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Assessment of land cover trajectories as an indicator of urban habitat temporal continuity

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HIGHLIGHTS

- Fast and simple method to identify priority urban habitats for conservation.
- The vegetation cover in Porto was significantly reduced from 1947 to 2019.
- Wooded habitats have persisted mainly in public parks and gardens.
- Herbaceous habitats are rare and extremely susceptible to new urban development.
- We propose the preservation, restoration, and expansion of stable habitats.

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ABSTRACT

The trends of land cover dynamics in cities have major repercussions on urban biodiversity, ultimately impacting the health and well-being of human urbanites. Older habitats, i.e., habitats with higher spatiotemporal continuity, allow ecosystem processes promoting the maintenance of diverse microhabitats and the accommodation of diverse species over time, contributing for a richer biodiversity at the city level and beyond. Thus, we implement the analysis of land cover trajectories proposed as an indicator of habitat continuity, in order to identify priority areas for conservation or restoration that can be integrated in urban planning. The method was applied in the city of Porto, Portugal, and it was used to detect the main habitat patches associated with trends of vegetation persistence, gain and loss, between 1947 and 2019. The most common change was related to urbanization processes, where artificial built elements replaced areas of tree, shrub and herbaceous cover, completely dominating the territory in 2019 (66% cover). Moreover, tree and shrub cover had a relatively balanced dynamics, keeping a constant area (20–25%) while newly created wooded habitats compensated for the lost ones. Finally, areas covered with herbaceous vegetation suffered a drastic loss and fragmentation over the period of 72 years (from 40% in 1947 to 10% in 2019). This analysis of land cover evolution allowed for the identification of areas that have been continuously covered by vegetation, displaying a greater stability and higher probability of habitat continuity. This information, with its spatially explicit character, can help to inform future planning, design and management policies at a local scale and contribute to the conservation, restoration and expansion of urban habitats and their biodiversity.

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1. Introduction

Habitat loss and transformation is one of the major threats to biodiversity and urbanization is one of the main culprits (McKinney, 2002; Simkin et al., 2022; Tilman et al., 2017). Human populations are frequently settled in areas with rich soils and vegetation, often near waterbodies, which can exhibit high levels of biodiversity (Chapman & Underwood, 2009; Ives et al., 2016; Kowarik, 2011). However, urbanization causes the destruction, degradation, fragmentation and isolation of natural habitats, alteration of natural disturbance regimes and introduction of non-native species (Gaertner et al., 2017; Kowarik, 2011). On the other hand, urbanization is responsible for the emergence of new habitats in these areas, creating a diverse mosaic of semi-natural and man-made habitats, home to unique assemblies of flora and fauna species (Müller et al., 2013; Spotswood et al., 2021).

Urban biodiversity is crucial for ecosystem functioning (Harrison et al., 2014; Millennium Ecosystem Assessment, 2005; Zari, 2018) and it is beneficial to the health, well-being, and quality of life of urban residents (Aerts et al., 2018; Schebella et al., 2019). Preservation and promotion of urban biodiversity is fundamental if cities want to tackle climate challenges and achieve sustainability (Huang et al., 2018; Xie & Bulkeley, 2020). As such, it is thus essential that urban planning and landscape design integrate actionable ecological knowledge (Garrard et al., 2018; Zhou et al., 2019), ideally including spatially explicit data and at the appropriate spatial and temporal scales (Aronson et al., 2017; Borgström et al., 2006; Jarvis & Young, 2005; Opdam et al., 2013). With urban biodiversity conservation in mind, planners and decision makers must plan for the availability of relevant habitats, specific to each geographic context. In addition to the creation of new urban green spaces, it is imperative to preserve the oldest habitats (Kowarik & von der Lippe, 2018; McKinney, 2002). While newly created habitats can show high species richness, they are commonly occupied by non-native species and generalist native species that can tolerate a diverse range of conditions and disturbance (Callaghan et al., 2019; Ducatez et al., 2018). If habitat quality is maintained, habitat continuity (i.e. the persistence of a particular habitat type through time) supports the occurrence of specialist species in these old habitats (Lõhmus & Liira, 2013; Nordén et al., 2014; Raduła et al., 2020) and consequently promotes a higher species diversity at the city level.

However, despite the call for action of international experts and institutions (e.g. ICLEI, 2020; Secretariat of the Convention on Biological Diversity, 2012), many cities face multiple challenges to manage and preserve their biodiversity (Cooper et al., 2021; Wilkinson et al., 2013). In this context, there is still a lack of ecological knowledge on the functioning of the urban ecosystem and, frequently, the available data is inadequate to meet the requirements of urban planners, designers and managers (Jarvis & Young, 2005; Zhou et al., 2019), especially at the local level (McDonnell & Hahs, 2013; Wilkinson et al., 2013). Remarkably, this is the level where planning and management decisions directly translate into site interventions that modify urban habitats (Brown, 2017; Opdam et al., 2013). There is often a disconnection between the spatial and temporal scale of ecological information and the practices and policies that influence the urban environment (Borgström et al., 2006; Opdam et al., 2013; Uchida et al., 2021), which makes it difficult to integrate scientific data in spatial planning and urban habitat conservation programs (Brown, 2017; Łopucki & Kiersztyn, 2015; Opdam et al., 2013). Spatialization of ecological data is considered a fundamental tool to improve the communication between multidisciplinary scientific teams and non-expert decision-makers and to contribute to generate knowledge that is useful to inform urban planning, design and management (Jarvis & Young, 2005; Pedersen et al., 2004).

Habitat availability (in space and time) is an important determinant of species dynamics and several authors have highlighted the importance of its temporal aspect (Cadenasso et al., 2006; Hanski, 1999), particularly in urban ecosystems (Alberti et al., 2020; Müller et al., 2013; Ramalho & Hobbs, 2012). However, it is most common that in urban ecology studies, habitat availability is evaluated from a merely static perspective, which disregards the intrinsic dynamism of urban environments. Few urban ecological studies include the temporal component (Ossola et al., 2021); this can be assessed as patch continuity, habitat age, site history, or land-use legacy (e.g. Gustavsson et al., 2007; Johnson et al., 2018). The temporal aspect of habitat availability can influence the occurrence of diverse organisms over time through a multitude of processes. For example, it interferes with processes typically associated with metapopulation or metacommunity dynamics, such as dispersal, (re)colonization, and extinction, by causing time-lagged responses to landscape change, which in turn can cause a phenomenon of extinction debt or immigration (or species) credit (Lira et al., 2019; Swan et al., 2021; Uroy et al., 2021). Furthermore, habitats with higher continuity, i.e., those older and more stable, offer better conditions for specialist species, as these are often associated with habitat niches that take time to develop, such as tree cavities or wood decay (Janssen et al., 2017; Nordén et al., 2014).

In the scope of biodiversity conservation in an urban context, the following questions arise: Which, and where, are the areas that present a higher probability of habitat continuity in time? How can they be rapidly identified in a context of insufficient biodiversity data and low financial resources? In this regard, here we propose the analysis of land cover trajectories as an indicator of habitat continuity, in order to identify priority areas for conservation or restoration that can be integrated in urban planning. Using the city of Porto, in Portugal, as a case study, our research objectives were: 1) to determine and spatially represent the land cover trajectories from the mid-20th century to a recent date; 2) to assess the main changes in vegetation patches in terms of persistence, gain or loss; 3) to identify the habitat patches subject to a lesser degree of habitat disruption and, consequently, exhibiting a higher degree of continuity.

2. Methods

2.1. Study area

This study focuses on the Municipality of Porto, located in the northern region of Portugal (Fig. 1). Porto presents itself as a relatively small city (42 km^2, 232 000 inhabitants; INE, 2022), but it is the center of a dynamic metropolitan area comprised of 17 municipalities and around 1.7 million residents (AMP, 2022). Its privileged location at the intersection of the Douro River and the Atlantic Ocean and the mild temperate climate (*Csb* regime, Koppen-Geiger classification; Peel et al., 2007) conduced to a rich human history, as well as rich biodiversity levels (Guilherme et al., 2015).

2.2. Land cover trajectories

This study follows the work of Guilherme et al. (2022), which classified the land cover in Porto for the years 1947, 1979 and 2019, based on aerial photography and satellite imagery, using an adaptation of the UrHBA method (Farinha-Marques et al., 2017). UrHBA methodology is spatially explicit, based on the visual interpretation of the landscape and it excels at capturing urban habitat diversity at a fine scale. The mapping and classification of land cover was made in vectorial format in a GIS environment, with the older images being previously orthorectified (Guilherme et al., 2022). In a context of limited time and resources, and due to the limitations of historic remote sensing data, the mapping procedure was streamlined – all elements were mapped as patches, even if with a linear shape; the minimum dimension for patches to be distinguished was 2000 m^2, and the minimum width was 5 m. The classification was also simplified to five land cover categories: 1) Artificial Built Elements, ABE, 2) Trees and Shrubs, TRS, 3) Herbaceous, HER, 4) Sparsely Vegetated - Terrestrial, SPV, and 5) Sparsely Vegetated - Aquatic, AQU (see Guilherme et al., 2022 for more details). In an urban context, ABE consists of impervious surfaces and buildings; TRS

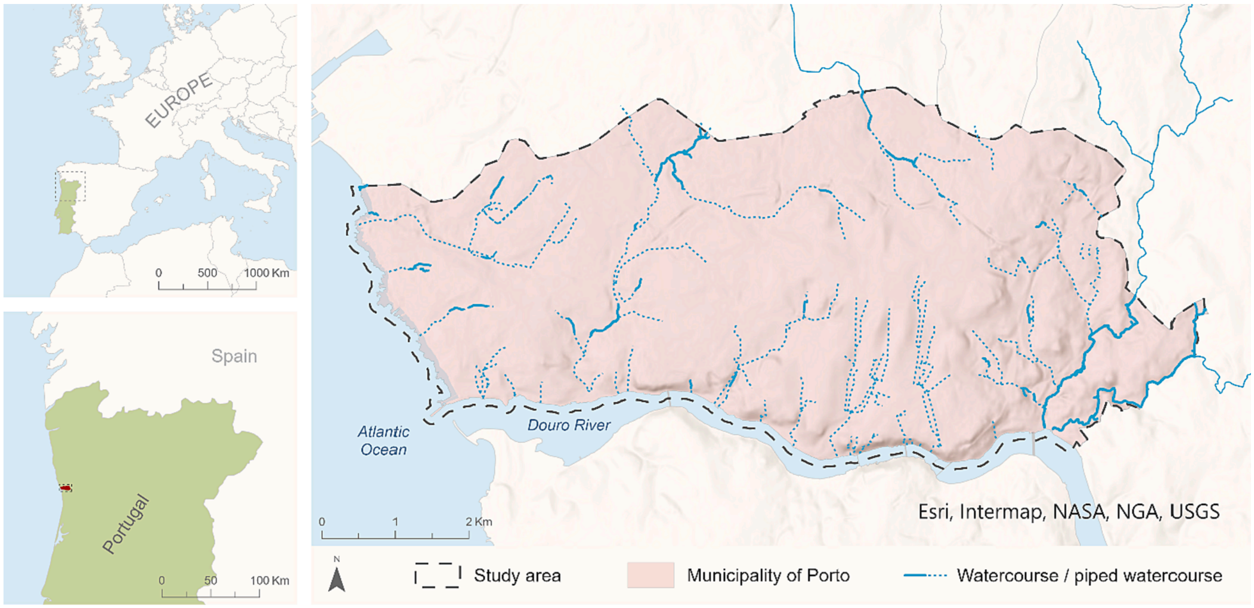

Fig. 1. Location of the study area, in northern Portugal.

identifies all tree and shrub cover, which can occur in parks, gardens, forested areas, scrub/bushland, riparian galleries, wooded crops, tree-lined streets, etc.; HER marks all areas covered by herbaceous vegetation, including both areas only temporarily covered with herbaceous vegetation and areas with frequent arrest to successional processes to maintain a permanent herbaceous cover, such as lawns, meadows, allotment or vegetable gardens, herbaceous crops, pastures, semi-natural grasslands, spontaneous vegetation in vacant lots, etc.; SPV consists of every type of non-vegetated soil, such as bare soil, sand, and rock; AQU comprises water surfaces, such as the sea, rivers, lakes, and ponds (including artificial water elements).

