Landscape and Urban Planning 228 (2022) 104570

Contents lists available at ScienceDirect

Landscape and Urban Planning

journal homepage: www.elsevier.com/locate/landurbplan

Research Paper

Wildness and habitat quality drive spatial patterns of urban biodiversity

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HIGHLIGHTS

- We tested wildness and habitat quality suitability as urban biodiversity indicators.
- Both indicators predicted biodiversity, particularly in urban green and blue spaces.
- Wildness and habitat quality correlated with biodiversity.
- Peripheral urban green and blue spaces showed higher biodiversity than centric ones.
- Rewilding initiatives on large centric areas could enhance urban biodiversity.

ABSTRACT

Urban green and blue spaces (UGBS) are key for biodiversity conservation. Many studies focus on UGBS benefits for well-being, but how UGBS ecological and quality influence urban biodiversity is still poorly understood. We analysed the predictive accuracy of urban wildness (UW) and habitat quality (HQ) spatial patterns to biodiversity in the city of Vitoria-Gasteiz, Basque Country. Using GIS techniques, we mapped relative UW as a landscape quality, considering remoteness, challenging terrain, and perceived naturalness. We further evaluated HQ using the InVEST habitat quality module, including data on habitat sensitivity to threats (e.g. population density, light and noise pollution, accessibility) and suitability for biodiversity support, based on a parametrization by expert consultation. We compared UW and HQ to observed species richness obtained from crowd-sourced databases as a biodiversity proxy. UW and HQ models predicted general biodiversity urban patterns, being particularly adequate in UGBS. Peripheral UGBS were associated with higher UW and HQ and positively correlated to biodiversity, as opposed to the smaller-sized centrally located UGBS, more exposed to threats. Both predictors significantly explained biodiversity, and HQ better accounted for threat susceptibility in UGBS. Our findings suggest that small-sized UGBS, such as parks and squares, fail to effectively support urban biodiversity, due to their high exposure and vulnerability to threats, particularly in centric areas. Emphasizing efforts in larger centric UGBS with rewilding strategies (e.g. lowering management frequency) and reducing exposure to threats is essential to increase the habitat quality of UGBS and thus support urban biodiversity.

1. Introduction

Given the continuous urban development around the world with anthropogenic activities increasingly shaping ecological processes and patterns, there is a growing need to enhance the multiple functions and benefits supplied by urban green and blue spaces (UGBS) (Mansur et al., 2022; Reyes-Riveros et al., 2021). UGBS supplies multiple functions and contributions to human well-being, including biodiversity support, urban cooling, runoff control, air and water quality improvement, opportunities for recreation and building social cohesion (Baró et al., 2019; Díaz et al., 2018; Gómez-Baggethun & Barton, 2013). There is growing evidence that exposure and access to UGBS areas encompass multiple health benefits: mental health, lowering disease and mortality risk, among others (Ribeiro et al., 2021; Engemann et al., 2019; Twohig-

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https://doi.org/10.1016/j.landurbplan.2022.104570
Received 14 April 2022; Received in revised form 26 August 2022; Accepted 3 September 2022
Available online 14 September 2022
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Bennett & Jones, 2018; Triguero-Mas et al., 2017). By providing people with opportunities to connect with nature, UGBS are key to decreasing the 'extinction of experience' *sensu* Soga and Gaston (2016) and the alienation from nature, which are major drivers of biodiversity loss and environmental injustice (Noss, 2020; Lepczyk et al., 2017; Soga and Gaston, 2016; Shanahan et al., 2015; Dunn et al., 2006).

UGBS have a key role in supporting biodiversity, by harbouring native and non-native species and thus promoting conservation both at regional and global scales (Lepczyk et al., 2017). Yet, biodiversity patterns in cities rely on the size, quantity and quality characteristics of UGBS, ultimately influencing the provision of related health and well-being benefits (Houlden et al., 2021; Marselle et al., 2019; Lepczyk et al., 2017; Fuller et al., 2007). Assessing urban biodiversity is, therefore, key to better understanding the role of UGBS in providing functions and contributions to well-being in urban socio-ecosystems. Even if biodiversity and UGBS are highly intertwined, the spatial and ecological characteristics of UGBS influencing urban biodiversity remain poorly understood (Marselle et al., 2021; Schwarz et al., 2017).

Urban planners usually perceive UGBS as multifunctional structures and it is often assumed that more UGBS will support more ecosystem services and host higher biodiversity (Schwarz et al., 2017). But although urban areas have the potential to provide habitat and sustain wildlife (Russo & Holzer, 2021) as human-altered habitats, their ecological functioning is in continuous interaction with anthropogenic impacts. Moreover, the way societies conceive different ecological processes as beneficial or detrimental to human well-being depends on social-ecological contexts which ultimately determines how societies might value the conservation of urban biodiversity (Marselle et al., 2021; Dunn et al., 2006). Preserving and enhancing urban biodiversity and raising public awareness of its societal values requires an understanding of the drivers and effects of changes to urban biodiversity (Knapp et al., 2021; Sallustio et al., 2017; Terrado et al., 2016). Thus, there is a clear need to analyse UGBS not only in terms of their quantity but also in UGBS quality attributes related to the conservation value of the naturalness present and their capacity to harbour urban biodiversity (Reyes-Riveros et al., 2021; Lepczyk et al., 2017).

UGBS may include areas with different management regimes, ranging from heavily managed formal UGBS such as urban parks to barely managed semi-natural and ruderal areas (Aguilera et al., 2019). There is a growing interest in understanding how to reduce intensive management in UGBS and restore relatively 'wild', self-managing ecosystems to avoid and reverse urban biodiversity loss (Aguilera et al., 2019; Müller et al., 2018). Wildness is the relative quality of being wild or undomesticated, encompassing a broad spectrum of landscape contexts, anthropogenic influence and scales: from remnant patches of urban vegetation to vast pristine areas, disturbance regimes and biogeochemical processes (Noss, 2020). Wildness, as a landscape character can be spatially represented. Yet, it cannot be directly separated from non-wildness, but rather expressed from less to more 'wild' along a continuum of anthropogenic impacts (Zoderer et al., 2020; Carver & Fritz, 2016). There have been many recent wildness mapping studies (Müller et al., 2015) almost exclusively referred to non-urban landscapes, often minimally affected by human action (Zoderer et al., 2020; Radford et al., 2019; Carver et al., 2012, 2013). Even though only one study has applied wildness mapping in cities (Müller et al., 2018), wildness can be a cost-effective indicator for management to be used in urban contexts: to enhance biodiversity, screening for priority conservation areas, while improving human well-being in cities (Mansur et al., 2022; Jalkanen et al., 2020; Noss, 2020; Kowarik, 2018; Müller et al., 2018).

In addition, habitat quality (hereafter HQ) indicates the capacity of an ecosystem to deliver the resources and conditions needed for wildlife and is a key determinant of biodiversity (Terrado et al., 2016; Hall et al., 1997). As a way to assess HQ, ecological spatial models are gaining research interest to evaluate habitat attributes, such as key resources and the constraints impairing the use of resources (i.e. anthropogenic influence). One of these models is the Habitat Quality module from the Integrated Valuation of Environmental Services and Trade-offs (InVEST-HQ) which has been used to analyse the degradation status of non-urban landscapes (Di Febbraro et al., 2018; Terrado et al., 2016), to identify specific conservation targets (Bhagabati et al., 2014), and to study natural or protected areas (Moreira et al., 2018; Sallustio et al., 2017). As biodiversity patterns are inherently spatial (Sharp et al., 2020) the use of spatially explicit indicators, such as urban wildness or HQ can play an important role in predicting the suitability of a landscape to host biodiversity and ecological functions. The use of these models is cost-effective, can be done remotely and can be linked to characteristics of the ecosystem's structure and functioning, such as naturalness (Müller et al., 2018; Di Febbraro et al., 2018). Although the application of InVEST HQ model has proved to be useful to assess biodiversity conservation status at large scales, it often oversimplifies the variability of habitat types at smaller scales as is the case of urban regions (Sallustio et al., 2017). To our knowledge, InVEST HQ has not yet been applied to estimate HQ for urban land uses, which can be often considered in the model as threats to biodiversity instead of potential habitat sources (Wu et al., 2019; Han et al., 2019; Sallustio et al., 2017). Although wildness and HQ has been extensively assessed in large scales and often low anthropized areas, there are limitations on their applications at a finer resolution in smaller geographical areas (Cao et al., 2019; Sallustio et al., 2017). Areas with high land cover heterogeneity such as urban areas are usually oversimplified in landscape assessments of wildness and HQ, by homogenizing land uses and habitats types at coarser scales (Cao et al., 2019; Sallustio et al., 2017). This may lead to the underrepresentation of locally important areas for biodiversity conservation and affect management opportunities to enhance naturalness and wildness. Cities are increasingly important areas worldwide and UGBS are key on the relationship societies – nature, but have traditionally been seen as opposite to wilderness. Therefore, new approaches are necessary to better address the opportunities and challenges associated with rewilding UGBS in cities. Identifying factors influencing the biodiversity and their relationship with wildness and habitat quality at smaller scales at the urban end of the wilderness *continuum* is a research avenue that calls for further exploration (Carver & Fritz, 2016; Dymond et al., 2003; Kowarik, 2011).

