, 2019; Gómez-Villarino et al., 2020; Madureira & 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 Directorate-General for Environment, 2021; Lai et al., 2018). In the urban context, GI provides all-around benefits for the city environment, infrastructure, and resident communities (Sturiale & Scuderi, 2019; GómezVillarino et al., 2020; Madureira & Andresen, 2013). The existence of high-quality GI, such as city parks, provides services for climate crisis adaptation firstly, by storm prevision, excess water storage, which mitigates the effects of floods (Madureira & Andresen, 2013). Secondly, 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 & Feng, 2019; Taylor & Hochuli, 2014; Annerstedt van den Bosch et al., 2015; Hegetschweiler et al., 2017). Different types of GI, however, and its spatial arrangement provide different sets of benefits, therefore the relevance of 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 of 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 is 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, 2005b; 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 online digital biodiversity databases. 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

The three case study cities Amsterdam, Rotterdam and Groningen were selected that had 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<sup>2</sup> 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 latter development plan, the “Ecological vision” report focusing on ecology, biodiversity and green connectivity was developed (Gemeente Amsterdam, 2012). The latter 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<sup>2</sup> and a population count of 651,157 (population density 2,963/km<sup>2</sup> in the whole municipality), this city is known for its one of the largest ports in Europe. In 2018, the municipality of Rotterdam as a part of the college targets 2018–2022 released an environmental program and since then more than 20ha 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* - With a core urban area of 147.3 Km<sup>2</sup> and population of 233,218 (population density 1,246/km<sup>2</sup>). In 2018 as part of the Greenplan 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 & Strootman Landshapsarchitecten, 2020). The latter includes implementing greening initiatives such as extending the tree cover (planting 1000 trees a year) and implementing 30 000m<sup>2</sup> of new green areas by creating and repurposing currently unused (gray infrastructure) space (Gemeente Groningen & Strootman Landshapsarchitecten, 2020). Moreover, in collaboration with residents municipality launched a “Groningen climate-proof” initiative by encouraging (and subsidizing) citizens, 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, but also more diverse plant species in order to increase the biodiversity and resilience of GI in the city (Gemeente Groningen & 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.
- Class 4 - water - was dedicated to water bodies such as wetlands, ponds, lakes, rivers, canals, and seas.

Then the land use classes were rasterized, 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<sup>2</sup> per hexagon were created in order to estimate the spatial distribution of green areas in relation to species occurrences and richness (Fig. 1). The rasters 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, and to reduce sampling bias (Hughes et al., 2020), we group occurrence data from species in three large groups: arthropods, birds, and mammals. The downloaded data covered species occurrence records from 2017 - to 2021 from Amsterdam, Groningen and Rotterdam. The data was explored to remove duplicates and missing coordinates, and using the three grid sizes, we calculated species occurrences and species richness per group. To further account for the sampling bias, by reducing 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., Class1 + 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<sub>wv</sub>) across all fitted models for each variable  $v$  (Burnham & Anderson, 2002). An AIC<sub>wv</sub> 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<sub>wv</sub> across all models in which the variable occurred. Higher AIC<sub>wv</sub> 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 bias in parameter estimation that may occur selecting a single best model (Burnham & Anderson, 2002). The set of models with a gamma value

higher than four was selected for each analysis to approximate the true model (Burnham & Anderson, 2002). All the analyses were performed in R software (R Core Team, 2020) using packages MuMIn (Bartoń, 2020), arm (Gelman & 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%), while Rotterdam and Amsterdam had a higher percentage of land covered with water bodies (Amsterdam = 47.9%, Rotterdam = 33.3% and Groningen = 6.35%). In terms of GI patch metrics the cities have similar mean GI patch sizes, between 0.9 (Groningen and Rotterdam) and 0.11 Km<sup>2</sup> (Amsterdam). Groningen has the smallest number of GI patches (902) which might be related to its smaller area; yet Rotterdam (4150) has 1000 more patches than Amsterdam (2939) despite its smaller size. The shape of the patches is another relevant metric, as areas with less edge have more core area - 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 edges 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<sup>2</sup> for all the cities (Fig. 2).