The method is based on the concept of land cover, which refers to the physical material that covers the surface of the land (Haines-Young, 2009; Jarvis & Young, 2005) and is easily applicable in a multitude of geographic contexts as it transcends local and cultural conditions (Farinha-Marques et al., 2017). It seems particularly relevant in urban ecological studies as it is associated with habitat types (Farinha-Marques et al., 2017) and it can be directly transformed through most urban interventions, not necessarily requiring a change in land use type (Haines-Young, 2009). At a fine scale, land use classification, being strongly associated with socio-economic activities, could fail to capture small habitat patches as the vegetation categories can occur in a variety of land use types; likewise, areas classified with the same land use type could encompass different vegetation covers.

Land cover trajectories are a common method to assess trends of landscape change over time, usually derived from satellite images or aerial photograph, at a particular location and at multiple dates (Gbanie et al., 2018; Liu & Cai, 2012). As such, the land cover layers for 1947, 1979 and 2019 (in vectorial format; Guilherme et al., 2022) were overlapped, using the "intersect" tool in ArcMap Desktop 10.5. This generated a new layer of information regarding the trajectory of land cover for the entire surface of the city. To reduce errors and/or biases and to simplify the data analysis, this process included the elimination of sliver polygons (smaller than 100 m^2), i.e., they were absorbed into larger adjoining polygons; this was achieved with the "eliminate" tool (ArcMap Desktop 10.5). In the newly created layer, each polygon is then described by a sequence of three land cover categories, i.e., a land cover trajectory, reflecting its evolution from 1947, to 1979 and to 2019, as illustrated in Fig. 2. The resulting data allowed for the analysis of the transition in land cover considering three different time intervals: 1) from 1947 to 1979; 2) from 1979 to 2019; and 3) the complete sequence from 1947, to 1979, then to 2019.

2.3. Changes in vegetation cover

Focusing particularly on vegetated areas, we analyzed in more detail the land cover transformations that occurred in the study area, for trajectories related to TRS cover, HER cover and general vegetation cover (combining both TRS and HER in the same analysis). The areas were classified according to the chronological occurrence of vegetation as follows: 1) persistent cover - present in 1947, 1979 and 2019; 2) gained cover - absent in 1947, but present in 2019; 3) disturbed cover - present in 1947 and 2019, but absent in 1979; 4) gained then lost cover - absent in 1947 and 2019, but present in 1979; 5) lost cover - present in 1947 and absent in 2019.

The analysis of vegetation evolution led to the identification of the areas with persistent vegetation cover, i.e., areas that were covered with vegetation in 1947, 1979, and 2019. The areas with persistent

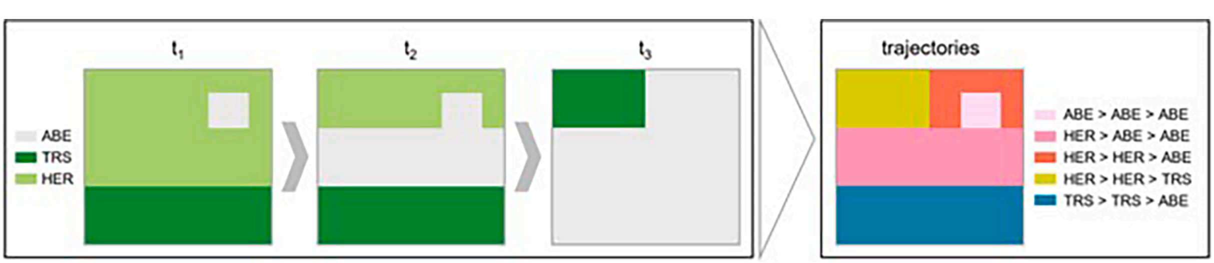

Fig. 2. Demonstrative example of the method for the classification of trajectories, using UrHBA categories in vectorial format.

vegetation were considered to have a higher probability of habitat continuity and, consequently, the areas with the most concentration of persistent vegetation were identified as the most important for habitat conservation and regeneration. Besides habitat continuity, these areas were also prioritized based on vulnerability to future land cover change, which was assessed based on their framing within urban planning instruments (Câmara Municipal do Porto, 2021a, 2021b), current land use, ownership, and accessibility. On one extreme, the areas with persistent vegetation that are included in environmental plans, that are currently integrated in green spaces, on the public domain, and publicly accessible, are considered the least vulnerable (and thus, the least priority). On the other extreme, the areas located in zones intended for urban development, that have no current use, are private domain, and are not publicly accessible, are considered the most vulnerable to habitat destruction and we consider they should be the top priority for conservation. For this purpose, vegetation patches located in public parks and gardens were considered to be the least susceptible to habitat destruction, as these areas tend to be preserved on the long term, since their conservation is usually included in urban plans and policies. Vegetation patches integrated on the green spaces of public and private services facilities (such as schools and hospitals) were classified as moderately susceptible, as these are relatively valued but are more prone to destruction when compared to public parks and gardens, as they are not included in environmental plans and policies. On the other hand, vegetation mainly located in undeveloped land or private green spaces was considered the most vulnerable to habitat loss, as these areas are frequently subject to urban pressure, being classified as areas destined to urban development.

3. Results

3.1. Land cover trajectories

The study of the dynamics of land cover transformation in Porto revealed 106 trajectories, amongst 125 possible ones, considering the set of five land cover categories. Of these trajectories, 11 stood out as predominant, covering in total more than 80% of the territory (Fig. 3). As an overall trend, derived by urbanization, over 62% of the study area has retained or acquired a higher degree of artificialization, i.e., marked by the coverage of Artificial Built Elements (ABE). Nevertheless, a considerable proportion of land has also maintained a less artificial land cover, namely Trees and Shrubs (TRS) and Herbaceous (HER), throughout the analyzed period of 72 years.

The city of Porto revealed a relatively dynamic land cover evolution, where almost 60% of the territory has experienced at least one change in its land cover since 1947 (Table 1). A smaller portion has been through

Table 1
Different situations of land cover dynamics and variation (and its distribution throughout the city).

Land cover dynamics	Area (ha)	Proportion (%)	Land cover variation	Area (ha)	Proportion (%)
Land cover never changed	1722.40	41.4%	Same land cover in 1947, 1979, and 2019	1722.40	41.4%
Land cover changed once	1853.44	44.6%	Same land cover in 1947 and 1979	956.21	23.0%
			Same land cover in 1979 and 2019	897.23	21.6%
Land cover changed twice	585.91	14.0%	Same land cover in 1947 and 2019	267.07	6.4%
			Different land cover in 1947, 1979, and 2019	318.84	7.6%

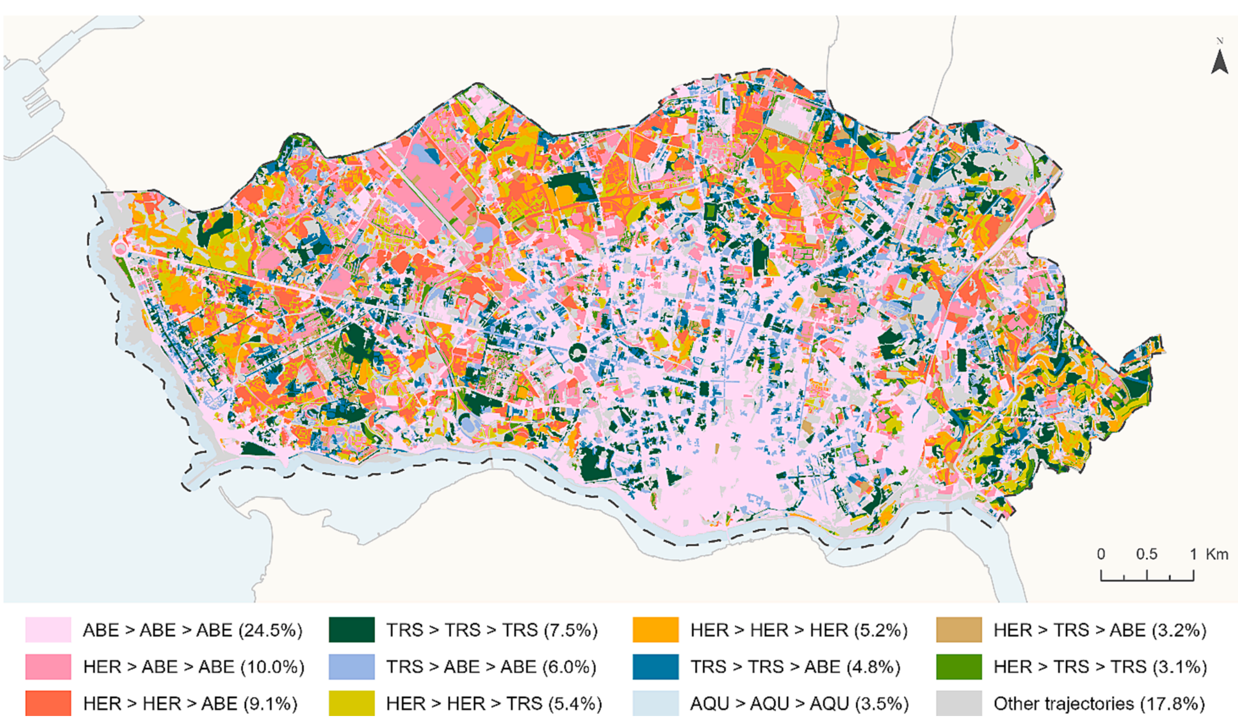

Fig. 3. Land cover trajectories from 1947, to 1979 and to 2019 (1947 > 1979 > 2019). Land cover categories: ABE, Artificial Built Elements; TRS, Trees and Shrubs; HER, Herbaceous; SPV, Sparsely Vegetated Terrestrial; AQU, Sparsely Vegetated Aquatic.

multiple transformations, with 7.6% of the study area exhibiting three different land cover for the three analyzed moments.

The city of Porto has experienced a marked increase in urbanization since the mid-20th century. From 1947 to 1979, the area covered by Artificial Built Elements (ABE) has increased by 47.7% (while maintaining 85.4% of their previous patches; Fig. 4); this growth was supported by the replacement of areas previously covered by herbaceous vegetation (HER, 494.60 ha), but also by trees and shrubs (TRS, 286.18 ha; Fig. 4; Table 2). From 1979 to 2019, the increment in ABE is also notorious; this time, ABE cover has incremented by 37.66%, while retaining almost 90% of previous areas. The destruction of vegetation patches continues to be the main source of land cover change, with 420.03 ha of TRS cover and 462.96 ha of HER cover being replaced by ABE cover (Fig. 4; Table 2). Artificial Built Elements (ABE) is the least dynamic category (besides AQU), since it retains almost all of the previous cover, experiencing mainly irreversible gains throughout the years (Fig. 4; Table 2). The most significant exception is the change from ABE to TRS (both from 1947 to 1979 and from 1979 to 2019), which is generally related to the implementation and growth of tree cover in streets and squares, in residential areas and in previously industrial areas (following the demolition of factory buildings).

The area covered by TRS cover has remained relatively constant for the analyzed period. However, vegetation patches with this cover experienced a fair number of changes (Fig. 4; Table 2). In fact, only approximately half of the TRS cover was maintained in each of the transitions (from 1947 to 1979 and from 1979 to 2019), while the rest was transformed mainly to ABE, but also HER cover.

As also seen in Table 2 and Fig. 4, for both time intervals, areas of HER cover have also been gradually substituted, in considerable proportions, by areas of TRS cover (309.84 ha for 1947–1979, 282.13 ha for 1979–2019). The category of HER cover went from being the predominant class, in 1947 (40% cover), to one of the least abundant land cover categories (10% cover), being surpassed by both ABE and TRS.