Research on urban ecology is usually mostly focused on large metropolitan regions and capital cities, despite the fast growth and dominance of small and medium-sized cities in many regions such as Europe (Borsekova et al., 2018; Boulton et al., 2018). Hence, the assessment of how UGBS and its capacity to harbour biodiversity in mid-sized cities is still understudied (Boulton et al., 2018). Here, we analyse the suitability of the spatially-explicit HQ and wildness models, as predictors of biodiversity when adapted to a mid-size urban area. Particularly, we: (i) spatially assessed relative urban wildness as a landscape quality following Müller et al. (2018); (ii) adapted the InVEST habitat quality module for the assessment of habitat quality on the urban landscape, considering the effects of anthropogenic impacts on the urban landscape and particularly across UGBS; (iii) identified potential UGBS that may have a key role in terms of biodiversity maintenance by using the outputs from (i) and (ii); and (iv) analysed the correlation between wildness, HQ, and species richness as biodiversity proxies.

2. Materials and methods

2.1. Study area

We conducted our study in Vitoria Gasteiz, a middle-sized European city, and the administrative capital of the Basque Country (248.087 inhabitants, Eustat, 2020), located in the North of the Iberian Peninsula (Fig. 1). Vitoria-Gasteiz is situated at the centre of a transitional biogeographic region (Mediterranean and Eurosiberian) (De la Hera, 2019) with an extensive plain delimited by mountains connecting the Cantabria (West) and the Pyrenean (East) mountain ranges, two

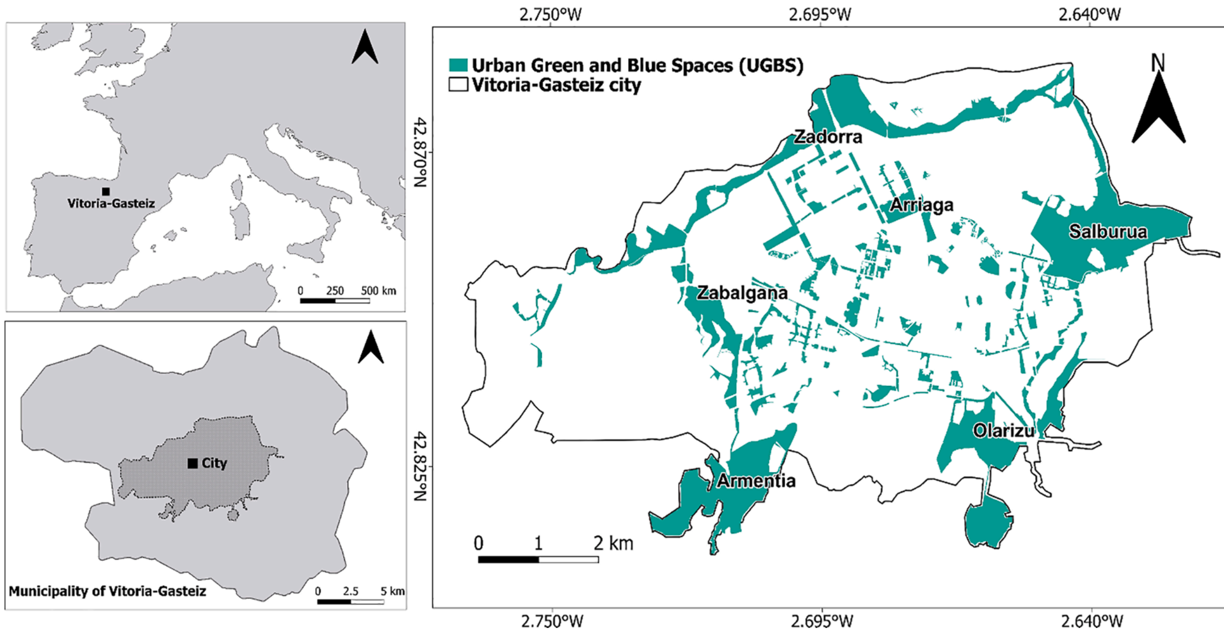

Fig. 1. Location of Vitoria-Gasteiz municipality, city and main urban green and blue spaces within city boundaries defined by urban land uses and delimited by the green belt. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

biodiversity hotspots maintaining the ecological connectivity of northern Iberia, and part of the Pan-European Ecological Network (CEA, 2014). Vitoria-Gasteiz has a remarkable share of urban and periurban green and blue spaces (Fig. 1), including a 35 km-long green belt surrounding the core city (Orive & Lema, 2012). The green belt was created during the 1990s as a way to transform peripheral degraded areas into UGBS along with enhancing biodiversity by restoring habitats and increasing ecological connectivity (Newman & Cabanek, 2020). It consists of a system of *peri*-urban semi-natural green spaces, promenades, wetlands, streams, and ponds in the interface between the city and the countryside (Monclús, 2018). Because of their natural value, Salburua wetlands, Vitoria Mounts and the River Zadorra have been protected as community interest sites and included as part of the European Natura 2000 network (CEA, 2014, Fig. 1). With 20 m^2 of UGBS per inhabitant, Vitoria-Gasteiz is considered a "green" city (CEA, 2014). Because of its pioneering and ambitious greening strategies, Vitoria-Gasteiz gained international recognition and was awarded in 2012 as a European Green Capital (CEA, 2014). In addition, Vitoria-Gasteiz offers a great test-bed for research given the high availability environmental data, including the spatial data required in our modelling approach.

2.2. Urban wildness spatial model

We mapped urban wildness (UW) at the city level along a *continuum* of historically human-modified landscapes, based on three parameters building on Carver et al. (2013), Müller et al. (2015), Müller et al. (2018), and Radford et al. (2019): i) Perceived naturalness, ii) Remoteness and iii) Challenging terrain.

Perceived naturalness was defined as the vegetation and land cover pattern created by land (un)management, appearing *natural* to the casual observer (Carver et al., 2013). To estimate perceived naturalness for the whole study area, we joined the following spatial datasets: Urban Land Use Map 2020, Green Belt Land Use Map 2020, and Nature-Based Solutions Inventory (2020) (Table S1, Supplementary material) on a composite land use dataset (hereafter 'land use map') at a resolution of 10 m. Land uses were reclassified into 18 naturalness classes following Müller et al. (2018) (Table S2, Supplementary material). Naturalness values ranged from 1, representing 'completely sealed areas', to 18 as 'land cover under the least human influence'. All datasets were provided by the Environmental Studies Centre (CEA), a public agency of the Vitoria-Gasteiz City Council. Brownfields and vacant land were classified based on the perceived naturalness as "Recreative areas" or "Relatively extensive open landscapes" according to the reconversion plan or use assigned for each area in the inventory of NBS provided by the City Council.