A total of 2'258,509 species occurrences were included in the analyses of biodiversity (Amsterdam = 826,818, Rotterdam = 1'356,926 and Groningen = 74,765 occurrences). With Rotterdam having the higher density of occurrences per square kilometer (Amsterdam = 2566, Rotterdam = 3143.6, Groningen = 893.2 occurrences/Km<sup>2</sup>). In terms of taxonomic groups, the birds accounted for more than 50% of the total occurrences and comprising the majority of observations for all the cities surveyed (68.4% Birds, 28.9% Arthropods, 28.9% 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<sup>2</sup> with more than 18000 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 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 that included agriculture/pasture cover, distance to the core best explained species richness at all scales. Models explaining bird species richness, specifically, that include green cover were more

significant explaining species richness but the effect was not as evident at smaller scales (0.2 Km<sup>2</sup>). Mammal species richness was positively affected by urban cover (Fig. 4), while distance to the city core is only significantly negative at the largest scale (1.0 Km<sup>2</sup>).

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. Arthropods species richness was negatively correlated with urban cover and agriculture/pasture cover, while it was positively related to the distance to the core, and green cover had an effect that varied from negative (0.2 Km<sup>2</sup>) to positive (1.0 Km<sup>2</sup>) depending on the scale. Bird species richness in Amsterdam was best explained at large scale, despite larger within variable variation, with all variables having a positive effect except for distance to the core that was negatively correlated. Models at medium scale (0.5 Km<sup>2</sup>), and including all variables, best explained mammal species richness with water, urban and green cover having a positive effect, and distance to the core a negative effect (Fig. 4).

Models explaining species richness in Rotterdam at large scale (1.0 Km<sup>2</sup>), best explained bird and arthropods species richness but the effect was not significant. Arthropods species richness was only significantly negatively affected at small scale (0.2 Km<sup>2</sup>) and only in relation to agriculture/pasture cover. Models including green cover (positive effect) and distance to the core (negative effect) best explained bird species richness. In the case of arthropods species richness, similar to the case of Amsterdam and Groningen, variation across scales was higher, with distance to the core having a significant negative effect at large scales, while agriculture/pasture and urban cover had a positive 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 wellbeing 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 -specially 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, Pautasso, and Figueiredo 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 this approach was possible - and to what extent- in cities, and found that changing scale by 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, in our study at 1 Km<sup>2</sup> grid size spatial heterogeneity was already lost.

Species distribute in relation to how they use the resources and are used as a resource, and this is determined by their size and patterns of aggregation, as well as their mobility (Condit 2000; Braaker et al. 2014; Beauguard, Brischoux, and Angelier 2021). Consequently, the scale at which the effect of city heterogeneity can be assessed varies from one species to another. We found that in the three cities studies, 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 studies 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; negative effect of urban areas on bird diversity at 50 km<sup>2</sup> (Ferenc et al. 2014). Studies performed at 1 km<sup>2</sup> -a scale explored in our study- found negative effect of urban cover on mammal (Gallo et al. 2017; Hursh et al. 2023) and arthropod richness (Fenoglio, Rossetti, and Videla 2020), in line with our findings in the case of arthropods and contrary to our findings in the case of mammals and for two of our cities (Groningen and Amsterdam), and a positive effect on species richness when looking at specifically 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<sup>2</sup>) 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). Previous studies have highlighted the importance of the type and characteristics of green spaces as an important factor for biodiversity (Das, Das, and Pereira 2023; Grade, Warren, and Lerman 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 type of organisms surveyed -as well as its ecology according to 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. In addition, other studies have shown the importance of green infrastructure type and connectivity to biodiversity (Grade, Warren, and Lerman 2022; Gallo et al. 2017; Lepczyk et al. 2017). In this context, the challenge remains as to how to develop integrative biodiversity assessments that accurately inform the design and management of biodiversity inclusive cities (Aronson et al. 2017; Garrard et al. 2018; Ronchi and Salata 2022; Apfelbeck et al. 2020). We propose a multiscale approach that includes: i) the tracking of heterogeneity, ii) biodiversity data coverage, and iii) species ecology across spatial scales.

## Declarations

Ethics approval

not applicable

Competing interests

No competing interests.

Authors' contributions

C.X. Garzon and G. Savickyte equally contributed to all parts of the manuscript.

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not applicable

Availability of data and material

The data that support the findings of this study will be openly available in the Dryad Digital Repository.