The categories of Sparsely Vegetated – Terrestrial (SPV) and Sparsely Vegetated – Aquatic (AQU) are strongly associated with the coastal and river lines (beaches, cliffs, sea, and watercourses), that have remained relatively stable at the city level (Fig. 4; Table 2). However, it should be noted that a relatively large surface of SPV cover has been replaced by ABE, especially in the period 1979–2019. The changes seen in the category of SPV are related to tidal variation, construction in previously

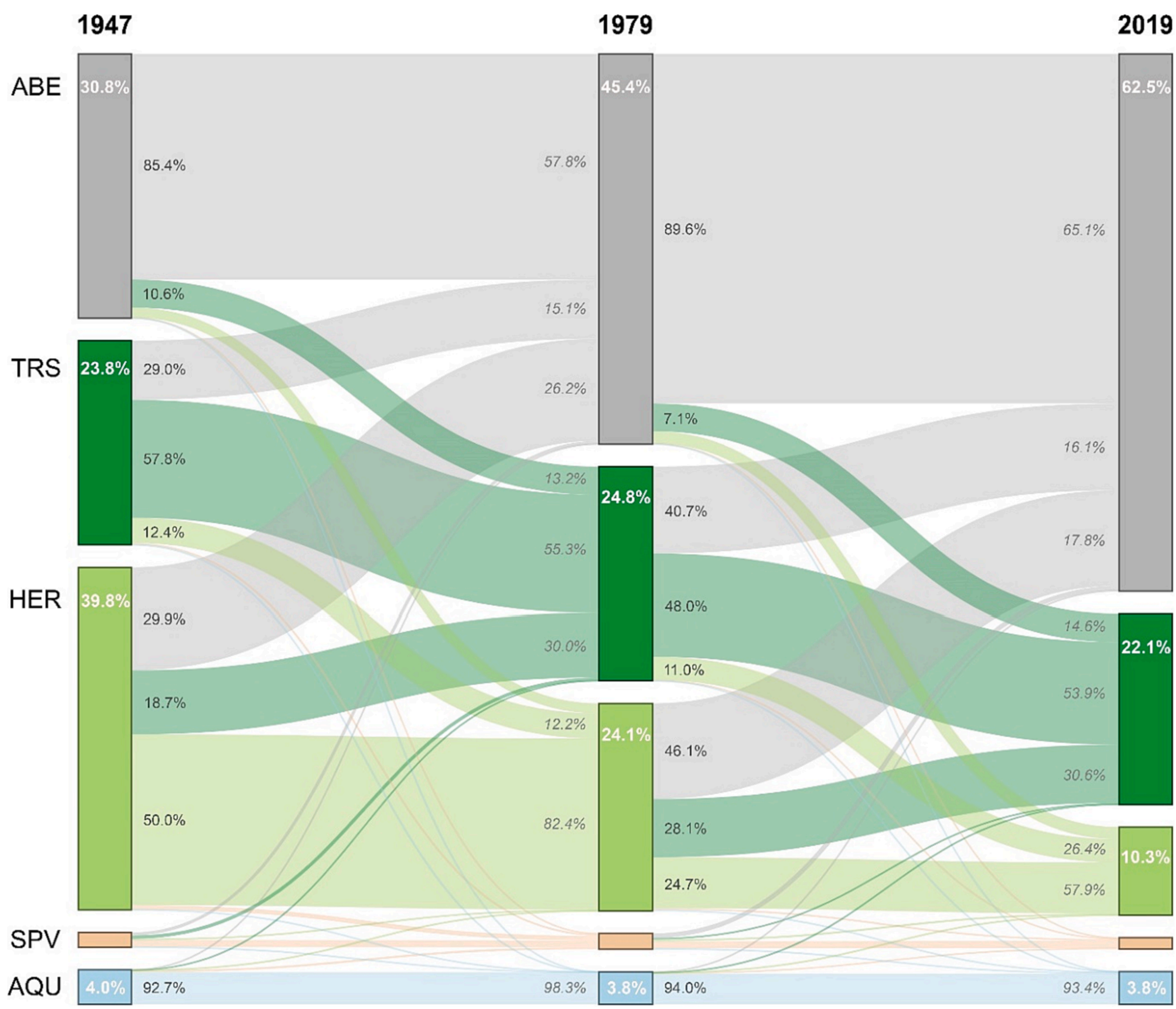

Fig. 4. Diagrammatic representation of the evolution of land cover in Porto, from 1947, to 1979 and to 2019 (created with "SankeyMATIC", at www.sankeymatic.com). Land cover categories: ABE, Artificial Built Elements; TRS, Trees and Shrubs; HER, Herbaceous; SPV, Sparsely Vegetated Terrestrial; AQU, Sparsely Vegetated Aquatic. White values refer to the proportions of each land cover category for each of the three years; dark values indicate: 1) the proportion of the original land cover that followed that path (regular text on the right side of the bars), 2) the proportion of the target land cover that followed that path (italicized text on the left side of the bars); only the values that relate to transformations that occurred in over 100 ha appear in the figure.

Table 2
Land cover conversion matrix from 1947 to 1979 and from 1979 to 2019 (area, ha).

Land cover		1979 ABE	1979 TRS	1979 HER	1979 SPV	1979 AQU	Total (1947)	Change
1947	ABE	1092.96	135.89	48.07	2.53	0.41	1279.86	+ 47.7%
	TRS	286.18	570.99	122.13	9.01	0.19	988.50	+ 4.5%
	HER	494.60	309.88	827.07	22.36	0.33	1654.24	− 39.3%
	SPV	14.81	16.02	5.90	33.44	1.68	71.85	+ 7.8%
	AQU	1.55	0.29	0.22	10.08	154.15	166.29	− 5.7%
Total (1979)		1890.10	1033.07	1003.39	77.42	156.76	4160.74	

Land cover		2019 ABE	2019 TRS	2019 HER	2019 SPV	2019 AQU	Total (1979)	Change
1979	ABE	1693.55	134.20	58.70	2.77	0.88	1890.10	+ 37.7%
	TRS	420.03	496.09	113.27	3.19	0.49	1033.07	− 10.9%
	HER	462.84	281.71	247.79	6.39	4.66	1003.39	− 57.3%
	SPV	22.02	7.74	8.42	34.90	4.34	77.42	− 32.3%
	AQU	3.79	0.45	0.09	5.14	147.29	156.77	+ 0.6%
Total (2019)		2602.23	920.19	428.27	52.38	157.67	4160.74	

vacant lots, abandonment of sediment extraction sites and regeneration of previously deforested sites, whereas for AQU it mainly reflects tidal variation, the artificialization of coastal and riverfront and the built-up of bridges. For the remainder of this article, SPV and AQU evolution will not be discussed further, as it is outside the main scope of this research.

3.2. Changes in vegetation cover

Now focusing on vegetation patches, the results revealed substantially different patterns of land cover transformation for TRS and HER categories. Regarding arboreal vegetation (Fig. 5.a), over 40% of the study area was covered by TRS at some point between 1947 and 2019. While a considerable proportion of the territory faced the loss of TRS cover (14.3% of the study area), this was somehow compensated by the gain in TRS cover in other areas (12.7% of the study area).

A significant portion of the city (7.5%) has been permanently covered by TRS since 1947, mainly organized in large patches (Fig. 5.a). To a similar extent, some areas (6.7% of the study area) experienced a gain of TRS cover between 1947 and 1979 but were later occupied by artificial elements between 1979 and 2019.

In contrast with the more balanced changes in tree cover, the evolution of herbaceous vegetation identifies a clear dominant trend towards the loss of the majority of patches of HER cover (Fig. 5.b). While almost half of the city was covered with herbaceous vegetation at some point between 1947 and 2019, the loss of HER cover occurred in about a third of the study area (32.8%) for the same period. In about 10% of the territory, HER cover was replaced by TRS. However, the most common change was from HER to ABE (in over 20% of the study area). On the contrary, only 5.2% of the city area corresponds to persistent herbaceous patches, from 1947 to 2019. Some new herbaceous patches were created in the analyzed period, but the gain of HER cover is not expressive at the city scale.

Considering all vegetated areas (Fig. 5.c), whether covered by herbaceous or wooded vegetation, it becomes clear that some large areas have been spared from construction since, at least, 1947, and they are mainly concentrated in the western, eastern, and northern sections of the city (amounting to 25.3% of the study area). On the other hand, more than one third of the territory (35.1%) has experienced the complete loss of vegetation cover; this trend is very noticeable in the periphery of the city, due to the urban sprawl of the second half of the 20th century (Fig. 5.c). In this overall analysis, the areas that gained vegetation cover were almost negligible.

This analysis of land cover trajectories allowed to detect which areas have been continuously covered by vegetation, displaying a higher stability and higher probability of habitat continuity. As highlighted in Fig. 6, around 25% of the territory has been consistently covered by vegetation, but approximately half of this area experienced significant habitat changes, transitioning from wooded to herbaceous vegetation or vice-versa. Patches of relatively stable habitat are restricted to 12.7% of the city surface and include wooded and herbaceous habitats in similar proportions. Habitats with continuous tree and shrub cover are scattered throughout the entire study area, while the habitats with continuous herbaceous cover are concentrated in the western, eastern, and northern sections.

This research allowed for the identification of the main sections of the city that have a greater conservation value based on the criteria of habitat temporal continuity, while also considering their susceptibility to habitat loss, based on their protection status in urban planning programs, current land use, ownership, and accessibility (Fig. 7). This spatially explicit representation can be particularly useful for communication and knowledge dissemination. Indeed, while the areas with high probability of habitat continuity are evenly distributed throughout the city, the eastern and northern sections face a lack of protection of these important green spaces. The majority of the older vegetation patches in the northern and eastern section is located in private undeveloped land, unlike the most western section, where they are generally integrated in parks and gardens.

4. Discussion

The global trend of increasing urbanization and loss of vegetated areas (Güneralp et al., 2020; Radwan et al., 2021; Richards & Belcher, 2020) was also experienced in Porto. This phenomenon of declining vegetation cover being associated with urban expansion and densification has been reported by numerous authors in cities and metropolitan areas around the world (e.g. Lennert et al., 2020; Zaitunah et al., 2021; Zubair, 2021). Some authors have also investigated the relation between urban expansion and vegetated areas for the particular case of Porto. For example, Madureira et al. (2011) analyzed the evolution of the green structure in Porto and highlighted its severe decline and fragmentation throughout the 20th century, especially for agricultural areas. In a comparison between multiple cities, Bagan and Yamagata (2014) noticed a similar pattern for Porto, where the increase in urban area resulted in the destruction of forest, cropland and grassland.