Remoteness was mapped by combining distance from mechanised access (shortest walking distances from main roads) and noise exposure to any pixel on the map as in Carver et al. (2013) and Müller et al. (2018). We reclassified the previous composite map based on the land use data into a cost surface building on Müller et al. 2018 (Table S3, Supplementary material). We estimated the time it takes a pedestrian to move across each pixel based on the assumed travel times for each land use type. The minimum time needed by a pedestrian to access any pixel in the study area from a vehicle access point was then calculated by the Path distance tool of ArcGIS. The weighted means of noise values (Lden) retrieved for the study area (CEA, 2017) were used for the calculation of noise exposure. The pixels without noise data were assigned 30 dB, comparable to a quiet garden (Müller et al., 2018). We reclassified these into inverse values, so high values show low decibel values making sure that the indicator correlated positively to UW. Next, we combined both datasets, remoteness from mechanised access and noise exposure to create the final remoteness map.

Challenging terrain was depicted by combining terrain ruggedness index and occurrence of wetlands, again building on previous approaches (Müller et al., 2018; Carver et al., 2013). The ruggedness index was considered as the standard deviation (SD) of terrain curvature within a 250 m radius of the observer since it has been generally assumed that people perceive that area of their surroundings (Müller et al., 2018; Scottish Natural Heritage, 2014; Carver et al., 2012). We calculated the ruggedness index from a high-resolution (10 m cell size) digital elevation model (DEM) data available for the study site (CEA, 2017), with a higher score indicating steep and rough terrain. At this resolution, the DEM would also capture anthropogenic infrastructure (e.g. buildings), so when calculating the SD of the terrain curvature such buildings would tend to return high values (Müller et al., 2018). To avoid the misleading effect of anthropogenic infrastructure on the index, we first excluded all pixels containing built infrastructure (e.g. roads, railways, buildings), from the dataset, and then the SD was calculated for each pixel using a 250 m neighbourhood buffer. As a previously identified limitation, the method captures ruggedness rather than the challenging character of the terrain more generally, so flatter areas such as wetlands and bogs tend to be underestimated (Müller et al., 2018; Scottish Natural Heritage, 2014). Thus, to cover all challenging terrain in terms of topographic attributes, we considered the occurrence of wetlands from the land use map to the terrain ruggedness index. Therefore, if a pixel cell led within a wetland, we added the average SD of terrain ruggedness to the pixel cell value (Müller et al., 2018).

We obtained the final relative UW map by Vitoria-Gasteiz by combining these three indicator maps (perceived naturalness, remoteness, and challenging terrain) using equal-weighted simple addition. We then rescaled each indicator map and the resulting map to a 0–1 range using Eq. (1) following Carver et al. (2013) and Müller et al. (2018):

$$Sij = \frac{(Xij - OmV)^*(NMV - NmV)}{(OMV - OmV + NMV)} \quad (1)$$

where *Sij* refers to the standardized value of cell *j* in map *i*, *Xij* to the current value of cell *j* in the map *I*, *OmV* to the old minimum value, *OMV* to the old maximum value, *NmV* to the new minimum value, and *NMV* to the new maximum value.

We performed a Jack-knife analysis to assess the sensitivity - robustness and weigh the uncertainty of the final UW model by checking the effect that leaving one input variable out of the model would have on the results (Quenouille, 1956). We calculated three alternative maps by excluding one input map, respectively. Then we calculated the correlation coefficients of the three resulting maps when compared to the original (full) UW map by using the r.regression.line module in GRASS 7.8.5 (QGIS.org, 2021). All the spatial analysis was performed in QGIS 3.18.2 (QGIS.org, 2021) and ArcGIS 10.7.1 (ESRI, 2019).

2.3. Habitat quality spatial model

We calculated the indicator for habitat quality (HQ) using the spatially-explicit InVEST Habitat Quality Model (v.3.8.9; Stanford University, Stanford, CA, USA) which estimates the relative degradation extent and status of different habitat types in a given region. This model is based on the relative sensitivity to different threats, distance to these potential threats, and location of protected areas. The HQ approach assumed that habitats with higher quality will support higher biodiversity and is a general estimator of biodiversity suitability (Sharp et al., 2020). Therefore, we estimated HQ as a function of i) the suitability (*Hj*) of each land use to provide habitat for biodiversity (as local species richness), ii) anthropogenic threats which might impair HQ, and iii) the sensitivity of each land use type to each threat. HQ models can be used for a particular species or general patterns of biodiversity associated to habitat quality, regardless any particular taxonomic or functional group, which makes it a suitable rapid tool for general biodiversity assessment studies (Sharp et al., 2020). Since not all habitats are affected by different threats, in the same way, we characterised the sensitivity of habitat types to various threats. We considered habitat suitability to be affected by: i) the relative impact of each threat (W*r*, weighted relative importance of each threat), ii) the distance between habitat and the source of the threat (including the maximum threshold distance, *Max. D*), and iii) the relative sensitivity of each habitat type to the threat (*Sjr*). Further details on the InVEST HQ model and parameterisation can be found in Sharp et al. (2020), Sallustio et al. (2017), and Terrado et al. (2016).

Since our work was based on an urban context, the HQ models were adapted from the InVEST approach by considering the spatial distribution of habitats including a broad range of managed land use classes as potential habitat providers to urban biodiversity rather than merely threats (Sharp et al., 2020). To define the potential habitats, we used the categorization of the 18 naturalness classes described in section 2.2. (Table S2, supplementary data). Instead of a binary (i.e. 'natural' vs 'unnatural') approach (Müller et al., 2018), we considered the relative habitat suitability score for each land use class organized along a naturalness *continuum*. To account for threats, we used spatial information of railways and traffic intensity sorted by main, secondary, and residential roads (Vitoria-Gasteiz City Council, 2020). We also included raster information on noise pollution decomposed by dB intensity (Vitoria-Gasteiz City council, 2018) and population density data (Vitoria-Gasteiz city council, 2016) as additional potential threats.

A relative habitat suitability score *Hj* (scaled from 0 to 1), was assigned to each land use class, where 1 indicates land use classes with the highest suitability for biodiversity (Sharp et al., 2020). *Hj* and all threat parameters were determined through expert consultation (following Kuhnert et al., 2010). This information was gathered from a purposedly designed structured survey administered to 21 international experts, on urban ecology, biodiversity conservation, and urban environmental management who suggested values to parametrize our urban landscape model (see Questionnaire S1, in Supplementary material) (Tables S5 and S6).

Since the list of 18 naturalness classes was too extensive for an expert survey, we grouped these classes into 8 categories of urban habitats based on urban green infrastructure typology according to Hansen et al. (2017) (Table S4, Supplementary material). During the administering of the survey, we did not share any results or feedback among the group of experts as in Terrado et al. (2016), so our method relied on experts having a good understanding of the questions. To estimate the model uncertainty and to determine parametrization scores from each descriptor variable, we first calculated the mean (μ), standard deviation (SD), and coefficient of variation (CV). We then identified and excluded from the analysis extreme values in the valuation of any descriptors from the expert survey by z-score deviation. The HQ score uncertainty was calculated with the Zonal Statistic tool (QGIS) as the average HQ and their coefficient of variation for each habitat typology class (Di Febbraro et al., 2018). The final values used as input parameters to build the HQM are reported in Tables S5 and S6 from the supplementary material.