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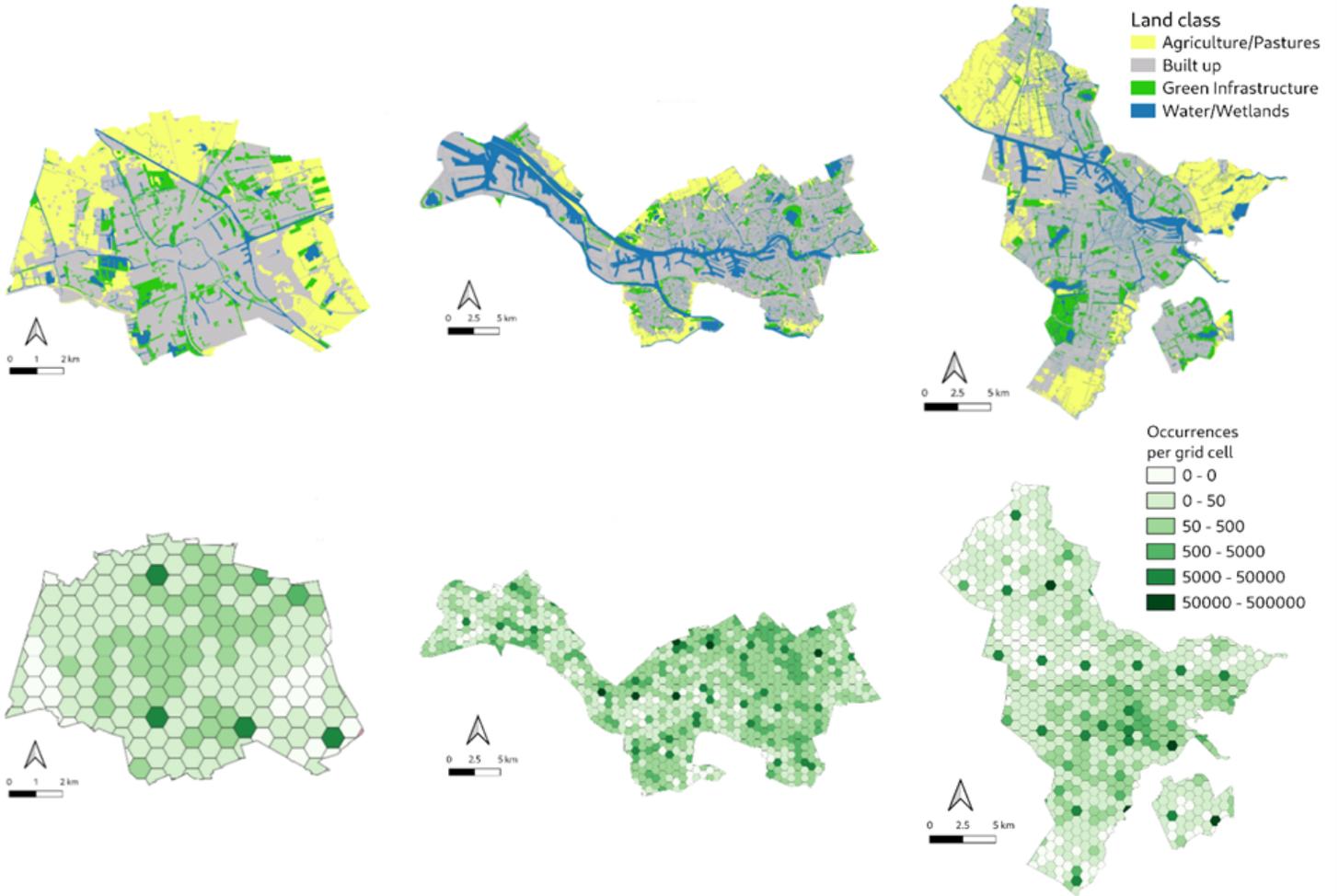
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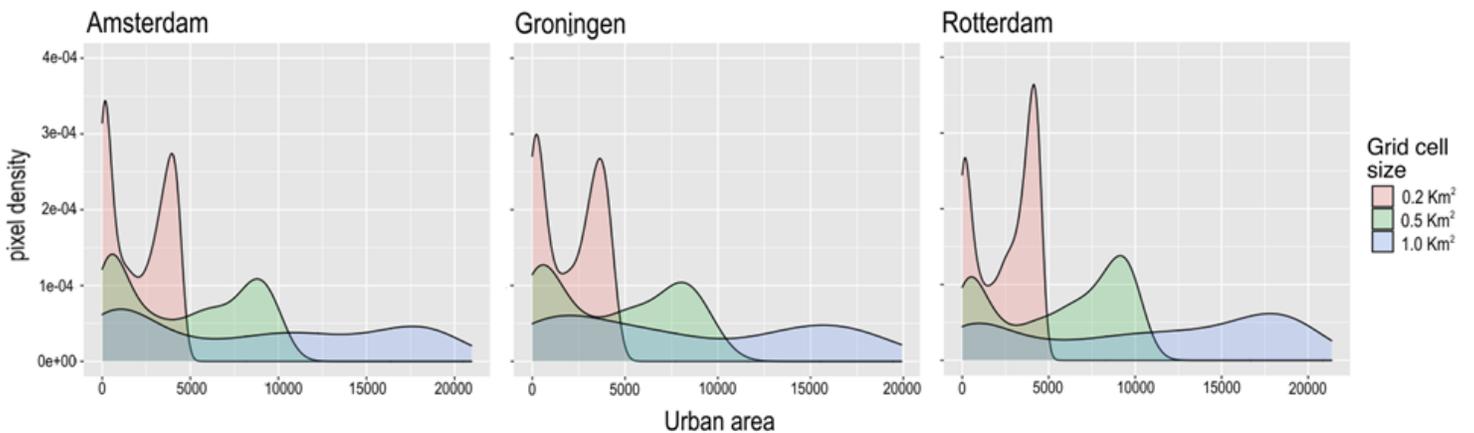
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## Figures