Nevertheless, except for a few cases, the research on urban site history or urban land cover trajectories is still very incipient, especially at the local level. The local landscape level is defined by Opdam et al. (2013) as the “areas where physical landscapes interact directly with

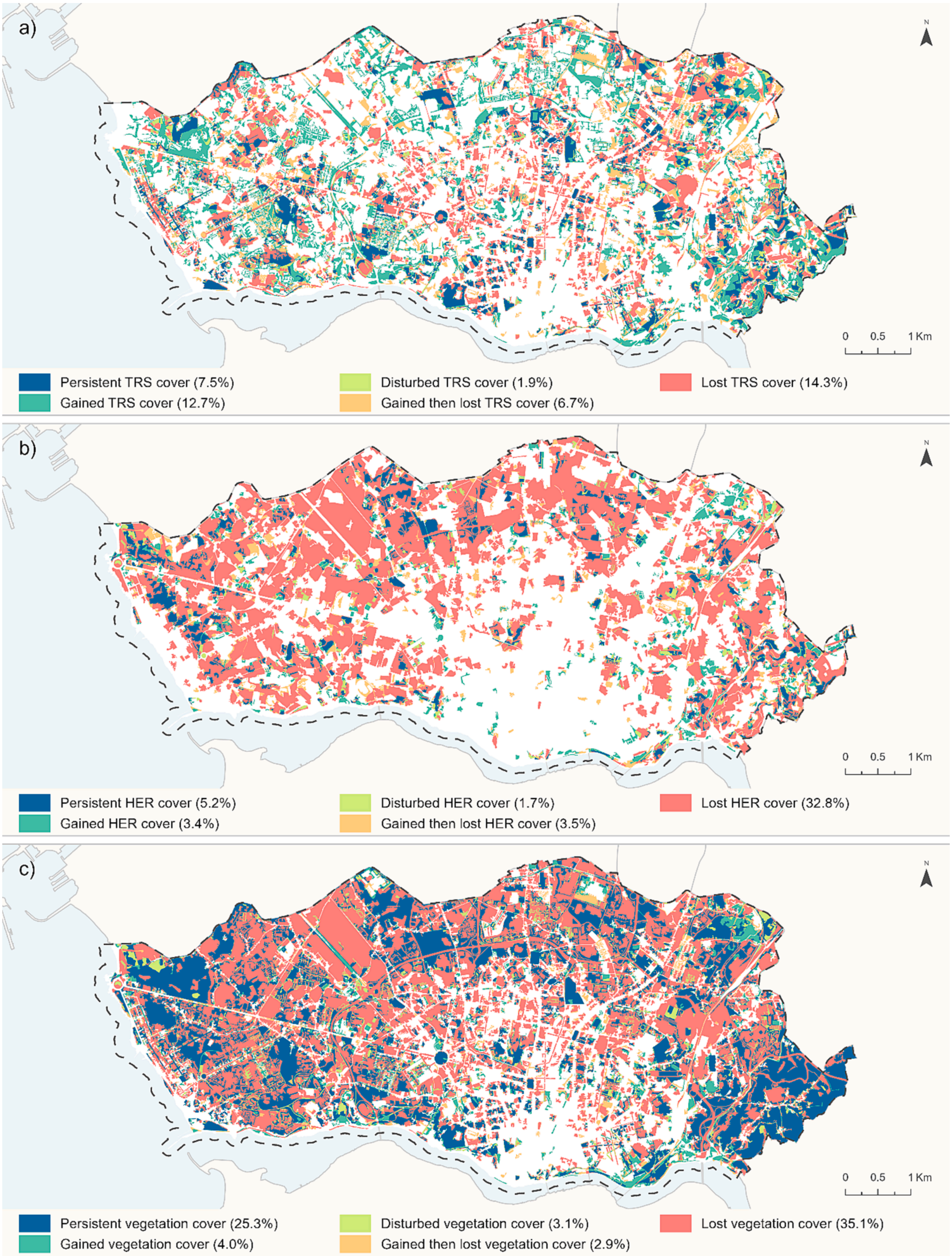

Fig. 5. Patterns of land cover change involving: a) TRS, Trees and Shrubs, b) HER, Herbaceous, and c) overall vegetation.

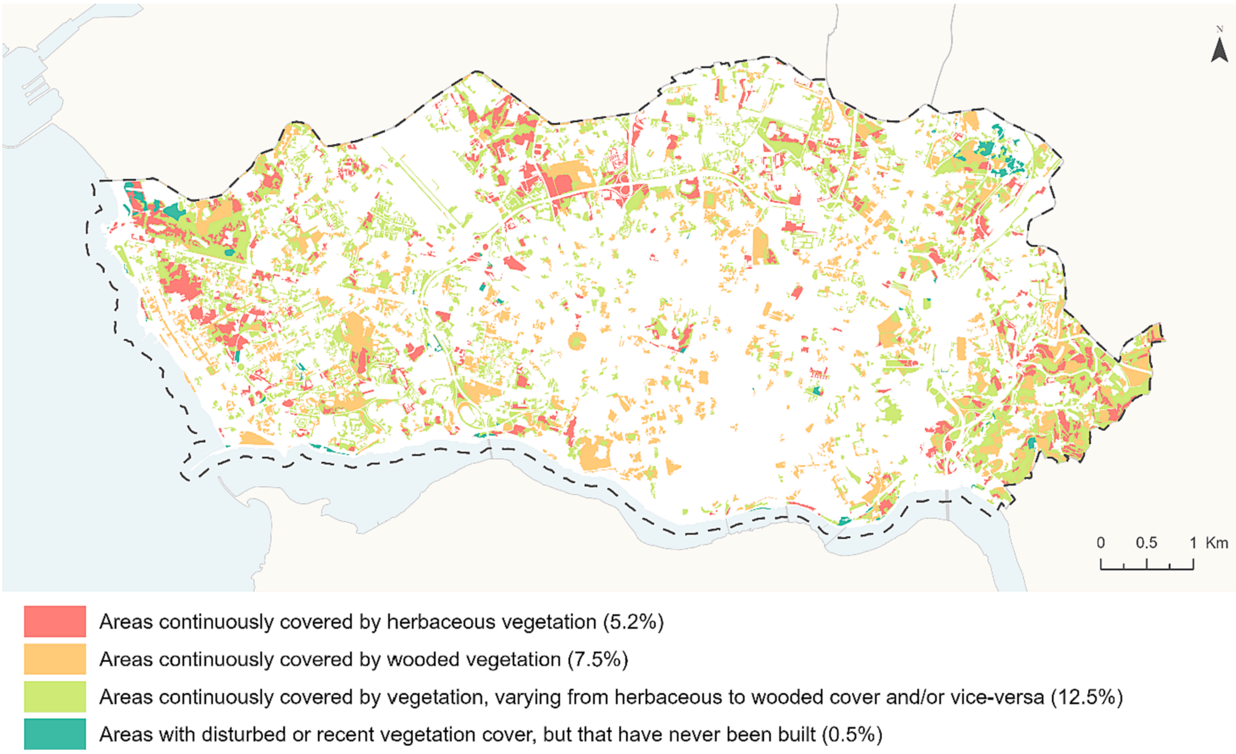

Fig. 6. Vegetated areas with higher stability and higher probability of habitat continuity.

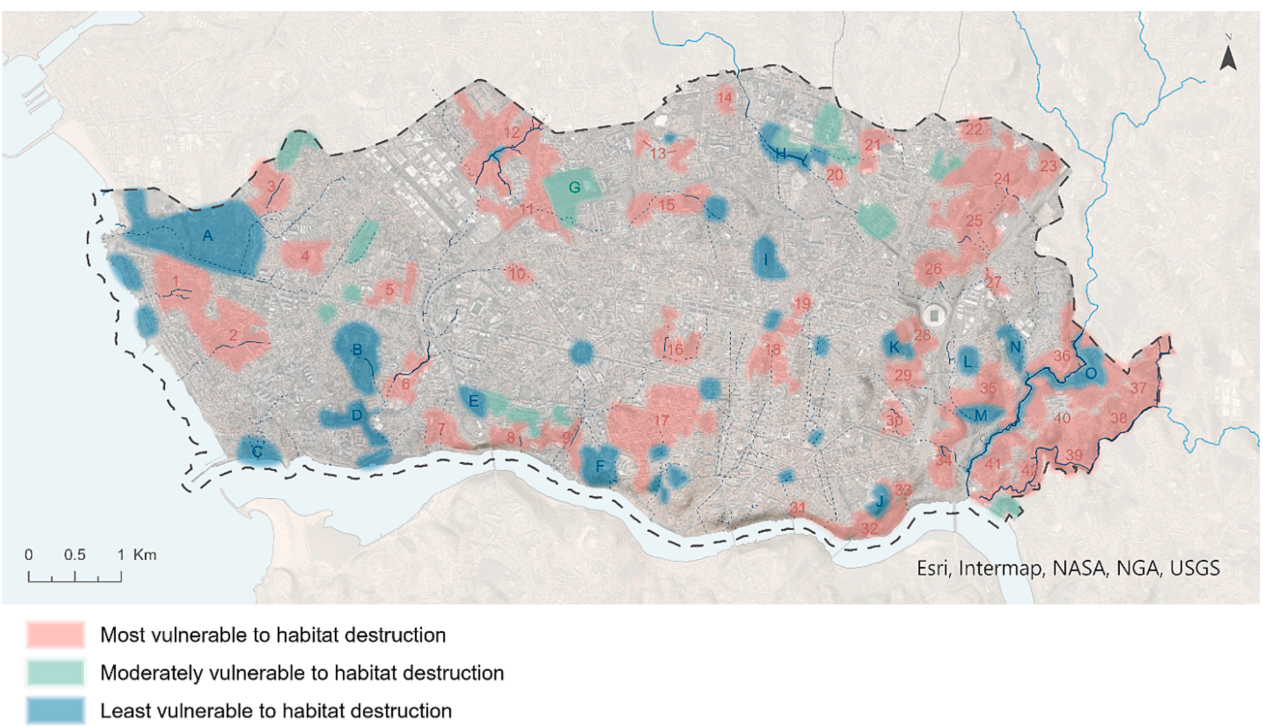

Fig. 7. Main sections of the city with higher habitat continuity, graded by their vulnerability to habitat destruction; for a complete legend see appendices (source of background imagery: Direção-Geral do Território, 2018).

social networks of land owners, managers, and landscape users". This is particularly relevant in urban landscapes, where the ecological dynamics are constantly affected by local scale decisions such as the planning, design and management of individual sites and infrastructures (Brown, 2017; Kremer et al., 2016). The study of land cover trajectories of Porto unraveled the history of each site and allowed to infer the temporal continuity of each vegetation patch, which was achieved at a very fine scale for the entire city. Past land use and land cover have an impact on ecosystem functioning, as they influence current landscape conditions and affect future responses to environmental change (Foster et al., 2003; Ramalho & Hobbs, 2012). The current assemblies of urban biodiversity are a result of the interaction between landscape heterogeneity and connectivity, which are influenced by the age and history of each site or habitat patch (Alberti et al., 2020; Alberti & Wang, 2022; Cadenasso et al., 2006; Foster et al., 2003). Older habitats, i.e. those that are more stable and that underwent the least transformation, are considered to harbor higher biodiversity levels (Janssen et al., 2017; Nordén et al., 2014; Raduła et al., 2020). For this reason, habitat age or temporal continuity can be used as a valuation criteria for the conservation of urban habitats (Jarvis & Young, 2005; Wittig & Schreiber, 1983).

The analysis of land cover change in Porto showed a highly dynamic urban landscape. Only around 40% of the study area has remained relatively stable in terms of land cover during the study period; this value seems to be a bit lower than the average reported for other southern European cities (e.g. Frondoni et al., 2011) and it mostly regards land that was already urbanized in the mid-20th century. The most noticeable land cover transformation in Porto is related to urbanization processes that drastically reduced and fragmented vegetation patches, which is comparable to other urban areas (Tello et al., 2020; Zaitunah et al., 2021).

Tree and shrub (TRS) cover revealed a dynamic history in Porto since 1947, despite maintaining a constant proportion in the city throughout the years – a fair portion of the old TRS patches were destroyed (597 ha or 14.3% of the study area), but others were created elsewhere in a similar extent (529 ha or 12.7%). Large patches of forestry and wooded crops were gradually replaced by urbanization (ABE), especially in the periphery. It is also noteworthy the loss of TRS cover in smaller spaces, spread throughout the center and in the western extremity (the oldest parts of the city), which once formed a network of tree-covered private gardens and tree-lined streets which was lost over time. The areas that experienced a gain in TRS cover reflect two main trends, both of which occurred mainly in patches previously dominated by herbaceous vegetation: 1) the abandonment of former agricultural fields, where the natural course of ecological succession originated wooded habitat patches; 2) the construction of residential buildings surrounded by wooded green spaces. Around 314 ha (or 7.5% of the study area) consists of land continuously covered by trees and shrubs. Most of this area corresponds to older public parks and gardens (e.g., Crystal Palace Gardens and Passeio Alegre Garden; Fig. 7), or old farms that have been converted into public parks more recently (e.g., Covelo Park and S. Roque Park; Fig. 7), but some forestry still remains in the eastern and north-eastern parts of the city (particularly in the areas of Areosa and Areias; Fig. 7). The inherent nature of forested areas, with periodic clearings and the use of monospecific cultures, can have negative effects on biodiversity levels (Liu et al., 2018; Pohjanmies et al., 2021), so its conservation value is probably limited. In contrast, the wooded habitat patches located in parks and gardens present a higher stability and are expected to harbor a more diverse and abundant array of species (Nielsen et al., 2014; Vasquez & Wood, 2022). Unlike what happened with the herbaceous cover, a significant amount of new TRS habitats was created between 1947 and 2019. This tree and shrub cover that developed more recently in vacant lots or previously cultivated land, with a multi-layered vegetation structure (which is not commonly observed in public parks and gardens) due to the lack of management, can also develop interesting levels of biodiversity, but faces additional challenges associated with the presence of invasive species, little social value and low acceptance by local residents (Filibeck et al., 2016; Gaertner et al., 2017).