The total threat level value in each raster cell *x* of habitat typology *j* was given by D_{xj} (Eq. (2)), where *y* indexes all grid cells on *r*'s map and Y_r corresponds to the set of raster cells of *r*'s map. If S_{jr} (relative sensitivity of each habitat type to the threat) equals 0, then D_{xj} is not a function of threat *r*. Threats were normalized so the sum of all considered threats equals 1 (Sharp et al., 2020):

$$D_{xj} = \sum_{r=1}^{R} \sum_{y=1}^{Yr} \left(\frac{Wr}{\sum_{r=1}^{R} Wr}\right) r_y r_{xi} \beta_x S_{jr} \quad (2)$$

The raster cell's degradation score was then translated to habitat quality (Q_{xj}) scores along with H_j using scaling parameters (*z* and *k*) in Eq. (3) (Sharp et al., 2020). Q_{xj} is equal to 0 if H_j is equal to 0. Q_{xj} can never be greater than 1. As raster cells degradation score increases, habitat quality decreases and vice versa:

$$Q_{xj} = H_j \left(1 - \left(\frac{D_{xj}^z}{D_{xj}^z + k^z}\right)\right) \quad (3)$$

2.4. Spatial correlation with urban biodiversity data

We evaluated the spatial correlation of the UW and HQ models with observed biodiversity data considering the correlation coefficient (R), the adjusted coefficient of determination (R^2), and the regression of the residuals, with the 'r.regression.line' and 'r.regression.multi' modules in GRASS 7.8.5 (QGIS.org, 2021). Biodiversity data relates to observed species richness (as a proxy of alpha biodiversity, i.e. mean species diversity at a local scale (Whittaker, 1972)) from occurrence records of mammals, birds and butterflies obtained from the 'Ornitho.eus' database (www.ornitho.eus) for the period January 1994 to December 2020. Birds and butterflies are common core indicators of biodiversity according to the City Biodiversity Index (SCBD, 2012), and all groups selected are well represented across citizen science and monitoring programs in the city (Albaina et al., 2020; De la Hera, 2019). Each record included a species occurrence at a specified geographic location along with the date and number of individuals. We first cleaned the species occurrence records retrieved from the 'ornitho' dataset by removing geospatial errors, duplicated, and museum registers. To correlate observed species richness with HQ and UW values, we first joined the cleaned species occurrences to a grid cell polygon layer of 100 × 100 m covering the whole study area. We then extracted the total count of species occurrences, species richness, and the mean values of our explanatory variables (i.e. UW and HQ) matching this grid cell scale. The responses of species richness to UW and HQ were analysed separately for each taxonomic group by Gaussian generalized linear models (GLM) first, for the whole city and then spatially constrained to UGBS. We selected the best models based on their statistical significance (F and p values <0.05) and goodness of fit (adjusted R^2), checking for normality and homoscedasticity through visual inspection of the residuals. To achieve a better fit for the whole city scale model, species richness was log10-transformed. To estimate the strength of the relationship between species richness, UW and HQ, we performed non-parametric Spearman's correlation tests. To test how different or complimentary were UW and HQ at predicting urban species richness, i.e. if the model had a detectable effect on richness, we regressed that variable (e.g. UW) against the residuals of related variables (i.e. richness as a function of habitat quality) and vice versa. All the statistical analyses were conducted in RStudio 1.4.1717 (R Core Team, 2021) and GRASS 7.8.5 (QGIS.org, 2021).

3. Results

We found that UGBS with higher UW and HQ values had the highest biodiversity values for all considered taxonomic groups. These UGBS were generally represented by large-sized peripheral areas, with high perceived naturalness, remoteness, challenging terrain (for UW), and high habitat suitability (for HQ). Both UW and HQ estimators were positively correlated to each other and associated with biodiversity.

3.1. Spatial patterns of urban wildness

UW mapping showed that peripheral areas were associated with higher wildness values, and in general lower values towards the city centre (Fig. 2). Areas with higher UW corresponded to challenging terrain zones or with barriers to access, particularly wetlands, streams, foothill forests, and hills mostly located at the edges of the city and distant from the urban centre (i.e. Ramsar's Salburua wetlands in the East and mountains to the South). Several large-sized UGBS, like large parks and freshwater ecosystems (i.e. rivers and wetlands), also showed relatively high UW even when located near the city centre (Fig. 2). Build-up infrastructure from residential and industrial areas was mapped as least wild, corresponding to 'completely sealed' or 'mostly sealed areas' according to Müller et al. (2018) classification. UW decreased towards higher impervious coverage and was particularly low in large industrialized with few or absent UGBS.

The sensitivity analysis of the UW model, comparing the differences in UW between the original (full inputs) and three alternative maps after excluding one of the three input parameters (perceived naturalness, remoteness and challenging terrain), showed a stable mapping trend and high correlation (correlation when excluding: Naturalness R = 0.75; Remoteness R = 0.97; Ruggedness R = 0.97) (Appendix A).

3.2. Spatial patterns of urban habitat quality

There was a clear urban–rural gradient in relation to the HQ spatial pattern, where most urban core areas were associated with low HQ values, while peripheral and large-sized UGBS showed the highest HQ to support biodiversity (Fig. 3). These areas mostly corresponded to the Ramsar wetland Salburua (to the East, Fig. 3), foothill forests and hills (to the South and West, Fig. 3). Streams showed high HQ values but were mostly restricted to the watercourse and not in the surrounding environment (i.e. floodplain). Furthermore, these ecosystems were particularly affected towards the city centre depicting very low HQ (Fig. 3). Vitoria-Gasteiz showed a large coverage of UGBS, mostly related to green corridors or parks near the city centre (e.g. Zabalgana and Park Arriaga, Fig. 1) that were however associated with low HQ values, some of them ranging between 0.12 and 0.15 (Fig. 3).

The coefficients of variation (CV) that expressed the uncertainty for the expert-based parametrization of habitat suitability for HQ, showed that freshwater ecosystems, natural, semi-natural areas and feral areas were generally perceived as more natural and similarly valuated (CV freshwater ecosystems = 14 %, CV natural, semi-natural areas and feral areas = 11 %, Appendix S5). Whereas, the habitat suitability valuation uncertainty was much higher for more anthropized areas, and the

Fig. 2. Urban wildness map: relative wildness for Vitoria-Gasteiz city on a 10 m resolution.

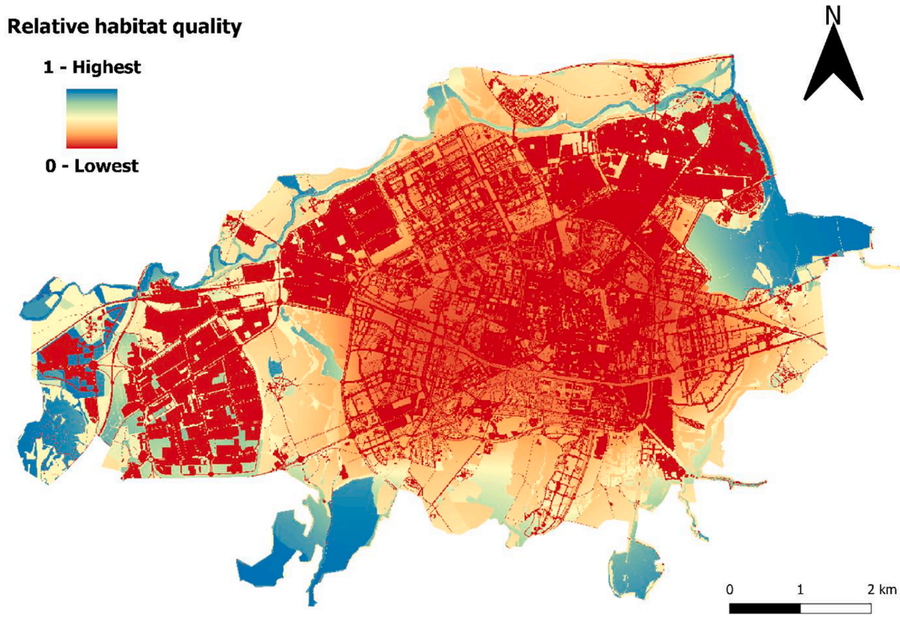

Fig. 3. Urban habitat quality (HQ) map for Vitoria-Gasteiz city: Landscape-scale relative HQ mapping on a 10 m resolution. Habitat quality is based on a continuous scale from 0 to 1, where the lowest values indicate lower habitat quality (warm colours) and the higher values indicate higher habitat quality (cold colours).

highest level of uncertainty corresponded to predominantly sealed areas (CV = 238 %) (Appendix S5). Other land uses such as agricultural land, parks or grasslands were associated with intermediate habitat suitability CV values of 30 %, 24 % and 30 % respectively (Appendix Table S5). Furthermore, the coefficient of variation concerning the threats for HQ was highly variable depending on the land use class and type of threat (Table S6).