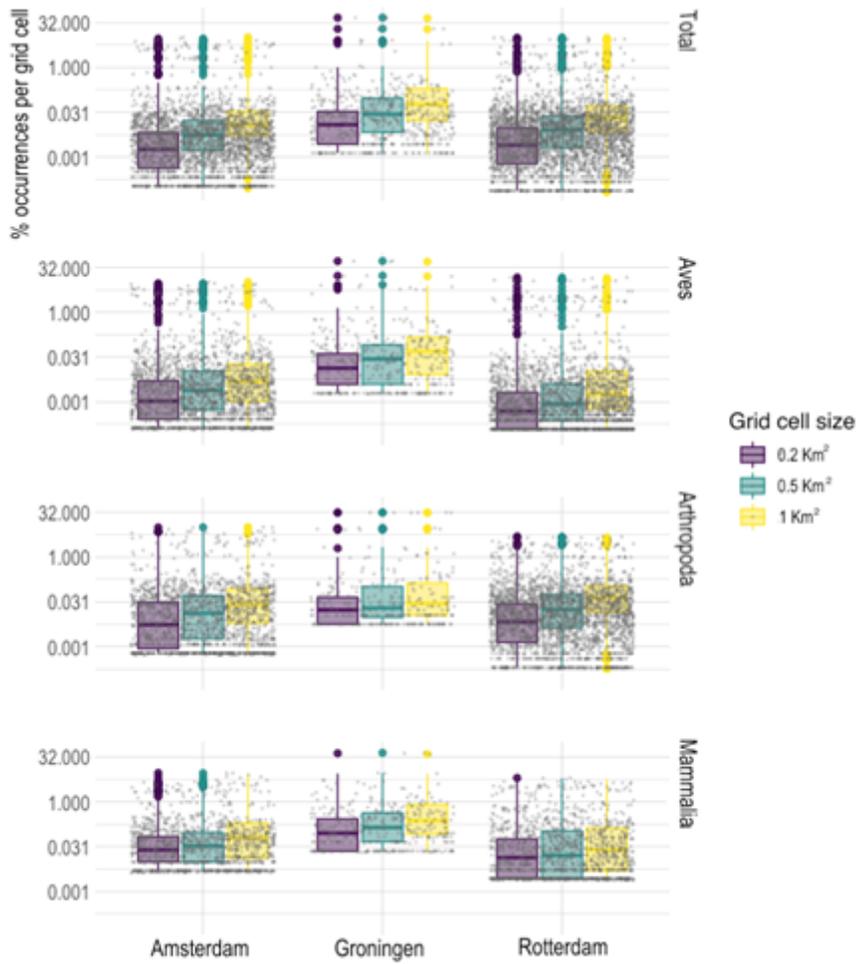
**Figure 1**

Maps, from left to right, of Groningen, Rotterdam and Amsterdam with the resulting land classification (above) and an example of the grid cell approach (0.5 Km<sup>2</sup> grid cell size) with total GBIF occurrences from 2017 to 2021 for the three taxonomic groups studied (Aves, Arthropoda and Mammalia). The scale bar reflects the difference in size across cities.