The areas occupied with herbaceous (HER) cover have suffered a drastic reduction since 1947, which was particularly severe between 1979 and 2019; in the overall picture, around one third of the territory (1367 ha) experienced the loss of herbaceous habitats. The development of new built structures and impervious surfaces in Porto took the place of previously open spaces, which are usually covered with herbaceous vegetation, in a similar scenario as other geographic contexts (e.g. Tello et al., 2020). The creation of new herbaceous patches was insignificant (only 3.4% of the study area, 141 ha) and corresponds to areas where previously forested land was cleared for future urban development and areas where old industrial or residential urban fabric was demolished and not yet replaced. Patches with persistent herbaceous vegetation occur in only 5.2% of the city (216 ha) and most of them express an agricultural past and were historically managed in the form of arable land or pastures. However, nowadays, most of these areas correspond to vacant lots without current defined use (where the herbaceous vegetation is kept by infrequent mowing or grazing), some have been absorbed by parks or gardens and converted to lawns, and only a small fraction retains an agricultural profile. The larger and older patches of herbaceous habitats are concentrated in the western extremity (the surrounding area of Nevogilde and Ervilheira streams), in the northern area (around Quinta da Prelada, Viso and Requesende), at the northeastern zone (around Contumil), and in the eastern portion of the city (particularly around the areas of Pinheiro de Campanhã, Tirares, Granja and Furamontes; Fig. 7). Unlike most of the older wooded habitats, which are integrated in large parks and gardens, most of these old herbaceous patches are located in vacant lots and small cultivated areas, thus being quite vulnerable to being displaced by new urban developments.

In order to maintain sufficient levels of ecosystem functioning, to promote urban biodiversity and ultimately to benefit the health and well-being of the residents of Porto (Aerts et al., 2018; Zari, 2018), vegetation patches and habitats should be preserved as they are becoming increasingly scarcer. Generally speaking, the biodiversity of different habitat types is enhanced when these habitats are older (Nordén et al., 2014), which can be due to a more complex vegetation structure and a more diverse collection of microhabitats that can harbor a higher diversity of species, as well as also factoring the time that certain species can take to colonize new habitat patches after dispersion (Janssen et al., 2017). The influence of habitat persistence varies according to several species life traits, especially their habitat specificity, longevity, and dispersal capability (Lira et al., 2019). Specialist, long-lived, and/or dispersal-limited species tend to show a more delayed response to habitat change (e.g. Krauss et al., 2010; Löfvenhaft et al., 2004; Sang et al., 2010). A similar pattern was discovered in a previous study focused on the patches conserving natural vegetation within the whole metropolitan area of Porto, where even small areas kept a substantial diversity of amphibian and reptile species with limited dispersal capabilities (Ribeiro et al., 2009). The maturity of vegetation, particularly trees, besides influencing the occurrence of microhabitat niches (Janssen et al., 2017), has also implications on the provision of several other ecosystem services beneficial for humans (Jonsson et al., 2020).

While the relationship between the age or continuity of urban habitats and their biodiversity levels is fairly understudied, several authors have investigated this association in other wooded and herbaceous habitats. It has been shown that forests and woodlands with a higher continuity in time exhibit a higher species richness, especially due to the occurrence of specialist species; these are associated with particular habitat niches that take time to develop as the forest stand matures, such as rough bark, tree cavities or deadwood (Janssen et al., 2017; Nordén et al., 2014; Siitonen & Saaristo, 2000). The city of Porto has, so far, succeeded at preserving a substantial portion of its older wooded habitats, by including them in public parks and gardens that provide multiple benefits for the citizens. A higher level of biodiversity in urban parks is positively associated with direct gains in the health and well-being of their users (Cameron et al., 2020; Schebella et al., 2019; Wood et al., 2018). Thus, in order to maximize the benefits of these old tree-covered habitats, it is necessary that their management considers the promotion of biodiversity and other ecosystem services. For example, it is important to allow trees to reach maturity, encourage the occurrence of multiple layers of vegetation and support the creation of diverse microhabitats, often contradicting traditional maintenance practices (Han et al., 2022; Le Roux et al., 2014; Nielsen et al., 2014; Threlfall et al., 2017).

Historical parks with restricted access, but belonging to institutions of public interest, have also an important role in the preservation of old woodlands in Porto, and they generally present lower disturbance levels due to reduced intensity of use. Here, this type of green space has its main origins in old recreational farms and manor house gardens and they can present a high biodiversity potential (Lõhmus & Liira, 2013). These areas are not publicly accessible, but are generally owned by institutions of public interest, such as universities, schools, health facilities, local government, etc. While these are likely to be maintained in the long run, their green spaces are not considered in spatial planning and are thus more susceptible to habitat destruction than public green spaces, and, consequently, its future preservation and adequate management should also be promoted. Likewise, some residential gardens are also relevant for the maintenance of older wooded habitats, especially near the city center where the urban matrix is also from an earlier period. These areas also have biodiversity conservation interest, especially when considered collectively (Doody et al., 2010; Goddard et al., 2010; Thompson et al., 2003). These are particularly susceptible to urban development and, therefore, quite challenging to preserve. We suggest the engagement of the population towards biodiversity conservation practices and the implementation of restrictions to further land development in these areas, accompanied by benefits for the involved homeowners (Goddard et al., 2010; Parris et al., 2018).

Similarly to wooded habitats, there is also a relation between herbaceous habitat age or continuity and its species richness (Inoue et al., 2021; Raduła et al., 2020), although it is still an understudied topic. In fact, herbaceous habitats, particularly grasslands, have been neglected by ecologists, land managers and planners, possibly due to the misconception that these are associated with land degradation after deforestation and other disturbances (Veldman et al., 2015). Urban grasslands are particularly affected and are often left out of conservation plans (Hüse et al., 2016). We have revealed a similar pattern in the city of Porto – while most of the relevant woodland patches have been safeguarded in parks and gardens, most of the old herbaceous patches have been lost due to urbanization or lack of maintenance. As an exception, a few herbaceous patches have also been integrated in public parks (namely City Park and Oriental Park; Fig. 7) and are likely to be maintained in the long run. However, these have been converted into lawns with dubious ecological value. The high frequency of mowing in these habitats is related to lower levels of plant, microorganism and invertebrate species richness (Chollet et al., 2018; Helden et al., 2018; Norton et al., 2019). The diversification of urban herbaceous habitats, so as to include different plant compositions, different heights and different uses, has the potential to generate considerable benefits for biodiversity. When adequately managed, urban herbaceous habitats, in the form of meadows, semi-natural grasslands, arable fields, etc., can provide valuable ecosystem services and harbor high biodiversity levels (Klaus, 2013; Onandia et al., 2019). Considering their extreme scarcity and lack of protection in the city of Porto, we believe that older herbaceous habitats should be considered as top priority regarding urban biodiversity conservation. To improve the biodiversity interest of herbaceous habitats in Porto, we suggest the promotion of a diverse vegetation community, aided by replanting native species and controlling invasive species, and the reinforcement of habitat connectivity amongst patches. The prospective loss of these vegetation patches at the city scale would contribute not only to the general trends of habitat destruction, but also to the trends of species extinctions that have already been described at an international level. For example, the European Environment Agency (2021) has recently reported the decline of farmland birds and grassland butterflies in EU member states, with both fauna groups being strongly associated with herbaceous habitats.

The most effective and cost-efficient strategy for urban habitat conservation involves the preservation of older habitat patches, but regeneration or restoration can be an alternative or complementary solution (McKinney, 2002). When considering this solution, attention must be given to the history of former land uses of each targeted vegetation patch, as well as its landscape context, as they can influence the biodiversity outcomes and other ecosystem services (Foster et al., 2003; Vandewalle et al., 2014). Habitat restoration should also take advantage of spatial–temporal connectivity of the patch (Uroy et al., 2021; Zhao et al., 2021), i.e. the restored habitat should emerge at, or near, a place where that habitat type has occurred previously, to maximize the flow of biological resources to the new habitat patch. This is particularly important in the case of low habitat availability at a larger scale and for species with low dispersal capability (Huang et al., 2020; Martensen et al., 2017). This way, older habitats, which can act as refuge for some species, can serve as a source for recolonization of the newly created favorable habitats.

The municipality of Porto has showed a growing interest in environmental issues in recent years, with plans to expand and connect its green infrastructure and with some interventions based primarily on the adaptation to climate change and, on a lesser degree, on the conflict between native and exotic species (Câmara Municipal do Porto, 2020, 2021a, 2023). However, the long-term plan for biodiversity conservation is still to be concretized, as most goals, actions and timeframes are not explicit yet. Although there are prospects for creating new public green spaces, the prioritization criteria are not discernible. This research could serve as a starting point for the definition of a future habitat and biodiversity conservation plan or program, as it provides crucial information for habitat restoration or expansion in Porto and recommendations for the design and management of green areas. Figs. 5-7 can help to identify priority areas for these types of interventions on wooded or herbaceous habitats, which should preferably occur near older vegetation patches, sites that have never been built and/or places where vegetation cover was recently lost.

The use of land cover trajectories seems to be an appropriate method to infer the degree of urban habitat continuity, with the main advantage of being a relatively simple and fast procedure, if adequate data is available, and that does not require expert knowledge. This method produces spatially explicit information, and its format has a high aptitude for the transfer of knowledge between the multiple actors in the design, planning and management of urban environments, from ecologists to landscape architects, urban planners, decision makers, and other stakeholders. For this exercise, we examined the evolution of land cover in Porto using three different time points, for a total period of 72 years. Although it was not possible due to limited resources and lack of available data, for improved results we suggest expanding the timeframe into the past and using shorter time intervals, to better capture the rich history and the fast-paced dynamics of this urban area. This method of spatial prioritization for the conservation of urban habitats can be enriched with information about habitat structure, composition, species richness and abundance of multiple taxa. Other authors have also used habitat connectivity (Lv et al., 2019; Mollashahi et al., 2020; Zhao et al., 2021), climate beneficiation (Norton et al., 2015), social values (Whitehead et al., 2014) and biodiversity quality (Jalkanen et al., 2020) to define priority habitats in cities.

There are a multitude of factors influencing the outcome of land cover evolution in a city, but urban planning, design and management have a central role in paving the way for more biodiverse and sustainable cities (Hersperger et al., 2018; Huang et al., 2018). City planners, designers and managers need to acknowledge that urban areas have the potential to harbor high levels of biodiversity and need to start integrating biodiversity concerns in their plans, policies and programs (Parris et al., 2018; Zari, 2018). Planning, design and management decisions that affect the pattern and composition of the urban landscape can also impact urban biodiversity, and, hence, the quality of life of human urbanites (Zari, 2018); it should be based on frequently updated and comprehensive information, displayed in a spatial framework, which must also include ecological data (Jarvis & Young, 2005). Similarly, ecologists need to address the needs of urban planners and managers regarding biodiversity conservation, translate them into ecologically relevant questions, and provide appropriate feedback that is useful at the scale of intervention (McDonnell & Hahs, 2013). Ultimately, the transformation of the urban landscape should be supported by multidisciplinary teams, but always including landscape and/or urban ecologists (Jarvis & Young, 2005; McDonnell & Hahs, 2013; Parris et al., 2018), so that science and research translates into real life practices and interventions, moving towards the greener, biodiverse and sustainable cities of the future.