UW depicted higher values than HQ in large-sized green spaces and peripheral areas including forests, hills, wetlands and rivers but also in centrally-located areas such as urban parks had higher UW values compared to HQ (in red Fig. 4). Particularly, freshwater ecosystems (river and wetlands) and the surrounding areas depicted higher UW compared to HQ (Fig. 4 in red). While in the case of HQ, peripheral and industrial areas showed higher values than UW (Fig. 4 in blue).

3.3. Compared wildness and habitat quality spatial patterns

Even if UW and HQ were positively correlated (Adjusted $R^2 = 65$ %) and explained a similar amount of biodiversity variation, they showed spatial differences for biodiversity estimations in particular areas, with some areas with a significantly higher valuation for UW than for HQ (in red Fig. 4) and vice versa (in blue Fig. 4).

3.4. Wildness and habitat quality as proxies for biodiversity support

The ‘ornitho’ occurrence registers for Vitoria-Gasteiz consisted of 50,168 ind of 250 species of birds, 2721 ind of 48 species of mammals and 2063 ind of 97 species of butterflies. The most frequently observed bird species were mostly generalists and songbirds like *Pica pica* (Eurasian Magpie) or *Turdus merula* (Eurasian Blackbird), and the

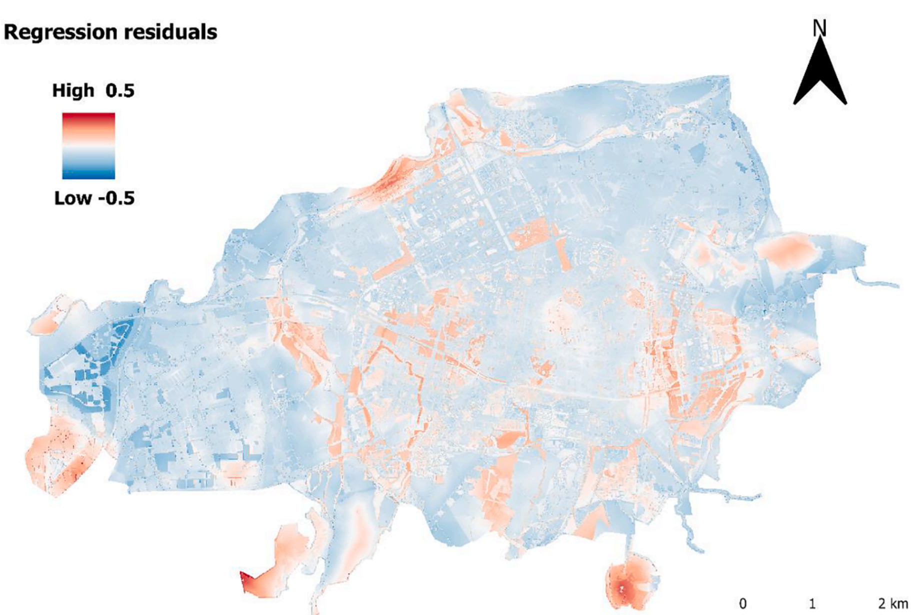

Fig. 4. Comparative mapping of urban wildness (UW) and habitat quality (HQ) explaining urban biodiversity patterns. Red areas indicate higher UW values compared to HQ, while blue areas indicate higher HQ values compared to UW. The higher the colour intensity the higher the differences between UW and HQ. Residual map based on the regression of UW as a function of habitat quality (HQ). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

worldwide dominant *Passer domesticus* (House sparrow) (Fig. 5). The most common butterflies were *Pararge aegeria* (Speckled Wood), *Pieris rapae* (Small White) and *Colias crocea* (Clouded yellow). Among the most common mammals, we found *Sus scrofa* (Wild Boar), *Oryctolagus cuniculus* (European Rabbit) and *Vulpes vulpes* (Red Fox), mostly mid-sized generalists and omnivores. However, other mammals observed in the case study area include the endangered *Arvicola sapidus* (Southern Water Vole) and the critically endangered *Mustrela lutreola* (European mink) according to the IUCN red list (IUCN, 2021).

Species richness of the different considered taxonomic groups (birds, butterflies and mammals) was positively correlated to both, UW and HQ values for the whole study area (Fig. 6 and Table S7 – Supplementary material). The Spearman's correlation for the strength of the relationship between species richness and UW, showed a strong positive correlation for birds ($p < 0.001$, $R = 0.26$), butterflies ($p < 0.01$, $R = 0.33$) and mammals ($p < 0.001$, $R = 0.34$), respectively. Similarly, the Spearman's correlation for richness and HQ, was also strongly and positively correlated for birds ($p < 0.001$, $R^2 = 0.32$), butterflies' ($p < 0.01$, $R^2 = 0.29$) and mammals ($p < 0.001$, $R^2 = 0.37$).

The goodness of fit increased substantially when focusing our analysis on UGBS instead of the whole study area (Fig. 6). Here, the relationship between species richness and both modelled predictors UW and HQ tested positive for all the considered taxonomic groups showing considerably higher correlation values and significance (Fig. 6).

The Spearman's correlation of richness against UW was positively correlated for birds ($p < 0.001$, $R^2 = 0.39$), butterflies ($p < 0.001$, $R^2 = 0.79$), and mammals ($p < 0.01$, $R^2 = 0.61$). Likewise, species richness and HQ were positively correlated for birds ($p < 0.05$, $R = 0.2$), butterflies ($p < 0.001$, $R = 0.65$) and mammals ($p < 0.01$, $R = 0.62$). Overall, the richness of all considered taxonomic groups showed a consistent positive response to UW and HQ.

Residual regressions analysis showed that UW and HQ were not influencing richness independently, but rather capturing the same relations (Table S8, Supplementary material). All the different taxonomic groups used to test the relationship between HQ, UW and richness, showed a clear trend to increase as our indicators increased.

4. Discussion

4.1 Habitat quality and wildness to biodiversity

Our results suggest that both UW and HQ models can be adequately adapted to urban environments to capture an area's capacity to support biodiversity. UW and HQ were significantly positively correlated with the biodiversity for all the considered taxonomic groups, and thus correlated to each other. The selection of these taxonomic groups was appropriate since they are common in urban environments and easily identifiable, making them suitable for citizen science programs. The most frequent species within each group, particularly in the case of generalists birds and small to medium-sized mammals, are in general highly recorded in crowdsourced databases due to their visibility and widespread presence in UGBS (Sultana & Storch, 2021). However, the biodiversity in UGBS is largely conditioned by the spatial and vegetal conformation of the UGBS (Reyes-Riveros et al., 2021; Lepczyk et al., 2017; Turner & Gardner, 2001). Some of the most frequent bird species found in the city such as magpies and blackbirds are commonly associated with weedy patches and native shrubs in central areas within the city and for which the amount of vegetation is a major determinant of density, diversity and distribution (Forman, 2014). Butterflies are usually the least species-rich in areas with intensive management and design such as the case of urban parks (Aguilera et al., 2019; Forman, 2014). The list of frequent butterfly species observed in our case study, coincided partially with previous research showing the decline of species richness in butterflies related to management intensity (Aguilera et al., 2019). In the case of large wild mammals, despite being less likely to be observed on an urban scale compared to birds and butterflies, the presence of mid-sized generalists within the city may be indicating that part of the UGBS may act as wildlife corridors, entering the city (Forman, 2014).