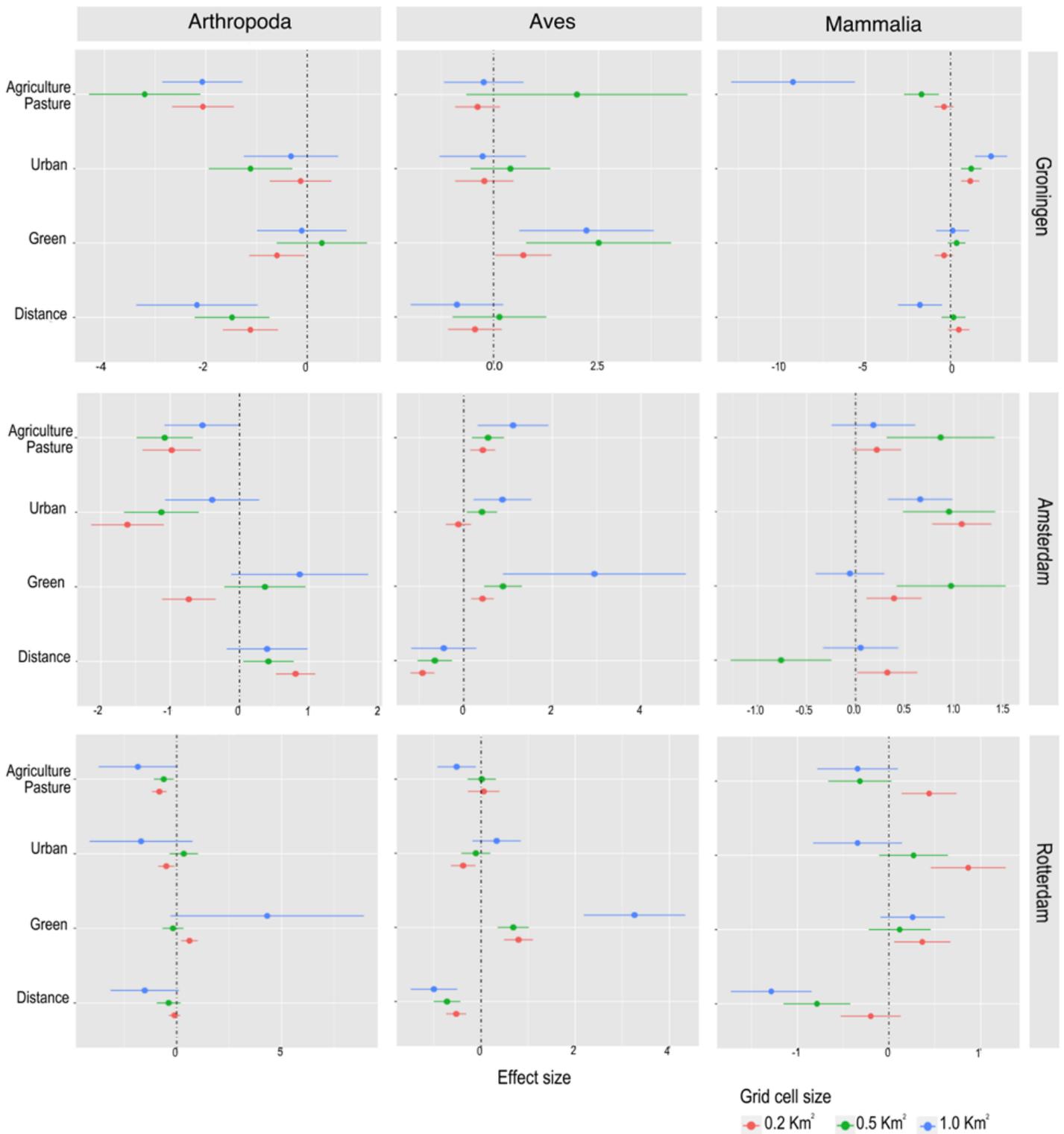
**Figure 2**

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



**Figure 3**

Distribution of occurrences (in logarithmic scale) per grid cell size and across cities.



**Figure 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.