5. Conclusion

Due to their nature, urban areas face a lack of vegetation, green spaces and habitats to support their biodiversity and ultimately the health and well-being of their citizens. In this study, we were able to identify the main urban habitat patches in the city of Porto, as well as their land cover history since the mid-20th century. Although around a third of the surface of the city is currently covered by vegetation, most of the vegetated patches have experienced significant disturbance and frequent transformations between herbaceous and wooded habitats. Persistently wooded areas are rather scarce, but most are already protected in large parks and gardens. Areas continuously covered by herbaceous vegetation are even rarer in Porto and they are constantly disregarded by politicians, urban planners and even conservationists, being left out of urban planning and conservation programs.

Spatially explicit historical land cover data, such as the information presented here, can be easily integrated in spatial planning projects. It should, however, be complemented with more detailed information about habitat structure and species composition. We should aim to preserve, restore and, ideally, expand older habitats, whether occupied by trees, shrubs, or herbaceous vegetation. Older habitats allow for the formation of diverse microhabitats and the accommodation of diverse species over time, contributing for a richer biodiversity at the city or regional level. Regarding the dynamics of urban ecosystems, the conservation of herbaceous habitats seems to be particularly challenging. However, if managed properly, these habitats can provide fundamental ecosystem services and harbor important species that are becoming increasingly rare.

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CRediT authorship contribution statement

Filipa Guilherme: Conceptualization, Methodology, Formal analysis, Investigation, Writing – original draft, Writing – review & editing, Visualization, Funding acquisition. **José Alberto Gonçalves:** Resources, Writing – review & editing. **Miguel A. Carretero:** Conceptualization, Methodology, Writing – review & editing, Supervision, Funding acquisition. **Paulo Farinha Marques:** Conceptualization, Methodology, Writing – review & editing, Supervision, 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.

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Appendix A List of the mains priority areas for the conservation of habitats with higher continuity (complementing Fig. 7)

Areas comprising mainly parks and gardens	Areas comprising mainly undeveloped land and/or private property		
A – City Park	1 – Ribeira de Nevogilde	15 – Regado	29 – 25 de Abril
B – Serralves Park	2 – Ribeira da Ervilheira	16 – Burgães / Lapa	30 – Godim
C – Passeio Alegre Garden	3 – Ribeira de Aldoar	17 – Cedofeita	31 – Fontainhas
D – Pasteleira Park	4 – Mata de Aldoar	18 – Santa Catarina	32 – Quinta da China
E – Botanical Garden	5 – Pinheiro Manso	19 – Lima	33 – Formiga
F – Crystal Palace Garden	6 – Lordelo	20 – Lamas	34 – Pinheiro de Campanhã
G – Prelada Park	7 – Ribeira do Ouro	21 – Areosa Ocidental	35 – Corujeira / Monte da Bela
H – Asprela Park	8 – Via Panorâmica	22 – Areosa Oriental	36 – Tirares / Pego Negro
I – Covelo Park	9 – Ribeira de Vilar / Pena	23 – Ranha	37 – Areias
J – Nova Sintra Park	10 – Francos	24 – Currais	38 – Furamontes
K – São Roque Park	11 – Prelada	25 – Contumil	39 – Granja
L – Corujeira Park	12 – Viso / Requesende	26 – Salgueiros	40 – Azevedo
M – Quinta da Bonjóia	13 – Telheira	27 – São Roque da Lameira	41 – Quinta da Revolta
N – future Alameda de Cartes Park	14 – Azenha	28 – Lameira de Cima	42 – Rio Torto
O – Oriental Park			

References

Aerts, R., Honnay, O., & Nieuwenhuyse, A. V. (2018). Biodiversity and human health: Mechanisms and evidence of the positive health effects of diversity in nature and green spaces. *British Medical Bulletin, 127*, 5–22. https://doi.org/10.1093/bmb/ldy021

Alberti, M., Palkovacs, E. P., Roches, S. D., De Meester, L., Brans, K. I., Govaert, L., Grimm, N. B., Harris, N. C., Hendry, A. P., Schell, C. J., Szulkin, M., Munshi-South, J., Urban, M. C., & Verrelli, B. C. (2020). The Complexity of Urban Eco-evolutionary Dynamics. *BioScience, 70*(9), 772–793. https://doi.org/10.1093/biosci/biaa079

Alberti, M., & Wang, T. (2022). Detecting patterns of vertebrate biodiversity across the multidimensional urban landscape. *Ecology Letters, 25*, 1027–1045. https://doi.org/10.1111/ele.13969

Aronson, M. F. J., Lepczyk, C. A., Evans, K. L., Goddard, M. A., Lerman, S. B., MacIvor, J. S., Nilon, C. H., & Vargo, T. (2017). Biodiversity in the city: Key challenges for urban green space management. *Frontiers in Ecology and the Environment, 15*(4), 189–196. https://doi.org/10.1002/fee.1480

Bagan, H., & Yamagata, Y. (2014). Land-cover change analysis in 50 global cities by using a combination of Landsat data and analysis of grid cells. *Environmental Research Letters, 9*, Article 064015. https://doi.org/10.1088/1748-9326/9/6/064015

Borgström, S. T., Elmqvist, T., Algelstam, P., & Alfsen-Norodom, C. (2006). Scale Mismatches in Management of Urban Landscapes. *Ecology and Society, 11*(2), 16.

Brown, I. T. (2017). Managing cities as urban ecosystems: Fundamentals and a framework for Los Angeles, California. *Cities and the Environment (CATE), 10*(2), 4.

Cadenasso, M. L., Pickett, S. T. A., & Grove, J. M. (2006). Dimensions of ecosystem complexity: Heterogeneity, connectivity, and history. *Ecological Complexity, 3*, 1–12. https://doi.org/10.1016/j.ecocom.2005.07.002

Callaghan, C. T., Major, R. E., Wilshire, J. H., Martin, J. M., Kingsford, R. T., & Cornwell, W. K. (2019). Generalists are the most urban-tolerant of birds: A phylogenetically controlled analysis of ecological and life history traits using a novel continuous measure of bird responses to urbanization. *Oikos, 128*, 845–858. https://doi.org/10.1111/oik.06158

2022 AMP. (2022). *AMPorto*. http://portal.amp.pt/pt/ (accessed 21 March 2022).

Câmara Municipal do Porto. (2020). *Environment quick guide*. Porto: Câmara Municipal do Porto.

Câmara Municipal do Porto. (2021a). *Plano Diretor Municipal. Planta de Ordenamento. Carta da Estrutura Ecológica Municipal*. Porto: Câmara Municipal do Porto.

Cameron, R. W. F., Brindley, P., Mears, M., McEwan, K., Ferguson, F., Sheffield, D., Jorgensen, A., Riley, J., Goodrick, J., Ballard, L., & Richardson, M. (2020). Where the wild things are! Do urban green spaces with greater avian biodiversity promote more positive emotions in humans? *Urban Ecosystems, 23*, 301–317.

Chapman, M. G., & Underwood, A. J. (2009). Comparative effects of urbanization in marine and terrestrial habitats. In M. J. McDonnell, A. K. Hahs, & J. H. Breuste (Eds.), *Ecology of cities and towns: A comparative approach* (pp. 51–70). Cambridge University Press.

Chollet, S., Brabant, C., Tessier, S., & Jung, V. (2018). From urban lawns to urban meadows: Reduction of mowing frequency increases plant taxonomic, functional and phylogenetic diversity. *Landscape and Urban Planning, 180*, 121–124. https://doi.org/10.1016/j.landurbplan.2018.08.009

Cooper, D. S., Wood, E. M., Katz, N. D., Superfisky, K., Osborn, F. M., Novoselov, A., Tarczynski, J., & Bacasen, L. K. (2021). Large Cities Fall Behind in "Neighborhood Biodiversity". *Frontiers in Conservation Science, 2*(October), 1–7. https://doi.org/10.3389/fcosc.2021.734931

Doody, B. J., Sullivan, J. J., Meurk, C. D., Stewart, G. H., & Perkins, H. C. (2010). Urban realities: The contribution of residential gardens to the conservation of urban forest remnants. *Biodiversity and Conservation, 19*, 1385–1400. https://doi.org/10.1007/s10531-009-9768-2

Ducatez, S., Sayol, F., Sol, D., & Lefebvre, L. (2018). Are urban vertebrates city specialists, artificial habitat exploiters, or environmental generalists? *Integrative and Comparative Biology, 58*(5), 929–938. https://doi.org/10.1093/icb/icy101

Câmara Municipal do Porto. (2023). *Estratégia de expansão | Ambiente*. https://ambiente.cm-porto.pt/estrutura-verde/estrategia-de-expansao (accessed 15 October 2023).

European Environment Agency. (2021). *Abundance and distribution of selected species in Europe*. https://www.eea.europa.eu/ims/abundance-and-distribution-of-selected (accessed 30 September 2022).

Farinha-Marques, P., Fernandes, C., Guilherme, F., Lameiras, J. M., Alves, P., & Bunce, R. G. H. (2017). Urban Habitats Biodiversity Assessment (UrHBA): A standardized procedure for recording biodiversity and its spatial distribution in urban environments. *Landscape Ecology, 32*, 1753–1770. https://doi.org/10.1007/s10980-017-0554-3

Filibeck, G., Petrella, P., & Cornelini, P. (2016). All ecosystems look messy, but some more so than others: A case-study on the management and acceptance of Mediterranean urban grasslands. *Urban Forestry & Urban Greening, 15*, 32–39. https://doi.org/10.1016/j.ufug.2015.11.005

Foster, D., Swanson, F., Aber, J., Burke, I., Brokaw, N., Tilman, D., & Knapp, A. (2003). The Importance of land-use legacies to ecology and conservation. *BioScience, 53*(1), 77–88. https://doi.org/10.1641/0006-3568(2003)053[0077:TIOLUL]2.0.CO;2

Frondoni, R., Mollo, B., & Capotorti, G. (2011). A landscape analysis of land cover change in the Municipality of Rome (Italy): Spatio-temporal characteristics and ecological implications of land cover transitions from 1954 to 2001. *Landscape and Urban Planning, 100*, 117–128. https://doi.org/10.1016/j.landurbplan.2010.12.002

Gaertner, M., Wilson, J. R. U., Cadotte, M. W., MacIvor, J. S., Zenni, R. D., & Richardson, D. M. (2017). Non-native species in urban environments: Patterns, processes, impacts and challenges. *Biological Invasions, 19*, 3461–3469. https://doi.org/10.1007/s10530-017-1598-7

Garrard, G. E., Williams, N. S. G., Mata, L., Thomas, J., & Bekessy, S. A. (2018). Biodiversity sensitive urban design. *Conservation Letters, 11*(2), 1–10. https://doi.org/10.1111/conl.12411

Gbanie, S. P., Griffin, A. L., & Thornton, A. (2018). Impacts on the urban environment: Land cover change trajectories and landscape fragmentation in post-war Western Area, Sierra Leone. *Remote Sensing, 10*, 129. https://doi.org/10.3390/rs10010129

Goddard, M. A., Dougill, A. J., & Benton, T. G. (2010). Scaling up from gardens: Biodiversity conservation in urban environments. *Trends in Ecology and Evolution, 25*(2), 90–98. https://doi.org/10.1016/j.tree.2009.07.016

Guilherme, F., Carretero, M. A., & Farinha-Marques, P. (2015). Porto. In J. Kelcey (Ed.), *Vertebrates and invertebrates of European cities: Selected non-avian fauna* (pp. 95–117). Springer. 10.1007/978-1-4939-1698-6_4.