We found that UGBS located in the periphery of the city, generally large-sized, were associated with similar high range values for both, UW and HQ and therefore the highest potential to support urban biodiversity. These areas were mostly represented by, freshwater ecosystems, large-sized parks, and foothills, where human access and threats were lower, and they are currently protected as community interest sites in the Natura 2000 network, such as the Ramsar wetland Salburua and the Zadorra river (CEA, 2014). Parks and other residential UGBS, patchier, small-sized and centric (Fig. 1) were characterized by low values of both proxies of biodiversity support UW and HQ, being particularly low when considering HQ. We found contrasting patterns of UW and HQ values between these areas, suggesting that not only the location but the size and shape of UGBS may have an important role in influencing biodiversity (Figs. 2 and 3). These findings are consistent with previous research highlighting the positive relation between UGBS characteristics

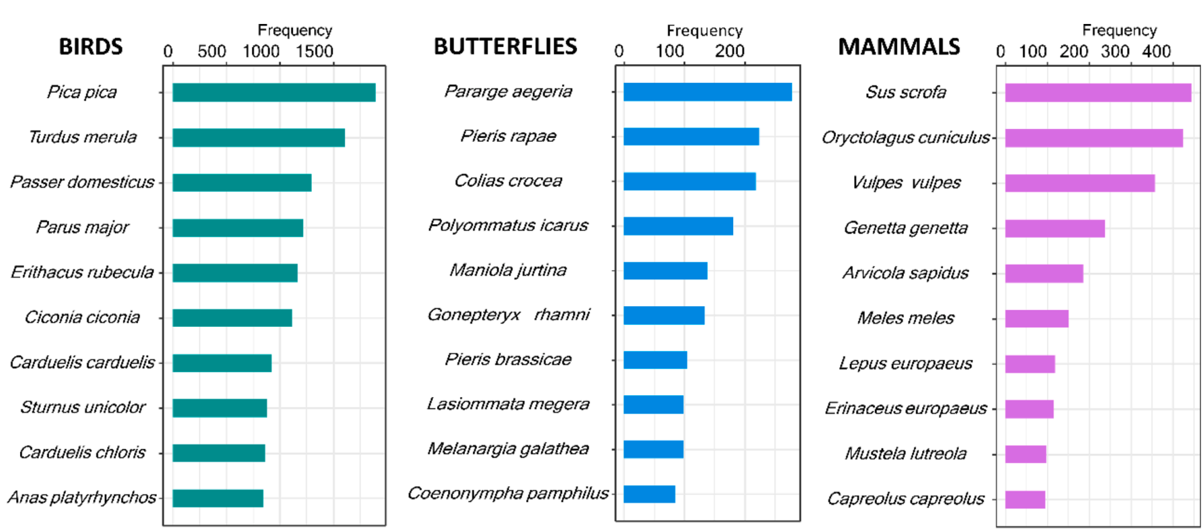

Fig. 5. Ordered list of the 10 most common species of birds, butterflies and mammals, in the 'ornitho' biodiversity dataset as occurring in Vitoria-Gasteiz (year: 1994–2020).

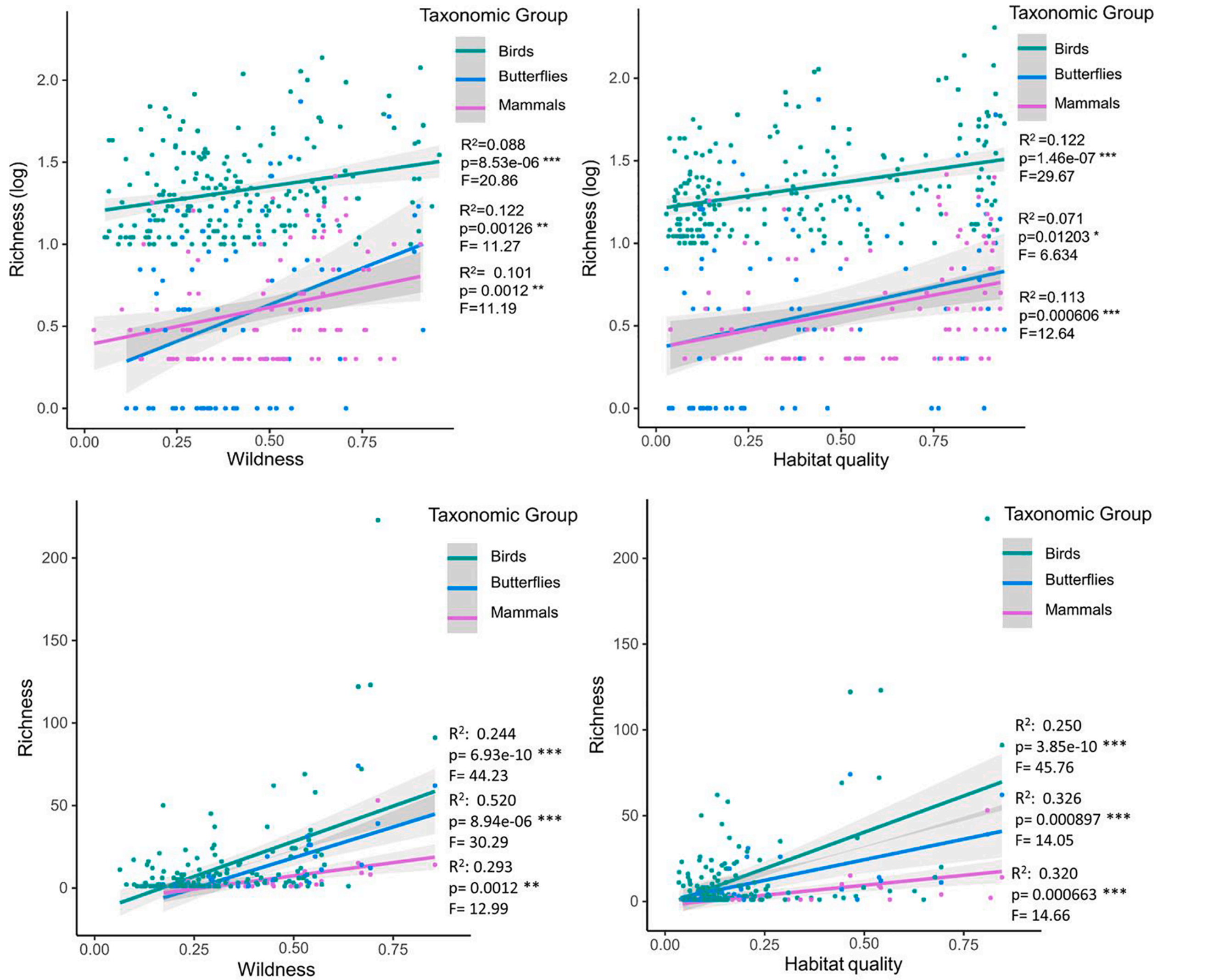

Fig. 6. Upper panel: Whole-city patterns, regression analysis of richness against UW (left) and HQ (right), and fitted linear model results for each taxonomic group. Lower panel: Regression analysis of richness against UW and HQ constrained to UGBS and fitted linear model results for each taxonomic group (R^2 and significance values of $p < 0.05$ and confidence intervals (shaded area).

on species richness, as size and habitat quality of UGBS in cities are key local factors to biodiversity (Aronson et al., 2017; Lepczyk et al., 2017; Goddard et al., 2010). Larger UGBS with a lower perimeter:area ratio have less border effect, i.e. are less exposed to threats and effects of the surrounding cover types (Turner & Gardner, 2001). This may explain why larger areas had both higher UW and HQ: the larger the UGBS were, the more distance from the central areas to the edge, i.e. the higher the distance to potential threats. Contrastingly, smaller-sized or edged shapes with a higher perimeter:area ratio, UGBS are potentially more exposed to threats and the interference cover of human activities (higher edge effect) (Turner & Gardner, 2001). Studies in large-sized cities evidenced that the perimeter:area ratio and indicators related to shape and connectedness to surrounding areas (edge effect) are better predictors of bird biodiversity than merely the size of UGBS (Garizábal-Carmona & Mancera-Rodríguez, 2021; Shih, 2018). Despite the positive effect of UGBS size on species richness, it remains poorly understood how size thresholds on individual patch size may influence biodiversity conservation in urbanized landscapes (Lepczyk et al., 2017). There are further current discussions about how land sharing (extensive sprawling urbanization) vs land sparing (intensive and compact urbanization) urbanisation initiatives along a landscape fragmentation *continuum* may influence biodiversity in urban contexts (Stott et al., 2015). Therefore, effective strategies to increase biodiversity in urban contexts should account not only for the size but for the shape and connectedness of UGBS (Turner & Gardner, 2001). As increasing the size of UGBS in consolidated cities can be challenging, due to land availability constraints (Stott et al., 2015), previous research suggests that increasing the tree coverage and vegetation stratum complexity is a good alternative to improve UGBS functionality (Garizábal-Carmona & Mancera-Rodríguez, 2021; Müller et al., 2018; Sandström et al., 2006). However, such measures should be carefully considered as management for one taxonomic group may be detrimental to others (e.g. increasing tree coverage would generally benefit birds but not butterflies, as these tend to prefer semi-open rather than shaded spaces) (Warren et al., 2021; Sandström et al., 2006).