Guilherme, F., Moreno, E. G., Gonçalves, J. A., Carretero, M. A., & Farinha-Marques, P. (2022). Looking Closer at the Patterns of Land Cover in the City of Porto, Portugal, between 1947 and 2019—A Contribution for the Integration of Ecological Data in Spatial Planning. *Land, 11*, 1828. https://doi.org/10.3390/land11101828

Güneralp, B., Reba, M., Hales, B. U., Wentz, E. A., & Seto, K. C. (2020). Trends in urban land expansion, density, and land transitions from 1970 to 2010: A global synthesis. *Environmental Research Letters, 15*, Article 044015. https://doi.org/10.1088/1748-9326/ab6669

Gustavsson, E., Lennartsson, T., & Emanuelsson, M. (2007). Land use more than 200 years ago explains current grassland plant diversity in a Swedish agricultural landscape. *Biological Conservation, 138*, 47–59. https://doi.org/10.1016/j.biocon.2007.04.004

Haines-Young, R. (2009). Land use and biodiversity relationships. *Land Use Policy, 26S*, S178–S186. https://doi.org/10.1016/j.landusepol.2009.08.009

Han, D., Wang, C., Sun, Z., She, J., Yin, L., Bian, Q., & Han, W. (2022). Microhabitat preferences of butterflies in urban parks: Both vegetation structure and resources are decisive. *Urban Forestry & Urban Greening, 71*, Article 127552. https://doi.org/10.1016/j.ufug.2022.127552

Hanski, I. (1999). Habitat connectivity, habitat continuity, and metapopulations in dynamic landscapes. *Oikos, 87*(2), 209–219. https://doi.org/10.2307/3546736

Harrison, P. A., Berry, P. M., Simpson, G., Haslett, J. R., Blicharska, M., Bucur, M., Dunford, R., Egoh, B., Garcia-Llorente, M., Geamănă, N., Geertsema, W., Lommelen, E., Meiresonne, L., & Turkelboom, F. (2014). Linkages between biodiversity attributes and ecosystem services: A systematic review. *Ecosystem Services, 9*, 191–203. https://doi.org/10.1016/j.ecoser.2014.05.006

Helden, A. J., Morely, G. J., Davidson, G. L., & Turner, E. C. (2018). What can we do for urban insect biodiversity? Applying lessons from ecological research. *Zoosymposia, 12*, 51–63. https://doi.org/10.1603/ice.2016.109052

Hersperger, A. M., Oliveira, E., Pagliarin, S., Palka, G., Verburg, P., Bolliger, J., & Grădinaru, S. (2018). Urban land-use change: The role of strategic spatial planning. *Global Environmental Change, 51*, 32–42. https://doi.org/10.1016/j.gloenvcha.2018.05.001

Huang, C.-W., McDonald, R. I., & Seto, K. C. (2018). The importance of land governance for biodiversity conservation in an era of global urban expansion. *Landscape and Urban Planning, 173*, 44–50. https://doi.org/10.1016/j.landurbplan.2018.01.011

Huang, J.-L., Andrello, M., Martensen, A. C., Saura, S., Liu, D.-F., He, J.-H., & Fortin, M.-J. (2020). Importance of spatio–temporal connectivity to maintain species experiencing range shifts. *Ecography, 43*, 591–603. https://doi.org/10.1111/ecog.04716

Hüse, B., Szabó, S., Deák, B., & Tóthmérész, B. (2016). Mapping an ecological network of green habitat patches and their role in maintaining urban biodiversity in and around Debrecen city (Eastern Hungary). *Land Use Policy, 57*, 574–581. https://doi.org/10.1016/j.landusepol.2016.06.026

ICLEI. (2020). *Policy Activation Brief, Fostering Urban Biodiversity through Local Actions*.

Inoue, T., Yaida, Y. A., Uehara, Y., Katsuhara, K. R., Kawai, J., Takashima, K., Ushimaru, A., & Kenta, T. (2021). The effects of temporal continuities of grasslands on the diversity and species composition of plants. *Ecological Research, 36*, 24–31. https://doi.org/10.1111/1440-1703.12169

Ives, C. D., Lentini, P. E., Threlfall, C. G., Ikin, K., Shanahan, D. F., Garrard, G. E., Bekessy, S. A., Fuller, R. A., Mumaw, L., Rayner, L., Rowe, R., Valentine, L. E., & Kendal, D. (2016). Cities are hotspots for threatened species. *Global Ecology and Biogeography, 25*, 117–126. https://doi.org/10.1111/geb.12404

Jalkanen, J., Vierikko, K., & Moilanen, A. (2020). Spatial prioritization for urban Biodiversity Quality using biotope maps and expert opinion. *Urban Forestry & Urban Greening, 49*, Article 126586. https://doi.org/10.1016/j.ufug.2020.126586

Janssen, P., Fuhr, M., Cateau, E., Nusillard, B., & Bouget, C. (2017). Forest continuity acts congruently with stand maturity in structuring the functional composition of saproxylic beetles. *Biological Conservation, 205*, 1–10. https://doi.org/10.1016/j.biocon.2016.11.021

Jarvis, P. J., & Young, C. H. (2005). The mapping of urban habitat and its evaluation. *Urban Forum of the United Kingdom Man and the Biosphere Programme*.

Johnson, A. L., Borowy, D., & Swan, C. M. (2018). Land use history and seed dispersal drive divergent plant community assembly patterns in urban vacant lots. *Journal of Applied Ecology, 55*, 451–460. https://doi.org/10.1111/1365-2664.12958

Jonsson, M., Bengtsson, J., Moen, J., Gamfeldt, L., & Snäll, T. (2020). Stand age and climate influence forest ecosystem service delivery and multifunctionality. *Environmental Research Letters, 15*, Article 0940a8.

Klaus, V. H. (2013). Urban grassland restoration: A neglected opportunity for biodiversity conservation. *Restoration Ecology, 21*(6), 665–669. https://doi.org/10.1111/rec.12051

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

Kowarik, I., & von der Lippe, M. (2018). Plant population success across urban ecosystems: A framework to inform biodiversity conservation in cities. *Journal of Applied Ecology, 55*(5), 2354–2361. https://doi.org/10.1111/1365-2664.13144

Krauss, J., Bommarco, R., Guardiola, M., Heikkinen, R. K., Helm, A., Kuussaari, M., Lindborg, R., Öckinger, E., Partel, M., Pino, J., Pöyry, J., Raatikainen, K. M., Sang, A., Stefanescu, C., Teder, T., Zobel, M., & Steffan-Dewenter, I. (2010). Habitat fragmentation causes immediate and time-delayed biodiversity loss at different trophic levels. *Ecology Letters, 13*, 597–605. https://doi.org/10.1111/j.1461-0248.2010.01457.x

Kremer, P., Hamstead, Z., Haase, D., McPhearson, T., Frantzeskaki, N., Andersson, E., … Elmqvist, T. (2016). Key insights for the future of urban ecosystem services research. *Ecology and Society, 21*(2), 29.

Le Roux, D. S., Ikin, K., Lindenmayer, D. B., Blanchard, W., Manning, A. D., & Gibbons, P. (2014). Reduced availability of habitat structures in urban landscapes: Implications for policy and practice. *Landscape and Urban Planning, 125*, 57–64. https://doi.org/10.1016/j.landurbplan.2014.01.015

Lennert, J., Farkas, J. Z., Kovács, A. D., Molnár, A., Módos, R., Baka, D., & Kovács, Z. (2020). Measuring and predicting long-term land cover changes in the functional urban area of Budapest. *Sustainability, 12*, 3331. https://doi.org/10.3390/SU12083331

Lira, P. K., de Souza Leite, M., & Metzger, J. P. (2019). Temporal lag in ecological responses to landscape change: Where are we now? *Current Landscape Ecology Reports, 4*, 70–82. https://doi.org/10.1007/s40823-019-00040-w

Liu, C. L. C., Kuchma, O., & Krutovsky, K. V. (2018). Mixed-species versus monocultures in plantation forestry: Development, benefits, ecosystem services and perspectives for the future. *Global Ecology and Conservation, 15*, e00419.

Liu, D., & Cai, S. (2012). A spatial-temporal modeling approach to reconstructing land-cover change trajectories from multi-temporal satellite imagery. *Annals of the Association of American Geographers, 102*(6), 1329–1347. https://doi.org/10.1080/00045608.2011.596357

Löfvenhaft, K., Runborg, S., & Sjögren-Gulve, P. (2004). Biotope patterns and amphibian distribution as assessment tools in urban landscape planning. *Landscape and Urban Planning, 68*, 403–427. https://doi.org/10.1016/S0169-2046(03)00154-3

Lõhmus, K., & Liira, J. (2013). Old rural parks support higher biodiversity than forest remnants. *Basic and Applied Ecology, 14*, 165–173.

Łopucki, R., & Kiersztyn, A. (2015). Urban green space conservation and management based on biodiversity of terrestrial fauna – A decision support tool. *Urban Forestry & Urban Greening, 14*, 508–518. https://doi.org/10.1016/j.ufug.2015.05.001

Lv, Z., Yang, J., Wielstra, B., Wei, J., Xu, F., & Si, Y. (2019). Prioritizing green spaces for biodiversity conservation in Beijing based on habitat network connectivity. *Sustainability, 11*, 2042. https://doi.org/10.3390/su11072042

Madureira, H., Andresen, T., & Monteiro, A. (2011). Green structure and planning evolution in Porto. *Urban Forestry & Urban Greening, 10*(2), 141–149. https://doi.org/10.1016/j.ufug.2010.12.004

Martensen, A. C., Saura, S., & Fortin, M.-J. (2017). Spatio-temporal connectivity: Assessing the amount of reachable habitat in dynamic landscapes. *Methods in Ecology and Evolution, 8*, 1253–1264. https://doi.org/10.1111/2041-210X.12799

McDonnell, M. J., & Hahs, A. K. (2013). The future of urban biodiversity research: Moving beyond the "low-hanging fruit". *Urban Ecosystems, 16*, 397–409. https://doi.org/10.1007/s11252-013-0315-2

McKinney, M. L. (2002). Urbanization, biodiversity and conservation. *BioScience, 52*(10), 883–890.

Millennium Ecosystem Assessment. (2005). Ecosystems and human well-being: Synthesis. *Island Press*. https://doi.org/10.1088/1751-8113/44/8/085201

Mollashahi, H., Szymura, M., & Szymura, T. H. (2020). Connectivity assessment and prioritization of urban grasslands as a helpful tool for effective management of urban ecosystem services. *PLoS ONE, 15*(12), Article e0244452. https://doi.org/10.1371/journal.pone.0244452

2022 INE. (2022). *Plataforma de divulgação dos Censos 2021 – Resultados Provisórios*. https://www.ine.pt/scripts/db_censos_2021.html (accessed on 21 March 2022).

Müller, N., Ignatieva, M., Nilon, C. H., Werner, P., & Zipperer, W. C. (2013). Patterns and trends in urban biodiversity and landscape design. In T. Elmqvist, M. Fragkias, J. Goodness, B. Güneralp, P. J. Marcotullio, R. I. McDonald, S. Parnell, M. Schewenius, M. Sendstad, K. C. Seto, & C. Wilkinson (Eds.), *Urbanization, biodiversity and ecosystem services: Challenges and opportunities*. Springer. 10.1007/978-94-007-7088-1.

Nielsen, A. B., van den Bosch, M., Maruthaveeran, S., & van den Bosch, C. K. (2014). Species richness in urban parks and its drivers: A review of empirical evidence. *Urban Ecosystems, 17*, 305–327. https://doi.org/10.1007/s11252-013-0316-1

Nordén, B., Dahlberg, A., Brandrud, T. E., Fritz, Ö., Ejrnaes, R., & Ovaskainen, O. (2014). Effects of ecological continuity on species richness and composition in forests and woodlands: A review. *Ecoscience, 21*(1), 34–45. https://doi.org/10.2980/21-1-3667

Norton, B. A., Bending, G. D., Clark, R., Corstanje, R., Dunnett, N., Evans, K. L., Grafius, D. R., Gravestock, E., Grice, S. M., Harris, J. A., Hilton, S., Hoyle, H., Lim, E., Mercer, T. G., Pawlett, M., Pescott, O. L., Richards, J. P., Southon, G. E., & Warren, P. H. (2019). Urban meadows as an alternative to short mown grassland: Effects of composition and height on biodiversity. *Ecological Applications, 29*(6), 1095–1115. https://doi.org/10.1002/eap.1946

Norton, B. A., Coutts, A. M., Livesley, S. J., Harris, R. J., Hunter, A. M., & Williams, N. S. G. (2015). Planning for cooler cities: A framework to prioritise green infrastructure to mitigate high temperatures in urban landscapes. *Landscape and Urban Planning, 134*, 127–138. https://doi.org/10.1016/j.landurbplan.2014.10.018

Onandia, G., Schittko, C., Ryo, M., Bernard-Verdier, M., Heger, T., Joshi, J., Kowarik, I., & Gessler, A. (2019). Ecosystem functioning in urban grasslands: The role of biodiversity, plant invasions and urbanization. *PLoS ONE, 14*(11), Article e0225438. https://doi.org/10.1371/journal.pone.0225438

Opdam, P., Nassauer, J. I., Wang, Z., Albert, C., Bentrup, G., Castella, J. C., McAlpine, C., Liu, J., Sheppard, S., & Swaffield, S. (2013). Science for action at the local landscape scale. *Landscape Ecology, 28*, 1439–1445. https://doi.org/10.1007/s10980-013-9925-6

Ossola, A., Cadenasso, M. L., & Meineke, E. K. (2021). Valuing the role of time in urban ecology. *Frontiers in Ecology and Evolution, 9*, Article 620620. https://doi.org/10.3389/fevo.2021.620620

Parris, K. M., Amati, M., Bekessy, S. A., Dagenais, D., Fryd, O., Hahs, A. K., Hes, D., Imberger, S. J., Livesley, S. J., Marshall, A. J., Rhodes, J. R., Threlfall, C. G., Tingley, R., van der Ree, R., Walsh, C. J., Wilkerson, M. L., & Williams, N. S. G. (2018). The seven lamps of planning for biodiversity in the city. *Cities, 83*, 44–53. https://doi.org/10.1016/j.cities.2018.06.007

Pedersen, A. O., Nyhuus, S., Blindheim, T., & Krog, O. M. W. (2004). Implementation of a GIS-based management tool for conservation of biodiversity within the municipality of Oslo, Norway. *Landscape and Urban Planning, 68*(4), 429–438. https://doi.org/10.1016/S0169-2046(03)00148-8

Peel, M. C., Finlayson, B. L., & McMahon, T. A. (2007). Updated world map of the Köppen-Geiger climate classification. *Hydrology and Earth System Sciences, 11*, 1633–1644. https://doi.org/10.5194/hess-11-1633-2007

Pohjanmies, T., Eyvindson, K., Triviño, M., Bengtsson, J., & Mönkkönen, M. (2021). Forest multifunctionality is not resilient to intensive forestry. *European Journal of Forest Research, 140*, 537–549. https://doi.org/10.1007/s10342-020-01348-7

Raduła, M. W., Szymura, T. H., Szymura, M., Swacha, G., & Kącki, Z. (2020). Effect of environmental gradients, habitat continuity and spatial structure on vascular plant species richness in semi-natural grasslands. *Agriculture, Ecosystems and Environment, 300*, Article 106974. https://doi.org/10.1016/j.agee.2020.106974

Radwan, T. M., Blackburn, G. A., Whyatt, J. D., & Atkinson, P. M. (2021). Global land cover trajectories and transitions. *Scientific Reports, 11*, 12814. https://doi.org/10.1038/s41598-021-92256-2

Ramalho, C. E., & Hobbs, R. J. (2012). Time for a change: Dynamic urban ecology. *Trends in Ecology and Evolution, 27*(3), 179–188. https://doi.org/10.1016/j.tree.2011.10.008

Ribeiro, R., Sillero, N., Carretero, M. A., & Llorente, G. A. (2009). Herp-Robinsons: Amphibians, reptiles, metropolitan areas and island ecology. *Conservation Biology and beyond: From Science to Practice. 2nd European Congress of Conservation Biology. Book of Abstracts*.

Richards, D. R., & Belcher, R. N. (2020). Global changes in urban vegetation cover. *Remote Sensing, 12*, 23. https://doi.org/10.3390/RS12010023

Sang, A., Teder, T., Helm, A., & Pärtel, M. (2010). Indirect evidence for an extinction debt of grassland butterflies half century after habitat loss. *Biological Conservation, 143*, 1405–1413. https://doi.org/10.1016/j.biocon.2010.03.015

Schebella, M. F., Weber, D., Schultz, L., & Weinstein, P. (2019). The wellbeing benefits associated with perceived and measured biodiversity in Australian Urban Green Spaces. *Sustainability, 11*, 802. https://doi.org/10.3390/su11030802

Secretariat of the Convention on Biological Diversity. (2012). *Cities and biodiversity outlook. Action and policy*. Montreal: Secretariat of the Convention on Biological Diversity.

Siitonen, J., & Saaristo, L. (2000). Habitat requirements and conservation of Pytho kolwensis, a beetle species of old-growth boreal forest. *Biological Conservation, 94*, 211–220. https://doi.org/10.1016/S0006-3207(99)00174-3

Simkin, R. D., Seto, K. C., McDonald, R. I., & Jetz, W. (2022). Biodiversity impacts and conservation implications of urban land expansion projected to 2050. *Proceedings of the National Academy of Sciences, 119*(12). https://doi.org/10.1073/pnas.2117297119

Spotswood, E. N., Beller, E. E., Grossinger, R., Grenier, J. L., Heller, N. E., & Aronson, M. F. J. (2021). The biological deserts fallacy: Cities in their landscapes contribute more than we think to regional biodiversity. *BioScience, 71*(2), 148–160. https://doi.org/10.1093/biosci/biaa155

Swan, C. M., Brown, B., Borowy, D., Cavender-Bares, J., Jeliazkov, A., Knapp, S., Lososová, Z., Padullés Cubino, J., Pavoine, S., Ricotta, C., & Sol, D. (2021). A framework for understanding how biodiversity patterns unfold across multiple spatial scales in urban ecosystems. *Ecosphere, 12*(7), Article e03650. https://doi.org/10.1002/ecs2.3650

Tello, E., Marull, J., Padr, R., Cattaneo, C., & Coll, F. (2020). The Loss of Landscape Ecological Functionality in the Barcelona Province (1956–2009): Could Land-Use History Involve a Legacy for Current Biodiversity? *Sustainability, 12*, 2238.

Thompson, K., Austin, K. C., Smith, R. M., Warren, P. H., Angold, P. G., & Gaston, K. J. (2003). Urban domestic gardens (I): Putting small-scale plant diversity in context. *Journal of Vegetation Science, 14*, 71–78.

Threlfall, C. G., Mata, L., Mackie, J. A., Hahs, A. K., Stork, N. E., Williams, N. S. G., & Livesley, S. J. (2017). Increasing biodiversity in urban green spaces through simple vegetation interventions. *Journal of Applied Ecology, 54*, 1874–1883. https://doi.org/10.1111/1365-2664.12876

Tilman, D., Clark, M., Williams, D. R., Kimmel, K., Polasky, S., & Packer, C. (2017). Future threats to biodiversity and pathways to their prevention. *Nature, 546*, 73–81. https://doi.org/10.1038/nature22900

Uchida, K., Blakey, R. V., Burger, J. R., Cooper, D. S., Niesner, C. A., & Blumstein, D. T. (2021). Urban biodiversity and the importance of scale. *Trends in Ecology & Evolution, 36*(2), 123–131. https://doi.org/10.1016/j.tree.2020.10.011

Uroy, L., Alignier, A., Mony, C., Foltête, J.-C., & Ernoult, A. (2021). How to assess the temporal dynamics of landscape connectivity in ever-changing landscapes: A literature review. *Landscape Ecology, 36*, 2487–2504. https://doi.org/10.1007/s10980-021-01277-9

Vandewalle, M., Purschke, O., de Bello, F., Reitalu, T., Prentice, H. C., Lavorel, S., Johansson, L. J., & Sykes, M. T. (2014). Functional responses of plant communities to management, landscape and historical factors in semi-natural grasslands. *Journal of Vegetation Science, 25*, 750–759. https://doi.org/10.1111/jvs.12126

Vasquez, A. V., & Wood, E. M. (2022). Urban parks are a refuge for birds in park-poor areas. *Frontiers in Ecology and Evolution, 10*, Article 958572. https://doi.org/10.3389/fevo.2022.958572

Veldman, J. W., Buisson, E., Durigan, G., Fernandes, G. W., Le Stradic, S., Mahy, G., Negreiros, D., Overbeck, G. E., Veldman, R. G., Zaloumis, N. P., Putz, F. E., & Bond, W. J. (2015). Toward an old-growth concept for grasslands, savannas, and woodlands. *Frontiers in Ecology and the Environment, 13*(3), 154–162. https://doi.org/10.1890/140270

Whitehead, A. L., Kujala, H., Ives, C. D., Gordon, A., Lentini, P. E., Wintle, B. A., Nicholson, E., & Raymond, C. M. (2014). Integrating biological and social values when prioritizing places for biodiversity conservation. *Conservation Biology, 28*(4), 992–1003. https://doi.org/10.1111/cobi.12257

Wilkinson, C., Sendstad, M., Parnell, S., & Schewenius, M. (2013). Urban governance of biodiversity and ecosystem services. In T. Elmqvist, M. Fragkias, J. Goodness, B. Güneralp, P. J. Marcotullio, R. I. McDonald, S. Parnell, M. Schewenius, M. Sendstad, K. C. Seto, & C. Wilkinson (Eds.), *Urbanization, Biodiversity and Ecosystem Services: Challenges and Opportunities*. Springer. 10.1007/978-94-007-7088-1.

Wittig, R., & Schreiber, K. F. (1983). A quick method for assessing the importance of open spaces in towns for urban nature conservation. *Biological Conservation, 26*, 57–64. https://doi.org/10.1016/0006-3207(83)90048-4

Wood, E., Harsant, A., Dallimer, M., Cronin de Chavez, A., McEachan, R. R. C., & Hassall, C. (2018). Not All Green Space Is Created Equal: Biodiversity Predicts Psychological Restorative Benefits From Urban Green Space. *Frontiers in Psychology, 9*, 2320. https://doi.org/10.3389/fpsyg.2018.02320

Xie, L., & Bulkeley, H. (2020). Nature-based solutions for urban biodiversity governance. *Environmental Science and Policy, 110*, 77–87. https://doi.org/10.1016/j.envsci.2020.04.002

Zaitunah, A., Samsuri, & Sahara, F. (2021). Mapping and assessment of vegetation cover change and species variation in Medan, North Sumatra. *Heliyon, 7*, Article e07637. https://doi.org/10.1016/j.heliyon.2021.e07637

Zari, M. P. (2018). The importance of urban biodiversity – an ecosystem services approach. *Biodiversity International Journal*, *2*(4), 357–360. 10.15406/bij.2018.02.00087.

Zhao, H., Liu, D., Li, F., Liu, X., Niu, J., He, J., & Liu, Y. (2021). Incorporating spatio-temporal connectivity for prioritized conservation of individual habitat patches in a dynamic landscape. *Ecological Indicators, 124*, Article 107414. https://doi.org/10.1016/j.ecolind.2021.107414

Zhou, W., Fisher, B., & Pickett, S. T. A. (2019). Cities are hungry for actionable ecological knowledge. *Frontiers in Ecology and the Environment, 17*, 135. https://doi.org/10.1002/fee.2021

Zubair, O. A. (2021). Investigating urban growth and the dynamics of urban land cover change using remote sensing data and landscape metrics. *Papers in Applied Geography, 7*(1), 67–81. https://doi.org/10.1080/23754931.2020.1817136