Both proxies, UW and HQ were in general higher in the periphery of the urban areas indicating a higher potential to support biodiversity. Some centric areas (e.g. centric parks Fig. 2) were identified with high UW but low HQ, and in general, HQ values were high exclusively in areas located in the periphery of the city. These differences may be due to the inputs used to estimate both indicators. While UW takes into account structural descriptors of land use and terrain (perceived naturalness, remoteness and challenging terrain) (Müller et al., 2018, 2015), HQ includes assessing the expert perception of the habitat suitability of different land uses and their susceptibility to different anthropogenic impacts. Thus, HQ provides not only structural descriptors that can be obtained from land use maps and digital elevation models but incorporate functional descriptors of habitat suitability and vulnerability to threats to spatially describe the capacity to support biodiversity (Sharp et al., 2020; Di Febbraro et al., 2018; Sallustio et al., 2017). HQ seems to better consider potential threats (i.e. roads, night light, population density, railways and noise), their distribution and potential impact on the capacity of the different land use classes to harbour biodiversity. Contrastingly, other areas were identified with higher UW than HQ. For instance, industrial areas with low UW values showed medium HQ values since the spatial distribution and addition of threats had a lower influence in these areas as they are far from the city centre (e.g. night-light sources were fewer and population density was considerably lower than towards the city centre).

In general, we found high UW and HQ values in freshwater ecosystems (e.g. Zadorra river) indicating an overall high capacity to support biodiversity. Preserved wetlands, streams and surrounding areas (floodplains) typically support high biodiversity and are ecological corridors that may connect UGBS to larger ecosystems (e.g. peripheral lakes or mountains) (Mansur et al., 2022). However, these ecosystems were highly vulnerable in urban areas (Stroud et al., 2022) and were in our study identified as with higher UW than HQ due to the high vulnerability of these ecosystems to potential impacts. This vulnerability seemed to be particularly high for rivers possibly due to their higher edge effect which lowers the distance to threats (Sharp et al., 2020). Mapping of HQ allows identifying how these potential threats can affect the habitat suitability to support biodiversity as well as the connectivity of these ecological corridors.

4.2. Methodological considerations

Both UW and HQ are simple indicators, yet robust in describing the spatial patterns of biodiversity in the urban context, though they have different potential uses. UW seems to be more suitable when functional data is not available as it incorporates indicators related to the landscape structure and the input requirement is relatively low (i.e. digital elevation model, land use, mechanized access). Besides, both models may partially address some limitations associated with NDVI/NDWI indicators, which are widely used in urban environmental studies to quantify exposure to UGBS but are less adequate for measuring their quality. Such indicators are sensitive to seasonal variability and do not differentiate between green space typologies as considered in this study, thus limiting their applicability in the context of urban biodiversity conservation (Trethewey & Reynolds, 2021).

On the other hand, HQ is a more robust model to evaluate impacts affecting the functionality of the landscape, since it includes information on threats and their relative impact on biodiversity conservation objectives. These differences in UW and HQ, provide more accurate information to managers by identifying areas that can be categorized with high UW that are not so likely to support high biodiversity due to the influence of threats. Furthermore, our outcomes allow identifying key areas for urban biodiversity protection, (un)suitable places for new urban development or where efforts need to focus on reducing threats impacts. However, both models do not integrate size or connectivity descriptors that are highly associated to habitat quality for biodiversity, which would be a good improvement to further model development. Acquisition of extensive information on (urban) biodiversity status is a time and resource-consuming endeavour (Jalkanen et al., 2020) and since the input requirements to assess general biodiversity proxies in our spatial models were relatively low, they are particularly suitable for places where biodiversity information is deficient or monitoring efforts are scarce. The combination with expert consultation, a widely-used method to obtain information when data is limited (Terrado et al., 2016; Kuhnert et al., 2010), makes this approach cost-effective to evaluate potential urban biodiversity support while incorporating habitat suitability and sensitivity to threats to the different habitats.

These models have however some limitations based on their different assumptions on biodiversity support, data quality, and potential modelling uncertainties, as happens with other biophysical models (e.g. Aznarez et al., 2021; Bagstad et al., 2013). UW was in general a robust indicator of biodiversity, and this was supported by the results of the Jack-knife test, where it remained with stable values even when excluding one of its three basic components (perceived naturalness, remoteness and challenging terrain) (Appendix A). However, from these three indicators, the maps excluding perceived naturalness had lower performances, indicating that perceived naturalness is a key input contributing the most to UW character. Yet, other studies suggest that remoteness was more important in influencing UW (Müller et al., 2018), which indicates that the potential role of each indicator may vary depending on the habitat distribution of the study area.

Expert consultation was effective when lacking threat data to enable spatial comparisons and use relative scores to rank habitat suitability and quality indicators. However, different expertise backgrounds from the consulted experts, considering the suitability of the urban habitats to different types of biodiversity, along with the different anthropogenic threats to habitat suitability may influence the results (Di Febbraro et al., 2018). Despite the subjectivity and variability in expert considerations (Sallustio et al., 2017) introducing moderate uncertainty in our HQ predictions, the overall assessment proved to be consistent with the relative UW outcomes and coherent with the species richness indicators. The parametrization scores obtained from the international expert consultation can be further tested to any other urban context, as the survey was designed for urban biodiversity and land uses in general. The crowd-sourced data of the occurrence of taxonomic groups used to assess potential associations with UW and HQ indicators may have different biases that hinder their suitability as predictions of species richness as overrepresented data in easily accessible UGBS, highly populated areas or circulation ways (Sultana & Storch, 2021; Petersen et al., 2021; Callaghan et al., 2020). For instance, bird richness was higher due to an artefact in the greater number of observations (Fig. 5) since it is a more charismatic group, widely used in citizen science and reported in 'ornitho.eus'. However, these data allow studying large spatial scales based on large databases from multiple observations which overcome at least part of these biases and made these data suitable for urban biomonitoring programs (Callaghan et al., 2020).

UW and HQ were positively correlated and robust proxies to predict biodiversity for all considered taxa in our study. This suggests that at least with the data availability we had for biodiversity, land use, habitat suitability and potential threats, in a middle-sized city such as Vitoria Gasteiz, these databases were reliable to estimate biodiversity at urban scales. However, there is a substantial unexplained variation between our proxies and biodiversity that is attributable at least partially to other non-contemplated factors, e.g. seasonal variability, unassessed threats or habitat conditions, biogeographical filters, the effect of invasive species. Yet, further research considering small (e.g. micro-scales) and unconventional habitats with aggregated distribution patterns in the environment should consider possible limitations in the use of these models developed for larger areas (Knapp et al., 2021; Soanes et al., 2019). This approach should allow to improve the currently limited availability of biodiversity data and inventory completeness with an adequate aggregation scale for different taxonomic groups across the urban landscape, ultimately assessing spatial patterns at fine scale resolution. Recognizing and measuring the value of small spaces and unconventional habitats to urban biodiversity (i.e. from brownfields and cemeteries to cavities within buildings and infrastructure) calls for exploration along with novel conservation opportunities for management (Knapp et al., 2021; Soanes et al., 2019). Our results suggest that even if the obtained outcomes may be sensitive to biases, which is often the case in ecological modelling, they provide valuable and robust insight into identifying urban areas with high potential to support biodiversity and to pinpoint conservation areas.

4.3. Implications for urban biodiversity management

In our study, UGBS close to the city centre showed low and very low UW and HQ values, suggesting that cities such as Vitoria-Gasteiz, with 20 m^2 UGBS per inhabitant, do not necessarily encompass high HQ to biodiversity as these UGBS may be ecologically very poor, although still providing nature contributions to people. Coinciding with other studies (Mansur et al., 2022; Soanes et al., 2019; Turner & Gardner, 2001; Forman, 2014) our results suggest that although small and fragmented UGBS, are important for species persistence, biodiversity would benefit from decreasing perimeter:area ratio, along with increasing the size, quality and connectivity of UGBS from *peri*-urban areas to core areas. As the Vitoria-Gasteiz municipal Biodiversity Conservation Strategy points out, the overall biological diversity of the flora and fauna in urban parks has decreased significantly due to intensive management practices (i.e. the frequency with which the grass is being cut or pruned and dead trees removed) (CEA, 2014). Another reason adding to the low HQ for biodiversity support is the specific and structural simplicity of UGBS, which mostly consists of the vertical structure of green spaces (i.e. lawn and trees) (Garizábal-Carmona Garizábal Mancera-Rodríguez, 2021; Aronson et al., 2017; CEA, 2014). Previous studies suggest that classical parks with lawns and tall deciduous trees usually harbour few breeding species (Forman, 2014). Increasing HQ for biodiversity in urban socio-ecosystems is critically needed (Soanes et al., 2019), which is also correlated to UW according to our findings. This includes actions aiming to improve the conditions for human-wildlife coexistence, by accommodating dynamic natural processes (Mansur et al., 2022), providing resources such as refuge sites (e.g. holes in tree chunks, adding shrub cover), feeding sources or reducing the impact of different threats impact to habitat suitability, such as night light or fragmentation (Soanes et al., 2019; Forman, 2014). In other words, to increase biodiversity, cities should bring some wildness back. Reducing management (through rewilding *sensu* Perino et al., 2019), without affecting people's accessibility to UGBS could be a cost-effective way to enhance HQ in the city and therefore general biodiversity (Mansur et al., 2022; Müller et al., 2018). A variety of wildness-friendly actions such as reducing pruning or mowing would be suitable to increase the environmental quality of UGBS (Kowarik, 2018) while reducing maintenance costs, ultimately benefitting urban dwellers through encouraging interaction with nature. Vacant land and brownfields may likewise be an opportunity for increasing the network of green infrastructure in urban landscapes (Kabisch & Haase, 2013). Due to the scattered and low density of urban residential developments, Vitoria-Gasteiz has a considerable share of vacant and abandoned land, which has been included in the municipal Green Strategy. Yet, previous research warns that the repurposing of such areas may have undesired consequences for biodiversity (Macgregor et al., 2022; Broughton et al., 2020). Therefore, the ecological value of vacant land and brownfields along with its potential for urban wildness for increasing both, local biodiversity and recreational purposes, should be carefully considered (Macgregor et al., 2022; Kabisch & Haase, 2013).

As the size and connectivity of UGBS influence biodiversity, it is also closely related to public access to recreation in natural environments (Senetra et al., 2018). Experiencing nature in cities is a necessary condition for conservation since people with more exposure and access to nature are more interested in its conservation (Callaghan et al., 2020). As an increasing majority of the human population lives in cities, urban biodiversity is becoming the main people's interaction with nature (Dunn et al., 2006). However, considering the link between contact with nature and conservation measures, it is important to address the inequitable distribution of urban nature, and specifically, biodiversity, as it may contribute to lower levels of participation from minorities in environmental leadership initiatives (Mansur et al., 2022; Dunn et al., 2006). Here, our results may be also used to indicate areas within the city with little or no UGBS, and relate it to the demand side for such UGBS (i.e. through population density at the neighbourhood scale) as areas with very low UW and HQ values coincided with highly impervious areas. Further research should address how urban biodiversity patterns are distributed among vulnerable urban dwellers (Mansur et al., 2022). However, understanding the way that urban dwellers value and experience nature among contrasting perspectives is key to identifying socio-ecological trade-offs and feedback (Mansur et al., 2022; Hill et al., 2021). For instance, urban dwellers often perceive rewilding actions (i.e. low maintenance regime) negatively in the city, seeing such areas as abandoned, particularly when applied to urban contexts (Kowarik, 2018; Botzat et al., 2016). On the other hand, some species present in urban surroundings can be seen as pests (e.g. wild boar, seagulls, pigeons), leading to new human-wildlife conflicts or reinforcing existing ones (Forman, 2014; Dunn et al., 2006). Management actions should be carefully considered and designed to successfully communicate that unmanaged areas are intended by planners and not the product of neglect (Kowarik, 2018; Müller et al., 2018; Botzat et al., 2016). Since most of the world population including decision-makers and financial resources reside in urban areas, urban biodiversity conservation actions will ultimately rely on the ability of the population to experience and maintain the connection with urban nature (Callaghan et al., 2020; Soga & Gaston, 2016; Dunn et al., 2006).

5. Conclusion

UW and HQ indicators have been found to be appropriate for predicting the potential for biodiversity support in an urban context and therefore be integrated as key layers into urban planning. However, it should be noted that their use and individual suitability will depend on the data availability and study focus. UW is more suitable when focused on landscape structure indicators if functional data is not available, while HQ adds to this spatial information the effects of anthropogenic impacts on biodiversity and landscape functionality. Overall, this research has developed easily adaptable and replicable indicators that can provide information at a fine urban scale, accounting for different categories of UGBS and considering the anthropogenic effects on the quality of UGBS. The methodology used was based on open-access software, public data and crowdsourced data for all the phases of the modelling approach. This approach can provide a detailed level of information on urban biodiversity conservation in cities with limited monitoring capacities. Our study suggests that strategies aiming to increase biodiversity in urban landscapes should be based on improving wildness by enhancing structural dimensions of UGBS (i.e. perimeter: area ratio, size and connectivity), reducing management frequency and intensity along with evaluating potential anthropogenic pressures on different habitats.

Urban dwellers would benefit from enhanced contact with nature by increasing these drivers of urban biodiversity, but this would rely on the capacity to adequately communicate the aim and intentions behind rewilding actions. We, highlight that UGBS planning should focus not only on access and quantity but also on the ecological quality and particularly on the support for biodiversity that will ultimately enhance urban dwellers' experiences in their nearby nature.

CRediT authorship contribution statement

Celina Aznarez: Conceptualization, Methodology, Data curation, Validation, Formal analysis, Investigation, Visualization, Writing – original draft, Funding acquisition. **Jens-Christian Svenning:** Conceptualization, Methodology, Resources, Supervision, Writing – review & editing. **German Taveira:** Conceptualization, Methodology, Formal analysis, Writing – review & editing. **Francesc Baró:** Conceptualization, Methodology, Resources, Supervision, Writing – review & editing. **Unai Pascual:** Conceptualization, Methodology, Resources, Supervision, Writing – review & editing, Funding acquisition.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

Acknowledgements

We are grateful to the Environmental Studies Centre (CEA) from the Vitoria-Gasteiz council, especially to Asier Sarasua and Aitor Albaina for their collaboration and assistance with data collection and for providing valuable information from the city and greenbelt. We are also grateful to all the experts who kindly participated in the survey design and filled out the questionnaire for model parametrization, along with providing interesting discussion points. We thank Juan Pablo Pacheco for his feedback on the manuscript. CA was supported by the Doctoral INPhINIT–INCOMING program, fellowship code (LCF/BQ/DI20/11780004), from "La Caixa" Foundation (ID 100010434). JCS considers this work a contribution to his VILLUM Investigator project "Biodiversity Dynamics in a Changing World" funded by VILLUM FONDEN (grant 16549). This work was supported by the Basque Centre for Climate Change "Unit of Excellence" (Spanish Ministry of Economy and Competitiveness; MDM-2017-0714).

This research contributes to ICTA-UAB "María de Maeztu" Programme for Units of Excellence of the Spanish Ministry of Science and Innovation (CEX2019-000940-M).

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.landurbplan.2022.104570.

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